# LYNX (*FELIS LYNX*) OF RIDING MOUNTAIN NATIONAL PARK: AN ASSESSMENT OF HABITAT AVAILABILITY AND POPULATION VIABILITY

**BY**

**MARCY NYLEN-NEMETCHEK**

A thesis/practicum submitted to
the Faculty of Graduate Studies
in Partial Fulfilment of the Requirements
for the Degree of

MASTER OF NATURAL RESOURCES MANAGEMENT

Natural Resources Institute
University of Manitoba
Winnipeg, Manitoba
R3T 2N2

August 1999

0-612-41666-6

THE UNIVERSITY OF MANITOBA

FACULTY OF GRADUATE STUDIES
*****
COPYRIGHT PERMISSION PAGE

# LYNX (FELIS LYNX) OF RIDING MOUNTAIN NATIONAL PARK: AN ASSESSMENT OF HABITAT AVAILABILITY AND POPULATION VIABILITY

BY

MARCY NYLEN-NEMETCHEK

A Practicum submitted to the Faculty of Graduate Studies of The University

of Manitoba in partial fulfillment of the requirements of the degree

of

MASTER OF NATURAL RESOURCES MANAGEMENT

# ABSTRACT

An assessment of lynx (*Felis lynx*) habitat availability and population viability was performed for Riding Mountain National Park (RMNP), Manitoba, Canada from 1997 to 1999. The purpose of this study was to demonstrate and verify a methodology for estimating the viability of lynx as a part of the ecological assessment of RMNP.

Verification of a lynx Habitat Suitability Index (HSI) model was performed for RMNP. Site specific lynx home-range-level attributes were obtained for three components of lynx habitat; foraging, denning, and interspersion, and a home-range-level habitat map was generated for RMNP. Snow-tracking was used as a practical means of verifying the HSI model. When using HSI model output at a fine resolution (maps of HSI classes in increments of 0.10) lynx tracks occurred less than expected in the lowest HSI class ($G = 30.974$, $df = 8$, $p = 0.05$) and all other HSI classes were used as expected. When a coarser HSI resolution was used (HSI's mapped to classes of 0.30), lynx tracks occurred less than expected in the lowest HSI class ($G = 11.402$, $df = 2$, $p = 0.05$) and the highest HSI class was used more than expected ($p = 0.20$). A number of factors including observer bias, poor tracking conditions, and tracking time frame may have influenced these results.

Lynx forage habitat availability appeared to be the limiting factor on lynx viability in RMNP based on home-range-level foraging, denning, and interspersion habitat maps. Habitat manipulations are recommended to increase the quality and quantity of lynx forage-type habitat in RMNP through means such as prescribed burns.

Population viability for lynx was assessed for RMNP using a modelling framework. This framework was used to develop a map of viable, marginal, and non-viable home ranges and to index the number of lynx in RMNP. Based on current habitat, RMNP is estimated to support 427 lynx home ranges whereby 170 are viable, 255 are marginal, and 2 are non-viable. A viable home range is believed to consistently contribute to population viability ($\lambda > 1$) even when resources become limiting. Snow-tracking data showed that groups of lynx tracks were in closer proximity to viable home ranges than solitary lynx tracks ($U = 717$, n (group of lynx) = 28 and n (single lynx) = 79,

$p = 0.005$). Assuming that groups of lynx during mid-winter represented family groups, it appeared that the viability model predicted areas that favoured lynx productivity and survival in RMNP.

Historical lynx data, habitat alteration caused by human intervention, and vegetation successional trajectories were also examined and their relevance to lynx viability was discussed. This information will aid Park managers in understanding the population viability goals for RMNP lynx in the future.

# ACKNOWLEDGMENTS

I would like to thank Riding Mountain National Park and the Riding Mountain Biosphere Reserve who provided the funding support for this research. Logistical support supplied by the Park was tremendous in terms of technical and personnel contributions. I would like to specifically thank the Park's Data Management Specialist, Sean Frey, for the Geographical Information System (GIS) support, and the Warden Service of RMNP for their assistance in data collection. Warden Service members who assisted in the data collection were Patrick Rousseau, Jacques Saquet, Gordon Pylypuik, Sharon Erwin, Debbie Kilfoyle, Laird Linklater, Glenn Schmidt, Barry Desmoulin, Terry Hoggins, Roger Baird, Clem Labbe, Art Cochrane, Mark Kochems, and Gabe Boros.

Boise Cascade Corporation provided extensive in-kind support both by technical and personnel contributions. Additional support was provided by the Alumni Association of the University of Manitoba (U of M), the Graduate Student's Association (U of M), and Graduate Studies (U of M) by their financial contributions to send me to the 4th Annual Wildlife Society Conference in Snowmass, Colorado to present this study. The University of Winnipeg also provided me with some equipment necessary for the field work.

I am very grateful to my committee members: Dr. Richard Baydack, Dr. Gary Roloff, Warden Patrick Rousseau, Mr. David Walker, and Mr. Jack Dubois.
Dr. Baydack, my student advisor, provided guidance throughout the thesis/practicum and was a source of constant encouragement. Mr. David Walker provided constructive criticism and offered valid suggestions to improve the study. Mr. Jack Dubois provided assistance that helped initiate the study. A special thanks goes out to Dr. Gary Roloff who spent countless hours guiding and editing my work, and without his contributions, the study couldn't have been done. In addition, Warden Patrick Rousseau initiated the study, spent weeks gathering the data with me, provided most of the logistical means to do the field work, and was a constant source of humour.

I would also like to thank Alan Tokaryk and Sara Goulet who volunteered their time to assist with data collection, Geraldine David who helped me prepare for my oral defence by providing computer assistance, Debi Forlanski who provided GIS support, and Wybo Vanderschuit who provided his vegetation expertise.

I would finally like to thank my family, Glenn Nylen, Dianne Nylen, and Kerri Nylen for their encouragement and understanding, and my brother, Jason Nylen whose memory inspired me. I would especially like to thank my husband, Donovan Nemetchek who stood by me, encouraged me, had faith in me, and provided unconditional support.

# TABLE OF CONTENTS

**Page**

## LIST OF APPENDICES

# LIST OF TABLES

# LIST OF FIGURES

# CHAPTER 1

# STUDY OVERVIEW

## INTRODUCTION

In Canadian National Parks the "maintenance of ecological integrity through the protection of natural resources shall be the first priority" (National Parks Act 1989). Thus, the concept of ecological integrity, though difficult to define, has been instilled into Canadian National Park management by legislation. Although there have been innumerable attempts to define the concept of ecological integrity, Woodley's (1993) definition will form the basis for this study:

> *"Ecological integrity is defined as a state of ecosystem development that is optimized for its geographic location. For parks and protected areas this optimal state has been referred to by such terms as natural, naturally evolving, pristine and untouched. It implies that ecosystem structures and functions are unimpaired by human-caused stresses, that native species are present at viable population levels and, within successional limits, that the system is likely to persist. Ecosystems with integrity do not exhibit the trends associated with stressed ecosystems. Parks and protected areas are part of larger ecosystems and determinations of integrity in national parks must consider these larger ecosystems."*

To help address the requirement of maintaining ecological integrity, Canada's federal parks are developing an ecosystem-based approach to management (Department of Canadian Heritage 1994, Riding Mountain National Park Round Table 1996). Riding Mountain National Park (RMNP), therefore, developed an Ecosystem Conservation Plan (Ecosystem Conservation Plan Team 1997). The Ecosystem Conservation Plan compliments Canadian park policy in stating that management decisions must be based upon research and science (Department of Canadian Heritage 1994). The purpose of the Ecosystem Conservation Plan is to "protect, restore and monitor...natural heritage within the Park to ensure ecological integrity" (Ecosystem Conservation Plan Team 1997). The plan set objectives for ecosystem management and protection which included establishing and implementing species inventories and monitoring programs (Ecosystem Conservation Plan Team 1997). Riding Mountain National Park must also be managed in a context that is broader than its political boundaries (Riding Mountain National Park Round Table 1996). Since ecosystems do not abruptly cease at the Park's jurisdictional boundaries, information sharing, cooperation, and partnerships with other individuals and organizations are vital to effective ecosystem management (Haufler et al. 1996).

The identification and monitoring of ecological integrity is a primary goal of the Ecosystem Conservation Plan (Ecosystem Conservation Plan Team 1997). Ecological integrity may be analyzed in a variety of ways, including consideration of habitat potential and species viability (Woodley 1993). Lynx (*Felis lynx*) were chosen as one species to help identify and monitor ecological integrity of RMNP for a variety of reasons. They are indigenous to the area, they require a diversity of forest successional stages for various parts of their life cycle (Koehler and Aubry 1994), and they are a top-level carnivore that closely associates with the snowshoe hare (*Lepus americanus*) cycle thus, lynx viability exhibits a strong link to the integrity of the food web.

## STUDY AREA

### Riding Mountain National Park - Historical Setting

Approximately 12,000 years ago, glaciers began retreating from the southwestern regions of Manitoba, including the RMNP area (Pettipas 1970, Colwill and Jamieson 1972, Buchner et al. 1983, Ecosystem Conservation Plan Team 1997). Twelve thousand to 10,000 years ago, RMNP was colonized by white spruce (*Picea glauca*) forest (Ecosystem Conservation Plan Team 1997). Ten thousand to 3,000 years ago, there was a grassland expansion (Colwill and Jamieson 1972, Buchner et al. 1983) and roughly 2,500 to 3,000 years ago to present, the RMNP area has been aspen (*Populus tremuloides*) parkland and mixed conifer and deciduous forest ecosystems, essentially becoming a boreal region (Reeves 1970, Colwill and Jamieson 1972).

A long human history has influenced the development of RMNP's ecosystem (Ecosystem Conservation Plan Team 1997). Aboriginal people have inhabited the RMNP area for 6000 years or more (Buchner et al. 1983, Riding Mountain National Park Round Table 1996). Hunting, gathering, and burning were practiced by the aboriginal people (Ecosystem Conservation Plan Team 1997). Assiniboine and Cree people were involved in the fur trade in the RMNP area since the 1600's. The earliest Europeans arrived in the mid-18th century (Ecosystem Conservation Plan Team 1997). During this time, several species of wildlife were exploited. For example, bison (*Bison bison*) were completely extirpated from the region by 1880 (Ecosystem Conservation Plan Team 1997).

The period of greatest European expansion occurred in the 1880's by which time the agricultural lands had been surveyed into townships by the Dominions Land Act (Warkentin 1967, Colwill and Jamieson 1972, Jamieson 1974). Timber harvesting was extensive in the RMNP area "to the extent that no undisturbed stands remain" (Ecosystem Conservation Plan Team 1997). The Riding Mountain Forest Reserve was established in 1895 to protect the timber resources. At this time, some timber harvesting continued within the reserve boundaries, along with reforestation, haying, and grazing. These activities continued until about 1970 (Ecosystem Conservation Plan Team 1997).

The RMNP area was formally established as a national park in 1930, and opened officially in 1933. At present, the town of Wasagaming has become a tourist resort, and a network of trails and campsites are scattered throughout the Park. The area surrounding the Park has almost exclusively been turned into agricultural land. The Ecosystem Conservation Plan Team (1997) stated that this long human history has "significantly influenced the development of the Riding Mountain's regional ecosystem as it exists today."

## Riding Mountain National Park - Present Situation

Riding Mountain National Park (Figure 1.1) currently encompasses 297,600 ha in western Manitoba and is the meeting point of three varied landscapes including the Manitoba Lowlands, the Saskatchewan Plain, and the Manitoba Escarpment (Riding Mountain National Park Round Table 1996). These landscapes support grasslands, aspen-oak (*Quercus* spp.) forest, and mixed-wood forest life zones (Riding Mountain National Park Round Table 1996). The Park exists as an island of wilderness almost


Figure 1.1 The regional setting of Riding Mountain National Park, Manitoba.

completely surrounded by agricultural land (Figure 1.2).

Riding Mountain National Park characterizes a portion of the Boreal Forest Region (Rowe 1972). It is also the site of an overlap of three major vegetation communities including boreal mixed-wood forest, prairie grassland, and forest-prairie transition (Wang 1995). As such, the Park has a diverse biota.

The vegetation has been classified using a variety of methods (Bailey 1968, Rowe 1972, Park Resource Management Team 1979, Wang 1995). Rowe (1972) described the majority of the Park as the Mixedwood Section of the Boreal forest region, although a small section in the southwestern area of RMNP is categorized as the Aspen-Oak Section. Grasses and forbes dominate the vegetation cover on areas interspersed throughout the Park, mainly in the western regions. Trembling aspen, balsam poplar (*Populus balsamifera*), white birch (*Betula papyrifera*), white spruce, and balsam fir (*Abies balsamea*) characterize the Mixedwood Section (Wang 1995) and occupy imperfectly to well-drained areas. In drier areas, jackpine (*Pinus banksiana*) and black spruce (*Picea mariana*) predominate, and black spruce and tamarack (*Larix laricina*) characterize the poorly drained and wetland areas. On the edge of the Manitoba Escarpment, the Aspen-Oak Section is mainly characterized by balsam poplar in the moister areas, and bur oak (*Q. macrocarpa*) tends to be found along rivers, in areas with shallow dry soils, or on south or west-facing slopes. Alluvial soils are populated by white elm (*Ulmnus americana*), green ash (*Fraxinus pennsylvanica*) and Manitoba maple (*Acer negundo*).

## Lynx in Riding Mountain National Park

Lynx range maps for North America (Hall and Kelson 1959, Burt and Grossenheider 1980) include RMNP, however the status of the lynx population in RMNP has been uncertain (Carbyn and Patriquin 1983). General lynx population trends of Manitoba can be approximated using reports from the Hudson Bay Fur Company since 1735 (Elton and Nicholson 1942). However, data on the current lynx population of RMNP are unavailable. Area-specific records from Green (1932) suggested that lynx were absent in the Park from the early 1900's. In fact, Green (1932) claimed that lynx were extinct from RMNP for two decades prior to his report. Also, Soper (1953)

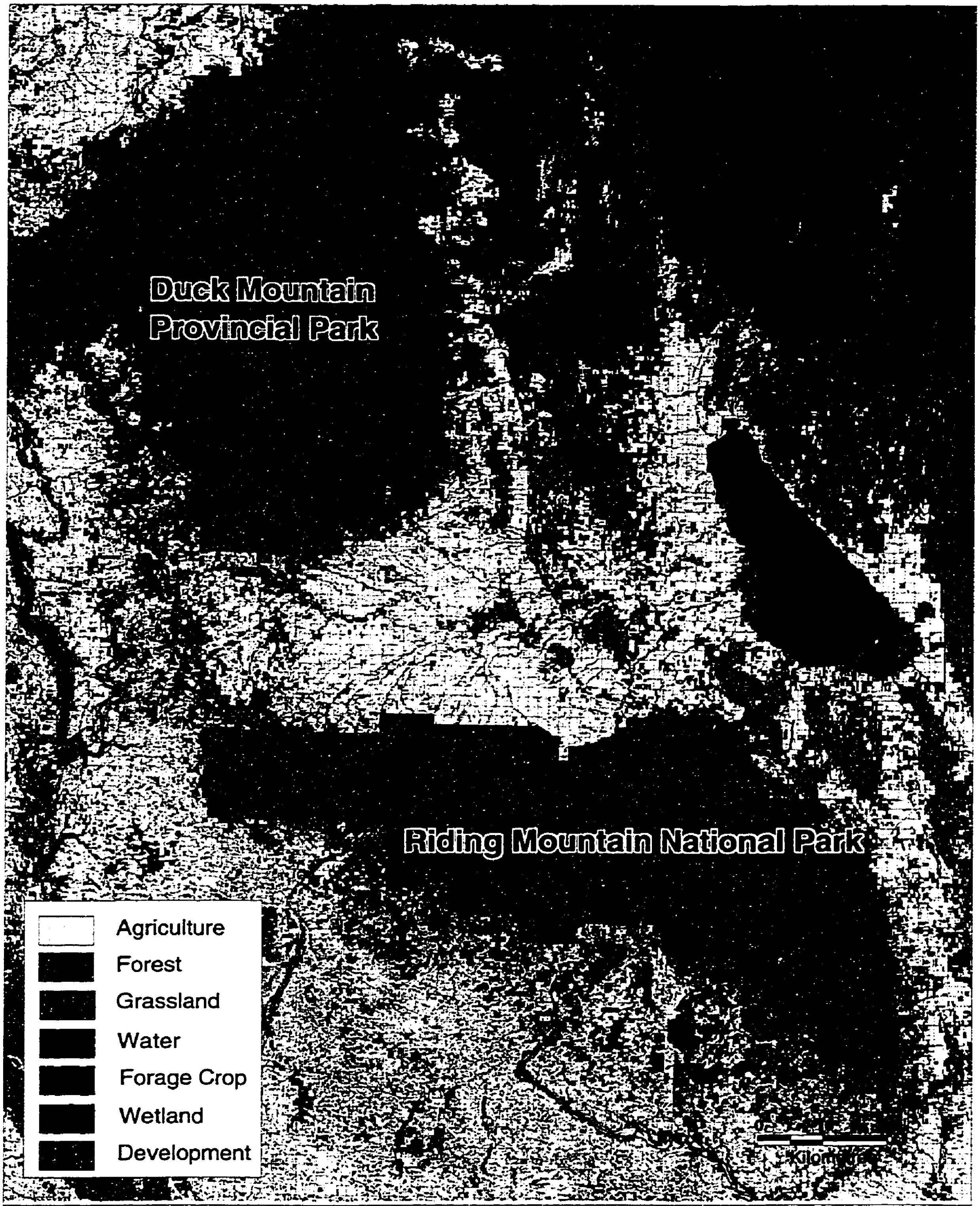



Figure 1.2 A classified Landsat image of Riding Mountain National Park, Manitoba (1993).
Source: Prairie Farm Rehabilitation Administration (PRFA) and Manitoba Remote Sensing.

conducted a study in the 1940's and found limited evidence of lynx. Soper (1953) suggested that only a few individuals sporadically migrated into the Park. Observers reported seeing lynx on several occasions in the early- to mid-1970's (Carbyn and Patriquin 1983) and warden accounts of lynx during this same time period suggested that lynx were sporadically distributed across the Park. Manitoba lynx harvest information also provides some information on lynx population trends (Manitoba Department of Natural Resources n.d.) (Figure 1.3).

Carbyn and Patriquin (1983) captured five lynx and radio-collared three in RMNP during the 1970's. Since then, no formal data have been collected. At present, range maps include RMNP as part of the lynx's distribution. Currently, there are reports of sightings on a regular basis from Park staff, residents, and tourists. The current management plan for RMNP (Riding Mountain National Park Round Table 1996) includes the lynx as a carnivore present within RMNP. Carbyn and Patriquin (1983) questioned the Park's ability to maintain a stable lynx population under a combination of high trapping pressure and low hare densities. The authors concluded that reinvasion from lynx outside RMNP may be essential for maintaining a resident population when such conditions exist.

## GAPS IN THE LITERATURE

A limited number of lynx studies have occurred in regions comparable to RMNP, however, none have specifically addressed lynx habitat (Elton and Nicholson 1942, Mech 1973, Koonz 1976, Mech 1977, Mech 1980, Carbyn and Patriquin 1983). Habitat-based studies have been performed in other geographic locations and some of that information was used to develop the lynx habitat model used in this study (Koehler 1990, Murray et al. 1994, Poole et al. 1996). This habitat-based lynx study will provide lynx habitat use and population viability information relative to the prairie region of Canada.

Lynx information specific to RMNP is limited to Carbyn's and Patriquin's (1983) study and some informal tracking information. There were studies conducted on snowshoe hare ecology from 1977 to 1982 (Leonard 1980, Poll 1981, Ristau and Bergeson n.d., Bergeson 1982) and in 1987-88 (Bergeson 1988). Since hare are the

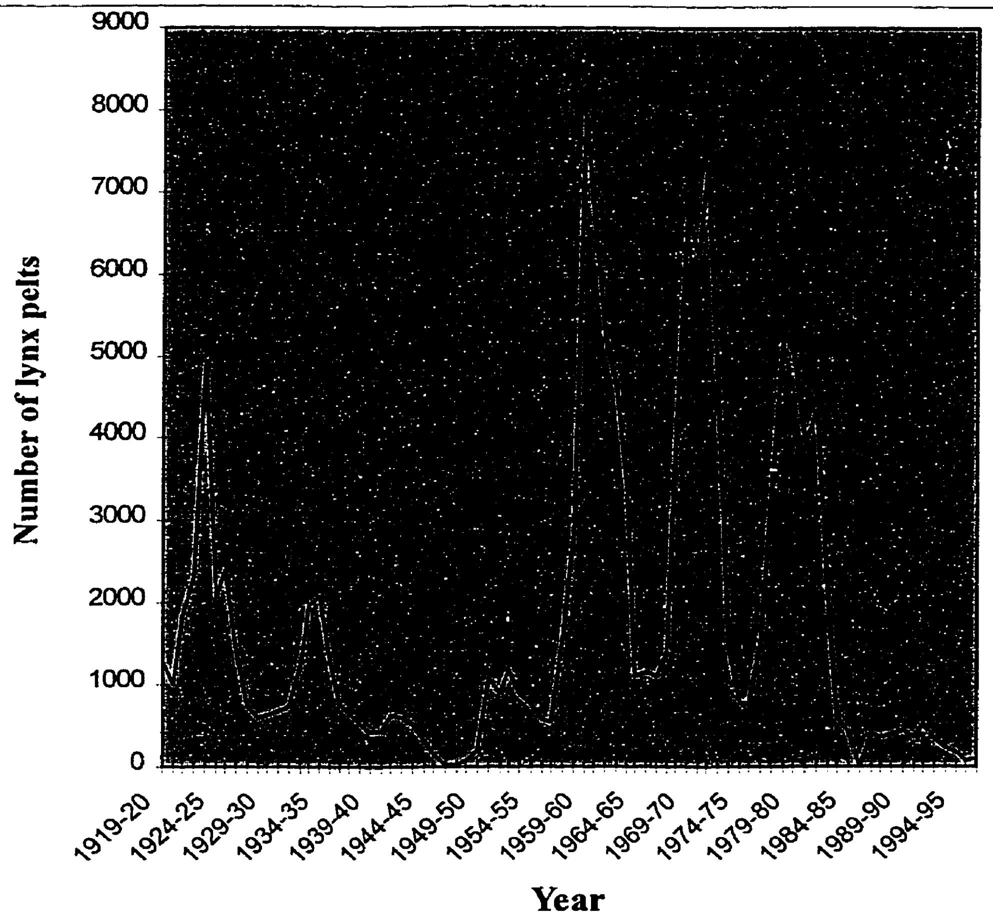



**Figure 1.3 Manitoba lynx harvest data from 1919 to 1997 (data from Manitoba Department of Natural Resources n.d.).**

primary food source of lynx, information from the hare study was relevant to this study.

## PURPOSE AND OBJECTIVES

The managers of RMNP are mandated to consider ecological integrity in their resource management plans. Part of the definition of ecological integrity includes consideration of species viability (Woodley 1993). The purpose of this study was to contribute information about lynx habitat availability and population viability in RMNP and use the information as a component of RMNP's ecological assessment. Objectives of this study were:

1) to verify Roloff's (1998) lynx Habitat Suitability Index (HSI) modelling framework for RMNP;
2) to apply and evaluate the habitat-based population viability approach of Roloff and Haufler (1997); and
3) to develop a process for land managers to evaluate and monitor lynx population numbers to assist in their evaluation of ecological integrity.

## PRACTICUM STRUCTURE

The practicum is presented in five chapters. Chapter two consists of a literature review of lynx ecology. Chapters three and four are written as journal articles related to objectives one and two respectively, to be submitted for publication after practicum completion. Chapter five is the summary, conclusions, and recommendations for lynx management to RMNP that address objective three.

# CHAPTER 2

# LYNX ECOLOGY

## INTRODUCTION

In North America, lynx occur predominantly in the boreal forests of Canada and Alaska, however their range extends southward into the western mountains of the United States (Koehler and Aubry 1994). Lynx distribution, therefore, includes RMNP where they are the most common resident felid species. The prehistoric range of lynx appears to be generally intact (Quinn and Parker 1987) with the exception of some southern portions of their range (e.g., the southern Rocky Mountains of Colorado; Seidel et al. 1998).

Riding Mountain National Park contains varied landscapes including the Manitoba Lowlands, Saskatchewan Plain, and the Manitoba Escarpment (Riding Mountain National Park Round Table 1996). The Park has been divided into three life zones including grasslands, aspen-oak, and mixed-wood ecosystems (Riding Mountain National Park Round Table 1996). In addition to these varied features, the Park is surrounded by a "sea of farmland" (Figure 1.2). All these factors influence lynx numbers by providing various types and combinations of habitat. Natural processes and events and human alteration of the landscape within and surrounding the Park also impact lynx numbers. A requisite for assessing the effects of these impacts on lynx is better documentation of the habitat requirements of lynx and their primary prey, the snowshoe hare (Elton and Nicholson 1942, Seton 1953, Keith 1963, Nellis et al. 1972).

## GENERAL HABITAT REQUIREMENTS

Lynx distribution is tied primarily to the boreal forest (Koehler and Aubry 1994), however, their habitat can be extremely varied. Lynx have been observed in vegetation cover types consisting of white spruce, black spruce, paper birch, willow (*Salix* spp.), trembling aspen, poplar, balsam fir, jackpine, Engelmann spruce (*P. engelmannii*), subalpine fir (*A. lasiocarpa*), and lodgepole pine (*P. contorta*) (Koehler and Aubry 1994). Similar habitat characteristics in each of these cover types include low topographic relief, continuous forest cover, and a mosaic of forest ages and types (Koehler and Aubry 1994).

Lynx require a variety of vegetation conditions to survive and reproduce successfully, both at the individual and population levels. Forest types that support high prey numbers are required for foraging. Mature forests or forests with certain structural components, such as deadfalls, are essential for denning and kitten cover. Open, low-stocked forests may serve as travel cover. The latter habitat is not essential for lynx survival, however, travel cover can function to close gaps between foraging and denning habitats (Koehler and Aubry 1994). Lynx habitat requirements thus focus on three features; foraging, denning, and interspersion (Koehler and Aubry 1994). An assessment of lynx habitat for a particular area should focus on the quality and quantity of these factors.

## Foraging

The approximate ten-year population cycle of lynx is almost certainly driven by the corresponding population cycles of snowshoe hare (Keith 1963). Lynx and hare population cycling occurs in RMNP (Poll 1981) with a peak occurring during the time of this study. Harvest records are assumed to coincide with the lynx population cycle. Harvest information for Manitoba is available (Manitoba Department of Natural Resources n.d) (Figure 1.3). These peaks average approximately three years (Elton and Nicholson 1942).

Lynx tend to select habitat where hares are most abundant (Koehler 1990) and early successional forests typically provide good hare habitat (Koehler and Aubry 1994). Snowshoe hare habitat is quite varied but a brushy understory, both deciduous or coniferous, that provides both winter food and cover is essential (Keith et al. 1984). Preferred winter food for hare consists of the small branches, twigs and stems of woody shrubs and saplings (Keith et al. 1984). Palatability is also a consideration and according to pellet analysis, unpalatable species known in RMNP include common snowberry (*Symphoricarpos albus*), Canada buffaloberry, (*Shepherdia canadensis)*, common

Labrador tea (*Ledum groenlandicum*), twining honeysuckle (*Lonicera dioica*), and cranberry (*Viburnum* spp.) (Leonard 1980). Leonard (1980) found that the most common forage species used by hares in a portion of RMNP were rose (*Rosa* spp.), aspen, willow, and alder (*Alnus ruguosa*). Although the predominant shrub species in RMNP is beaked hazelnut (*Corylus cornuta*), it did not constitute a significant portion of the hare diet.

In Washington, Koehler (1990) found that hare numbers were greater in 20 year-old stands of lodgepole pine (four to five times greater) than in older stands (43 to 82 years old). The 20 year-old stands had an average density of trees and shrubs (less than 2.5 cm diameter at breast height [dbh]) of 15,840 stems/ha. The stem density likely provided snowshoe hare with forage, escape, and thermal protection (Litvaitis et. al. 1985). Koehler (1990) also noted that the 20 year-old stands provided the greatest amount of browse for hares throughout the winter. Other studies have indicated that preferred hare habitat consists of 22,027 stems/ha in Alaska (Wolff 1980) and >16,000 softwood stems/ha in Maine (Litvaitis et al. 1985).

Another factor to consider in describing hare habitat is woody stem height (Koehler and Aubry 1994). This factor is especially important in winter since snow depths in boreal forests are typically greater than 1 meter (Koehler and Aubry 1994). Koehler and Aubry (1994) summarized studies from Minnesota (Pietz and Tester 1983), Nova Scotia (Parker et al. 1983), the Rocky Mountains (Dolbeer and Clark 1975, Wolfe et al. 1982) and central Wisconsin (Sievert and Keith 1985) and surmised that vegetation heights of approximately 1-3 meters were required for optimal hare foraging, depending on location of the study. Snow records from RMNP show annual snow accumulation of approximately one meter (Trottier et al. 1983, McGinn and Rousseau 1995, 1996, 1997).

Lynx will opportunistically prey on species other than snowshoe hare (Koehler and Aubry 1994). For example, a study conducted by Poole (1994) in the Northwest Territories found that gallinaceous birds and red squirrels (*Tamiasciurus hudsonicus*) were used as prey by lynx (Brand et al. 1976, Koehler 1990). Other studies have found lynx preying on ruffed grouse (*Bonasa umbellus*) (Brand et al. 1976), squirrels (Nellis and Keith 1968, Brand et al. 1976), various birds (Nellis and Keith 1968, Nellis et al. 1972, Brand et al. 1976), and occasionally ungulates (Bergerud 1971). Mice (*Peromyscus* spp.)

(Koehler 1990, McCord and Cardoza 1982, Nellis et al. 1972) and ptarmigan (*Lagopus* spp.) have been added to this list by McCord and Cardoza (1982) who suggested that lynx tend to prey on these species, particularly during the summer. Brand et al. (1976) also listed carrion as an alternate food source, however, carrion appeared as a prey item only when hare densities were low. Early records by Sheldon [reported in Elton and Nicholson (1942)] noted that during a hare low all the lynx that they examined were starving and that "the only fat lynx seen that winter was an old female whose stomach was filled with mice and one ground squirrel - an exceptional event." Koehler and Aubry (1994) suggested that since most alternative prey species are typically smaller than hares, the shift to alternate prey sources during a hare decline may still result in an energy deficit for lynx.

## Denning

Limited information exists about suitable denning habitat for lynx, however the information available provides varied results (e.g. denning occurs in hollow trees, stumps, or logs, in fallen logs or tangled spruce roots, and in almost any type of vegetation provided that there is adequate cover), as summarized by the Washington Department of Natural Resources (1996). Koehler and Aubry (1994) generalized these characteristics as dense, mature forested stands with fallen trees or upturned stumps. The physical structure of vegetation seems especially important to denning habitat quality. Roloff (1998) described a suitable denning site as consisting of inter-tangled, woody material that provided interstitial spaces under a vegetation canopy. Koehler's (1990) study in north-central Washington found that denning sites occurred in areas of mature (≥250 years old) forests. All denning sites recorded in Koehler's (1990) study were on north-northeast aspects and had an average of 40 downfall logs per 50 meters. The kittens used the logs as escape cover. In addition, important factors for denning sites include minimal human disturbance, foraging areas in close proximity, and stands that are a minimum of 1 ha in size (Roloff 1998). Travel corridors between denning sites and foraging areas are important landscape considerations (Koehler and Brittell 1990).

## Interspersion

Another lynx habitat feature is travel cover (termed interspersion) that permits lynx to move between resource patches. Lynx require cover for security and stalking prey. Coniferous or deciduous vegetation >2 meters in height with a closed canopy tend to be suitable travel cover (Brittell et al. 1989). Lynx have been observed crossing sparsely vegetated openings ≥100 meters in width, however, they avoided hunting in those areas (Koehler 1990). Large open areas tend to discourage travel thereby disrupting natural movement patterns (Koehler and Aubry 1994). In RMNP, lynx tracks were often observed along the forested edge of frozen lakes rather than traversing straight across.

## LYNX HABITAT IN RIDING MOUNTAIN NATIONAL PARK

Preferred lynx habitat includes a diverse forest environment and a mosaic of successional stages that offer habitat in both winter and summer seasons (Parker et al. 1983). Lynx range corresponds to the extent of suitable forests. Quinn and Thompson (1987) stated that lynx carrying capacity in a boreal mixed wood forest (diverse and mixed vegetation occupying varying topography and soils) may by relatively high when compared to a true boreal forest (monotypic coniferous vegetation occupying flat plains). Additionally, boreal mixed wood forest that is heterogeneous and disturbed by logging or burning is probably better lynx habitat than true boreal forests (Quinn and Thompson 1987).

Boreal forest vegetation diversity is dependant upon fire that reinitiates plant succession at intervals that creates a successional mosaic (Heinselman 1973, Fox 1978). It has been hypothesized that fire and plant succession primarily, though not exclusively, drive the hare cycle (Grange 1949, 1965). Generally, post-fire successional habitats provide optimal foods for hare. In the boreal forest, post-fire succession includes an increase in plant biomass and productivity of such species as jackpine or deciduous shrubs and trees such as birch, willows, and aspen (Grange 1949, 1965 Heinselman 1973, Fox 1978). In this region, these species constitute preferred hare food, where preferred suggests an increased nutritional value (Fox 1978). As a result of post-fire succession, an increase in hare habitat quality may increase the carrying capacity for hares in that area, thus increasing the hare population (Fox 1978). Since hares are the lynx's primary food source, fluctuations in the hare population may cause fluctuations in lynx populations (Nellis et al. 1972). Fox (1978) also referred to numerous case studies where the vegetation structure resulting from logging corresponded to an increase in wildlife species that preferred successional browse. Koehler (1990) similarly noted that early successional forests are beneficial to snowshoe hares. If snowshoe hare numbers remain stable in these areas, then lynx reproductive rates may remain relatively stable (Koehler 1990). Therefore, according to Fox (1978) there is a reasonable coincidence between the population fluctuations of the lynx cycle and that of forest and brush fires.

Murray et al. (1994) conducted a three-year study in southwestern Yukon which included cover types of boreal forests (primarily white spruce), deciduous forests (aspen species.), willows, and non-forested areas. In these areas, use of cover types by lynx was not proportional to availability. Lynx selected areas of very closed spruce (>76% canopy cover) in one year, avoided open cover types in two years, and avoided willow cover types in all years (Murray et al. 1994). The most heavily used cover type in all years (35-43% of the time) was open spruce (26-50% canopy cover).

Lynx habitat information specific to RMNP, or to Manitoba, is lacking. Throughout Carbyn and Patriquin's (1983) study, lynx were reported in the eastern portion of the Park which is characterized by the Manitoba Escarpment and also the site of a large forest fire that occurred in 1980 (Caners and Kenkel 1998). Carbyn and Patriquin (1983) also reported that wardens saw lynx on the western portion of the Park which is characterized by forests with interspersed grasslands.

## LYNX HOME RANGE

Carbyn and Patriquin (1983) estimated that home range sizes for three collared lynx in RMNP were 156 $km^2$ (15,600 ha) (average) for two females and 221 $km^2$ (22,100 ha) for one male. Mech (1980) reported an exception whereby male home range sizes in Minnesota were 145-243 $km^2$ (14,500-24,300 ha), however, the home range sizes for RMNP lynx were higher than most others reported in the literature (Carbyn and Patriquin 1983). Other studies have found a wide variation in lynx home range sizes with results from 8-783 $km^2$ (800-78,300 ha) (summarized in Koehler and Aubry 1994). Translocated lynx (such as in New York) have been documented as having even larger home ranges with a harmonic mean estimate of 1,760 $km^2$ (176,000 ha) (Brocke et al. 1992). The average home range size, discounting factors such as prey availability, has been estimated at approximately 16-20 $km^2$ (1,600-2,000 ha) (Quinn and Parker 1987).

Mech (1980), Koehler (1990), and Koehler and Aubry (1994) suggested that scarcity of prey may result in larger lynx home ranges. This type of situation appeared to occur in RMNP during the 1970's when female lynx homes range averaged 156 $km^2$ (15,600 $ha^2$) and a male RMNP lynx home range was 221 $km^2$ (22,100 ha) (Carbyn and Patriquin 1983). Poole (1994) also found food scarcity to be a factor in home range sizes. Poole (1994) noted that the smallest annual home range size occurred throughout the period of hare decline, and the largest home range size occurred during the second full year of hare scarcity.

Loss or fragmentation of suitable habitat could also explain large home range sizes. Building on this theory, lack of suitable habitat for snowshoe hare may also be a significant factor. Marginal or suboptimal habitat may lead to lower hare numbers or more dispersed hare populations, hence obligating lynx to maintain larger home ranges to satisfy life requisites.

## STRESSES

### Harvesting

In general, felids have a reduced reproductive capacity to quickly respond to a decline in prey population or to exploitation pressure (Eisenberg 1986). During a lynx population peak, when prey is abundant and kitten production is high, harvest regulations may be "liberal" (Quinn and Parker 1987), however, harvesting may have detrimental effects during times of prey scarcity when kitten survival is low (Carbyn and Patriquin 1983, Parker et al. 1983, Bailey et al. 1986, Quinn and Parker 1987, Koehler and Aubry 1994). Bailey et al. (1986) recognized the need for large refugia from trapping during periods of low lynx recruitment. Also, controlled harvesting should only be allowed during years of high population recruitment (Parker et al. 1983, Bailey et al. 1986, Quinn and Parker 1987). This stipulation is especially important in areas where habitat is limited and immigration is minimal (Parker et al. 1983).

Harvest records usually indicate a sex bias toward males (Berrie 1973) possibly because of their greater mobility, larger home ranges (Koehler and Aubry 1994), and dispersal patterns (Koonz 1976). Occasionally, there is an even sex ratio (Brand and Keith 1979) and sometimes proportionately more females (Bailey et al. 1986). A study conducted in Manitoba (Koonz 1976) found that during a population peak, young lynx were most frequently trapped. During these times, more males were trapped than females. As the overall population size decreased, females made up a higher proportion of the catch. Koonz (1976) speculated that this shift in proportion may be the result of more males being trapped in previous years thereby leaving an unequal distribution of females to males in the population. Other studies have suggested that when reproductive success is high, yearlings constitute a large portion of the harvest, especially during the early trapping season, declining thereafter (Quinn and Parker 1987). Young of the year do not seem to appear in the harvest until later winter when they leave the female (Quinn and Parker 1987).

Riding Mountain National Park's lynx population has been vulnerable to external, non-Park activities (Carbyn and Patriquin 1983). Carbyn and Patriquin (1983) found that lynx were vulnerable to trappers outside RMNP, and speculated that the Park may not be large enough to sustain a viable lynx population under certain types of trapping regimes. Each of the three radio-collared lynx in Carbyn and Patriquin's (1983) study were killed by trappers. The Park is limited to 2,976 $km^2$ (297,600 ha), and is surrounded by agricultural land, potentially making immigration difficult. In addition, Koehler (1990) indicated that marginal habitats may be especially vulnerable to exploitation. If lynx numbers are low and trapping pressure high, it is possible that the lynx population within the Park could become extirpated (Carbyn and Patriquin 1983).

Poole (1994) also provided a number of trapping management suggestions. Suggestions included that lynx trapping should be lessened or eliminated during a three-to-four year period (Brand and Keith 1979) or even up to five years (Bailey et al. 1986) when hares are scarce. This trapping regime may ensure that lynx populations remain above a certain threshold to reestablish the population during the subsequent hare peak. In addition, having untrapped refuges (Bailey et al. 1986) that are large enough to maintain a sufficient number of lynx seem important to replenish the population. It is not known how large these refuges need to be in order to sustain a viable lynx population (Bailey et al. 1986). Carbyn and Patriquin (1983) and Bailey et al. (1986) reported concerns that refuges several thousand $km^2$ were of insufficient size to sustain a viable population. Carbyn and Patriquin (1983) reported moderate to heavy trapping pressure around RMNP and Bailey et al. (1986) stated that trapping was "liberal" in the Kenai National Wildlife Refuge. Poole (1994) suggested that where refuges are present, changes to the trapping regimes may be necessary so that surviving lynx and new recruits may be protected thereby directing the harvest to nomadic and more vulnerable animals.

Lynx kittens are dependant on the female during the first year of life. Hunting success as an adult is determined by learned behavior within that first year. Effects of early winter trapping of female lynx on kitten survival is unknown. So, in addition to yearly restrictions on lynx harvesting, the seasonal timing of the harvest may also be an important consideration (Carbyn and Patriquin 1983).

## Competition

Lynx and coyotes (*Canis latrans*) inhabit the boreal forest sympatrically (Murray et al. 1994). Each species is present within and around RMNP. Both species have a similar size and mass, however, lynx feet are much larger (Murray and Boutin 1991). Murray and Boutin's (1991) study found that coyotes have a foot-load (ratio of body mass to foot area) 4.1-8.8 times higher than that of lynx resulting in coyotes having a greater sinking depth in snow. Lynx were found to use areas that had a greater snow depth than coyotes and as a result, competition between the two species appeared negligible (Murray and Boutin 1991).

Both coyotes and lynx prey on snowshoe hare (Murray and Boutin 1991). However, Keith et al. (1977) found that lynx depended solely on hares at all times in their study area, whereas coyotes only relied on hares when hares were abundant. In addition, felids and canids typically prefer different hunting habitats (Kleiman and Eisenberg

1973), although in the northern latitudes, these areas seem to correspond to habitats occupied by snowshoe hares (Murray et al. 1994). Poole (1994) conducted a study in the Northwest Territories and found that potential competitors of lynx also included the red fox (*Vulpes vulpes*), wolf (*Canis lupus*), black bear (*Ursus americanus*), wolverine (*Gulo gulo*), and numerous raptor species. Such species, with the possible exception of the wolverine, also inhabit RMNP. Therefore, it is reasonable to hypothesize that these same species also compete with lynx in RMNP.

Bobcats (*Lynx rufus*) are also a potential competitor to lynx, although similar to coyotes, they probably occupy different niches than lynx in part due to their anatomical differences. Parker et al. (1983) found that lynx paws can support twice the weight of bobcat paws in snow. Habitat alterations that favor a northern expansion of the bobcat range may prove detrimental for lynx (Koehler and Aubry 1994). At present, range maps do not include RMNP as inclusive of bobcat range, however, bobcat do range into southern Manitoba (Rolley 1987).

## Roadways

Roads and trails potentially increase lynx vulnerability to hunters and trappers (Bailey et al. 1986) and increase the chances of vehicle and lynx collisions, harassment, and other human-lynx interactions (Washington Department of Wildlife 1993). Vehicle mortality does occasionally claim lynx in RMNP (personal observation). Mortality may be high because lynx commonly travel along narrow roadways provided that adequate cover is available on either side of the road (Koehler and Brittell 1990).

# CHAPTER 3

## VERIFICATION OF A LYNX (*FELIS LYNX*) HABITAT ASSESSMENT METHOD FOR RIDING MOUNTAIN NATIONAL PARK

## INTRODUCTION

The geographic range of lynx is linked primarily to the distribution of the boreal forest region although their habitat is often varied (Koehler and Aubry 1994). General habitat structure can include species such as white spruce, black spruce, paper birch, willow, quaking aspen, poplar, balsam fir, and jackpine in Canada and the eastern United States. In the western United States, lynx often use stands of Engelmann spruce, subalpine fir, and lodgepole pine (Koehler and Aubry 1994). Some habitat characteristics are similar among these areas including low topographic relief, continuous forests, and a mosaic of forest successional stages (Koehler and Aubry 1994). Areas meeting these criteria often support snowshoe hares; the lynx's main prey (Koehler and Aubry 1994).

Habitat requirements for lynx can be categorized into three features; foraging, denning, and interspersion (Koehler and Aubry 1994). Vegetation conditions that support high prey numbers are required for foraging. Snowshoe hare habitat is quite varied but a brushy understory that provides both winter food and cover is essential (Keith et al. 1984). Denning and kitten cover is often provided by forests with certain structural components, such as many deadfalls and a closed vegetation canopy. Roloff (1998) described a suitable denning site as consisting of inter-tangled, woody material that provide interstitial spaces under a vegetation canopy. These denning sites should also be subjected to minimal human disturbance, be in close proximity to foraging areas, and be a minimum of 1 ha in size (Roloff 1998). Vegetated conditions that do not provide forage or denning habitat may serve as travel cover for lynx. Coniferous or deciduous vegetation >2 meters in height with a closed canopy offers suitable travel cover (Brittell et al. 1989).

Travel cover is not essential for lynx survival, however, these areas often link foraging and denning habitats (Koehler and Aubry 1994). Roloff (1998) quantified the foraging, denning, and interspersion components of lynx habitat into a Habitat Suitability Index (HSI) model. The purpose of this study was to verify the Roloff (1998) lynx HSI model as a habitat assessment tool for RMNP, Canada.

# STUDY AREA

See Chapter 1.

# METHODS

## Lynx Habitat Suitability Index Model

In Manitoba, lynx are classified as fur-bearing animals and are therefore subject to legal harvest (Manitoba Wildlife Act 1987). In RMNP, however, the National Parks Act (1989) restricts the hunting or possession of any wild animal or part thereof and thus, lynx receive full protection within the Park boundaries. Under these varied management regimes, the effects of land management decisions on lynx populations must be understood to ensure the persistence of this species. An understanding of these effects is especially critical in areas that provide sanctuary for lynx.

Roloff (1998) developed a lynx HSI model (Appendix A) to help managers evaluate the effects of land management activities on lynx. Roloff's (1998) HSI model uses the limiting factor approach whereby the most limiting resource for an organism is assumed to have the greatest impact on the individual. Roloff (1998), consistent with Koehler and Aubry (1994), portrayed the quality of three features of lynx habitat; foraging, denning, and interspersion. Part of the foraging component is a snowshoe hare sub-model. The snowshoe hare sub-model consists of two components; foraging and security (Roloff 1998). Mathematical computations of these components provide an overall index of hare habitat quality during winter. The winter period is considered to be the limiting season on snowshoe hare fitness (Roloff 1998). The snowshoe hare index is calculated from understory vegetation cover, browse abundance, and vegetation species composition at various height strata.

Roloff's (1998) model was initially developed for application in the Intermountain west region of the United States, but noted that the model framework can be applied elsewhere by calibrating the inputs and model relationships to the biogeoclimatic conditions of the specified region. This type of model calibration was performed for RMNP. Differences between Roloff's (1998) Intermountain model and the model calibrated for RMNP included an emphasis on 0-1 and 1-2 meter vegetation (corresponding to typical snow depths on RMNP), a reevaluation of palatable browse species for snowshoe hare, and a reduction in the average tree size for forested denning areas. Roloff's (1998) Intermountain model used vegetation measurements from three height strata to calculate snowshoe hare forage and cover habitat quality indices. The model for RMNP uses the lower two height strata to index hare habitat quality because the Park rarely accumulates greater than 1 meter of snow (Trottier et al. 1983, McGinn and Rousseau 1995, 1996, 1997). Also, Roloff's (1998) Intermountain palatable browse species list was calibrated to RMNP using Leonard's (1980) snowshoe hare study. Roloff's (1998) model defines potential denning stands as having an average overstory tree size of 36 cm. Since forests on RMNP are generally of smaller stature than Intermountain forests, a 25 cm diameter was used for this model attribute.

## Foraging

Habitat suitability for snowshoe hares is related to foraging and security cover (Roloff 1998). Important components of hare habitat are characterized by Roloff (1998) using measures of dense woody vegetation, stem diameter of browse species, coniferous cover, habitat interspersion, and patch size. Winter hare habitat quality was estimated at the map-polygon and home-range-level using a mathematical combination of these components for RMNP.

The snowshoe hare habitat assessment produced a GIS coverage in which each mapped polygon was assigned an HSI score. Subsequently, these polygons (or portions thereof) were combined into hare home ranges, the size of which depended on the quality of the habitat as portrayed by the HSI score. In this study, two types of home ranges were delineated from the snowshoe hare HSI map; viable and marginal. Viable snowshoe hare home ranges were delineated by establishing a threshold of 0.60 habitat quality and a habitat unit objective (where a habitat unit is the product of HSI score and area) of 5.0. A habitat unit objective of 5.0 indicates that a minimum of 5 ha of habitat at 1.00 quality are required for a functional home range. As habitat quality decreases, more area is required. Snowshoe hare home range functionality thresholds were estimated by Roloff (1998) using the methodology of Roloff and Haufler (1997). Home ranges that satisfied the viability criteria were expected to consistently support and produce hares (Roloff and Haufler 1997). In other words, only snowshoe hare habitats that averaged >0.60 quality contributed towards a "viable" snowshoe hare home range. This home range delineation process resulted in a GIS coverage that contained "viable" snowshoe hare home ranges.

In identifying and delineating marginal home ranges, viable home ranges were removed from the total available habitat and the process was repeated with lower habitat thresholds. For the marginal home range iteration, a minimum quality rating of 0.25 with a habitat unit objective of 5.0 was used. Marginal hare home ranges are expected to support and produce hares during good resource years, but they are the first to disappear when resources become limiting (Roloff and Haufler 1997). The output of the home range delineating process was a GIS coverage of viable and marginal snowshoe hare home ranges. This coverage was used as input for the lynx habitat model (Roloff 1998). The lynx model based the forage potential score for lynx on the number of viable and marginal snowshoe hare home ranges that were contained in each lynx home range.

## Denning

Roloff (1998) identified six aspects of lynx denning habitat that were used to evaluate denning habitat potential for RMNP. These included vegetation cover type, site conditions, canopy closure, area of the vegetation type, juxtaposition and interspersion to forage cover, and the quantity and arrangement of downed woody debris. To index the denning potential of a lynx home range, Roloff (1998) identified a 3-phase process: "1) identify vegetation types that provide vegetation structure and size deemed suitable for denning, 2) identify vegetation types that are properly arranged within a home range area, and 3) identify vegetation types that provide suitable denning micro-sites." These phases use the six components of lynx denning habitat to evaluate a home-range-sized area for denning habitat quality.

The phase 1 denning assessment for RMNP consisted of a summary and review of vegetation inventory and soils data. Due to a lack of site-specific information about denning habitat used by lynx in RMNP, Roloff's (1998) recommendations were used to identify potential denning habitat. Map polygons that had an average tree diameter $\geq$25 cm, an average basal area $\geq$3.72 $m^2$/ha, suitable denning soil, and an overstory canopy cover >50% were identified as potential denning areas for RMNP. Mapped polygons that satisfied these criteria were subsequently evaluated to determine which were a minimum of 2 ha in size. Map polygons that satisfied the Phase 1 denning criteria outlined in Roloff (1998) were subsequently evaluated for juxtaposition to forage.

Phase 1 polygons were then evaluated to determine the polygons that had a minimum of 50% of their perimeter adjacent to lynx denning, foraging, or travel habitat (Roloff 1998). Additionally, 30% of the potential denning patch had to be within 0.8 km of suitable summer forage habitat (Roloff 1998). Suitable summer forage habitat consisted of vegetation providing at least >20% vertical cover in the 0 to 1 meter height strata (Roloff 1998). Map polygons that met these spatial criteria satisfied the Phase 2 denning requirements. Denning areas that met the Phase 2 criteria were subsequently evaluated for the suitability of site conditions. Site conditions included an assessment of micro-site characteristics required for denning (closed canopy, suitable soil conditions, downed woody debris). Denning areas that satisfied the micro-site evaluation were carried forward as potential denning sites. Each lynx home range was scored for denning based on the number and distribution of these den sites (Roloff 1998).

## Interspersion

Interspersion also plays an important role in lynx habitat quality. Roloff's (1998) assumptions that lynx will travel through most cover types and open areas <100 meters in width were adopted and used in the RMNP analysis. Also, it is assumed that the number, size, and spatial distribution of barriers to lynx movements influence habitat quality (Roloff 1998). Roloff's (1998) model uses two processes to calculate an interspersion HSI index. First, "non-lynx" cover is identified based on vegetation structure information. Second, the quantity and spatial distribution of "non-lynx" habitat is evaluated at the home-range-level. An interspersion index for a lynx home range is assigned based on the average distance a lynx can travel in a home range without encountering a barrier to movement (Roloff 1998).

## Determination of the Lynx Home-Range-Level HSI

The quality for each lynx habitat feature (i.e., foraging, denning, and interspersion) was expressed as a home-range-level GIS grid. Following Roloff's (1998) model, forage, denning, and interspersion habitat scores were combined into one lynx HSI value for a home-range-sized area. The overall lynx habitat quality grid represented lynx HSI scores for a 250 ha area and was termed a habitat contour map (Roloff and Haufler 1997). The use of a 250 ha home range for lynx habitat assessments is discussed by Roloff and Haufler (1997) and has a relationship to estimating the viability of the home range. The habitat contour map formed the basis for delineating viable and marginal lynx home ranges following the process previously discussed for snowshoe hares. The output from the home range delineation process was a map of viable and marginal lynx home ranges for the study area (Roloff and Haufler 1997). The implications of the viability map are discussed in Chapter 4 of this document.

## Classifying Ecological Units for Habitat Modelling

The 297,600 ha Park was stratified into 52 classes based upon a combination of existing vegetation and soil delineations (Walker and Kenkel 1997, Lombard North Group Limited 1976). Since forage quality and vegetation species composition can be dependant upon soils, a combination of the vegetation and soils maps were used. These vegetation and soil combinations represented ecological land units that formed the foundation of the land classification scheme used in this study. The initial combination of soils and vegetation for RMNP yielded 322 different ecological land units. To narrow the focus of vegetation sampling, unique ecological land units external to RMNP (e.g. agriculture), water, and any ecological land unit that accounted for <50 ha of the Park were filtered out of the sample.

The filtering process resulted in a stratification that covered 85% of RMNP's area (254,000 ha). Logistics prevented sampling each ecological unit, thus, the 20 most common classes (in terms of area) were sampled. An additional ecological land unit was also added to represent an area that burned in 1980. The area consisted of regenerating jackpine. These 21 classes accounted for about 74% (220,980 ha) of the Park's total area. Ecological land units classified as water, agriculture, and grasslands were not sampled but are assumed to provide no suitable lynx or snowshoe hare habitat. These ecological land units accounted for 6% (16,850 ha) of the Park's area. The maximum mapping unit was 5,763 ha (aspen parkland on sand-clay). The mean polygon size was 5 ha.

The RMNP soils map (Lombard North Group Limited 1976) was in digital format and contained attributes that described geological material, landform, morphological expression, erosion potential, slope, and soil morphology. To map soils for the lynx habitat assessment, attributes that accounted for landform, morphological expression, and erosion were used (Table 3.1). Four soils groups were delineated for the lynx habitat assessment. These groups included clay-loam, clay, sand-silt, and sand-clay (Table 3.1). Each soils group consisted of several different soil texture types as identified in the Park's geographic information system (Table 3.1).

The existing vegetation was delineated using Walker and Kenkel's (1997) satellite image interpretation. Walker and Kenkel (1997) identified 15 vegetation types that were used for this study. These vegetation types included: 1) aspen parkland; 2) deciduous canopy-coniferous subcanopy; 3) eastern deciduous forest; 4) mixed canopy forest; 5) shrubland; 6) low canopy deciduous forest; 7) closed coniferous forest; 8) low shrub grassland; 9) open coniferous forest; 10) bur oak forest; 11) wetland; 12) shrubland; 13) regenerating coniferous forest; 14) grassland; and 15) agriculture (Table 3.2). Two types of ground verification were performed as an accuracy assessment of a satellite map (Walker and Kenkel 1996). These included testing the vegetation map against forest

## Table 3.1. Riding Mountain National Park soil texture information.

| SOIL TEXTURE DESCRIPTION | LANDFORM | MORPHOLOGICAL EXPRESSION | SLOPE | DRAINAGE | VEGETATION |
|---|---|---|---|---|---|
| **CLAY LOAMS** | | | | | |
| CIL-SL[a] (clay loam-sand loam) | Glacio-fluvial geology | Plains, terrace, and undulating | Gentle to moderate | Well to imperfectly drained | Poplar, grassland, dry-land shrubs, and white spruce |
| Cl-SCL (clay-sand coarse loam) | Glacial-till with with scarp landform | n/a[b] | Steep to very steep | Well-drained | Poplar, white birch, balsam fir |
| Cl-SL (clay-sand loam) | Glacial till with alluvial and morainal landform | Hummocky and undulating | Moderate to steep | Well to poorly drained | Poplar, spruce; flat areas can support wetlands |
| **CLAYS** | | | | | |
| Cl (clay) | Glacial till with morainal landforms | Undulating, hummocky, or scarp | Moderate to steep | Imperfectly to poorly drained | Poplar, spruce, and wetland |
| Cl-CIL (clay-clay loam) | Glacial till with morainal landforms | Hummocky, undulating, or plain | Gentle to steep | Well to poorly drained | Poplar, spruce, white birch, and wetland |
| CIL-L (clay loam-loam) | Glacial till with morainal landfomrs | Hummocky or plain | Moderate | Imperfectly drained | Poplar and wetland |
| HCL-Cl (heavy clay-clay) | Glacial till with morainal landform | Hummocky and undulating | Moderate to steep | Imperfectly drained | Spruce, poplar, black spruce and wetland |

[a] **Soil texture code as expressed in RMNP's soil map layer.**

[b] **Not available**

**TABLE 3.1 Riding Mountain National Park soil texture information (continued).**

| SOIL TEXTURE DESCRIPTION | LANDFORM | MORPHOLOGICAL EXPRESSION | SLOPE | DRAINAGE | VEGETATION |
|---|---|---|---|---|---|
| **SANDY SILTS** | | | | | |
| SiL-SL (silt loam-sand loam) | Alluvial geology and landform | Plain and fan | Moderate to steep | Well to imperfectly drained | Poplar and balsam fir |
| Sil (silt loam) | Glacial till with residual and undifferentiated landforms | Gorge | Steep to extremely steep | n/a | Poplar, white birch and oak |
| Sil-SL (silt loam-sand loam) | Alluvial geology with fan and plain landforms | n/a | Moderate to steep | Well to imperfectly drained | Poplar and balsam fir |
| S-Si (sand-silt) | Lacustrine geology with plain and undulating landforms | n/a | Gentle to moderate slopes | Imperfect to poorly drained | Poplar and spruce |
| S-SL (sand-sand loam) | Glacio-fluvial geology | Plain and undulating | Gentle to moderate | Rapid to imperfectly drained | Grassland, poplar, spruce, and wetland |

**[a] Soil texture code as delivered by RMNP's soil map layer.**

**[b] Not available**

**TABLE 3.1 Riding Mountain National Park soil texture information (continued).**

| SOIL TEXTURE DESCRIPTION | LANDFORM | MORPHOLOGICAL EXPRESSION | SLOPE | DRAINAGE | VEGETATION |
|---|---|---|---|---|---|
| **SANDY CLAYS** | | | | | |
| L-SCIL (loam-sand clay loam) | Glacio-till with morainal landform | Hummocky, gorge and undulating | Steep | Well-drained | Poplar, white birch, and spruce |
| SCIL (sand clay loam) | Glacio-till and glacial-lacustime geology with morainal landform | Ridged and plain | Moderate | Well to imperfectly drained | Poplar and oak |
| SCIL-L (sand clay loam-loam) | Lacustrine geology and landform | Undulating | Gentle | Well-drained | Poplar, spruce, and wetland |
| SCI-SL (sand clay-sand loam) | Glacial-till with morainal landform | Plain | Moderate | Poorly-drained | Poplar and balsam fir |
| SL-CI (sand loam-clay) | Glacial-till with morainal and undifferentiated landfomrs | n/a | Moderate to steep | Well-drained | Poplar, spruce, and white birch |
| SL (sand loam) | Glacial-till and glacial lacustrine geology, morainal and and lacustrine landforms | Hummocky, undulating, and ridged | Gentle to steep | Rapid to imperfectly drained | Poplar, white birch, black spruce, and oak |

[a] **Soil texture code as delivered by RMNP's soil map layer.**

[b] **Not available**

**TABLE 3.1 Riding Mountain National Park soil texture information (continued).**

| SOIL TEXTURE DESCRIPTION | LANDFORM | MORPHOLOGICAL EXPRESSION | SLOPE | DRAINAGE | VEGETATION |
|---|---|---|---|---|---|
| SL-SL (sand loam-sand loam) | Glacial-fluvial geology and landform | Undulating | Steep | Rapidly drained | Grassland |

[a] **Soil texture code as delivered by RMNP's soil map layer**

[b] **Not available**

## Table 3.2. Riding Mountain National Park vegetation types (based on Walker and Kenkel 1997).

| VEGETATION TYPE | DESCRIPTION |
|---|---|
| Aspen parkland | -trembling aspen stands with intermittent shrubby grasslands.<br>-some balsam popular stands and small wetlands. |
| Deciduous canopy - coniferous subcanopy | -trembling aspen forest canopy with a coniferous (white or black spruce) subcanopy and sapling subcanopy.<br>-herb-rich understory . |
| Eastern deciduous forest | -high canopy pure mature trembling aspen stands or mixed eastern deciduous forest with a herb-rich understory . |
| Mixed canopy forest | -mixed canopy trembling aspen and white spruce codominated forests.<br>-black spruce, paper birch, balsam poplar and jackpine are less common. |
| Low canopy deciduous forest | -trembling aspen forest <10 m tall with some balsam poplar and paper birch, tall shrubs with forb understory. |
| Closed coniferous forest | -mature, regenerating coniferous stands of white spruce, black spruce, or balsam fir.<br>-understory of feathermoss. |
| Low shrub grassland | -grasslands with low shrubs, graminoids, and forbes.<br>-some trembling aspen and white spruce in moister areas. |
| Open-coniferous forest | -conifer-dominated stands (black spruce and/or larch *Larix laricina*) including semi-treed bogs and fens.<br>-uplands are sparsely treed with white spruce. |
| Bur oak forest | -open to semi-closed savanna bur oak forest.<br>-some white spruce, trembling aspen and paper birch. |

**Table 3.2. continued.**

| VEGETATION TYPE | DESCRIPTION |
|---|---|
| Wetland | -non-forested.<br>-graminoids or cattail (*Typha latifolia*). |
| Shrubland | -dense, tall shrubs, primarily beaked hazel.<br>-dense regenerating trembling aspen stands. |
| Regenerating coniferous forest | -dense, regenerating coniferous forest dominated by jackpine, black spruce or white spruce. |
| Grassland | -grasslands with intermittent shrubs.<br>-dominated by forbs and graminoids. |
| Agriculture | -non-forested land outside Park. |

inventory maps and against ground sample sites. An analysis of the accuracy of the classification is currently being examined. A description of each existing vegetation type is presented in Table 3.2.

## Vegetation Sample Design

Three replicates of each eligible ecological unit were sampled. The three replicates were chosen based on accessibility (i.e., accessible via roads or trails) and a perceived conformity to the ecological unit criteria (i.e., the existing vegetation type and soils group were confirmed at each site). In each replicate, three sample plots were systematically located. Circular plots (78.5 $m^2$) were established at least 20 meters away from roads and trails and whenever possible, greater than 100 meters from the edge of a differing ecological land unit (Thomas et al. 1997). A minimum of 20 meters separated the center of the sample plots. Low sample sizes precluded rigorous comparative analyses of the vegetation data between replicates. Rather, the intent was to efficiently sample as many ecological units as possible to provide a reasonable description of snowshoe hare and lynx habitat structure.

## Horizontal Abundance of Browse and Security Cover

Vegetation data were collected according to Wolff (1980) and Thomas et al. (1997). A 1- $m^2$ horizontal cover board, divided into 64-5 by 5 cm squares, was used to estimate the thickness of hare browse in each sample plot. The bottom of the cover board was mounted at 0 and 1 meters from ground level and the number of squares at least partially covered by live woody vegetation were counted from 5 meters away. Since hares browse different vegetation species across their range and since palatability varies temporally (Thomas et al. 1997), all live woody vegetation less than 1 cm diameter (Koehler and Brittell 1990) was assumed to be potential browse (available for entire twig consumption or barking) unless the species was known to be unpalatable. Vegetation species were identified using twig and bud characteristics. Based upon pellet analysis, unpalatable species for RMNP include common snowberry, Canada buffaloberry, common Labrador tea, twining honeysuckle, and cranberry (Leonard 1980). Since the Leonard (1980) study occurred in the western portion of the Park and potentially excluded vegetation species from the eastern region (Poll 1981), personal communication with site experts, as well as personal observation were also used to identify additional unpalatable species for the sample.

The horizontal cover board was also used to estimate the thickness of security cover for hares. Live, dead, and inanimate (e.g., rocks) structures were included in the sample. For both forage abundance and security cover, four readings were taken at each plot. The direction of the first reading was randomly determined and subsequent readings were taken by rotating around the cover board in 90 degree increments. The number of squares intercepted by cover in each reading was divided by 0.64 to get a percentage of area covered (Thomas et al. 1997). The arithmetic mean of the four readings was used as the final output of each plot.

## Vertical Abundance of Browse and Security Cover

Vertical browse and cover were assessed using a spherical densiometer. Thomas et al's. (1997) methodology was used to measure vertical browse and cover at 0-1 and 1-2 meters. Vertical browse included all vegetation species reflected in the spherical densiometer excluding those species unpalatable to hare in RMNP or woody stems that were too thick (>1 cm) to be classified as browse. Vertical security cover included any obstructions reflected in the spherical densiometer. The concave mirror was divided into 24 squares. Each square was imagined as being divided into four quadrants. Each quadrant was represented by an imaginary dot in the center of each of the smaller squares. Thus, the total number of dots was 96 (Lemmon 1957). Any dots intercepted by vegetation were tallied with the resulting sum being divided by 0.96 (Thomas et al. 1997). The sample produced estimates of the percent area occupied by the various vertical vegetation layers.

## Understory species dominance

Based on the premise that ecological land units with a coniferous understory provide better hare habitat than deciduous understories, Roloff (1998) suggested that a subjective evaluation of the dominant vegetation type ≤3 meters tall be performed to index winter cover composition. Ecological land units were classified as "Deciduous" if the understory vegetation type ≤3 meters was composed of >60% deciduous species. A "mixed" classification was used if the understory species composition contained between 40% and 60% of either deciduous or coniferous species. "Coniferous" classification was assigned to ecological land units if >60% of the understory was coniferous species. "None" classification is given to the ecological land units that contain no understory vegetation (Roloff 1998). These values were subjectively determined by ocular assessment at each plot.

## Other Vegetation Measures

A tree-level data inventory did not exist for RMNP by ecological land unit. Therefore, overstory variables for each of the ecological units were approximated by RMNP vegetation management specialist (pers. comm., Wybo Vanderschuit, Vegetation Management Specialist, Riding Mountain National Park, Wasagaming, Manitoba). These variables included the average diameter of trees (cm), average basal area of trees ($m^2$/ha), percent canopy cover of the forested overstory, average height of the overstory (m), and the average density of trees >10 cm dbh.

## Snow-Tracking

Snow-tracking was used as a practical means of indexing lynx habitat use to verify the performance of Roloff's (1998) HSI model. The goal of snow-tracking was to determine if lynx relative abundance varied as expected according to habitat model output. The information was also used to establish baseline data for a lynx population monitoring program for the Park. Lynx monitoring began December 7, 1997 and was completed March 22, 1998. Many of the trails were traversed more than once during the tracking period. To prevent pseudo-replication of trails travelled more than once, a central date of March 3 was chosen whereby the travel day closest to March 3 was used in the analysis. March 3 was chosen since a winter storm ended on March 1 and thus fresh snow conditions were provided. It was assumed that good tracking conditions would occur on March 3, 48 hours after the fresh snowfall.

Tracking was performed via snow machine on 833 km of trails, roads, and cutlines throughout RMNP. A minimal amount of off-trail travel was performed in order to gain additional coverage in under-represented HSI classes. The Park's traditional wolf monitoring routes were generally followed, and numerous parameters were recorded including observer name, date, weather conditions, temperature, previous night cloud cover (none, partial, complete), and date of last snowfall. At each lynx crossing, the number of different lynx tracks, direction of travel, estimated track age (in hours), and hare abundance were recorded (Appendix B). All lynx tracks were recorded as independent animals provided they could not be visually connected to one another (O'Donoghue et al. 1997). If the lynx followed a trail, it was counted only once.

## Statistical Analysis

The statistical methodology used by Murray et al. (1994) was used to test the significance of observed habitat use patterns. William's corrected G-test (Sokal and Rohlf 1981) was used to determine if the observed distribution of lynx tracks significantly differed from the distribution of habitat classes sampled .

William's corrected G-test formula (Sokal and Rohlf 1981) is as follows:

$$G = 2\sum^{a} f_i \ln (f_i / \hat{f}_i)$$

where: $f_i$ = observed frequency of lynx tracks

$\hat{f}_i$ = expected frequency of lynx tracks

a = HSI classes

The William's correction to G is computed by:

$$q = 1 + \frac{a^2 - 1}{6n(a-1)}$$

$$G_{adj} = \frac{G}{q}$$

The data fit the criteria outlined by Thomas and Taylor (1990) (Design 1) and Alldredge and Ratti (1992) for establishing confidence limits using a Bonferroni z method (Dunn and Massey 1965, Neu et al. 1974) and thus, confidence limits were calculated to determine which individual habitat classes were different. The Bonferroni z equation (Neu et al. 1974) is:

$$p_i - z_{(1-a/2k)} \sqrt{p_i(1-p_i)/n} \leq p_i \leq p_i + z_{(1-a/2k)} \sqrt{p_i(1-p_i)/n}$$

Where: $p_i$ = proportion of tracks in each habitat class

a = significance level

k = number of classes

n = sample size

The criteria required for establishing confidence limits using Bonferonni z include an investigation of resource selectivity at the population level (not at the individual animal level), a known availability of habitat, the ability to depict habitat use and availability data as proportions, and a comparison of the relative number of tracks observed in each habitat type to the proportion of respective habitats available (Thomas and Taylor 1990). Also, it was assumed that there was a relationship between animal density and relative preference (Thomas and Taylor 1990). To maintain the statistical independence of lynx observations in this study, groups of lynx were counted as a single observation (Thomas and Taylor 1990).

# RESULTS

## Ecological Land Classification

Twenty-one ecological land units (Figure 3.1) were sampled that included aspen parkland on sand-clay, aspen parkland on clay, deciduous canopy-coniferous subcanopy on sand-clay, eastern deciduous on sand-clay, deciduous canopy, coniferous subcanopy on clay, eastern deciduous on clay-loam, mixed canopy on sand-clay, mixed canopy on clay, shrubland on clay, aspen parkland on clay-loam, low canopy deciduous on clay, shrubland on sand-clay, eastern deciduous on clay, low canopy deciduous on sand, eastern deciduous on clay, mixed canopy on clay-loam, shrubland on clay-loam, closed coniferous on clay, deciduous canopy-coniferous subcanopy on clay-loam, closed coniferous on sand-clay, low shrub grassland on clay, and regenerating conifer.

## Snowshoe Hare Habitat Analysis

Vegetation variables were classified as unsuitable, marginal, or optimum based upon the relationships depicted in Roloff (1998) (Table 3.3). Each ecological land unit was assigned an HSI score based upon Roloff's (1998) methodology. For example, if a vegetation variable received an "unsuitable" score, the HSI value assigned to that variable was 0, if "marginal", then the HSI score was between 0.01 and 0.99, and if "optimum," then the HSI score was 1.00.

Roloff's (1998) model assigned snowshoe hare HSI values to each ecological land unit based upon scores given to vegetation variables (Table 3.4 and Table 3.5). The overall hare forage component was assessed using the geometric mean of horizontal and vertical forage. All ecological land units except water and agriculture were deemed to provide at least some forage suitability to snowshoe hares (Table 3.5). The best forage score was for the regenerating jackpine ecological land unit with an HSI of 1.00 (Table 3.5). The lowest suitable forage score was the low shrub grassland-clay with an HSI of 0.17 (Table 3.5).

Roloff (1998) assessed snowshoe hare winter security cover based on three measures; understory species composition, horizontal cover, and vertical cover. All but

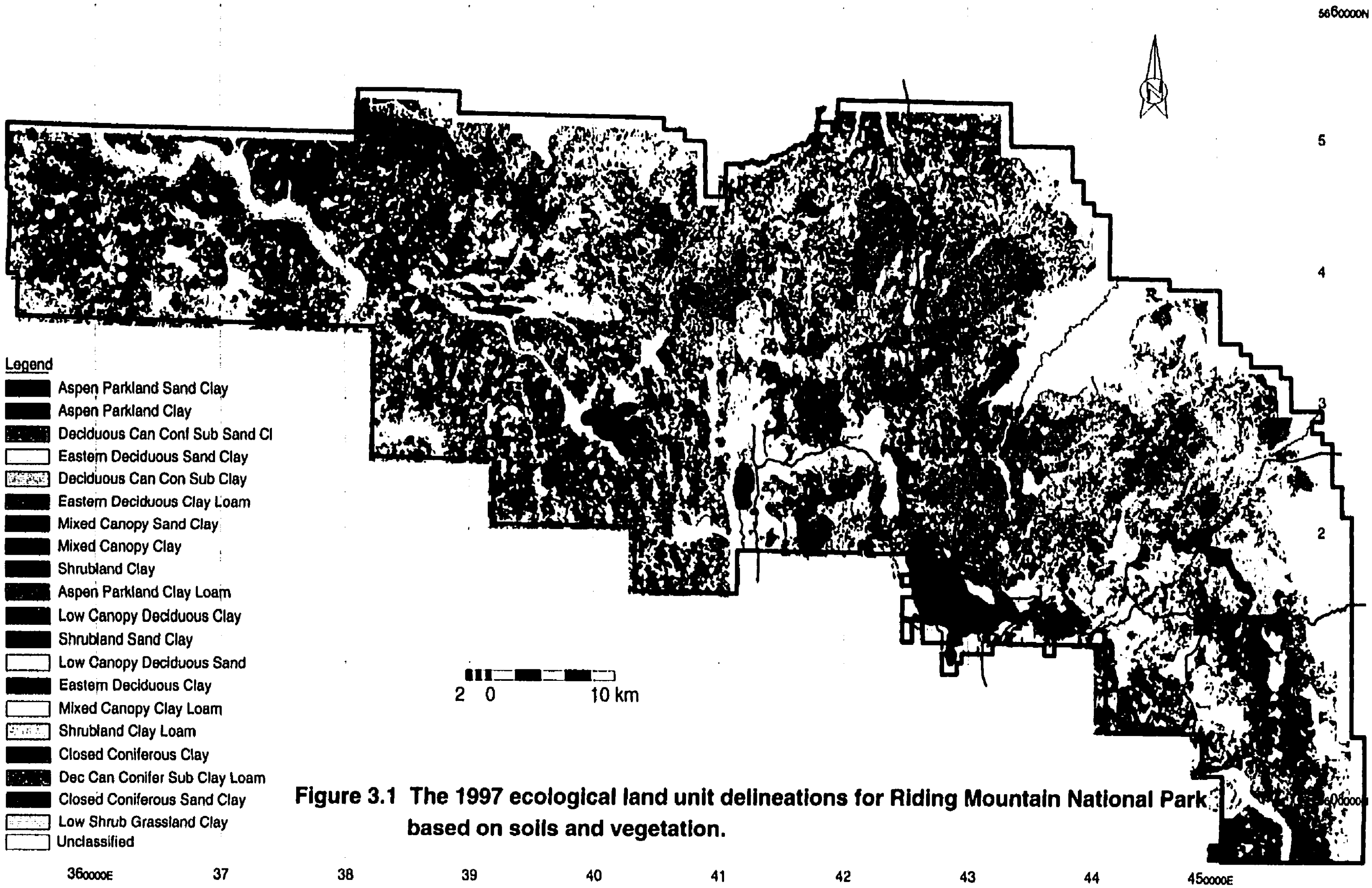



**Figure 3.1 The 1997 ecological land unit delineations for Riding Mountain National Park based on soils and vegetation.**

**Table 3.3. Snowshoe hare habitat variables and HSI scoring (based on Roloff 1998).**

| | Habitat Suitability Index Scoring[a] | | |
|---|---|---|---|
| **Vegetation Variable** | **Unsuitable** | **Marginal** | **Optimum** |
| Horizontal cover of palatable browse species, height strata 0-1 and 1-2 meters | ≤ 10% | 11-34% | ≥ 35% |
| Vertical cover of palatable browse species, height strata 0-1, 1-2, and 2-3 meters | ≤ 20% | 21-79% | ≥ 80% |
| Horizontal cover, height strata 0-1 and 1-2 meters | ≤ 40% | 41-89% | ≥90% |
| Vertical cover, height strata 0-1, 1-2, and 2-3 meters | ≤ 40% | 41-89% | ≥ 90% |
| Understory species composition | "none" | "deciduous" "mixed" | "coniferous" |

[a]HSI scoring: Unsuitable = 0.00, Marginal = 0.01-0.99, Optimum = 1.00

# Table 3.4. Mean vegetation structure in Riding Mountain National Park.

| ECOLOGICAL LAND UNIT | Horizontal cover at 0-1 meters (%) | Horizontal cover at 1-2 meters (%) | Vertical cover at 0-1 meters (%) | Vertical cover at 1-2 meters (%) | Forage at 0-1 meters (%) | Forage at 1-2 meters (%) |
|---|---|---|---|---|---|---|
| Aspen parkland - sand-clay | 95 | 84 | 32 | 25 | 31 | 24 |
| Aspen parkland - clay | 99 | 79 | 63 | 39 | 59 | 37 |
| Decid. can./conif/ subcan. - sand-clay | 79 | 66 | 26 | 17 | 20 | 12 |
| Eastern Deciduous - sand-clay | 89 | 79 | 26 | 23 | 25 | 22 |
| Decid. can./conif. subcan. - clay | 94 | 90 | 31 | 23 | 24 | 16 |
| Eastern Deciduous - clay-loam | 92 | 86 | 43 | 33 | 43 | 32 |
| Mixed Canopy - sand-clay | 87 | 80 | 40 | 20 | 29 | 11 |
| Mixed Canopy - clay | 96 | 64 | 33 | 24 | 30 | 22 |
| Shrubland - clay | 97 | 66 | 47 | 34 | 38 | 31 |
| Aspen Parkland - clay-loam | 99 | 90 | 65 | 51 | 51 | 46 |
| Low Canopy deciduous - clay | 98 | 43 | 16 | 9 | 16 | 9 |
| Shrubland - sand-clay | 97 | 67 | 48 | 16 | 37 | 16 |
| Low canopy deciduous - sand | 94 | 60 | 28 | 25 | 24 | 24 |
| Eastern Deciduous - clay | 89 | 48 | 19 | 15 | 11 | 10 |
| Mixed canopy - clay-loam | 98 | 61 | 30 | 31 | 16 | 19 |
| Shrubland - clay-loam | 99 | 57 | 44 | 33 | 35 | 27 |
| Closed coniferous - clay | 71 | 32 | 28 | 16 | 23 | 13 |
| Decid. can./conif. subcan - clay-loam | 94 | 36 | 11 | 10 | 11 | 10 |
| Closed coniferous - sand-clay | 77 | 31 | 24 | 21 | 18 | 15 |
| Low shrub grassland - clay | 98 | 3 | 7 | 0 | 2 | 0 |
| Jackpine (burn area) | 100 | 100 | 100 | 100 | 100 | 100 |
| Water, agriculture | 0 | 0 | 0 | 0 | 0 | 0 |

## Table 3.5. Snowshoe hare habitat quality values for Riding Mountain National Park.

| ECOLOGICAL LAND UNIT | Forage horizontal cover | Forage vertical cover | Forage HSI | Cover Species Composition | Security horizontal cover | Security vertical cover | Security HSI | Summer cover | Hare Winter HSI |
|---|---|---|---|---|---|---|---|---|---|
| Aspen parkland - sand-clay | 1.00 | 0.08 | 0.54 | 0.50 | 0.54 | 0.00 | 0.37 | 1.00 | 0.45 |
| Aspen parkland - clay | 1.00 | 0.04 | 0.70 | 0.50 | 0.47 | 0.15 | 0.39 | 1.00 | 0.53 |
| Decid. can./conif/ subcan. - sand-clay | 1.00 | 0.00 | 0.50 | 0.75 | 0.27 | 0.00 | 0.32 | 1.00 | 0.40 |
| Easter Deciduous - sand-clay | 1.00 | 0.04 | 0.52 | 0.50 | 0.53 | 0.00 | 0.36 | 1.00 | 0.43 |
| Decid. can./conif. subcan. - clay | 1.00 | 0.02 | 0.51 | 0.50 | 0.47 | 0.00 | 0.34 | 1.00 | 0.42 |
| Eastern Deciduous - clay-loam | 1.00 | 0.22 | 0.61 | 0.50 | 0.63 | 0.02 | 0.40 | 1.00 | 0.50 |
| Mixed Canopy - sand-clay | 1.00 | 0.05 | 0.53 | 0.50 | 0.34 | 0.00 | 0.29 | 1.00 | 0.39 |
| Mixed Canopy - clay | 1.00 | 0.07 | 0.53 | 0.50 | 0.18 | 0.00 | 0.21 | 1.00 | 0.34 |
| Shrubland - clay | 1.00 | 0.20 | 0.60 | 0.50 | 0.28 | 0.05 | 0.29 | 1.00 | 0.42 |
| Aspen Parkland - clay-loam | 1.00 | 0.54 | 0.77 | 0.50 | 0.75 | 0.39 | 0.54 | 1.00 | 0.64 |
| Low Canopy deciduous - clay | 0.88 | 0.00 | 0.44 | 0.50 | 0.02 | 0.00 | 0.07 | 1.00 | 0.18 |
| Shrubland - sand-clay | 1.00 | 0.10 | 0.55 | 0.50 | 0.18 | 0.05 | 0.24 | 1.00 | 0.36 |
| Low canopy deciduous - sand | 1.00 | 0.05 | 0.52 | 0.50 | 0.22 | 0.00 | 0.23 | 1.00 | 0.35 |
| Eastern Deciduous - clay | 1.00 | 0.00 | 0.50 | 0.50 | 0.07 | 0.00 | 0.14 | 1.00 | 0.26 |
| Mixed canopy - clay-loam | 1.00 | 0.00 | 0.50 | 0.75 | 0.24 | 0.00 | 0.30 | 1.00 | 0.39 |
| Shrubland - clay-loam | 1.00 | 0.12 | 0.56 | 0.50 | 0.15 | 0.03 | 0.21 | 1.00 | 0.35 |
| Closed coniferous - clay | 0.96 | 0.02 | 0.49 | 1.00 | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 |
| Decid. can./conif. subcan - clay-loam | 0.87 | 0.00 | 0.43 | 0.50 | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 |
| Closed coniferous - sand-clay | 0.95 | 0.00 | 0.47 | 1.00 | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 |
| Low shrub grassland - clay | 0.33 | 0.00 | 0.17 | 0.50 | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 |
| Jackpine (burn area) | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Water, agriculture | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

five ecological land units (closed coniferous-clay, deciduous canopy/coniferous subcanopy-clay-loam, closed coniferous-sand-clay, low shrub grassland-clay, and water/agriculture) were considered to provide at least some level of cover to snowshoe hares (Table 3.5). The winter vertical and horizontal cover for hare were combined using an arithmetic mean, and then geometrically combined with the understory composition component (Roloff 1998).

The snowshoe hare winter HSI was calculated from the winter habitat components (forage and cover) using a geometric mean (Roloff 1998). If either forage or cover was absent, then the final snowshoe hare HSI equalled 0.00. The ecological land units that received an HSI rating of 0.00 were closed coniferous-clay, deciduous canopy/coniferous subcanopy-clay-loam, closed coniferous-sand-clay, low shrub grassland-clay, and agriculture/water (Table 3.5). The regenerating jackpine ecological land unit was calculated as optimum hare habitat (Table 3.5). All other ecological land units provided varying levels of habitat suitability for hare (Table 3.5).

### Snowshoe Hare Home Ranges

Indices of snowshoe hare habitat suitability for the modelling area were generated and mapped (Figure 3.2). Using Roloff and Haufler's (1997) and Roloff's (1998) methodology, 444 viable hare home ranges (6,442 ha) were mapped for RMNP. Similarly, 474 marginal hare home ranges (227,917 ha) were mapped. These home ranges were mapped using the classified area in RMNP, thus these numbers may be conservative value. These viable and marginal home range maps were used as inputs into the lynx foraging model.

## LYNX HABITAT ASSESSMENT

### Lynx Foraging Component

Forage potential for each lynx home range was estimated based upon the number of viable, marginal, and non-viable snowshoe hare home ranges that each lynx home range encompassed (Roloff 1998). Figure 3.3 shows the lynx home-range-level foraging

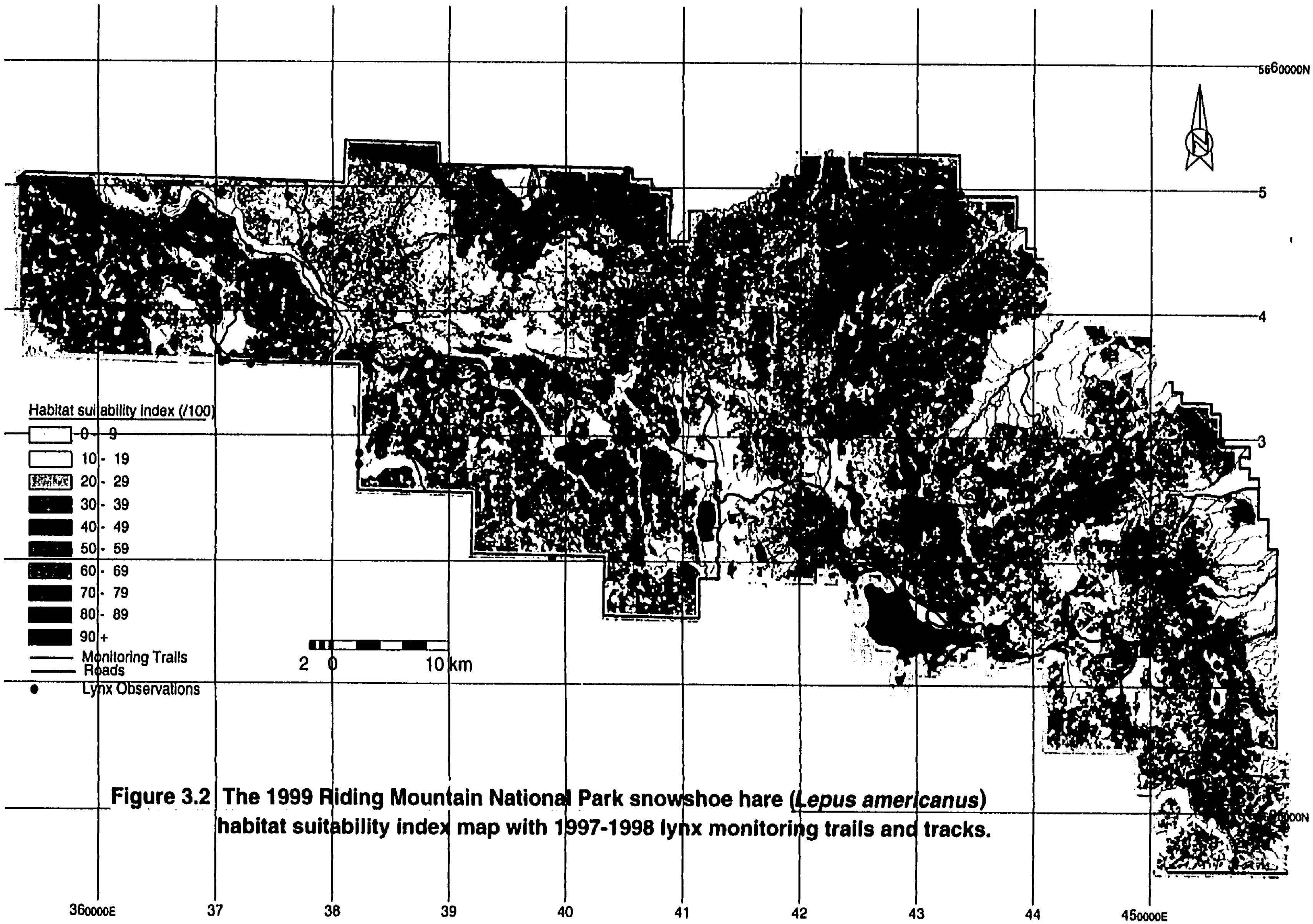



**Figure 3.2** The 1999 Riding Mountain National Park snowshoe hare (*Lepus americanus*) habitat suitability index map with 1997-1998 lynx monitoring trails and tracks.

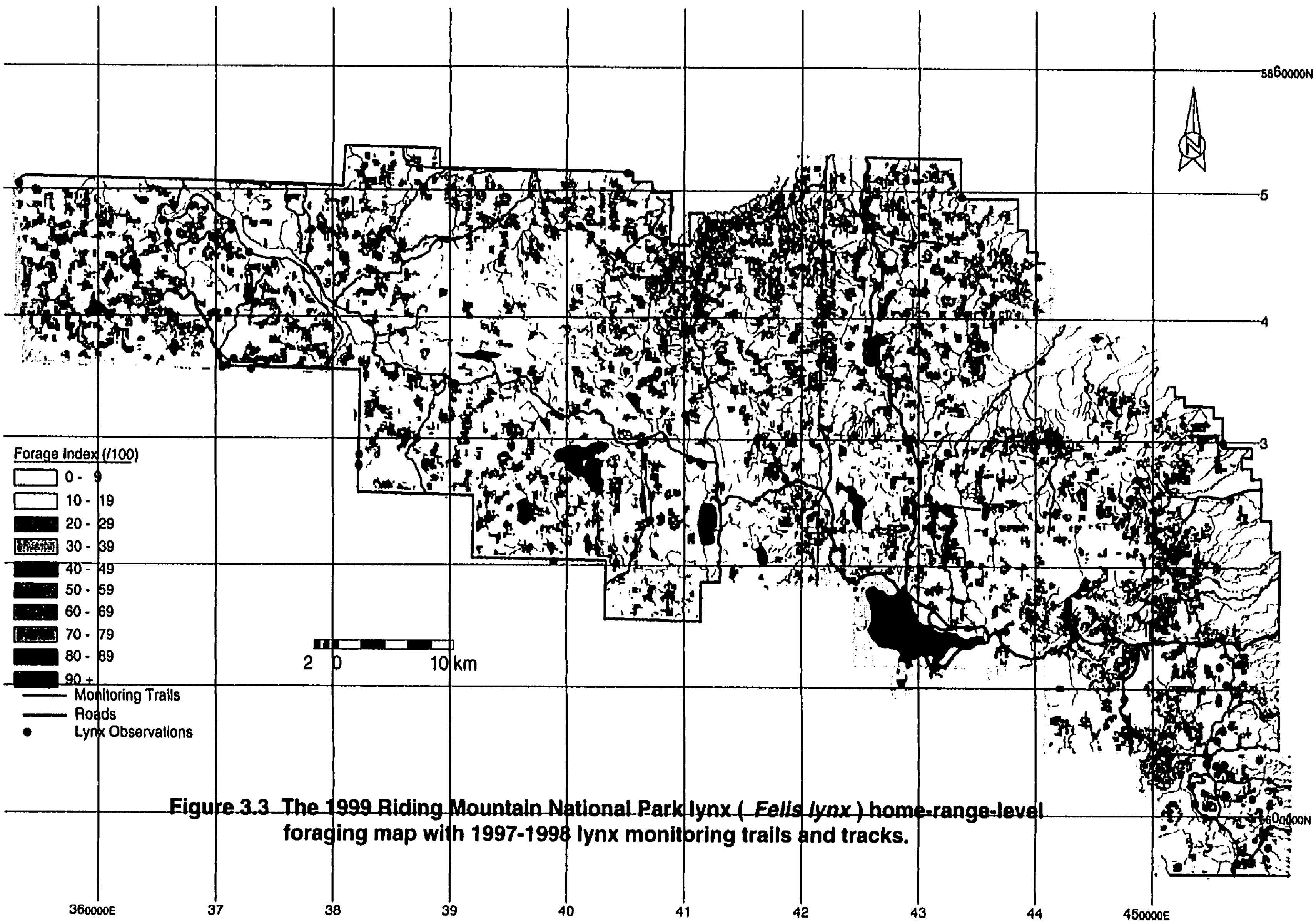



**Figure 3.3 The 1999 Riding Mountain National Park lynx ( *Felis lynx* ) home-range-level foraging map with 1997-1998 lynx monitoring trails and tracks.**

scores. Lynx forage habitat quality ranged from 0.00 - 1.00 HSI, with the majority of habitat in 0.30 - 0.59 HSI class (Table 3.6). Only 0.8% of the total classified area fell into the 0.90+ HSI forage class. High quality foraging "pockets" were interspersed throughout the Park (Figure 3.3). High quality foraging habitat often occurred in isolated patches (Figure 3.3).

## Lynx Denning Component

Denning attributes for RMNP and the critical denning values for each ecological land unit are presented in Table 3.7. The first phase of the lynx denning assessment included determining which ecological land units satisfied the vegetation cover type, site potential, canopy closure, and minimum area criteria (Roloff 1998). Using Roloff's (1998) denning criteria, the only ecological land units that met the Phase 1 denning criteria were aspen-parkland-sand-clay, mixed canopy-sand-clay, mixed canopy-clay, aspen parkland-clay-loam, closed coniferous-clay, deciduous canopy/coniferous subcanopy-clay-loam, and closed coniferous-sand-clay (Table 3.7). In the modelling process, polygons constituting these ecological land units were advanced to Phase 2 of the denning assessment.

Phase 2, the juxtaposition and interspersion of potential denning areas and Phase 3, the presence of dead and downed woody debris for denning sites, are home-range-level assessments (Roloff 1998). Three sets of criteria had to be established before potential denning sites in Phase 2 could be assessed at Phase 3. The perimeter of potential denning sites had to be at least 50% surrounded by suitable lynx denning, foraging, or travel cover. Also, at least 30% of the area surrounding the potential denning area had be within 0.8 km of suitable lynx foraging habitat. Suitable summer lynx forage HSI values were calculated from measurements of the vertical vegetation cover component between 0-1 meters in each ecological land unit. Optimum lynx summer forage habitat occurred in aspen parkland-clay, aspen parkland-clay-loam, and jackpine ecological land units. The result of this analysis was a GIS coverage of the polygons that met the denning criteria.

**Table 3.6 Lynx forage habitat qualities and amounts in Riding Mountain National Park.**

| HSI CLASS | AMOUNT OF FORAGE AREA (ha) | PERCENT OF CLASSIFIED AREA |
|---|---|---|
| 0.00-0.09 | 33,400 | 13 |
| 0.10-0.19 | 8,182 | 3 |
| 0.20-0.29 | 6,225 | 3 |
| 0.30-0.39 | 48,397 | 20 |
| 0.40-0.49 | 101,653 | 43 |
| 0.50-0.59 | 30,668 | 13 |
| 0.60-0.69 | 8,412 | 4 |
| 0.70-0.79 | 0 | 0 |
| 0.80-0.89 | 0 | 0 |
| 0.90+ | 1,967 | 1 |
| **TOTAL** | **238,904** | |

## Table 3.7. Lynx denning attributes and mean values for Riding Mountain National Park.

| | CRITICAL VALUE | | | |
|---|---|---|---|---|
| | 25 cm dbh | $\geq 3.72 m^2/ha$ | >50% | |
| ECOLOGICAL LAND UNIT | Diameter of overstory (cm) | Average basal area ($m^2/ha$) | Canopy closure (%) | Vertical cover between 0-1 meters[a] (%) |
| aspen parkland-sand-clay | 25 | 98 | 70 | 32 |
| aspen parkland-clay | 22 | 91 | 90 | 63 |
| decid.can/conif. subcan.-sand-clay | 24 | 54 | 40 | 26 |
| eastern deciduous-sand-clay | 23 | 133 | 90 | 26 |
| decid.can/con subcan.-clay | 23 | 91 | 80 | 32 |
| eastern deciduous-clay-loam | 20 | 126 | 100 | 43 |
| mixed canopy-sand-clay | 25 | 118 | 90 | 40 |
| mixed canopy-clay | 25 | 118 | 90 | 33 |
| shrubland-clay | 6 | 2 | 80 | 47 |
| aspen parkland-clay-loam | 27 | 92 | 50 | 65 |
| low canopy deciduous-clay | 13 | 133 | 90 | 16 |
| shrubland-sand-clay | 7 | 5 | 80 | 48 |
| low canopy deciduous-sand | 15 | 212 | 90 | 28 |
| eastern deciduous-clay | 18 | 163 | 100 | 19 |
| mixed canopy-clay-loam | 27 | 69 | 40 | 30 |
| shrubland-clay-loam | 8 | 4 | 70 | 44 |
| closed coniferous-clay | 26 | 85 | 80 | 28 |
| decid.can./conif. subcan.-clay-loam | 25 | 98 | 60 | 11 |
| closed coniferous-sand-clay | 27 | 80 | 70 | 24 |
| low shrub grassland-clay | 2 | 0 | 50 | 7 |
| jackpine (burned area)-various | 8 | 8 | 100 | 100 |

[a] Summer forage HSI value for lynx is based upon these values. Optimum lynx summer forage exists at $\geq$ 60% vertical cover and summer cover habitat quality is 0 when $\leq$ 20% vertical cover exists (Roloff 1998).

The number and arrangement of potential denning sites within a home-range-sized area was determined by Phase 3 of Roloff's (1998) model. In this phase, the HSI score for a home-range-sized area is based upon the average distance in a home range to a denning site. Optimal conditions occur when there is a denning site about every 16 ha (Roloff 1998). Figure 3.4 and Table 3.8 show that the majority of the Park was estimated to have 0.90+ HSI quality denning habitat suggesting that denning habitat was not limiting.

## Lynx Interspersion Component

Travel needs are addressed by Roloff's (1998) interspersion component. The first step was to determine the areas that were considered "non-lynx" cover. Subsequently, the amount and spatial distribution of map polygons that are "non-lynx" habitat within the home range was assessed (Roloff 1998). "Non-lynx" habitat was assigned to map polygons with a summer foraging or denning HSI of 0.00 that had permanent "openings" >91 meters in width and satisfied the criteria outlined in Table 3.9 (Roloff 1998). Subsequently, map polygons not already identified as forage, denning, or "non-lynx" habitat were delineated as travel cover (Roloff 1998).

After "non-lynx" and potential travel habitats were identified and mapped, the average nearest distance within a home range to travel barriers was calculated using the intersection points in a 100 X 100 meter grid (Roloff 1998). The interspersion map (Figure 3.5) was based on the 100 X 100 meter grid which corresponds to the maximum theoretical distance a lynx will travel without sufficient cover (Roloff 1998). The home-range-level habitat interspersion map (Figure 3.5) portrays suitable travel habitat for lynx in RMNP. Lower travel suitability appeared along the southern edges of the Park, especially the south-western quadrant. This area of the Park is interspersed by grassland, thus open areas are quite common. The northern and eastern perimeters of the Park appear to provide excellent lynx interspersion components (HSI = 0.90-0.99). The majority of the Park (57%) was classified as not limiting to lynx travel (Table 3.10).

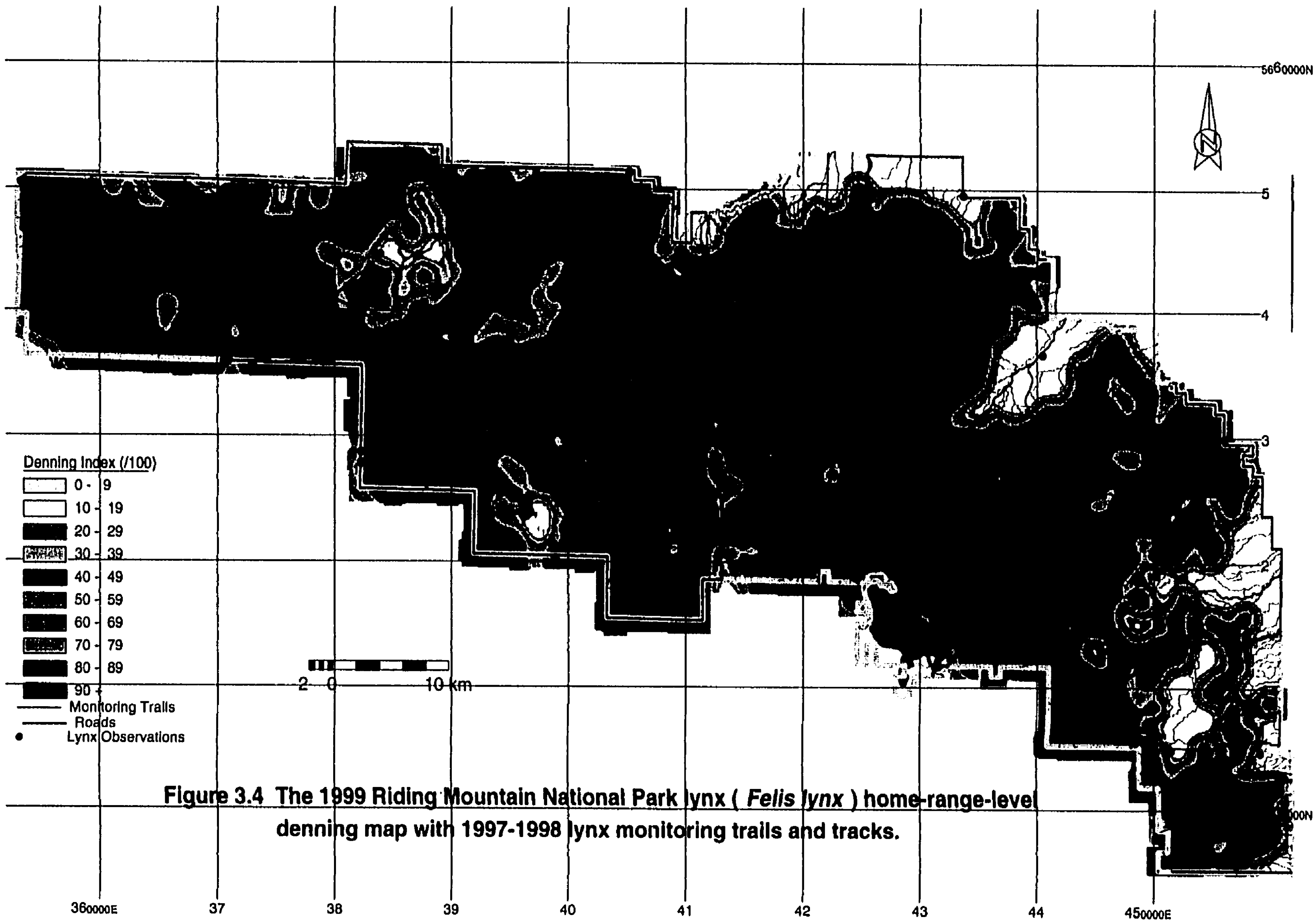



Figure 3.4 The 1999 Riding Mountain National Park lynx ( *Felis lynx* ) home-range-level denning map with 1997-1998 lynx monitoring trails and tracks.

**Table 3.8 Lynx denning habitat qualities and amounts in Riding Mountain National Park.**

| HSI CLASS | AMOUNT OF DENNING AREA (ha) | PERCENT OF CLASSIFIED AREA |
|---|---|---|
| 0.00-0.09 | 2,058 | 1 |
| 0.10-0.19 | 2,452 | 1 |
| 0.20-0.29 | 3,037 | 1 |
| 0.30-0.39 | 3,619 | 2 |
| 0.40-0.49 | 4,464 | 2 |
| 0.50-0.59 | 6,085 | 3 |
| 0.60-0.69 | 7,894 | 3 |
| 0.70-0.79 | 8,211 | 3 |
| 0.80-0.89 | 11,453 | 5 |
| 0.90+ | 189,593 | 79 |
| **TOTAL** | **238,866** | |

## Table 3.9. Mean values for lynx interspersion attributes for Riding Mountain National Park.

| | CRITICAL VALUE | | | |
|---|---|---|---|---|
| | <2 meters | <72 trees/ha | <50% | |
| ECOLOGICAL LAND UNIT | Average tree height (m) | Average number of trees/ha | Horizontal cover between 1-2 meters (%) | Vertical cover between 0-1 meters[a] |
| aspen parkland-sand-clay | 20 | 500 | 84 | 32 |
| aspen parkland-clay | 24 | 600 | 79 | 63 |
| decid.can/conif. subcan.-sand-clay | 18 | 300 | 66 | 26 |
| eastern deciduous-sand-clay | 15 | 800 | 79 | 26 |
| decid.can/con subcan.-clay | 21 | 550 | 90 | 32 |
| eastern deciduous-clay-loam | 20 | 1000 | 86 | 43. |
| mixed canopy-sand-clay | 23 | 600 | 80 | 40 |
| mixed canopy-clay | 23 | 600 | 64 | 33 |
| shrubland-clay | 4 | 200 | 66 | 47 |
| aspen parkland-clay-loam | 20 | 400 | 90 | 65 |
| low canopy deciduous-clay | 8 | 2500 | 43 | 16 |
| shrubland-sand-clay | 6 | 300 | 67 | 48 |
| low canopy deciduous-sand | 14 | 3000 | 60 | 28 |
| eastern deciduous-clay | 16 | 1600 | 48 | 19 |
| mixed canopy-clay-loam | 20 | 300 | 61 | 30 |
| shrubland-clay-loam | 6 | 200 | 57 | 44 |
| closed coniferous-clay | 20 | 400 | 32 | 28 |
| decid.can./conif. subcan.-clay-loam | 20 | 500 | 36 | 11 |
| closed coniferous-sand-clay | 22 | 350 | 31 | 24 |
| low shrub grassland-clay | 5 | 100 | 3 | 7 |
| jackpine (burned area)-various | 5 | 400 | 100 | 100 |

[a] Summer forage HSI value for lynx is based upon these values. Optimum lynx summer forage exists at ≥60% vertical cover and summer cover habitat quality is 0 when ≤20% vertical cover exists (Roloff 1998).

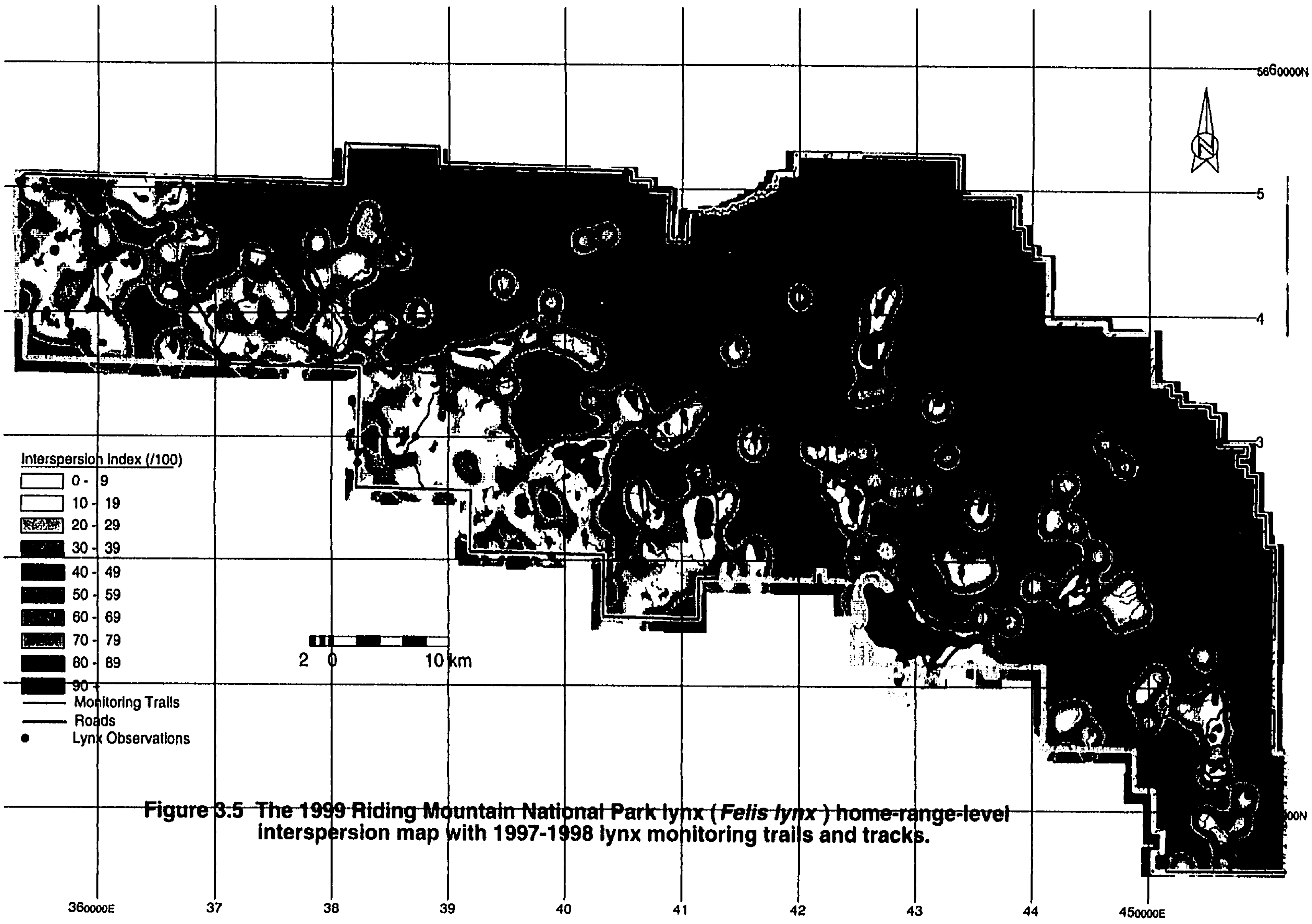



Figure 3.5 The 1999 Riding Mountain National Park lynx (*Felis lynx*) home-range-level interspersion map with 1997-1998 lynx monitoring trails and tracks.

**Table 3.10 Lynx interspersion HSI classes and amount of travel habitat in Riding Mountain National Park.**

| HSI CLASS | AMOUNT OF INTERSPERSION AREA (HA) | PERCENT OF CLASSIFIED AREA |
|---|---|---|
| 0.00-0.09 | 18,390 | 8 |
| 0.10-0.19 | 12,038 | 5 |
| 0.20-0.29 | 12,654 | 5 |
| 0.30-0.39 | 11,816 | 5 |
| 0.40-0.49 | 11,274 | 5 |
| 0.50-0.59 | 10,005 | 4 |
| 0.60-0.69 | 9,189 | 4 |
| 0.70-0.79 | 8,539 | 4 |
| 0.80-0.89 | 8,385 | 4 |
| 0.90+ | 136,577 | 57 |
| **TOTAL** | **238,867** | |

## Lynx HSI Analysis

Figure 3.6 manifests the spatial portrayal of lynx home-range-level habitat in RMNP according to Roloff's (1998) model. The geometric mean of all lynx habitat components, foraging, denning, and interspersion, provide a single lynx home-range-level HSI value (Roloff 1998) for each map pixel. Each pixel represents the 250 ha of habitat area (which corresponds to the lynx allometric home range) (Roloff and Haufler 1997). Habitat quality is generated in the typical HSI format ranging from 0.0 to 1.0 denoting unsuitable to optimum lynx home-range-level habitat, respectively.

The majority of the Park was classified as 0.00-0.09, and 0.20 to 0.60 HSI (Table 3.11). Much of the Park was classified as poor to moderate lynx habitat. Only 145 ha of the Park was considered optimal lynx habitat quality. The spatial distribution of higher quality lynx habitat is patchy (Figure 3.6). Optimum habitat patches occur sporadically throughout the Park, however, there is a concentration of optimum habitat in the region of the 1980 burn (Figure 3.6).

## Snow-Tracking

Snow-tracking was performed on 833 km of trails in 1997 and 1998. The majority of the trails were traversed in February and March of 1998, however, the entire tracking period extended from December 7, 1997 to March 24, 1998. Three percent of the trails were traversed during December 7-20, 1998, 4% during January 18-31, 1999, 57% during February 15-28, 1999, 31% during March 1-14, 1999, and 5% from March 15-28, 1999. There were a total of 107 replicate lynx tracks. To satisfy the assumption of statistical independence (Thomas and Taylor 1990), groups of tracks were recorded as a single observation.

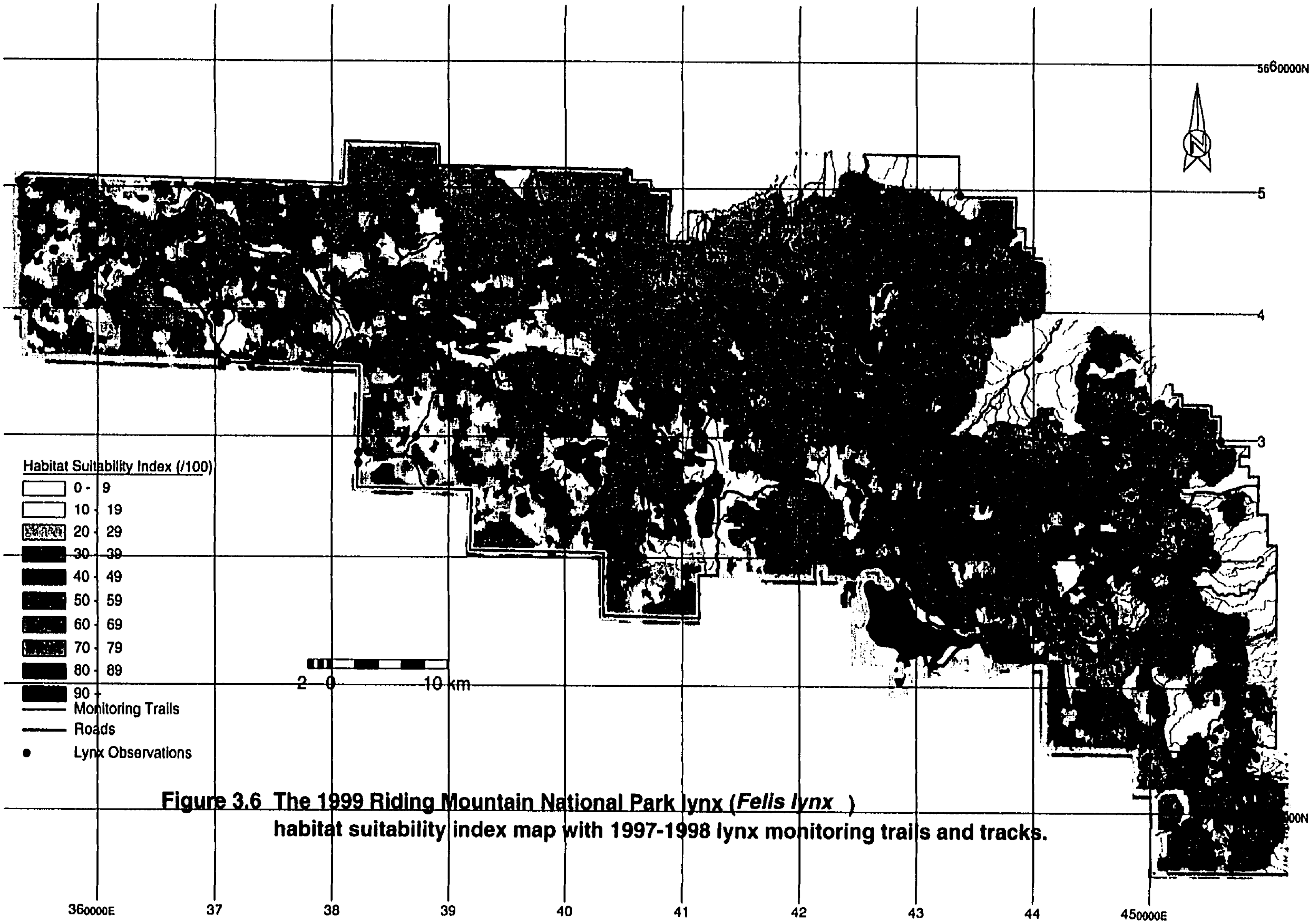



**Figure 3.6 The 1999 Riding Mountain National Park lynx (*Felis lynx* ) habitat suitability index map with 1997-1998 lynx monitoring trails and tracks.**

**Table 3.11. Area covered by each HSI class in Riding Mountain National Park.**

| HSI CLASS | AREA COVERED (ha) |
|---|---|
| 0.00-0.09 | 17,778 |
| 0.10-0.19 | 7,333 |
| 0.20-0.29 | 20,861 |
| 0.30-0.39 | 48,225 |
| 0.40-0.49 | 55,893 |
| 0.50-0.59 | 47,159 |
| 0.60-0.69 | 34,251 |
| 0.70-0.79 | 5,714 |
| 0.80-0.89 | 1,514 |
| 0.90-0.99 | 145 |
| **TOTAL** | **238,873** |

## Statistical analysis of lynx habitat utilization

Nine of the 10 lynx HSI classes were traversed during the tracking survey. G-tests were performed for two lynx habitat class resolutions. The assumptions of the G-test was that the sample was random and at least of nominal (categorical) scale. The first tests were conducted for HSI class increments of 0.10 at 80% confidence level (Tables 3.12). The G-test was significant (G = 30.974, df = 8, p = 0.05), however, Bonferonni confidence intervals indicated that only the 0.00-0.09 class was significant (Tables 3.12). The 0.00-0.09 class was used less than expected in proportion to the amount of that habitat class surveyed.

In order to determine if the HSI model functioned better at a coarser resolution, the habitat quality scores were clumped into three equal classes (0.30 HSI class increments). Again, the G-test was significant at 95% confidence levels (G = 11.402, df = 2, p = 0.05). In the 80% confidence limits, the 0.00-0.29 class was used less than expected and the 0.60-0.89 class was used more than expected in proportion to habitat availability (Table 3.13).

**Table 3.12. Lynx HSI class assessment for Riding Mountain National Park (class increment of 0.10, p = 0.20.)**

| | | | | 80% CONFIDENCE | | |
|---|---|---|---|---|---|---|
| HSI CLASS | LENGTH OF TRAIL (km$^2$) | # OF LYNX TRACKS | PROPORTION OF TRAIL IN EACH HSI CLASS | LOWER CONFIDENCE LIMIT | UPPER CONFIDENCE LIMIT | G-TEST |
| 0.00-0.09 | 154.10 | 6 | 0.1849 | 0.0058 | 0.1063 | 30.9740 |
| 0.10-0.19 | 25.33 | 6 | 0.0304 | 0.0051 | 0.1070 | |
| 0.20-0.29 | 90.49 | 8 | 0.1086 | 0.0168 | 0.1327 | |
| 0.30-0.39 | 174.34 | 24 | 0.2092 | 0.1332 | 0.3154 | |
| 0.40-0.49 | 173.12 | 23 | 0.2077 | 0.1252 | 0.3047 | |
| 0.50-0.59 | 114.44 | 20 | 0.1373 | 0.1010 | 0.2728 | |
| 0.60-0.69 | 83.23 | 12 | 0.0999 | 0.0426 | 0.1817 | |
| 0.70-0.79 | 15.28 | 4 | 0.0183 | -0.0046 | 0.0794 | |
| 0.80-0.89 | 3.02 | 4 | 0.0036 | -0.0046 | 0.0794 | |
| **TOTAL** | **833.35** | **107** | | | | |

**Table 3.13. Lynx HSI class assessment for Riding Mountain National Park (class increment of 0.30, p = 0.20.)**

| | | | | 80% CONFIDENCE | | |
|---|---|---|---|---|---|---|
| HSI CLASS | LENGTH OF TRAIL ($km^2$) | # OF LYNX TRACKS | PROPORTION OF TRAIL IN EACH HSI CLASS | LOWER CONFIDENCE LIMIT | UPPER CONFIDENCE LIMIT | G-TEST |
| 0.00-0.29 | 269.92 | 20 | 0.3239 | 0.1266 | 0.2472 | 11.4022 |
| 0.30-0.59 | 461.9 | 67 | 0.5543 | 0.5523 | 0.7001 | |
| 0.60-0.89 | 101.53 | 20 | 0.1218 | 0.1259 | 0.2480 | |
| **TOTAL** | **833.35** | **107** | | | | |

# DISCUSSION

Roloff's (1998) lynx HSI model provided RMNP managers with a pragmatic tool to assess, monitor, and project lynx habitat in the Park with at least 80% confidence. Foraging, denning, and travel habitat components provided the basis for the lynx HSI model (Roloff 1998). Analysis of all these components is essential for examining the quality of habitat available to lynx in a given region. The goal of this study was to provide empirical verification of Roloff's (1998) lynx HSI model and to provide RMNP with baseline data on lynx abundance.

The initial step in the lynx habitat modelling project was to assess and classify the ecological land units in RMNP. This analysis entailed combining the soil and vegetation attributes of RMNP to spatially portray the Park's ecological land descriptions (Figure 3.1). Specific attributes for each ecological land unit were then measured and modelled to determine the forage habitat potential for lynx (Figure 3.3). The best way of indexing lynx forage potential is to evaluate the quality of habitat available to snowshoe hare, the lynx's main prey (Elton and Nicholson 1942, Seton 1953, Keith 1963, Nellis and Keith 1968, Nellis et al. 1972). The vegetation sampling conducted in the Park provided data on the characteristics of horizontal and vertical cover and forage that is important to snowshoe hares (Table 3.5). In RMNP, regenerating jackpine was classified as the most suitable (HSI 1.00) hare habitat and closed coniferous-clay, deciduous canopy/coniferous subcanopy-clay-loam, closed coniferous-sand-clay, and low shrub grassland-clay were classified as unsuitable (HSI 0.00) habitat. Throughout the duration of this project, greater numbers of lynx were observed in areas of RMNP that contained abundant hares. The majority (96%) of lynx tracks were observed in areas where there were at least some hare tracks as evidenced from the snow-tracking surveys. This observation is consistent with other studies (Koehler et al. 1979, Parker et al. 1983, Ward and Krebs 1985, Bailey et al. 1986, Koehler 1990). The 0.90+ HSI forage class accounted for only 0.8% of the total classified area (Table 3.6). Therefore, only a small proportion of the Park provides optimal (1.00) quality hare habitat. This finding is significant since during hare population lows, it is the highest quality habitat that will continue to contribute individuals to the population. As such, these areas are vital to lynx during periods of resource scarcity.

Denning is another component of Roloff's (1998) lynx model. The majority (79%) of the classified area fell within 0.90+ HSI (Table 3.8) indicating that much of the Park currently provides high quality denning opportunities for lynx. No empirical data existed on the specific locations of lynx dens in RMNP and therefore, it is not possible to verify the denning assessment.

Interspersion is the final component in Roloff's (1998) model. The more travel barriers a lynx was apt to encounter within its home range, the poorer the HSI rating of that home range. Again much of the Park provides suitable travel habitat for lynx. Approximately 57% of the Park currently provides optimal interspersion habitat (0.90+ HSI). The lower interspersion HSI values are found throughout the rest of the Park (Table 3.10).

Based on data from the three components, foraging, denning, and interspersion, it appears that foraging habitat is the most limiting factor on RMNP since 79% of the classified area in RMNP is suitable denning habitat, and 57% provides suitable travel habitat for lynx. Less than 1% of the Park provides optimum lynx forage habitat. This result should be used to direct lynx management in RMNP by incorporating management practices, such as prescribed burning, that will increase the area of early successional forest in the Park providing areas of suitable hare and lynx forage habitat.

In RMNP, Roloff's (1998) lynx HSI model generated habitat maps that generally corresponded to the locations of lynx observations. When the habitat classes were grouped in 0.10 increments (i.e., 0.00-0.09, 0.01-0.19,...,0.91-1.00), a significant difference in habitat use versus availability was reflected (Tables 3.12). However, only the 0.00-0.09 HSI class was deemed significant using the Bonferonni z-statistic. Therefore, only the lowest HSI class was used differently (less) than expected. When the resolution was broadened to 0.30 HSI increments (i.e. 0.00-0.29, 0.30-0.59 and 0.60-0.89), the model more closely reflected the expected patterns in habitat use and availability (Tables 3.13). The 0.00-0.29 and 0.60-0.89 HSI class were both significant at the 80% confidence level where the low HSI class was used less than expected in proportion to habitat availability and the 0.60-0.89 HSI class was used more than expected in proportion to availability (Table 3.13), consistent with model predictions. Therefore, RMNP land managers can use Roloff's (1998) lynx HSI model as a tool to assess lynx habitat in RMNP with an expected confidence of at least 80%.

Several factors may have affected the results of these analyses. Error may have existed in mapping the vegetation and soils. Tracking conditions during the winter of 1997-1998 were less than ideal. Many observers were responsible for tracking lynx, thus observer bias could be a factor. In addition, some observers did not use a GPS unit and mapped the track locations on hard-copy maps. The tracking period was intended to occur around a one week period in early March, however, data were collected over a four month period. The majority of the data however, were collected over a 4-week period (February 15-March 14, 1999). The time lag between samples may have increased the likelihood that lynx in poorer quality habitat could have travelled more than lynx in better quality habitat and thus their tracks could have been counted more than once. The likelihood of this error could have increased since most of the tracking in the best quality habitats occurred over a shorter time frame than the poorer quality habitats. To mitigate some of these errors, data from the 1999 tracking season were collected from February 15 to March 15. Tracking conditions were better during this time and all observers had more experience at tracking lynx and all used a GPS unit to record lynx locations. These data will be analyzed to determine if the observed patterns from the 1997-1998 data change.

Other studies that included components of lynx habitat usage have also found that lynx used habitats disproportionately to habitat availability (Parker et al. 1983, Quinn and Thompson 1987, Koehler 1990, Murray et al. 1994, Poole et al. 1996). It is difficult to compare the results of these studies to those found in this study because of the varying parameters. The previous studies have examined lynx habitat preference generally in terms of canopy cover whereas understory characteristics played an integral role in the RMNP study. Other lynx habitat studies (Parker et al. 1983, Koehler 1990, Koehler and Brittell 1990, Murray and Boutin 1994, Poole et al. 1996) acknowledged the need for varying successional stages for lynx, this study incorporated those requirements. The 80% confidence of Roloff's (1998) model manifests the importance of these successional habitat requirements.

Roloff's (1998) model, as modified by this study, can also be used to project lynx habitat in RMNP. As used herein, Roloff's (1998) model provided a snapshot of the quality and quantity of habitat that exists in RMNP at the present time. Data from this study suggested that lynx foraging habitat is currently the limiting factor in RMNP. Optimum snowshoe hare habitat typically occurs in early successional forest (Koehler and Aubry 1994), however, historical and present factors have influenced natural successional patterns on RMNP. Historically, logging, grazing, and fire perpetuated early seral stages. Currently, decades of fire suppression have encouraged older successional stages. As a result, the Park is slowly progressing to mid-to-late-seral stages thereby decreasing the amount of early successional habitat. This trend could be detrimental to the RMNP lynx population by limiting the habitat available to snowshoe hare, thereby limiting lynx foraging potential. During the formal tracking survey and at other sporadic times, lynx are often observed in the regenerating jackpine ecological land unit that burned in 1980. Field observations in this area also demonstrated that it supports high numbers of hares. In addition, the HSI analysis indicated that much of the high quality lynx habitat (Figure 3.6) fell within this burned area, supporting the contention that lynx require early seral stages.

Park managers recognize the potential consequences of such trends and thus are examining the importance of ecological integrity and trying to find means to achieve it. Reintroduction of some natural processes, such as fire, is one way managers are trying to regain the ecological integrity of the Park. They have begun implementing prescribed burns to attempt to restore "natural" succession since traditionally, fire played an integral role in the successional patterns of RMNP. This practice should continue to prevent the Park from becoming a region dominated by a single successional chronosequence. Late successional forest do not appear to provide the level of suitable foraging habitat for long-term population success of snowshoe hares.

Transformations of the landscape will therefore occur due to natural events (such as fire), human manipulation (such as prescribed burns), or other practices unforeseeable at this time. The current habitat maps produced by this study will become outdated. It is recommended that Roloff's (1998) methodology be utilized to update these maps for future lynx and hare management issues.

An annual lynx monitoring program has been implemented in RMNP to monitor the changing lynx population and to further verify Roloff's (1998) lynx HSI modelling framework. At present, the hare population has been consistently high for the past several years (personal observation). Observations from Park staff indicated hare numbers were slightly decreasing in some areas of the Park during the winter of 1999. It is likely that the hare population will crash in 2000 or 2001. Thus, it is expected that the lynx population will start to decline within a few years following the hare decline as demonstrated by decades of fur harvest data (Elton and Nicholson 1942). According to Roloff's (1998) model, lynx should begin appearing less and less in areas designated as poor lynx HSI classes and become more concentrated in areas of high lynx HSI classes. This expected trend will allow further verification of the lynx HSI model by examining the number and distribution of lynx tracks in varying habitat classes. Annual lynx tracking should continue following the guidelines presented in Chapter 5 and Appendix C of this document. Tracking should occur on an annual basis for approximately ten more years, or until the completion of a full lynx population cycle. The statistical analysis presented in this study should be repeated annually to further assess the reliability of Roloff's (1998) lynx HSI modelling framework. This analysis will provide managers with information about how well the model performs at all stages of the lynx population cycle. Once the full cycle has been monitored, assuming Roloff's (1998) model continues to reasonably predict lynx habitat usage and population trends, then RMNP managers can rely on the model more heavily as a tool to predict lynx habitat availability and population trends in RMNP. Tracking should then be performed on a tri-annual basis as a continuing check of the model. Finally, the vegetation maps must be updated every 10 years to reflect changes to vegetation over time. Changing vegetation will affect lynx habitat availability.

It is also recommended that another study be initiated to more thoroughly document lynx habitat use and fitness. Telemetry studies could provide more detailed analysis of lynx habitat use versus availability and provide lynx demographic data by habitat class. Telemetry data may also provide information on whether the tracks collected in the monitoring survey are from lynx using areas as travel or foraging habitats.

Roloff's (1998) lynx HSI model should be used by RMNP land managers as a tool to assess, monitor, and project lynx habitat. It appears to be a pragmatic tool that can be used to guide land-management decisions that will affect lynx habitat and population trends within RMNP. However, the need for adaptive management in this, like every other wildlife management issue is recognized and advocated for use in assessing lynx habitat and population trends in RMNP.

# CHAPTER 4

# POPULATION VIABILITY ANALYSIS BASED ON HABITAT POTENTIAL FOR LYNX (*FELIS LYNX*) IN RIDING MOUNTAIN NATIONAL PARK

## INTRODUCTION

Parks Canada policy (Department of Canadian Heritage 1994) developed guiding principles to direct management in Canada's national parks and defined ecological integrity as a primary priority. By protecting or restoring ecological integrity, Parks Canada is able to fulfil its national and international responsibilities of heritage conservation and protection. The document states that management decisions must be based upon knowledge gained from scientific research and monitoring. In response to the policy, land managers at RMNP developed an Ecosystem Conservation Plan (ECP) (Ecosystem Conservation Plan Team 1997) to "protect, restore, and monitor...natural...heritage within the Park." Parks Canada stated that a component of ecological integrity is "managing ecosystems in such a way that ecological processes are maintained and genetic, species, and ecosystem diversity are assured for the future" (Ecosystem Conservation Plan Team 1997). The ECP outlined long-term ecosystem management goals which partially included retaining the ecological integrity so that RMNP can "accommodate normal fluctuations of natural phenomena...(and) healthy regional populations of...animals are sustained and normal interactions between populations are maintained (and) functional connections between habitat in the regions are conserved or restored." The ECP also notes the importance of maintaining biodiversity in the RMNP region and monitoring key species that may be indicators of the ecological integrity of the region. Examining the population viability of lynx conforms to these goals.

Population viability examines the question, "how much is enough?" (Soule 1987).

Population viability is described as a state that maintains vigour and potential for evolutionary adaptation (Soule 1987). It is a question that intrigues resource managers, is incredibly difficult to answer, and is essential for conservation. Determining population viability for any species is an arduous task in that it requires predicting organism persistence based on biotic and abiotic factors in a spatial and temporal regime (Soule 1987). Assessment of habitat is a vital component of population viability analysis. In the past, viability assessments typically emphasized genetics and demographics. Only recently has habitat been incorporated into these models.

A framework for examining population viability based upon habitat potential was developed by Roloff and Haufler (1997) and was used for indexing the viability of lynx in RMNP. Previously, the only quantitative means of linking habitat requirements to lynx fitness was through studies that associated lynx demographics to snowshoe hare abundance (Koehler et al. 1979, Parker et al. 1983, Koehler 1990) . Similarly, no data existed on the source and sink population dynamics of lynx and thus, there was no means to quantitatively frame a research hypothesis regarding lynx population viability for RMNP. The lack of information on lynx population demographics and movements necessitated a habitat-based approach for addressing population viability for RMNP. The intent of the lynx HSI model (Roloff 1998) and the population viability assessment method proposed by Roloff and Haufler (1997) offered these tools.

Studies have demonstrated habitat relationships for lynx (Koehler et al. 1979, Koehler 1990, Murray et al. 1994, Mowat et al. 1996, Poole et al. 1996), however, quantitative methods to map and assess habitat conditions have not existed until recently. To that end, a lynx HSI model was developed by Roloff (1998). Roloff's (1998) model was developed with the intention of providing resource planners and managers with a tool that indexed relative fitness of lynx based on habitat. Verification of Roloff's (1998) model was presented and discussed in Chapter 3 of this document. Verification tests of the lynx HSI model indicated that the model was useful for describing lynx distribution and thus is a useful tool for RMNP. Roloff's (1998) lynx HSI model provided the basis for the lynx population viability assessment presented herein.

This study was performed to help managers understand the habitat potential of RMNP for lynx and how that habitat potential may relate to population viability for the species. Lynx are an indigenous species to RMNP, and thus are inherent to RMNP and its greater ecosystem. Without lynx, the Park would be missing a component of ecological integrity. The objective of this study was to index the viability of RMNP's lynx population using the approach of Roloff and Haufler (1997). The result offers a quantifiable, repeatable index of lynx population viability that can be used to guide resource management decisions.

## STUDY AREA

See Chapter 1.

## METHODS

### Lynx home ranges in RMNP based upon habitat potential

Using Roloff's (1998) HSI model, lynx habitat potential (i.e., the quantity, quality, and spatial distribution of habitat) was identified and mapped for RMNP (Chapter 3 of this document). The HSI modelling process generated the data required for applying Roloff and Haufler's (1997) viability framework. Roloff and Haufler's (1997) approach requires a home-range-level representation of habitat potential for the planning landscape (termed a habitat contour map). The habitat contour map consists of grid cells, where each cell represents the center of a lynx home range (Roloff and Haufler 1997). For RMNP, each grid cell (30 X 30 meters) was assigned a home-range-level habitat potential value from the HSI model based on lynx home range components (i.e., foraging, denning, and travel requirements; Roloff 1998) (Figure 3.6)

An initial step in using Roloff and Haufler's (1997) viability approach was to determine the criteria for evaluating the habitat contour map (Roloff and Haufler 1997). This process consisted of two components (Roloff and Haufler 1997). First, the viability and marginal habitat quality thresholds were identified. These thresholds corresponded to home-range-level HSI scores. Home ranges consisting of habitats above the viability threshold were assumed to be of sufficient quality to consistently contribute young to the population, even during lean resource years (Roloff and Haufler 1997). Home ranges that contained habitats between the marginal and viable habitat quality thresholds were assumed to sporadically contribute young to population viability in response to resource fluctuations (Roloff and Haufler 1997). The viable and marginal HSI thresholds are best inferred from studies that relate fitness indicators, such as survival rate, fecundity, litter size, and pregnancy rates, to habitat quality. Roloff and Haufler (1997) examined studies that documented the relationship between prey abundance (as a surrogate for habitat quality), home range size, and the above mentioned fitness indicators for lynx. Based on a subjective evaluation of lynx fitness indicators to habitat quality, Roloff and Haufler (1997) set the lynx habitat viability threshold at 0.70 HSI for their example analysis. The underlying assumption to the viability analysis is that home ranges above this threshold have a long-term growth rate >1.0. For lack of any better estimates, this value was used as the home range viability threshold for this study. The marginal habitat quality threshold of 0.25 used in Roloff and Haufler's (1997) example was also used for this study. Thus, it was assumed that home ranges <0.25 HSI did not contribute to long-term population growth regardless of resource condition (Roloff and Haufler 1997).

The second component of Roloff and Haufler's (1997) viability framework consisted of estimating the amounts of habitat required for functional home ranges. One required estimate is the smallest possible area that lynx will use if habitat is optimum. The assumption is made that if the habitat model is properly calibrated, optimum HSI scores correspond to conditions in which no resources are limiting and the home range used by the organism is smallest (Roloff and Haufler 1997). As habitat quality decreases from optimum, lynx use larger areas and vice versa. Roloff and Haufler's (1997) approach for calculating the allometric home-range-size for lynx was used to establish the minimum habitat amount threshold. The allometric home range was calculated using Harestad and Bunnell's (1979) equation for carnivores. Using this equation, Roloff and Haufler (1997) calculated a minimum lynx home range as 250 ha based on the average weight of female lynx (8.5 kg as presented in Quinn and Parker 1987). A study conducted in Manitoba from 1972 to 1975 (Koonz 1976) found the average length of female lynx associated with this region to be 91 cm (n=42), which is longer than the 82 cm reported in Quinn and Parker (1987). Carbyn and Patriquin (1983) also reported that two females radio-collared in RMNP were 13.6 and 10.9 kg respectively, again, larger than the 8.5 kg average reported by Quinn and Parker (1987). Thus, Harestad and Bunnell's (1979) allometric equation for carnivores was recalculated for RMNP using a weight of 12.25 kg (average weight of Carbyn and Patriquin's (1983) female lynx). This process resulted in an allometric home range of 399 ha for RMNP. It was assumed that areas smaller than this would be insufficient to meet the spatial requisites of lynx regardless of the habitat quality (Roloff and Haufler 1997).

Utilizing the quality and quantity thresholds, a habitat unit objective of 399 habitat units was used as a criterion for evaluating the habitat contour map (Roloff and Haufler 1997). A habitat unit is defined as the product of HSI score and the corresponding area (U.S. Fish and Wildlife Service 1981). Grid cells (900 $m^2$) in the habitat contour map were aggregated, from highest to lowest quality sequentially, until the habitat unit objective was satisfied. Home ranges that met the habitat unit objective and had an average HSI score above 0.70 were deemed viable. Home ranges that met the habitat unit objective and had an average HSI score between 0.25 and 0.70 were marginal. The remaining habitat constituted non-viable home ranges.

Once the thresholds and habitat unit objective were established, the habitat contour map for RMNP was evaluated. Following Roloff and Haufler's (1997) framework, the number and functionality of lynx home ranges in the Park was estimated and mapped. In this analysis, the assumption was made that lynx home ranges do not overlap, so the viability estimate is conservative because studies (Saunders 1963, Brand and Keith 1979, Mech 1980, Carbyn and Patriquin 1983, Ward and Krebs 1985, Koehler 1990) have demonstrated that lynx home ranges often overlap.

## Viability Analysis

Assessing spatial viability is an important component to determine the contribution of a given habitat toward the long-term success of a population. An examination of viable home ranges depicted by Roloff and Haufler's (1997) framework on RMNP to an index of lynx productivity was performed. During the winter of 1997-1998, data on lynx abundance was collected as part of a tracking survey conducted in RMNP. These data were collected in part to verify Roloff's (1998) lynx HSI model. A full description of the tracking methodology can be found in Chapter 3 of this document. Information gathered for this study included location of lynx tracks, habitat class, as well as the number of lynx found at each track sighting. The lynx home range map was overlaid with the tracking results, noting individual and groups of animals. The data gathered from this study does not provide enough information to be able to prove that groups of tracks were those of females with kittens although anecdotal evidence suggests that this is typically the case, particularly in mid-winter. Quinn and Parker (1987) stated that lynx may not be as anti-social as once thought. Two groups of lynx monitored by Carbyn and Patriquin (1983) were females and their kittens. Carbyn and Patriquin (1983) also reported seeing two females travelling together with their kittens. Parker et al. (1983) reported that on Cape Breton Island a group of three or four lynx travelling together were a female with kittens although another adult lynx would occasionally join the family group for a short period of time. They also stated that two yearlings may travel together, or in late winter, two sets of tracks may be a male with a female (Parker et al. 1983). Koehler and Aubry (1994) summarized research and found that the lynx breeding season occurs from late March to early April at both northern and southern latitudes. Quinn and Parker (1987) stated that family groups break up at the onset of the short breeding season (mid-March and early April). Quinn and Parker (1987) also noted that adult lynx of the same sex seem hostile toward one another and thus keep exclusive home ranges. Hence, since the tracking period was prior to the breeding season, and based on anecdotal evidence from the literature, it is likely that groups of tracks in this study contained at least some kittens. Thus, it can be assumed that groups of lynx should be associated with viable home ranges.

To examine whether groups of lynx were more closely associated with viable home ranges than solitary lynx, the average distance between lynx locations (groups and solitary animals) and viable home ranges was calculated using GIS. If lynx locations fell within a viable area, the distance was 0. For lynx locations external to viable habitat, the nearest straight-line distance to viable habitat was recorded. A Mann-Whitney U-test (Bailey 1995) was used to determine if there was a significant difference in distance to viable home ranges for group and single tracks. The null hypothesis was that the group lynx tracks were the same distance from viable home ranges as single lynx tracks (i.e., the grouped lynx tracks and single lynx tracks have identical distributions or an identical mean distance to viable home range).

# RESULTS

## Habitat Potential

Lynx home-range-level habitat potential for RMNP was assessed in Chapter 3. The spatial portrayal of lynx home-range-level habitat by Roloff's (1998) HSI model (Figure 3.6) resulted in 239,873 ha of RMNP receiving an HSI score (see Chapter 3) (Table 3.1) The output of the HSI model provided the input for the habitat-based population viability analysis in the form of a lynx home-range-level HSI map (habitat contour map) (Figure 3.6).

## Viability Analysis

Roloff and Haufler's (1997) home-range-level viability criteria (0.70 HSI = viability threshold and 0.25 HSI = marginal threshold) and the habitat contour map were used to determine the functionality of and to delineate lynx home ranges. The analysis indicated that RMNP contains the habitat potential for 427 lynx home ranges (Figure 4.1). Of those home ranges, approximately 170 were viable, 255 were marginal, and 2 were not viable (Table 4.1) (Figure 4.1). This approximation may be slightly under-estimated because the mapping technique used resulted in some unclassified areas (see Chapter 3) that were not included in the viability analysis. Some of this unclassified area fell within a regenerating jackpine stand which is considered prime lynx habitat according to site experts.

All lynx track locations were within viable or marginal home range delineations. The majority of the Park (269,958 ha) was classified as viable or marginal (Figure 4.1).

The Mann-Whitney U-test (Bailey 1995) indicated that there was a significance difference in the mean distance to viable home ranges between groups of lynx and single lynx ($U = 717$, $n$ (group of lynx) = 28 and $n$ (single lynx) = 79, $p = 0.005$). The mean

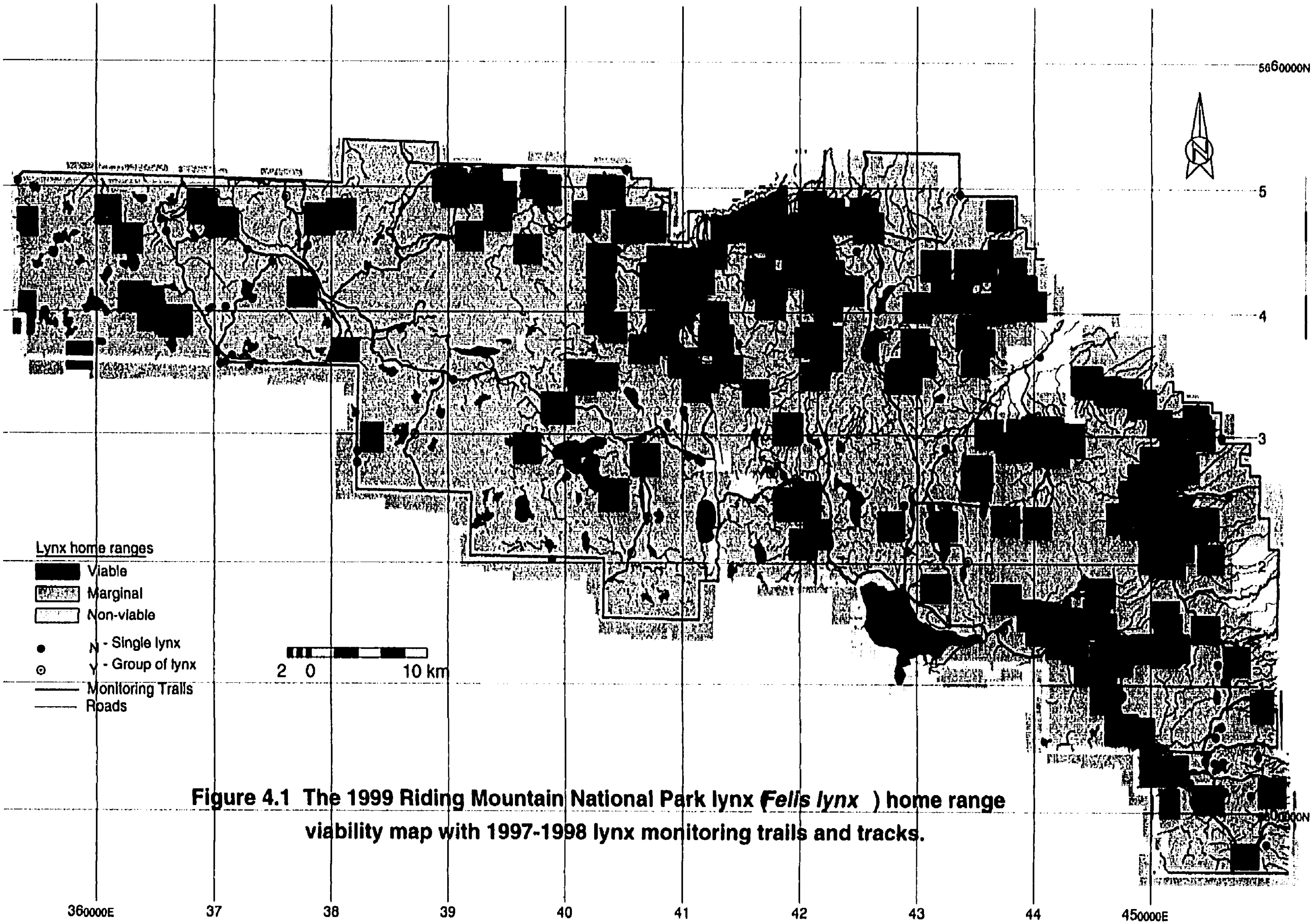



**Figure 4.1** The 1999 Riding Mountain National Park lynx (*Felis lynx*) home range viability map with 1997-1998 lynx monitoring trails and tracks.

**Table 4.1. The functionality of lynx home ranges in Riding Mountain National Park based upon habitat potential.**

| HOME RANGE FUNCTIONALITY | NUMBER OF HOME RANGES | AREA COVERED (ha) |
|---|---|---|
| Viable (HSI $\geq$ 0.07) | 170 | 87,299 |
| Marginal (HSI $\geq$ 0.25 < 0.07) | 255 | 182,659 |
| Non-viable (HSI > 0.25) | 2 | 5,382 |

distance of groups of lynx to viable home ranges was 522 meters. The mean distance of single lynx to viable home ranges was 1200 meters. The 95% confidence limit for groups of lynx was 177 to 868 meters. The 95% confidence limit for single lynx was 879 to 1520 meters. The null hypothesis that groups of lynx and single lynx would occur at similar distances from viable home ranges was rejected. Groups of lynx were found significantly closer to viable home ranges than single lynx.

# DISCUSSION

Lynx home range viability in RMNP was examined using Roloff and Haufler's (1997) population viability modelling framework. This framework appeared to be a useful tool to identify viable, marginal, and non-viable lynx home ranges within RMNP. As such, RMNP managers can use the framework to examine the past, present, and future viability of lynx for the Park.

Relative to RMNP's objectives regarding population viability management, Parks Canada Policy (Department of Canadian Heritage 1994) and management guidelines specific to RMNP (Riding Mountain National Park Round Table 1996, Ecosystem Conservation Plan Team 1997) provide a framework for "responsible management decisions" and the promotion of "the protection of ecosystems and natural habitat, (and) the maintenance and recovery of viable wild populations of species" (Department of Canadian Heritage 1994). Although the land management plan developed for RMNP calls for the sustenance of the "key species and processes", the plan currently excludes lynx. The reason for this omission is unclear, but may be due to a lack of information about lynx in RMNP. The lynx's historical range, which includes RMNP, is generally intact (Quinn and Parker 1987) and thus, lynx are part of the natural biota. The relationship between lynx numbers and the snowshoe hare population cycle has been documented for decades, particularly in the boreal regions of the lynx's range. Although the relationship between these two species is not completely understood, it is part of the natural processes that occur within the RMNP region and thus should be considered essential to the ecological integrity of the region. Riding Mountain National Park is a representative of the southern boreal plains and plateaux region of Canada. It is the mission of RMNP to conserve this region and encourage naturally evolving ecosystems (Riding Mountain National Park Round Table 1996). The ECP also emphasizes the population dynamics of indicator species. Thus, identifying species that are indicators of overall ecological integrity and setting up monitoring plans for those species is a Park priority. With these goals in mind, the viability of lynx in RMNP was examined.

Roloff and Haufler's (1997) framework helped estimate lynx viability for RMNP. Roloff and Haufler (1997) described a home range as the area needed to meet the life requisites of lynx. Due to the diversity of home range sizes reported in the literature (summarized in Koehler and Aubry 1994) as well as the constantly changing home range size due to habitat dynamics and fluctuations in prey, Harestad and Bunnell's (1979) allometric home range equation was used to estimate a minimum home-range-sized area for RMNP lynx (399 ha). The allometric home range was used in the analysis to "benchmark" what optimum habitat conditions mean to lynx. It provides the capability to develop habitat unit objectives because it "bounds" the system, i.e., it provides a link between optimum habitat and area requirements. This minimum area needed to provide a viable lynx home range, combined with the HSI analysis (Chapter 3 and Roloff 1998) using the framework described in Roloff and Haufler (1997) resulted in the spatial delineation of home ranges for RMNP. The viability analysis suggested that based on the Park's present habitat, RMNP is capable of supporting 427 lynx home ranges. An implicit assumption in the model (Roloff and Haufler 1997) is that lynx home ranges do not overlap. This assumption is refuted in the literature (Saunders 1963, Brand and Keith 1979, Mech 1980, Carbyn and Patriquin 1983, Ward and Krebs 1985, Koehler 1990) and therefore it is recognized that this analysis may have underestimated the number of home ranges. The amount and degree of spatial overlap of lynx home ranges varies geographically, temporally, and by the home range estimation technique (Saunders 1963, Brand and Keith 1979, Mech 1980, Carbyn and Patriquin 1983, Ward and Krebs 1985, Koehler 1990). In RMNP, Carbyn and Patriquin (1983) found inconclusive evidence that home ranges overlapped.

It is important to note that the framework (Roloff and Haufler 1997) discussed herein is habitat-based only, and not sensitive to hare fluctuations or density-dependent effects. This study indexed habitat potential as related to lynx viability. It did not establish lynx density and thus it is recommended that telemetry or another density estimation technique be used to estimate the actual number of lynx that the Park contains.

Setting population goals for lynx in RMNP in the absence of empirical data on productivity or fitness is an arduous task. Since habitat is a major component of the survival of any species, habitat-based analysis can be used to postulate the status of the lynx population into the future. For management purposes, it is useful to benchmark the Park's existing conditions against past and future habitat conditions and lynx numbers. Roloff's (1998) lynx HSI modelling framework performed on RMNP demonstrated that lynx foraging habitat was the most limiting habitat component (Chapter 3). Many anthropogenic factors have contributed to this situation. An examination of these elements will permit RMNP managers to understand the historical impacts on lynx habitat and ultimately guide decisions about future lynx viability.

Lynx are an indigenous species of RMNP, with trapping records for Canada dating back to the early 1700's (Elton and Nicholson 1942). Early successional forests are required by lynx for foraging (Koehler and Brittell 1990). Fire, although initially detrimental, will create early successional habitat that is conducive to good hare habitat and thus is an important component of good lynx habitat (Koehler and Brittell 1990). The RMNP area was described as "a disturbance forest, usually maintained in youth and health by frequent fire" (Rowe 1961). Whitaker et al. (1994) stated that Native people's activities and the fur trade had negligible effects on vegetation development in the RMNP area in comparison to the impacts from European settlement. The onset of European settlement resulted in the tilling of native grassland cover types and a reduction in the frequency of large prairie fires. Also, logging and railways were established, and towns and farms were developed (Whitaker et al. 1994). These direct and indirect effects on lynx habitat undoubtedly had a sudden, negative influence on the spatial structure and viability of lynx home ranges within RMNP.

During post-European settlement prior to 1895, poorly administered logging and deliberately set fires shaped the ecology of the RMNP area (Sentar Consultants Limited 1992). Most small fire locations were never recorded, however, in 1890 two large fires burned over 70% of the western portion of RMNP. Also, Tunstell (1940) stated that fires were especially severe between 1885 and 1889 during which several thousand acres burned. The creation of the Riding Mountain Forest Reserve in 1895 was done to protect the area but logging and the burning of brush piles were still permitted. It was during this time that many settlers arrived in the area. In the early 1900's, the region's white spruce was reported as scarce. The area was being replaced by poplar, which thrived in the semi-prairie conditions that resulted from extensive fires and logging. Reports from the 1908 Department of the Interior (reported in Sentar Consultants Limited 1992) stated that, with few exceptions, this area had little "mature timber" left, and was covered with 5-20 year old stands of aspen and some jackpine. The fires reportedly eliminated much of the white spruce in what is now the western portion of the Park. In 1895, approximately 13,000 ha were burned in the north-east portion of the Park, and in that same year, 1,800 ha were burned in the south-eastern part of what is now RMNP. The level of fire activity varied during the early 1900's. For example, the Department of the Interior reports (reported in Sentar Consultants Limited 1992) 42 fires in the Riding Mountain area in 1912 and almost no fires in 1914. Lynx were reported as being extirpated from the Park between 1910 and 1930, although it is known that lynx have persisted in RMNP since the first decade after the Park's official opening (Park Resource Management Team 1979) in 1933. By the 1920's, there was a well-developed warden service and fire roads and look-out towers and thus, the occurrence of fire was greatly reduced. Riding Mountain National Park was formally designated as a national park in 1930 and officially opened in 1933 (Ecosystem Conservation Plan Team 1997). By 1935, most timber harvesting was abolished from RMNP, however, logging activity was still permitted in fire-killed stands and in other areas where tree thinning was considered beneficial to the health of the forest (Tabulenas 1983). In the mid-1960's, timber harvesting stopped. From 1940-1978, 41,000 ha of area burned as the result of 167 fires, of which 85% were human-caused. The area burned was greatly reduced after 1930, with the exception of major burns in 1940 (Whitewater Lake), 1961 (Gunn Lake), and 1980 (Rolling River) (Caners and Kenkel 1998). Therefore, it is apparent that the fire regime has been altered significantly since European settlement. The change in this regime has altered the successional pattern of RMNP, ultimately changing the type and distribution of habitat available to lynx. These disturbances undoubtedly influenced lynx home range viability and lynx populations, especially since snowshoe hare habitat is typically enhanced by fire and logging (Quinn and Parker 1987, Koehler and Brittell 1990, Koehler and Aubry 1994). Lynx foraging habitat depends on snowshoe hare habitat quality and abundance (Chapter 3) and thus, if snowshoe hare habitat is altered across large areas, the effects will ripple throughout the lynx population and alter viability.

Little information exists on lynx population numbers in the RMNP region before European settlement. Elton and Nicholson (1942) summarized lynx harvest returns collected by the Hudson Bay Fur Company that dated back to the early 1700's. Riding Mountain National Park was part of an area classified as the "Winnipeg Basin." Harvest peaks were much higher during the periods of uncontrolled natural disturbances and prior to European settlement in the RMNP region in the late 1800's. Elton and Nicholson (1942) noted that the smallpox epidemic could account for the reduced harvest levels in the 1880's and 1890's, however, it can be hypothesized that major alterations to lynx and snowshoe hare habitat caused by human activity could have also contributed to fewer lynx numbers. Also noteworthy is that lynx harvests continued to decline for the first three decades of the 1900's, which corresponds to the period of altered habitat disturbance regimes (such as fire). Concurrently, there was an apparent extirpation of lynx in the RMNP area somewhere between 1910 and 1930. Five lynx were reported to have migrated from the Duck Mountains in 1939-40 and wardens reported seeing lynx occasionally in the winter of 1940 (Park Resource Management Team 1979). It was also reported that a "fair" number of lynx were seen in the Rolling River area by 1955, and several tracks were seen on the east escarpment in 1962. The document reported several lynx being seen in the Rolling River and east escarpment areas since 1975. Therefore, it appears that protection provided by the Park has aided in the lynx's recolonization of RMNP. Manitoba's lynx harvest (Figure 1.3) has also been recorded since 1919. These numbers correspond to the approximate 10-year cycle of lynx. The magnitude of harvest peaks, at least until the mid-1980's, correspond to the maximum numbers of lynx taken from the Manitoba region in the late 1800's. It is difficult to determine the reason (end of the fur trade, sociological or economic reasons) for the decline in lynx harvests during the late 1800's, however, it is known that habitat in RMNP has changed significantly over that time due to human intervention. Comparing lynx harvest data with landscape alterations, it can be hypothesized that lynx population viability has been decreased due to human disturbance through habitat alteration. This hypothesis could be tested using Roloff and Haufler's (1997) framework before and after European settlement within the RMNP area. This analysis would require the reconstruction of vegetation maps to the appropriate time periods. It would provide managers with an idea of lynx home range structure within the Park prior to human manipulation which could guide population viability goals within RMNP.

Lynx require a mosaic of landscape characteristics for optimum habitat conditions (Koehler and Aubry 1994). As demonstrated above, the composition of the Park's current landscape was influenced by logging, fire, grazing, and haying (Ecosystem Conservation Plan Team 1997). These factors have disappeared, with the exception of prescribed burning and the rare occurrence of wildfires. As a result, the Park is moving toward an old successional forest which is unsuitable for long-term lynx population persistence. Chapter 3 demonstrated that the foraging component of lynx habitat is the limiting factor in RMNP. Since European settlement in the RMNP area, fire suppression has resulted in the loss of a successional mosaic landscape deemed important to lynx. Further progression toward mono-aged forests, due to a lack of natural disturbances, will ultimately decrease the viability of RMNP's lynx population. As such, it is important to identify where subpopulations of lynx exist, determine the connectivity between these subpopulations, and establish what areas are viable. Provision of these viable areas is important for ensuring that the lynx population will persist during poor hare years. This information is important for making sound management decisions on landscape composition and configuration relative to lynx viability.

Population viability objectives should be set at the landscape level according to Roloff and Haufler (1997). Most populations, including lynx, often consist of spatially discontinuous subpopulations. Subpopulations refer to individuals that are not spatially disjunct from other individuals (Roloff and Haufler 1997). Lynx in RMNP probably constitute a subpopulation because of geographic isolation (Figure 1.2). Viable habitat, although patchy in distribution, is not separated by insurmountable movement barriers within the Park (Figure 3.5). As high quality habitat is lost, the ability for RMNP lynx to interact may change, but at present, this does not seem to pose a problem (Figure 3.5). Another potential subpopulation of lynx likely exists in Duck Mountain Provincial Park, approximately 5 km north of RMNP (Figure 1.2). Although this subpopulation is well within the dispersal distance of lynx, these two areas are separated by expanses of non-habitat (agricultural land with minimal forested area) (Figure 1.2). In addition, only a very small portion of the two parks lie within 5 km of one another. The distance between the two refugia is generally much greater. Thus, overall animal dispersal between the subpopulations is likely limited by landscape structure.

Roloff and Haufler's (1997) framework could also be used to project habitat conditions for lynx into the future thus assisting managers in estimating RMNP's potential for supporting viable lynx populations. Caners and Kenkel (1998) modelled the vegetation dynamics of RMNP. Eight vegetation types were identified, and successional trajectories determined. Caners and Kenkel (1998) portrayed the hypothesized, deterministic changes to the RMNP vegetation over time. Part of the successional changes in the Park were related to fire frequency although the extent was minimized by fire control measures. The extent of wildfire influence was unknown. Fire suppression since the 1920's has greatly reduced the fire frequency and extent in the Park. Bailey (1968) reported that the fire return period was 134 years in 1940 and was 409 years by 1965. The suppressed influence of fire will presumably lead the Park into late-successional stands, potentially threatening habitat for species that utilize new, regenerating forests such as snowshoe hare. It is known that lynx require a mosaic of successional stages for various life requisites (Koehler and Brittell 1990, Washington Department of Wildlife 1993, Koehler and Aubry 1994). As previously discussed, one requisite is associated with young, regenerating forests that support hares (Koehler and Brittell 1990, Koehler and Aubry 1994). Since lynx feed primarily on snowshoe hare, areas that support hare are vital for the persistence of a lynx population in RMNP. In RMNP, it appears that foraging habitat is the limiting factor for lynx. Therefore, the philosophy of managing RMNP outside of its historical disturbance regimes will most likely be detrimental to the persistence of the lynx population.

As a starting point to identify lynx population viability goals for RMNP, it is recognized that information is lacking on lynx productivity and fitness. These data are needed to generate a so-called reliable estimate of a minimum viable population. A persistence model presented in Belovsky (1987) (that ignores habitat condition), depicted the population size and area ($km^2$) needed for a 95% chance of persistence for various sized mammals for 100 and 1000 years respectively. According to this crude model, at just under 3000 $km^2$ RMNP is large enough to ensure the persistence of a lynx-sized animal for 100 years, but not for 1000 years. However, according to the model (Belovsky 1987) the Park does not contain a high enough lynx population to ensure that the species will persist over the next 100 years. Belovsky (1987) stated that this crude model can provide managers with a "rule of thumb" approach when designing and managing refuges. Other estimates to predict viable population sizes range from 50 to 5000 individuals (as summarized by Roloff and Haufler 1997), a range that is too broad for management purposes.

Based on the viability analysis presented herein and in the absence of lynx productivity and fitness data, it was assumed that the RMNP lynx subpopulation is a source area for lynx in Manitoba. This assumption was made because lynx were known to have recolonized the Park in about 1939 and have been present in varying numbers since then (Park Resource Management Team 1979). It is most likely that the long-term persistence of RMNP lynx require interchange with other wilderness regions such as the Duck Mountains to the northwest and the Interlake region to the northeast. This interchange is important in order to maintain genetic diversity and to provide sources of recolonization when detrimental environmental stochasities occur. In addition, lynx are vulnerable to trapping pressure, especially during periods of hare scarcity (Brand and Keith 1979, Parker et al. 1983, Bailey et al. 1986, Quinn and Parker 1987). Historically, trapping pressure has been extensive in areas surrounding the Park and Carbyn and Patriquin (1983) speculated that the Park's lynx population may become extirpated if trapping pressure was high during periods of low hare populations. Conservation efforts, such as provincial trapping moratoriums during lynx population lows may be important to ensure the sustainability of the RMNP lynx population. It is assumed that with these factors in place, RMNP's lynx population could be considered viable. It is therefore recommended that Park managers maintain at least the status quo in terms of lynx numbers, recognizing that the numbers will fluctuate dramatically during population peaks and falls. To maintain the existing population, a long-term monitoring project of both habitat and population numbers is needed. In time, managers will be able to relate habitat potential to the dynamics of the lynx population. If either habitat or demographics become more limiting, then adaptive management will need to be implemented.

The RMNP lynx population should be monitored throughout the entire population cycle. During periods of hare population lows, lynx are most vulnerable to various mortality factors such as disease, starvation, kitten mortality, and human influences such as trapping (Koehler and Aubry 1994). Also during these times, environmental stochasities could have the greatest impact on the lynx population because of the insular nature of the Park making immigration and emigration difficult if not impossible. It is during these times that the areas classified as high quality lynx habitat (0.70 - 1.00 HSI) (Figure 3.6) would be used by lynx. The viable home ranges during these times would be the areas that would produce kittens (or maintain them sufficiently to reach adulthood). Therefore, the monitoring program that has been established (Chapter 5) should focus especially on the population trends of lynx during their population low. If the population appears to be falling lower and lower at the bottom of each cycle, it suggests that some type of additional management must be implemented. This may include using Roloff and Haufler's (1997) framework to see where new viable home ranges could be established, or recommending lower trapping quotas for the surrounding regions, decreasing the trapping season around the perimeter of the Park, or other factors. Monitoring is the first step in providing managers baseline data to use when making these types of decisions.

It is recommended that managers pay close attention to maintaining or restoring viable lynx habitat to assist in maintaining a viable lynx population through low hare and lynx population cycles. The results from applying Roloff and Haufler's (1997) viability analysis approach to RMNP revealed that groups of lynx were associated with viable home ranges significantly more than solitary lynx. This finding strengthens evidence supporting the validity of Roloff and Haufler's (1997) framework. Viable home ranges are presumed to have a population growth rate >1.0 (Roloff and Haufler 1997). Marginal home ranges are presumed to not contribute to population viability during times of resource scarcity, and non-viable populations do not contribute to population viability.

Therefore, it is essential that managers maintain viable home ranges on a "no-net-loss" basis to ensure that the RMNP lynx population remains stable (relative to its natural population cycle). Roloff and Haufler's (1997) model appeared to be a useful tool for estimating viable lynx home ranges, therefore, this model should be used to assist managers in monitoring home range viability based upon habitat potential in RMNP.

# CHAPTER 5

# SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

## SUMMARY AND CONCLUSIONS

Riding Mountain National Park managers are legislated to manage the Park with ecological integrity as their primary goal (National Parks Act 1989). The viability of indigenous species is woven into the core of ecological integrity. Therefore, studying species viability is important to help assess the level of ecological integrity of a region. It also aids in Park management by emphasizing scientific research (a requirement of national park policy) in planning. This study demonstrated and verified a methodology for incorporating lynx into RMNP's ecological assessment.

A lynx habitat assessment methodology (Roloff 1998) was demonstrated and verified for RMNP. The study included gathering site-specific information on the foraging, denning, and interspersion habitat available to lynx in RMNP. A home-range-level output was generated for each of these components. It was estimated that foraging habitat was the limiting factor to lynx in RMNP since less than 1% of the classified area provided optimum foraging whereas approximately 79% provided optimum denning habitat and about 57% provided optimum interspersion habitat. It was also recognized that this limiting factor would become especially detrimental to lynx during poor resource years (i.e., low hare populations), or if early successional forests (i.e., snowshoe hare habitat) are taken out of the Park's landscape.

Foraging, denning, and interspersion habitat are all deemed critical to the long term population viability of lynx, therefore, these components were combined to determine home-range-level habitat suitability for lynx (Roloff 1998). The output was an estimate of the quality and quantity of habitat available to lynx in the form of a habitat suitability map (also termed habitat contour map). The majority of the Park's habitat was classified as moderate HSI value (between 0.30-0.59 HSI).

Roloff's (1998) model was verified by examining lynx track data gathered during the winters of 1997 and 1998. Using a fine habitat resolution (9 HSI classes) there was a significant difference between habitat use versus availability. However, only the lowest quality class (0.00-0.09 HSI) showed a significant difference (i.e., it was used less than expected). The analysis was performed on a coarser resolution (3 grouped habitat classes) and once again, a significant difference was found between habitat use versus availability. At p=0.20, the lowest and highest habitat classes were used differently than expected (the lowest HSI class was used less than expected and the highest HSI class was used more than expected in proportion to availability). Potential bias with the 1997-1998 survey could have influenced these results.

Further assessment of the RMNP landscape was performed by analyzing the lynx population viability of the Park based upon habitat potential. Roloff and Haufler's (1997) framework was utilized to establish the number and functionality of home ranges within Park boundaries. This was accomplished by combining the concept of habitat requirements and allometrics to establish a home-range-level lynx HSI map (Roloff and Haufler 1997). Marginal (0.25 HSI) and viable (0.70 HSI) habitat quality thresholds as suggested by Roloff and Haufler (1997) were used in the analysis. The habitat contour map generated by the HSI model was assessed for a habitat unit objective of 399 (described in Roloff and Haufler 1997) to delineate the Park into non–viable, marginal, and viable habitats. This analysis estimated that the Park contains approximately 170 viable, 255 marginal, and 2 non-viable home ranges. The home ranges that are considered viable (i.e., have an average HSI score of $\geq$0.7) are estimated to have a population growth rate >1.0. Marginal home ranges (HSI between 0.25 and 0.69) are estimated to contribute to the lynx population sporadically, depending on resource availability, and non-viable home ranges (<0.25 HSI) are estimated not to contribute to lynx population viability (Roloff and Haufler 1997). The Mann-Whitney U-test found groups of lynx tracks were significantly closer to viable home ranges than solitary lynx tracks. Therefore, Roloff and Haufler's (1997) framework appeared to reasonably reflect viable, marginal, and non-viable home ranges.

Historical disturbances to the Park that influenced vegetation structure and composition included logging, grazing, and changes to the fire regime. Anthropogenic disturbances undoubtedly had an influence on lynx habitat and populations. It is likely that the spatial arrangement and number of viable, marginal, and non-viable lynx home ranges were affected by these vegetation changes to the landscape. This seems to be evident in trapping records where a large decline in the number of pelts occurred concurrently with the large influx of European settlers into the present-day RMNP region. However, other factors, such as decreased trapping effort, could have added to this drop.

Managers can use Roloff and Haufler's (1997) framework to examine the effects of their management decisions on the viability of lynx home ranges in RMNP. This analysis could be enhanced by projecting the successional changes that the landscape will undergo and subsequently applying the framework to the projected GIS maps. Therefore, Roloff and Haufler's (1997) framework is useful for RMNP managers to help predict present and future lynx population trends.

Habitat potential modelling is a useful tool for land managers. Such analyses can help establish planning objectives based on the best available information without requiring extensive and expensive demographic data collection. This is not to imply that demographic data are not required or preferred to develop or verify habitat models nor that models will be entirely accurate, but in cases where baseline demographic data are scant or non-existent, habitat modelling can certainly provide a cursory step to develop and implement management strategies. This project was undertaken to assist RMNP managers in assessing habitat potential for lynx in the Park and to provide a planning approach that addresses the long-term survival of one of its indigenous species.

# RECOMMENDATIONS

Management of the lynx population in RMNP must focus on factors both internal and external to the Park boundaries. Habitat is a key factor to a species' survival and a factor that Park managers can control. Such management can include habitat enhancement, restoration, and monitoring. These factors are essential since without suitable habitat, no amount of management in any other regime will permit lynx to continue to exist in RMNP. Other factors are also important to the whole concept of species protection. For RMNP lynx, external factors play critical roles in lynx conservation. For example, trapping quotas and seasons around the perimeter of the Park should correspond to the lynx's natural population cycles (Quinn and Parker 1987). Another crucial aspect is the involvement of multiple stakeholders since wildlife considerations extend beyond political boundaries of the Park. Lynx, like all species, require genetic diversity to maintain a healthy and viable population. Travel linkages to areas outside the Park may be important to provide access to external resources but also to provide new population sources in the event of environmental stochasities within RMNP.

Teamwork is vital to the management of any large-scale species. RMNP would benefit from participating in open dialogue with various stakeholders about the lynx research occurring in RMNP and asking for their input. Interested parties may include Manitoba Department of Natural Resources, Duck Mountain Provincial Park, trappers, landowners, First Nations groups, various environmental groups, surrounding school groups, the Riding Mountain Biosphere Reserve, universities, and other lynx researchers. Working with these individuals and groups would create better understanding of RMNP lynx population trends, habitat requirements and usage, and benefits of a persistent lynx population.

Monitoring the lynx population within RMNP will help establish comparative data for land managers, both within and outside Park boundaries. It will establish whether lynx populations in RMNP are increasing or decreasing in synchrony with the predictable 10-year cycle (Elton and Nicholson 1942, Nellis et al. 1972, Brand et al. 1976, Brand and Keith 1979). If the cycling is not occurring as predicted, managers can make decisions regarding the aberration. These decisions may involve further study, habitat alteration, influencing the trapping regulations, or other manipulations. Another advantage is that by annual monitoring, managers can determine if the lynx population peaks remain at a constant level. If, for example, the magnitude of population peaks is declining, it is reasonable cause for further investigation into habitat availability, disease, over-trapping, human presence, or other stresses that may affect lynx.

## SPECIFIC RECOMMENDATIONS

### 1) Monitor lynx population trends and habitat usage.

There are two goals of a lynx monitoring program in RMNP. First is to index lynx numbers. The second is to index the status of lynx habitat in RMNP. The latter is to assess whether overall lynx habitat is increasing or decreasing, and to estimate how the spatial arrangement of habitat is changing. This step is described in Recommendation 2.

Monitoring lynx population trends is the most tangible measure of the health of the RMNP lynx population. Like most wildlife species, it is impossible to determine a complete count for a specific population (Quinn and Parker 1987). Thus, it is recommended that an annual snow-tracking monitoring program be continued. The framework for the annual monitoring has already been established by this project. A similar method, with slight modifications, is recommended for future monitoring (Appendix C).

A telemetry study should be initiated to more fully document lynx habitat use and fitness. This type of study could provide lynx demographic data, relative to habitat class usage specific to RMNP. The study could also examine how lynx are using the habitat classes as classified by Roloff's (1998) lynx HSI model. For example, are areas classified as prime denning areas or viable home ranges actually being used for denning? The same question could be applied to the foraging and interspersion habitats.

### 2) Monitor composition and structure of the ecological land units and use the information to develop updated lynx HSI maps.

Standardized and regular monitoring of the ecological land units classified in this study should be performed. Two types of monitoring would be required. First, the vegetation map for the Park would need to be periodically updated, and second, additional vegetation data should be collected to strengthen the vegetation database used in the habitat analysis.

It is recommended that the Park repeat the satellite image process to map existing vegetation within ten years and at regular ten-year intervals from that period on. In addition, ground-truthing of these images must be performed to monitor changes in species composition, horizontal cover, vertical cover, stem diameter, basal area, canopy cover, tree height, and tree density. It is suggested that vegetation monitoring be performed at permanently located plots (a minimum of 3 plots per ecological land unit) that are re-visited at 5-year intervals. These type of data would provide managers with data to predict the successional sequence of ecological land units.

Roloff's (1998) lynx HSI modelling framework should be applied to the data generated from the updated vegetation maps. The quality and quantity of hare and lynx habitat could then be temporally assessed thus facilitating appropriate management actions.

## 3) Use fire as a management tool in RMNP to help restore successional diversity within the Park.

This study emphasized the importance of a mosaic of successional stages required by lynx. In RMNP, lynx forage habitat was estimated to be the limiting factor on lynx habitat quality. Less than 1% of the Park was classified as optimum forage habitat. These are the areas that will be especially critical to lynx during the snowshoe hare population decline since that is where hares will most likely thrive during these years.

Burned (Fox 1978, Bailey et al. 1986, Quinn and Parker 1987, Koehler 1990) or logged (Parker et al. 1983, Keith 1990, Koehler and Aubry 1994) areas can provide good hare habitat. In RMNP, this observation was made during this study in the 1980 Rolling River Fire burn area, where hare sign was abundant (personal observation). Fire provided a mosaic of successional stages and cover types in the landscape which offered optimum lynx habitat. Fire exclusion in RMNP has permitted vegetation in the Park to "develop a disproportionately large area of old forest stands" (Ecosystem Conservation Plan Team 1997). Thus, burns could help restore a vegetation mosaic to the Park. Logging is not a management option in RMNP as dictated by legislation. Therefore, fire is the only feasible option that can be implemented. It is recommended that the current prescribed burning program continue and potentially be expanded. Presently, in RMNP, prescribe fires have only been allowed to burn in grasslands containing few trees. Permitting wooded areas to burn is also recommended. Fires that burn in a wooded area would provide new growth areas for foraging, intermingled with cover types suitable for denning and interspersion due to the patchy nature of burns. The Rolling River fire was an uncontrolled fire that burned through a wooded area of the Park. The best habitat for lynx in RMNP, according to the HSI model, occurred in that burned area because of the high quality foraging, denning, and interspersion values that was left behind by the fire. Permitting wildfires to burn would also be advantageous to hare and lynx habitat, however, it is understood that the political nature of such events submerges any such possibility.

The details of the types of prescribed burns is beyond the scope of this study, however, it is recommended that the burns be scattered throughout the RMNP's landscape leaving a patchwork of ideal hare and lynx habitat throughout the area. This would help to spread out suitable habitat, potentially enhancing the lynx population viability of RMNP.

## 4) Examine the historical habitat availability of lynx in the area that is currently RMNP.

Digitize the historical vegetation structure of the RMNP area at various time intervals, pre-European settlement. Construct maps dating as far back as the early 1700's so that lynx harvest data gathered from the Hudson's Bay Company could be compared to habitat availability within the area of RMNP. Apply Roloff's (1998) lynx HSI framework and Roloff and Haufler's (1997) population viability framework to examine the historical lynx habitat quality and quantity, and home range viability available within the RMNP area. This information could provide baseline information about the state of ecological integrity of RMNP area during that time. Managers could use the information as a comparative tool against which to measure the current state of the Park's ecological integrity. In other words, the quality and quantity of hare and lynx habitat could be examined prior and post European settlement. Lynx population numbers and trends could then be compared to the habitat availability. This information would provide one measure of the health of RMNP's current state of ecological integrity.

The information could also be used by managers to develop a goal for the quality and quantity of lynx habitat as well as the number of viable and marginal lynx home ranges that should exist in the Park for the long-term survival of a lynx population in the area.

## 5) Digitize projected successional changes and apply Roloff and Haufler's (1997) population viability model.

Succession will inevitably occur. The projected successional patterns for RMNP are presented by Caners and Kenkel (1998). As succession changes the landscape, lynx habitat will also change. For example, areas that were once classified as prime lynx foraging habitat will progress into varying quality and quantity levels of interspersion and denning habitat. As the landscape continues to change, the spatial arrangement and functionality of lynx home ranges will also change. As outlined in Chapter 4, it is recommended that the projected successional changes in RMNP be digitized to help assess what type of habitat will be available, not only to lynx, but to all species of RMNP. Frameworks like Roloff and Haufler's (1997) habitat potential model could then be used to estimate population trends of various species. Such analysis could help managers look into the future and plan management strategies, such as prescribed burns, that would best lead them toward the goal of ecological integrity.

## 6) Examine lynx habitat usage of the forested between RMNP and other wildlife refugia, such as Duck Mountain Provincial Park and the Interlake region.

Currently, it is unknown the extent that lynx use the travel corridors that exist outside RMNP's boundaries. Maintaining and perhaps restoring travel corridors between RMNP and Duck Mountain Provincial Park, as well as between RMNP and the Interlake region is very important in the preservation of many species, not just lynx. A special consideration for lynx is that they tend to avoid open areas >100 meters in width (Koehler 1990). This trend appears warranted in that RMNP lynx, based on subjective observation during our tracking surveys, tended to travel around large frozen lakes or open spaces. The components of suitable travel habitat can be found in Chapter 3. Population genetics data show that isolated populations with few individuals will eventually suffer from genetic depletion (Lande and Barrowclough 1987). Therefore, maintaining genetic interchange is essential for a healthy and viable population in the long run. Evidence suggests that the Park is becoming increasingly isolated from the surrounding landscape. Satellite photography of RMNP (Figure 1.2) shows the island geography of the Park. The reduction in vegetation areas between RMNP and the Duck Mountains is displayed in Figure 5.1 which shows diminishing travel corridors for wildlife between the two parks (Walker and Kenkel 1997). Thus, working with other agencies to progress toward sustaining and re-creating those corridors will aid in the distribution of genetic material

## 7) Initiate a study to examine the genetic health of RMNP's lynx population.

Further studies that focus on genetic variability should be conducted on lynx in RMNP. Such measures would provide information that could be used in making management decisions. For example, if the lynx monitoring program developed by this study revealed aberrations in the lynx population, habitat-based reasons could be examined because of the habitat-based study. Genetic information would allow managers to examine other reasons for the unexpected population changes. In addition, genetic information would be useful to develop a minimum viable popualtion estimate for RMNP. Therefore, an organized genetic testing study should be undertaken to ensure that the RMNP lynx population exhibits genetic diversity. One possible method would be to gather genetic material from lynx trapped along the border of RMNP. Another suitable method would be to use lynx hair snagging stations. Either of these methods would provide managers with useful data. Genetic data would provide managers with a scientific basis upon which to make important management decisions. Only scientific studies can help managers predict the necessary measures to ensure the survival of lynx, or any other species. The alternative to study is "wait and see", however, the ripple effects of a crippled link in the ecological chain is not an acceptable means of ensuring the ecological integrity of the region.

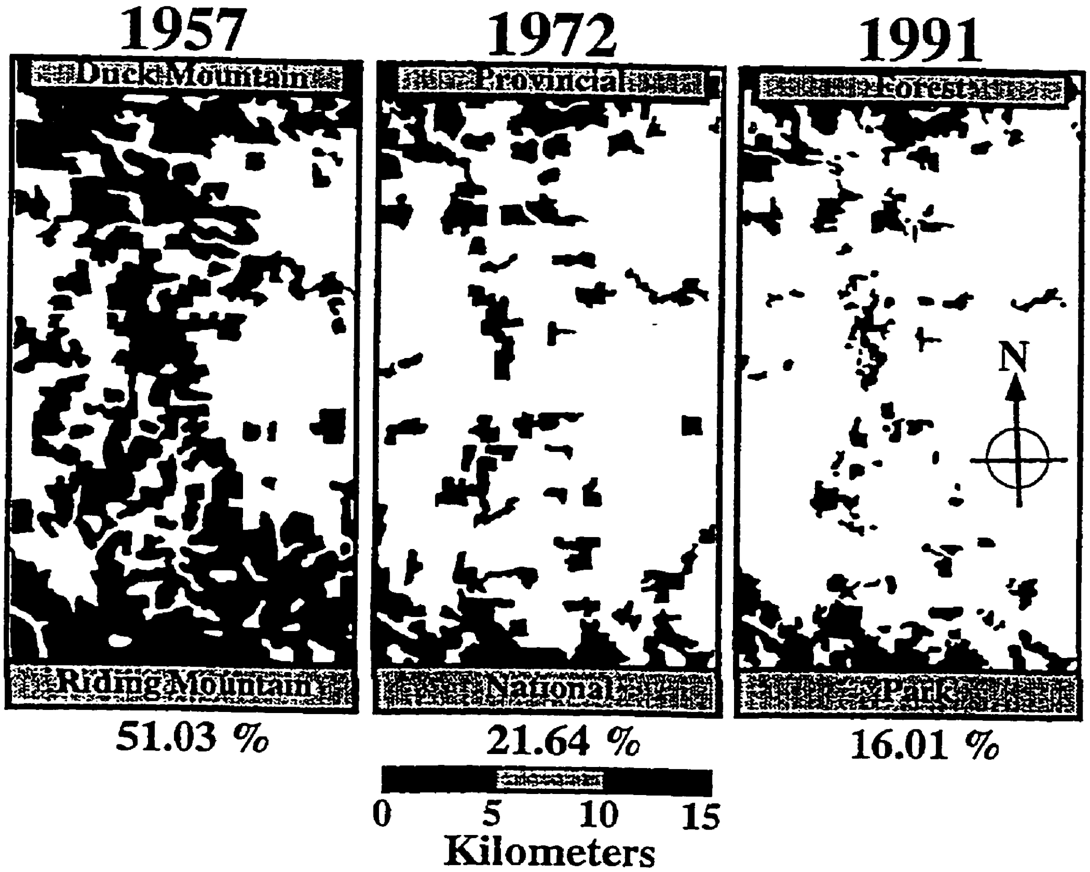



**Figure 5.1** The corridor linking Riding Mountain National Park with Duck Mountain Provincial Park (shaded grey) over three time intervals; 1957, 1972, and 1991. Forested patches are indicated in black and all other cover types (mostly farmland) are white. The percent coverage of the forest at each time interval is indicated (Source: Walker and Kenkel 1997).

**8) Provide copies of the population trend data to the Department of Natural Resources on an annual basis to aid them in setting trapping limits and seasons and participate in those decisions by making recommendations and providing expertise on the RMNP lynx population.**

Although it is clearly stated that killing any wildlife in the Park's boundaries is unlawful (National Parks Act 1989), trapping has been permitted in areas surrounding the Park. Lynx population trend data from RMNP should be shared with provincial authorities since the Park is likely an important lynx population source for the region. Future trapping regulations and seasons should be discussed with various stakeholders, especially the Manitoba Department of Natural Resources. The following are some factors to consider when providing input into these discussions.

Lynx population cycling is intrinsically linked with the snowshoe hare population cycle. Therefore, it is essential that managers take the natural hare cycle into consideration when setting limits for lynx harvesting (Quinn and Parker 1987). Managers must also track the magnitude of hare peaks because hare habitat could be diminished beyond the point of being able to sustain a viable lynx population. Another critical element to consider when establishing trapping seasons and limits is that studies (Brand and Keith 1979, Parker et al. 1983, Bailey et al. 1986) have indicated that lynx population resiliency is diminished when trapping occurs during periods of low lynx density and low population recruitment (Quinn and Parker 1987). Often, it is price that dictates the amount of trapping effort, making "even a single lynx capture...worthwhile for the trapper to continue trapping" (Quinn and Parker 1987). For example, in Manitoba, the average auction value for lynx in 1984-1985 was $700.00 (Manitoba Department of Natural Resources records).

Setting flexible harvest regulations is important for ensuring the sustainable use of lynx surrounding RMNP. This may mean setting a no harvest regulation during certain years, however, this analysis is beyond the scope of this study. Quinn and Parker (1987) suggested that flexible harvest regulations should be based upon a variety of factors including; 1) monitoring indices of trends of total annual lynx harvest; 2) kit numbers in the annual harvest; 3) snowshoe hare abundance trends based upon samples of small game licence returns; 4) questionnaires sent out to hunters and trappers (in the surrounding municipalities); and 5) reports from regional wildlife experts. Using information derived from RMNP's monitoring program, the Park can supply some of this information to Manitoba Natural Resources on an annual basis. The Park currently has data on the population cycles of snowshoe hare in the Park from the study that occurred from the late 1970's through mid-80's. It is suggested that continued monitoring of the hare population, as well as the data from the lynx monitoring would greatly benefit Manitoba Natural Resources in setting trapping regulations for lynx. Also, the habitat assessment tool used in this study can assist managers to index lynx and snowshoe hare abundance.

Setting harvest seasons and quotas requires extensive research. Brand and Keith (1979) stated that: 1) few, if any kits would be found in the trapping harvest during the declining phase of the lynx cycle; 2) that trapping mortality is additive to natural mortality, not compensatory, thus making lynx populations vulnerable to over-trapping during the low phase of their population; and 3) by stopping trapping for the first 3-4 years of the lynx population decline, economic benefits gained by trappers may bring greater returns. Quinn and Parker (1987) stated that lynx are very easy to trap and that populations are capable of becoming over-depleted to the point that historic population peaks are no longer attainable. In addition, season restrictions should be implemented in early winter to avoid trapping females who have dependent kittens since there is no evidence that kittens of that age will survive being orphaned (Slough and Mowat 1996).

## 9) Recommend that the Duck Mountain Provincial Park develop an annual monitoring program to compliment the data from RMNP.

Another source for information about population trends would be from Duck Mountain Provincial Park, which is probably an additional population source for lynx. Therefore, a lynx monitoring program should be implemented in that Park. It is suggested that it follow the same format as the monitoring program in RMNP so that the results would be directly comparable and thus more useful. This information would be useful for setting trapping limits, but also to help assess the two subpopulations. Although it is unknown at this time, it can be reasonably assumed that lynx from the two Parks migrated back and forth when a sufficient travel corridor existed between the Parks. There is no doubt that genetic interchange occurred between these areas prior to European settlement. It can also be assumed that such migration would be beneficial to help maintain a healthy gene pool of the lynx population in south-central Manitoba. Therefore, a comparable database regarding lynx populations in both areas would allow managers to examine population trends on a larger scale and make educated management decisions about lynx.

### 10) Ensure that there is stakeholder involvement when monitoring lynx population trends and participating in provincial harvest management.

Riding Mountain National Park managers alone cannot achieve ecological integrity of the RMNP region. They must work in partnerships as illustrated in this chapter. For instance, in addition to RMNP, provincial park officials from Duck Mountain Provincial Park, Manitoba Natural Resources, landowners, and Louisiana Pacific Forestry Company are examples of the groups needed to help maintain and potentially reconstruct the travel habitat that links RMNP to other wildlife sources. Other groups such as outfitters, Manitoba schools, Riding Mountain Biosphere Reserve, First Nations educational staff, conservation groups, and the Trappers Association need to understand the importance of managing trapping seasons or trapping limits during times of the hare and lynx population decline. Education is the key to cooperation and thus to successful management in any regime. Riding Mountain National Park has the forum to provide such information through interpretative programs or site experts at related meetings. It is obvious that RMNP managers already understand the importance of team-work and stakeholder involvement by the new approach taken with the Management Plan for RMNP (Riding Mountain National Park Round Table 1997).

Involving the public or other interest groups in monitoring the lynx population is another way of obtaining support and interest in Park management. Currently, there is a program in place where people are asked to fill out forms when they see an uncommon species in the Park such as wolf, river otter, wolverine, martin, fisher, or lynx. Records of these lynx sightings should be continued, collected, and kept with the annual lynx tracking report.

Relatively little is known about the lynx in RMNP. The home range size and distribution of 3 radio-collared lynx from Carbyn and Patriquin's (1983) study and historical trapping records for various regions in Manitoba as summarized in Elton and Nicholson (1942) can be examined. General habitat requirements from other studies can also provide information to managers. Lynx habitat availability and associated population viability was estimated for RMNP in this study. However, the number of lynx that is necessary to ensure long-term survival in the area is not known. Therefore, managers must use the best available knowledge to manage for a viable population of lynx in RMNP.

Adaptive management is a fairly new "buzz word" but in fact, the practice has been performed since humans began trying to manage nature. Managing lynx in and around RMNP is no exception. Although it is inevitable that further technology and research will reveal new management plans and techniques, it is recommended that RMNP's lynx habitat be managed by providing measures to help maintain, restore, and enhance it. Monitoring and documenting all information regarding lynx sightings, both during the organized tracking survey and informally by people (staff and others) filling out the lynx observation forms is encouraged. Also, Park officials should gather as much information about the species as possible. This document has provided some baseline data about the lynx of RMNP, as requested by Park managers. Build on this study.

# LITERATURE CITED

Alldredge, J. R., and J. T. Ratti. 1992. Further comparison of some statistical techniques for analysis of resource selection. Journal of Wildlife Management 56:1-9.

Bailey, N. T. J. 1995. Statistical methods in biology. Third edition. Cambridge University Press, Cambridge, United Kingdom. 225 pp.

Bailey, R. H. 1968. Notes on the vegetation in Riding Mountain National Park, Manitoba. Department of Forest and Rural Development, Forest Management Institute, Ottawa.
80 pp.

Bailey, T., E. Bangs, M. Portner, J. Malloy, and R. McAvinchey. 1986. An apparent overexploited lynx population on the Kenai Peninsula, Alaska. Journal of Wildlife Management 56:279-290.

Belovsky, G. E. 1987. Extinction models and mammalian persistence. Pages 35-51 *in* M. E. Soule, editor. 1987. Viable populations for conservation. Cambridge University Press, Cambridge.

Bergerud, A. 1971. The population dynamics of Newfoundland caribou. Wildlife Monographs 25:1-55.

Bergeson, G. 1982. Snowshoe hare study. Government of Canada - Park Service - Riding Mountain National Park. Wasagaming, Manitoba. 41 pp.

Bergeson, G. 1988. Snowshoe hare study - Riding Mountain National Park during winter 1987-1988. Riding Mountain National Park Unpublished Report. Wasagaming, Manitoba. 8 pp. + data sheets.

Berrie, P. M. 1973. Ecology and status of the lynx in interior Alaska. Pages 4-41 *in* R. L. Eaton, editor. The world's cats. Vol. 1. World Wildlife Safari, Winston, Oregon.

Brand, C. J., and L. B. Keith. 1979. Lynx demography during a snowshoe hare decline in Alberta. Journal of Wildlife Management 43:827-849.

Brand, C., L. Keith, and C. Fischer. 1976. Lynx responses to changing snowshoe hare densities in central Alberta. Journal of Wildlife Management 40:416-428.

Brittell, J. D., R. J. Poelker, S. J. Sweeney, G. M. Koehler. 1989. Native cats of Washington. Washington Department of Wildlife Unpublished Report, Olympia. Washington. 169 pp.

Brocke, R. H., K. A. Gustafson, and L. B. Fox. 1992. Restoration of large predators: potential and problems. Pages 303-315 *in*: D.J. Decker, M. E. Krasny, G. R. Goff, C. R. Smith, and D. W. Gross, editors. Challenges in the conservation of biological resources, a practitioner's guide. Westview Press, Boulder, Colorado.

Buchner, A. P., P. Carmichael, G. Dickson, I. Dyck, B. Fardoe, J. Haug, T. Jones, L. Jones, D. Joyes, O. Mallory, M. Mallot, D. Meyer, D. Miller, R. Nash, L. Pettipas, C. T. Shay, E. L. Syms, M. A. Tisdale, J. P Whelan. 1983. Introducing Manitoba prehistory: Papers in Manitoba archaeology popular series, Number 4. Manitoba Department of Cultural Affairs and Historical Resources. Manitoba. 225 ppl.

Burt, W. H., and R. P. Grossenheider. 1980. The Peterson field guide series: A field guide to the mammals: North America north of Mexico. Library of Congress Cataloging in Publication Data, Washington, DC. 289 pp.

Caners, R., and N. Kenkel. 1998. Modelling landscape-level vegetation dynamics in Riding Mountain National Park. Unpublished report prepared for Heritage Canada, Parks. Manitoba. 156 pp.

Carbyn, L. N. and D. Patriquin. 1983. Observations on home range sizes and social organization of Lynx, *Lynx canadensis*, in Riding Mountain National Park, Manitoba. Canadian Field-Naturalist 97:262-267.

Colwill, W. L., and S. M. Jamieson. 1972. Agassiz archaeological research: Report on preliminary survey of Riding Mountain National Park 1971. Unpublished Report for Department of Indian Affairs and Northern Development. Manitoba. 166 pp.

Department of Canadian Heritage. 1994. Guiding principles and operational policies. Ottawa, Ontario. 125 pp.

Dolbeer, R. A., and W. R. Clark. 1975. Population ecology of snowshoe hares in the central Rocky Mountains. Journal of Wildlife Management 39:535-549.

Dunn, O. J., and F. J. Massey Jr. 1965. Estimation of multiple contrasts using t-distributions. The American Statistician 19:573-583.

Ecosystem Conservation Plan Team. 1997. Riding Mountain National Park ecosystem conservation plan. Unpublished report for Parks Canada (Department of Canadian Heritage). Manitoba. 111 pp.

Eisenberg, J. F. 1986. Life history strategies of the felidae: variations on a common theme. Pages 293-303 *in* S. D. Miller and D. D. Everett, editors. Cats of the world: biology, conservation and management. National Wildlife Federation, Washington, DC.

Elton, C., and M. Nicholson. 1942. The ten - year cycle in numbers of the lynx in Canada. Journal of Animal Ecology 11:215-244.

Fox, J. F. 1978. Forest fires and the snowshoe hare - Canada lynx cycle. Oecologia 31: 349-374.

Grange, W. B. 1949. The way to game abundance - with an explanation of game cycles. Charles Scribner's Sons, New York, New York. 365 pp.

Grange, W. B. 1965. Fire and tree growth relationships to snowshoe rabbits. Proceeding of the the Annual Tall Timbers Fire Ecology Conference. 4:111-125.

Green, H. U. 1932. Mammals of the Riding Mountain National Park, Manitoba: a compilation of field notes and observations. Canadian Field - Naturalist 46:149-152.

Hall, E. R., and K. R. Kelson. 1959. The mammals of North America (Vol. 2). The Ronald Press Company, New York, New York. pp. 547-1083.

Harestad, A. S., and F. L. Bunnell. 1979. Home range and body weight - a reevaluation. Ecology 60:389-402.

Haufler, J. B., C. A. Mehl, and G. J. Roloff. 1996. Using a coarse filter approach with species assessment for ecosystem management. Wildlife Society Bulletin 24:200-208.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quarternary Research 3:329-382.

Jamieson, S. M. 1974. A synthesis of archaeological research in Riding Mountain National Park to 1973. Unpublished Report for Department of Indian and Northern Affairs. Manitoba. 241 pp.

Keith, L. B. 1963. Wildlife's ten - year cycle. University of Wisconsin Press. Madison, WI. pp. xvi + 201.

Keith, L. B. 1990. Dynamics of snowshoe hare populations. Pages 119-195 *in*: H. H. Genoways, editor. Current mammalogy. Plenum Press, New York, New York.

Keith, L. B., J. R. Cary, O. J. Rongstad, and M. C. Brittingham. 1984. Demography and ecology of a declining snowshoe hare population. Wildlife Monographs (90):1-43 pp.

Keith, L. B., A. W. Todd, C. J. Brand, R. S. Adamcik, and D. H. Rusch. 1977. An analysis of predation during a cyclic fluctuation of snowshoe hares. Pages 151-175 *in*: T. J. Peterle, editor. Proceedings of XIII International Congress of Game Biologists, Atlanta, Georgia. The Wildlife Society and Wildlife Management Institute, Washington, D. C.

Kleiman, D. F., and J. F. Eisenberg. 1973. Comparisons of canid and felid social systems from an evolutionary perspective. Animal Behaviour 21:637-659.

Koehler, G. M., M. G. Hornocker, and H. S. Hash. 1979. Lynx movements and habitat use in Montana. The Canadian Field Naturalist 93:441-442.

Koehler, G. M. 1990. Population and habitat characteristics of lynx and snowshoe hares in north central Washington. Canadian Journal of Zoology 68:845-851.

Koehler, G. M., and J. D. Brittell. 1990. Managing spruce fir habitat for lynx and snowshoe hares. Journal of Forestry 88:10-14.

Koehler, G. M., and K. B. Aubry. 1994. Lynx. Pages 74-98 *in:* L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski, editors. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. USDA Forest Service General Technical Report RM-254.

Koonz, W. H. 1976. A biological investigation of lynx in Manitoba. Unpublished report for Manitoba Department of Renewable Resources and Transportation Services, Research Branch 76-2. Manitoba. 35 pp.

Lande, R., and G. F. Barrowclough. 1987. Effective population size, genetic variation, and their use in population management. Pages 69-86 *in*: M. E. Soule. editor. Viable populations for conservation. Cambridge University Press, New York, New York.

Lemmon, P. E. 1957. A new instrument for measuring forest overstory density. Journal of Forestry 55(9):667-668.

Leonard, R. D. 1980. The distribution and abundance, food habits, and interspecific relationships of the snowshoe hare during winter 1978-79 in Riding Mountain National Park. Government of Canada - Parks Service - Riding Mountain National Park. Manitoba. 95 pp.

Litvaitis, J., J. Sherburne, and J. Bissonette. 1985. Influence of understory characteristics on snowshoe hare habitat use and density. Journal of Wildlife Management 49: 866-873.

Lombard North Group Limited. 1976. Riding Mountain National Park biophysical land inventory: soil field data. Unpublished contract report for Parks Canada Prairie Report. Manitoba. 312 pp with 16 1:50 K maps.

Manitoba Department of Natural Resources. n.d. Manitoba lynx harvest data (pelt returns) (1919 - 1997), Manitoba.

Manitoba Wildlife Act. 1987. Queen's Printer for the Province of Manitoba. Manitoba. c.W130. 45 pp.

McCord, C. M., and J. E. Cardoza. 1982. Bobcat and lynx. Pages 728-766 *in*: J. A. Chapman and G. A. Feldhamer, editors Wild mammals of North America: biology, management, economics. The John's Hopkins University Press, Baltimore, Maryland.

McGinn, R. A., and P. Rousseau. 1995. The 1995 snowpack survey in the Clear Lake Watershed. Unpublished Report, Riding Mountain National Park document. Manitoba. 21 pp.

McGinn, R. A., and P. Rousseau. 1996. The 1996 snowpack survey in the Clear Lake Watershed. Unpublished Report, Riding Mountain National Park document. Manitoba. 60 pp.

McGinn, R. A., and P. Rousseau. 1997 The 1997 snowpack survey in the Clear Lake Watershed. Unpublished Report, Riding Mountain National Park document. Manitoba. 55 pp.

Mech, L. D. 1973. Canadian lynx invasion of Minnesota. Biological Conservation 5:151-152.

Mech, L. D. 1977. Record movement of a Canadian lynx. Journal of Mammalogy 58:676-677.

Mech, L. D. 1980. Age, sex, reproduction, and spatial organization of lynxes colonizing northeastern Minnesota. Journal of Mammalogy 61:261-267.

Mowat, G., B. G. Slough, and S. Boutin. 1996. Lynx recruitment during a snowshoe hare population peak and decline in southwest Yukon. Journal of Wildlife Management 60:441-452.

Murray, D. L., and S. Boutin. 1991. The influence of snow on lynx and coyote movements: does morphology affect behaviour? Oecologia 88:463-469.

Murray, D. L., S. Boutin, and M. O'Donoghue. 1994. Winter habitat selection by lynx and coyotes in relation to snowshoe hare abundance. Canadian Journal of Zoology 72:1444-1451.

National Parks Act. 1989. Queen's Printer for Canada, Ottawa, Ontario. 143 pp.

Nellis, C. H. and L. B. Keith. 1968. Hunting activities and success of lynxes in Alberta. Journal of Wildlife Management 32:718-722.

Nellis, C. H., S. P. Wetmore, and L. B. Keith. 1972. Lynx - prey interactions in central Alberta. Journal of Wildlife Management 36:320-329.

Neu, C. W., C. R. Byers, and J.M. Peek. 1974. A technique for analysis of utilization-availability data. Journal of Wildlife Management 38:541-545.

O' Donoghue, M., S. Boutin, C. J. Krebs, and E. J. Hofer. 1997. Numerical responses of coyotes and lynx to the snowshoe hare cycle. Oikos 80:150-162.

Park Resource Management Team. 1979. Riding Mountain National Park resource description and analysis. Unpublished Report for Parks Canada. Manitoba.

Parker, G. R., J. W. Maxwell, L. D. Morton, and G. E. J. Smith. 1983. The ecology of the lynx (*Lynx canadensis*) on Cape Breton Island. Canadian Journal of Zoology 61:770-786.

Pettipas, L. 1970. Early man in Manitoba. Pages 5-28 *in* W. M. Hlady, editor. Ten thousand years: archaeology in Manitoba. D. W. Friesen and Sons, Limited, Altona, Manitoba.

Pietz, P. J., and J. R. Tester. 1983. Habitat selection by snowshoe hares in north central Minnesota. Journal of Wildlife Management 47:686-696.

Poll, D. M. 1981. Snowshoe hare population dynamics and interspecific relationships in Riding Mountain National Park during winter 1979 - 1980. Canadian Wildlife Service - Unpublished report for Parks Canada-Prairie Region. Manitoba. 69 pp.

Poole, K. J. 1994. Characteristics of an unharvested lynx population during a snowshoe hare decline. Journal of Wildlife Management 58 (4):608-618.

Poole, K. G., L. A. Wakelyn, and P. N. Nicklen. 1996. Habitat selection by lynx in the Northwest Territories. Canadian Journal of Zoology 74:845-850.

Quinn, N., and G. Parker. 1987. Lynx. Pages 683-694 *in*: M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, editors. Wild furbearer management and conservation in North America. The Ontario Trappers Association, Toronto, Ontario.

Quinn, N., and J. Thompson. 1987. Dynamics of an exploited Canada lynx population in Ontario. Journal of Wildlife Management 51(2):297-305.

Reeves, B. 1970. Culture dynamics in the Manitoba grasslands 1000 B.C - A.D. 700. Pages 153-174 *in* W. M. Hlady, editor. Ten thousand years: archaeology in Manitoba. D. W. Friesen and Sons, Limited. Altona, Manitoba.

Riding Mountain National Park Round Table. 1996. Management Plan for Riding Mountain National Park. Department of Canadian Heritage, Winnipeg, Manitoba. 43 pp. + appendix.

Ristau, C., and G. Bergeson. n. d. Snowshoe hare study - RMNP. Government of Canada - Park Service - Riding Mountain National Park. Manitoba. 18 pp. + track count result pages.

Rolley, R. E. 1987. Bobcat. Pages 671-681 *in*: M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, editors. Wild furbearer management and conservation in North America. The Ontario Trappers Association, Toronto, Ontario.

Roloff, G. J. 1998. Habitat suitability model for North American lynx. Unpublished Report, Timberland Resources, Boise Cascade Corporation, Boise, Idaho. 27 pp.

Roloff, G. J., and J. B. Haufler. 1997. Establishing population viability planning objectives based on habitat potentials. Wildlife Society Bulletin 25:895-904.

Rowe, J. S. 1961. Critique of some vegetational concepts as applied to forests of northwestern Alberta. Canadian Journal of Botany 39:1007-1017.

Rowe, J. S. 1972. Forest regions of Canada. Department of the Environment. Canadian Forestry Service. Ottawa, Ontario. 172 pp.

Saunders, J. K. 1963. Food habits of the lynx in Newfoundland. Journal of Wildlife Management 27:384-390.

Seidel, J., B. Andree, S. Berlinger, K. Buell, G. Byrne, B. Gill, D. Kenvin, D. Reed. 1998. Draft strategy for the conservation and reestablishment of lynx and wolverine in the southern Rocky Mountains. Unpublished Report - Colorado Division of Wildlife, U.S. Forest Service, National Park Service, U.S. Fish and Wildlife Service, New Mexico Game and Fish Department, Wyoming Game and Fish Department. 115 pp.

Sentar Consultants Limited. 1992. Riding Mountain National Park: A literature review of historic timber harvesting and forest fire activity. Unpublished Report for Riding Mountain National Park. Manitoba. 21 pp.

Seton, E. T. 1953. Lives of game animals, Vol. IV, Part II, Rodents. Boston, Massachussetts.

Sievert, P. R., and L. B. Keith. 1985. Survival of snowshoe hares at a geographic range boundary. Journal of Wildlife Management 49:854-866.

Slough, B. G., and G. Mowat. 1996. Lynx population dynamics in an untrapped refugium. Journal of Wildlife Management 60(4):946-961.

Sokal, R. R., and F. J. Rohlf. 1981. Biometry. W.H. Freeman and Company, New York, New York.

Soper, J. D. 1953. The mammals of Riding Mountain National Park. Canadian Wildlife Service Wildlife Management Bulletin Series 1. Manitoba, Canada. No. 7. 34 pp.

Soule, M. E. 1987. Viable populations for conservation. Cambridge University Press, Cambridge, Massachusettes. 189 pp.

Stelfox, J. G. 1976. Riding Mountain National Park: Guidelines for wildlife surveys and data recording. Unpublished Report for Parks Canada - Canadian Wildlife Service. Manitoba.

Stephenson, R. O. 1986. Development of lynx population techniques. Alaska Department of Fish and Game, Federal Aid Wildlife Restoration. Report W-22-2, W-22-3, W-22-4, and W-22-5, Job 7.12R. Alaska. 84 pp.

Tabulenas, D. T. 1983. Narrative human history, Riding Mountain National Park and area: Prehistory to 1980. Microfiche report series #102. Environment Canada. Manitoba.

Thomas, D. L., and E. J. Taylor. 1990. Study designs and tests for comparing resource use and availability. Journal of Wildlife Management 54:322-330.

Thomas, J. A., J. G. Hallett, and M. A. O'Connell. 1997. Habitat use by snowshoe hares in managed landscapes of northeastern Washington. Washington Department of Fish and Wildlife. Olympia, Washington. 58 pp.

Trottier, G. C., S.R. Rollans, and R. C. Hutchinson. 1983. Range, habitat and foraging relationships of ungulates in Riding Mountain National Park. Canadian Wildlife Service, Edmonton, Alberta. 224 pp.

Tunstell, G. 1940. Forest working plan, Riding Mountain National Park. Report prepared for Department of Mines and Resources, Lands, Parks and Forests Branch, Manitoba.

U.S. Fish and Wildlife Service. 1981. Standards for the development of habitat suitability index models. U.S. Dept. Interior, Fish and Wildlife Service Release Number 1-81, 103 ESM. Varied pages.

Walker, D., and N. Kenkel. 1996. Quantifying change in habitat diversity and complexity, RMNP: 1996 Annual Progress Report. University of Manitoba, Winnipeg, Manitoba. 39 pp.

Walker, D. J., and N. Kenkel. 1997. Vegetation map legend - Riding Mountain National Park. Unpublished Report for Parks Canada - Department of Canadian Heritage, Manitoba. 19 pp.

Wang, G. G. 1995. Statistical analysis of a vegetation data base for Riding Mountain National Park. Park Service, Environment Canada, Manitoba. 65 pp.

Ward, R., and C. Krebs. 1985. Behavioural responses of lynx to declining snowshoe hare abundance. Canadian Journal of Zoology 63:2817-2824.

Warkentin, J. 1967. Human history of the Lake Agassiz region in the 19th century. Pages 325-338 *in* W. J. Mayer-Oakes, editor. Life land and water: Proceedings of the 1966 conference on environmental studies of the glacial Lake Agassiz region. The University of Manitoba Press, Winnipeg, Manitoba. Canada.

Washington Department of Wildlife. 1993. Status of the North American lynx (*Lynx canadensis*) in Washington. Unpublished Report, Washington Department of Wildlife, Olympia, Washington. 95 pp.

Washington Department of Natural Resources. 1996. Lynx habitat management plan for DNR managed lands. Unpublished Report, Washington State Department of Natural Resources, Olympia, Washington.

Whitaker, J., P. C. Paquet, and C. Walker. 1994. An ecological literature review for native plant communities of the Riding Mountain area of Manitoba (Part 1). Unpublished report for: Department of Canadian Heritage - Parks Services. Manitoba. 63 pp.

Wolfe, M. L., N. W. Debyle, C. S. Winchell, and T. R. McCabe. 1982. Snowshoe hare cover relationships in northern Utah. Journal of Wildlife Management 46:662-670.

Wolff, J. O. 1980. The role of habitat patchiness in the population dynamics of snowshoe hares. Ecological Monographs 50:111-130.

Woodley, S. J. 1993. Assessing and monitoring ecological integrity in parks and protected areas. Thesis Report, Waterloo University, Waterloo, Ontario. 167 pp.

# APPENDIX A

# Habitat Suitability Model for North American Lynx

Gary J. Roloff
Wildlife Management Specialist
Timberland Resources
Boise Cascade Corporation

## Introduction

In 1995, concerns over lynx (*Felis lynx*) population viability in the conterminous United States prompted several resource groups to petition the species for listing under the Endangered Species Act. Since that time, lynx conservation has received considerable attention (e.g., Paquet and Hackman 1995, Washington Department of Natural Resources 1996). To date, lynx have not been listed as federally threatened or endangered, however, a formal proposal for listing is pending, management concerns persist, and several State listings (e.g., Washington, Colorado) were approved. Lynx are also listed on Appendix II of the Convention on International Trade of Endangered Species (CITES) and are currently identified as a sensitive species by several U.S. Forest Service regions (Macfarlane 1994).

Concerns surrounding the effects of land management activities on lynx populations in the conterminous United States necessitated development of a model that quantitatively assessed these impacts on habitat suitability. Although few data exist on lynx in the western mountains of the United States from which to build habitat models, considerable research has been conducted in Canada and Alaska (see review in Koehler and Aubry 1994). Lynx data from Canadian and Alaskan studies must be applied cautiously to the southern Rocky Mountains because of the unique features associated with the southern range of lynx distribution. These unique features of the southern Rocky Mountains include:

- The inherently peninsular and disjunct distribution of suitable habitats (Koehler and Aubry 1994).
- The lack of dramatic fluctuations in both lynx and snowshoe hare (*Lepus americanus*) populations (Dolbeer and Clark 1975, Sievert and Keith 1985, Koehler 1990, Koehler and Aubry 1994).
- Consistently low hare densities, comparable to hare population lows in Canada and Alaska (Koehler and Aubry 1994).
- Consistently low lynx densities making the effects of fur-harvests on populations in some areas additive rather than compensatory (Koehler and Aubry 1994).
- Higher human densities coupled with low lynx densities potentially causing both direct (e.g., fur harvest) and indirect (e.g., land development causing displacement) anthropogenic effects on population persistence to be of greater magnitude.
- The potential importance of immigration from the north for short-term population persistence (Koehler and Aubry 1994).

- The range overlaps of lynx, bobcat (*Felis rufus*), and coyote (*Canis latrans*) (Koehler and Aubry 1994), and the propagation of bobcat and coyote range extensions that typically accompany anthropogenic development.

The habitat suitability index (HSI) modeling concept (U.S. Fish and Wildlife Service 1981) was used to develop a lynx habitat model for the southern Rocky Mountains. In this model, habitat potential for a lynx home range was divided into foraging, denning, and travel requisites. The lynx model uses a limiting factor approach (a concept founded in ecological theory) in that the most limiting resource(s) is assumed to have the greatest impact(s) on the population. A premise of HSI models is that limiting factors can be portrayed using mathematical relationships between vegetation structures, spatial metrics, and indices of habitat quality (U.S. Fish and Wildlife Service 1981). Theoretically, these limiting factors can be expressed as an index to animal fitness.

**Overview**

Critical considerations prior to running the lynx model are the resolution, accuracy, and precision of the land classification system and associated vegetation attribute information for the planning landscape. The land classification must be capable of characterizing vegetation structures and spatial arrangements at a resolution compatible with lynx and snowshoe hare habitat use. The ideal stratification is a stand-based (minimum mapping unit around 2 ha) ecological classification system that integrates existing vegetation conditions and site potentials (e.g., geology, soils). An ecological classification system is recommended to reduce the variability in quantifying understory vegetation attributes since these attributes are primary components of the lynx model. Deviations from these baseline recommendations for land stratification will reduce the robustness and utility of the model output.

The lynx model is divided into three components: 1) foraging, 2) denning, and 3) interspersion (Fig. 1). The lynx model was specifically developed for the mountainous habitats of Washington, Idaho, and Montana, corresponding to the southern extension of lynx range in the Rocky Mountains, however, the model framework may be applied to other regions. In applying the model to other regions, each input variable must be calibrated to the biogeoclimatic conditions characteristic of the region. For example, model variables that index winter browse availability for snowshoe hares will differ across regions depending on average snow depths. These types of differences must be accounted for when applying this framework to other regions.

Lynx habitat in the western mountains consists primarily of 2 structurally different forest types occurring at opposite ends of the forest seral gradient: 1) early successional forest structures that contain high numbers of prey (especially snowshoe hare), and 2) late-successional forest structures for denning (Koehler and Aubry 1994). Second-growth forests with dense understories also may support abundant hare populations (John Weaver, Northern Rockies Conservation Cooperative, Missoula, MT, pers. comm.). Intermediate seral stages with sparse understories serve as travel cover, functioning to provide connectivity between foraging and denning patches (Koehler and Aubry 1994).

Literature reviews and consultation with experts on lynx and snowshoe hare ecology were used to develop the lynx model. The model is not stand-based, but rather, it is designed to evaluate habitat quality in an area that corresponds to the allometric home range of lynx (250 ha; Roloff and Haufler 1997). Within a 250 ha area, habitat quality is expressed on a scale of 0.00-1.00, denoting "poor to good" habitat, respectively. Subsequently, habitat units from each allometric home range are aggregated into viable, marginal, or non-viable areas, the size of which depends on habitat quality (Roloff and Haufler 1997).

## Lynx Foraging Component

Forage availability during the winter months appears to be the most important criterion in the determination of lynx home range size and degree of home range overlap (McCord and Cardoza 1982, Ward and Krebs 1985). Lynx populations covary with snowshoe hare numbers (Brand et al. 1976, Brand and Keith 1979, Parker 1981, Bailey et al. 1986), and lynx tend to choose habitats where hares are most numerous (Murray et al. 1994). Although prey switching has been documented in the southern Rocky Mountains, the underlying determinant of lynx fitness appears to be related to winter snowshoe hare abundance. Thus, the foraging component of the lynx model is based on winter snowshoe hare habitat quality. Snowshoe hare habitat is assessed using an HSI model, and the results of the hare model are incorporated into a lynx foraging assessment.

### Habitat Suitability Model for Snowshoe Hare

#### Overview

Important components of hare habitat have been reported for different vegetation types and include dense woody vegetation (Adams 1959, Monthey 1986, Koehler 1990, Keith et al. 1993), stem diameter of browse (Keith 1974), continuity of coniferous cover (Brocke 1975), habitat interspersion (Keith et al. 1993), the distance to lowland forest cover (Conroy et al. 1979), and patch size (Thomas et al. 1997). The snowshoe hare model is divided into two primary components: 1) foraging, and 2) security cover (Fig. 1). These components are mathematically combined into an overall index of winter hare habitat quality at the map-polygon and home range levels.

Winter foraging and security cover conditions are assumed to be limiting to hares (Hart et al. 1965, Dolbeer 1972, Keith and Windberg 1978, Pease et al. 1979, Keith et al. 1984, Boutin et al. 1986, Keith et al. 1993). In this model, summer habitats are not considered a limiting factor. To index the quality of snowshoe hare habitat, it is assumed that measures of understory cover and species composition in different height strata can be used (summarized by Ferron and Ouellet 1992). In support of this assumption, Thomas et al. (1997) demonstrated significant relationships between hare population indices and the horizontal and vertical cover of understory vegetation. Since few snowshoe hare studies have been conducted in the Pacific Northwest, the vegetation-hare relationships depicted in this model were inferred from Thomas et al. (1997). Studies conducted across North America were used to supplement Thomas et al.'s (1997) work..

**Snowshoe Hare Winter Food Component**

Winter availability of palatable browse is believed to be a limiting factor of snowshoe hare populations (e.g., Windberg and Keith 1976, Pease et al. 1979, Vaughan and Keith 1981, Sinclair et al. 1988, Sullivan and Sullivan 1988, O'Donoghue and Krebs 1992). In this model, the amount of winter browse for snowshoe hares is assessed using two different measures: 1) the amount of horizontal (or lateral) cover, and 2) the amount of vertical cover in palatable species. Both measures are used to represent the "thickness" of forage for hares. Horizontal and vertical cover correlate with understory stem density (Gysel and Lyon 1980, Litvaitis et al. 1985), although this relationship may be weak (Thomas et al. 1997). In the southern Rocky Mountains, forage for hares is quantified in three height strata; 0-1, 1-2, and 2-3 m to account for variations in availability as a result of changing snow depths and the ability of hares to "clip" down vegetation from unreachable heights (Keith et al. 1984, Sullivan and Moses 1986).

Horizontal cover is measured along the geometric plane that corresponds to the ground (i.e., the thickness if one stands and tries to look through a vegetation type) whereas vertical cover is measured along a geometric plane perpendicular to the ground (i.e., the thickness if one looks up). Woody browse constitutes live plant species that have been documented as hare food sources (Table 1).

Thomas et al. (1997) found that highest browse use occurred in vegetation types with 30 to 40% horizontal cover of live vegetation. Use of vegetation types for foraging declined as woody cover approached <20% (Ferron and Ouellet 1992, Thomas et al. 1997). These findings roughly correspond to other studies that found highest hare use during winter in vegetation types with ≥50% horizontal cover (Carreker 1985, Parker 1986). Thus, optimal foraging habitat for snowshoe hares in the Rocky Mountains is assumed to be provided by vegetation types with 35% horizontal cover of live vegetation (Fig. 2). Hare winter foraging habitat quality declines as horizontal cover decreases, and habitat is unsuitable when 10% horizontal cover of live vegetation is provided (Fig. 2). Horizontal cover for foraging habitat is measured for the 0-1, 1-2, and 2-3 m height strata.

Thomas et al. (1997) also associated vertical cover with the intensity of snowshoe hare browsing. Highest browse levels corresponded to about 80% vertical cover. Browse use approached zero as vertical cover declined to about 20%. In this model, vertical cover of live vegetation is optimum at ≥80% and provides no foraging habitat at 20% (Fig. 3). Similar to horizontal cover, vertical cover is measured across three height strata.

For both horizontal and vertical cover relative to snowshoe hare browsing potential, overall habitat quality is assessed independently for each strata (i.e., an increase in browse in one stratum cannot offset a decrease in another stratum). The rationale behind this logic is that snow levels dictate the heights at which hares can access browse, thus, the different strata cannot compensate for each other (i.e., if the 1-2 m strata is unavailable, the quality does not matter because hares cannot access it). Three HSI scores are calculated from figure 2: 1) horizontal cover 0-1 m tall ($HSI_{hare,wint,food,hcov,0-1}$), 2) horizontal cover 1-2 m tall

($HSI_{hare,wint,food,hcov,1-2}$), and 3) horizontal cover 2-3 m tall ($HSI_{hare,wint,food,hcov,2-3}$). Similarly, three HSI values are derived from figure 3: 1) vertical cover 0-1 m tall ($HSI_{hare,wint,food,vcov,0-1}$), 2) vertical cover 1-2 m tall ($HSI_{hare,wint,food,vcov,1-2}$), and vertical cover 2-3 m tall ($HSI_{hare,wint,food,vcov,2-3}$). The equation for calculating hare foraging HSI's from horizontal and vertical cover is presented in equations 1 and 2, respectively.

*Equation 1:*

$$\left(\frac{HSI_{hare,wint,food,hcov,0-1} + HSI_{hare,wint,food,hcov,1-2} + HSI_{hare,wint,food,hcov,2-3}}{3}\right) = HSI_{hare,wint,food,hcov}$$

*Equation 2:*

$$\left(\frac{HSI_{hare,wint,food,vcov,0-1} + HSI_{hare,wint,food,vcov,1-2} + HSI_{hare,wint,food,vcov,2-3}}{3}\right) = HSI_{hare,wint,food,vcov}$$

The foraging habitat quality for snowshoe hare is based on the arithmetic mean of $HSI_{hare,wint,food,hcov}$ and $HSI_{hare,wint,food,vcov}$ (Equation 3). An arithmetic mean was selected because some foraging habitat can be provided (i.e., the foraging HSI > 0.00) if *only* horizontal or vertical foraging cover is present. For example, densely-stocked woody species with single-stem growth forms that do not have spreading crowns [e.g., aspen (*Populus tremuloides*)] will tend to exhibit suitable horizontal cover during winter months whereas the vertical cover provided by this vegetation community may be marginal. Using the arithmetic relationship in Equation 3, horizontal or vertical foraging cover can equal 0.00 and the foraging HSI can still be >0.00. Both horizontal and vertical foraging cover are weighted equally in the winter food component (Equation 3).

*Equation 3:*

$$\left(\frac{HSI_{hare,wint,food,hcov} + HSI_{hare,wint,food,vcov}}{2}\right) = HSI_{hare,wint,foo}$$

<u>Snowshoe Hare Winter Security Cover Component</u>

The presence of adequate winter security cover has been recognized as the primary determinant of hare habitat quality (Buehler and Keith 1982, Wolfe et al. 1982, Sievert and Keith 1985, Rogowitz 1988). In this model, cover is defined as any structure (*live or dead*) that provide security for snowshoe hares. Winter security cover for hares is assessed using three measures of structure and composition: 1) understory vegetation composition (Severaid 1942, deVois 1962, Bookhout 1965a,b, Buehler and Keith 1982, Orr and Dodds 1982), 2) horizontal cover in three height strata (Brocke 1975, Wolfe et al. 1982), and 3) vertical cover in three height strata (Wolff 1980).

*Understory Vegetation Composition*

Snowshoe hares appear to select habitats based on vegetation structure as opposed to species composition (Litvaitis et al. 1985, Ferron and Ouellet 1992) and will use most forest cover types if adequate understory vegetation exists. However, some researchers have demonstrated that vegetated stands <3 m tall dominated by conifer species provide better habitat as opposed to stands dominated by deciduous species (deVos 1962, Buehler and Keith 1982, Orr and Dodds 1982, Monthey 1986). A subjective evaluation of the dominant understory vegetation type ≤3 m tall is used to index winter cover composition ($HSI_{hare,wint,cov,dom}$). The following criteria were developed to calculate $HSI_{hare,wint,cov,dom}$:

| Understory Cover Dominance Class[a] | | | |
|---|---|---|---|
| Deciduous | Mixed | Coniferous | None |
| HSI=0.50 | HSI=0.75 | HSI=1.00 | HSI=0.00 |

[a]Based on a subjective evaluation of understory cover ≤3 m tall. "Deciduous" = >60% understory in deciduous species; "Mixed" = 40% ≤ Deciduous/Coniferous Cover ≤60%; "Coniferous" = >60% understory in coniferous species. If no understory cover exists, the HSI defaults to 0.00.

*Horizontal Security Cover*

Brocke (1975) suggested that horizontal cover is the single most important stimulus in selecting cover to avoid predation. Parker (1986) found that snowshoe hare population indices were related to horizontal cover in the 1-2 and 2-3 m height strata. Winter horizontal cover ($HSI_{hare,wint,cov,hcov}$) includes both live and dead vegetation and inanimate objects (e.g., rocks, root wads). Optimum horizontal cover is assumed to be provided at ≥90%, and horizontal cover ≤40% is deemed unsuitable winter habitat (Carreker 1985, Ferron and Ouellet 1992)(Fig. 4). Separate horizontal cover HSI's are calculated for height strata 0-1, 1-2, and 2-3 m and these are subsequently combined using an arithmetic mean to produce $HSI_{hare,wint,cov,hcov}$ (Equation 4).

*Equation 4:*

$$\left(\frac{HSI_{hare,wint,cov,hcov,0-1} + HSI_{hare,wint,cov,hcov,1-2} + HSI_{hare,wint,cov,hcov,2-3}}{3}\right) = HSI_{hare,wint,cov,hcov}$$

*Vertical Security Cover*

Vertical vegetation cover is also considered an important component of hare habitat quality (Wolff 1980). Vertical cover is defined as the percent cover of live or dead material. Again, multiple strata are used to account for variations in cover availability as a result of changing snow depths. Optimal vertical cover occurs at ≥90%, and vertical cover ≤40% provides unsuitable habitat ($HSI_{hare,wint,cov,vcov}$)(Fig. 5). Separate vertical cover indices are calculated for height strata 0-1 ($HSI_{hare,wint,cov,vcov,0-1}$), 1-2 (HSI

$_{hare,wint,cov,vcov,1\text{-}2}$), and 2-3 m ( HSI $_{hare,wint,cov,vcov,2\text{-}3}$) and are subsequently combined using an arithmetic mean to produce HSI $_{hare,wint,cov,vcov}$ (Equation 5). An arithmetic mean was selected because if vertical cover is provided in one stratum, the vegetation type provides functional security cover for at least a portion of the winter (i.e., until snow covers it). All vertical cover strata are weighted equally in the winter vertical cover component (Equation 5).

*Equation 5:*

$$\left(\frac{HSI_{hare,wint,cov,vcov,0-1} + HSI_{hare,wint,cov,vcov,1-2} + HSI_{hare,wint,cov,vcov,2-3}}{3}\right) = HSI_{hare,wint,cov,vcov}$$

Winter security cover for snowshoe hares (HSI $_{hare,wint,cov}$) is computed by first establishing whether suitable security cover exists (as the arithmetic mean of HSI $_{hare,wint,cov,hcov}$, and HSI $_{hare,wint,cov,vcov}$)(Equation 6). Subsequently, the arithmetic mean from the cover calculation is geometrically combined with the understory composition component (HSI$_{hare,wint,cov,dom}$)(Equation 6). This mathematical relationship will cause HSI$_{hare,wint,cov}$ to score as unsuitable if appropriate cover conditions are not provided.

*Equation 6:*

$$\left[\left(\frac{HSI_{hare,wint,hcov} + HSI_{hare,wint,vcov}}{2}\right) \cdot HSI_{hare,wint,dom}\right]^{0.5} = HSI_{hare,wint,cov}$$

Calculating the Snowshoe Hare Winter HSI

Winter habitat conditions are assumed to limit snowshoe hare populations, and thus, winter HSI values drive the final HSI calculation. Winter habitat components (forage and cover) are integrated into one habitat value using a geometric mean. If the winter HSI for one habitat component equals 0.00, the final HSI equals 0.00 (i.e., suitable forage *and* cover must be present to provide hare habitat). Equation 7 is used to calculate the snowshoe hare winter HSI (HSI$_{hare,wint}$).

*Equation 7:*

$$(HSI_{hare,wint,food} \times HSI_{hare,wint,cov})^{0.5} = HSI_{hare,wint}$$

## Calculating the Lynx Forage Component

The index HSI$_{hare,wint}$ provides a map-polygon level assessment of snowshoe hare habitat quality. The next step in the modeling process for lynx is to relate the polygon-level depiction of hare habitat quality to the allometric home range of lynx (250 ha). It is assumed that for each allomteric home area to support lynx, some minimum level of foraging habitat (i.e., snowshoe hare habitat) is required. These habitats must themselves be of sufficient quality to support consistent and abundant numbers of snowshoe hares. Applying the methodology of Roloff and Haufler (1997), home range functionality thresholds were established for snowshoe hares based on an evaluation of hare studies.

Similar to relationships demonstrated for other wildlife species, the home range of Lagomorphs appears negatively associated with habitat quality (Boutin 1984, Hulbert et al. 1996). It is assumed that hares will exhibit smallest home ranges when habitat conditions are optimum and that hares have largest home ranges or become nomadic in unsuitable habitats (see Roloff and Haufler 1997). Although few home range studies have quantified habitat quality and estimates of annual home range sizes for hares are uncommon, existing literature and allometric theory were used to estimate home range functionality thresholds for snowshoe hares (Roloff and Haufler 1997).

The smallest documented home range for snowshoe hares is 1.4 ha for females (mid-summer to fall)(Ferron and Ouellet 1992). Ferron and Ouellet's (1992) estimate is smaller than the allometric home range for snowshoe hares [4.5 ha, assuming an average mass of 1,400 g (Boutin et al. 1986) and using the equation for primary consumers from Harestad and Bunnell 1979], but note that Ferron and Ouellet (1992) did not estimate an annual range. Studies conducted over longer time periods have demonstrated larger home ranges. For example, Dolbeer and Clark (1975) estimated a home range of 8.1 ha using mark-recapture techniques from mid-April to early September in Colorado. Similarly, Sievert and Keith (1985) documented annual home ranges >10 ha in Wisconsin. Neither of these studies occurred in what would be considered optimum habitat conditions (Dolbeer and Clark 1975, Sievert and Keith 1985), thus, for an annual estimate of snowshoe hare home range in optimal habitat, the allometric scale (4.5 ha) seems to be a reasonable minimum area threshold (Fig 6).

The maximum documented home range (excluding nomadic individuals) is 16 ha (Behrend 1962). Behrend's (1962) study occurred at the southern edge of snowshoe hare range in presumably fragmented and thus sub-optimal habitat (Sievert and Keith 1985). Sievert and Keith (1985), working in fragmented habitats in Wisconsin, also documented home ranges >10 ha in size. Based on assumptions between home range size and expected productivity (see Fig. 6), this model assumes that hares exhibiting home ranges of 16 ha or larger are not contributing to the viability of the population (see Roloff and Haufler 1997).

Habitat quality thresholds (Roloff and Haufler 1997) were inferred by comparing home range size to hare productivity under the assumption that larger home ranges correspond to lower quality habitats and thus lower productivity (Fig. 6). The maximum annual productivity of snowshoe hares (18 young/female) has been recorded from the center of their geographic range (i.e., central Alberta) in what many assume to be an area of high quality habitats (Cary and Keith 1979). This model assumes that maximum reproductive output corresponds to optimum habitat conditions, that habitat quality scores scale linearly with reproductive output, and that the maximum documented home range corresponds to habitat quality in which a female only replaces herself annually (i.e., 2 offspring per year assuming a 50-50 sex ratio)(Fig. 6). Using documented productivity rates and estimates of home range area, a viability relationship was established for snowshoe hare (Fig. 6).

A 4.5 ha (corresponding to the allometric home range of hares) area-weighted HSI is calculated from the polygon-level assessment of snowshoe hare habitat quality. A moving window approach is used to generate a habitat contour map of snowshoe hare habitat potential that depicts home-range-level habitat quality (Roloff and Haufler 1998). Using the viability relationship developed for snowshoe hares (Fig. 6) and the home-range-level output from the snowshoe hare model, the forage potential of each lynx home range is scored according to the number of viable, marginal, and non-viable hare ranges it encompasses (Fig. 7). Hare home ranges above the viability threshold (0.60 HSI, Fig. 6) count double towards the home range tally whereas marginal home ranges (between 0.25 and 0.60 HSI, Fig. 6) count one each. Non-viable home ranges do not contribute towards the forage score. It is estimated that lynx require about 600 g of food/day (or a hare every 2 days) to subsist during winter (Brand et al. 1976). Assuming that the winter season starts in November and extends through April (about 180 days), this would imply that 90 hares would support a lynx through winter. Thus, 90 hare home ranges in a lynx home range was considered optimum (Fig. 7). The resulting HSI score from tallying hare home ranges and applying the sum to figure 7 is the foraging score for the 250 ha lynx home range.

## Lynx Denning Component

Delineation of potential lynx denning habitat is a 3-phase process: I) identify vegetation types that provide vegetative structure and size deemed suitable for denning, II) identify vegetation types that are properly arranged within a home range area, and III) identify vegetation types that provide suitable denning micro-sites (Fig. 1). Components of suitable lynx denning habitat include: 1) vegetation cover type, 2) mesic site conditions, 3) canopy closure, 4) the area of the vegetation type, 4) juxtaposition and interspersion, and 5) the amount and arrangement of downed woody debris. These stand- and site-based components are integrated into a single estimate of denning habitat quality ($HSI_{lynx,den}$) for the home range area. Management for denning habitat should also emphasize minimizing human disturbance.

### PHASE I: Vegetation cover type, site potential, canopy closure, and the area of the vegetation type

Potential denning sites are initially delineated by vegetation cover type and site conditions. Vegetation types classified as forested with an average diameter of 36 cm providing ≥ 3.72 $m^2$/ha of basal area on mesic sites satisfy denning requirements in the southern Rocky Mountains. Lynx are presumed to use mature subalpine fir (*Abies lasiocarpa*), spruce (*Picea engelmannii*), lodgepole pine (*Pinus contorta*), and mixed conifer cover for denning in Washington (Brittell et al. 1989). Phase I stands must also have >50% canopy closure (where "canopy" is defined as trees >5 m in height) and be a minimum of 2 ha in size. Also, rock crevices, caves, and overhanging banks may be used for denning sites (Hoover and Willis 1987).

These types of den sites *that have demonstrated* their suitability as lynx denning habitat in the past are evaluated using the criteria outlined in Phase II.

**PHASE II: Juxtaposition and Interspersion**

Denning sites (the 2 ha patch, rocks, crevices, or banks) must be in close proximity to forage cover (Koehler and Brittell 1990). At least 50% of the perimeter of 2 ha patches identified as potential denning sites in "Phase I" must be adjacent to some form of "suitable lynx habitat" (i.e., habitat identified as denning, foraging, or travel). Also, 30% of the land within 0.8 km of the potential denning sites must be in suitable summer foraging habitat. Suitable summer foraging habitat is based on the habitat potential score calculated for the 0-1 m vertical cover measurement in the snowshoe hare model.

Snowshoe hares forage on a variety of herbaceous vegetation during the spring and summer months (Wolff 1978), and thus, hare forage is not limiting. It is assumed, however, that snowshoe hares are more vulnerable to predation in open areas (Adams 1959, Dolbeer and Clark 1975, Sievert and Keith 1985), and thus, vegetation cover for security is the limiting factor during spring and summer. Snowshoe hare security cover can be estimated based on the amount of cover (both live and dead) 0-1 m tall. The HSI for summer cover ($HSI_{hare,sum,cov}$) is derived from a measure of vertical cover provided by all objects within the 0-1 m height strata. Optimum summer cover for hares exists in stands providing ≥60% vertical cover, and summer cover habitat quality is 0.00 when ≤20% vertical cover exists (Fig. 8).

Map polygons with a *summer* forage HSI value >0.10 satisfy the forage requirements. A map polygon may provide both suitable forage and denning, in which case the denning area is counted towards the 30% foraging. If these criteria are satisfied, the map polygon is a potential denning site and an assessment of downed woody debris is performed. Map polygons not identified as potential denning habitat through the first 2-phases are assigned a $HSI_{lynx,den,stand}$ value of 0.00. These sites are not evaluated for Phase III attributes. Also, denning structures can be constructed in association with sites that satisfy the Phase I and II criteria.

**PHASE III: Dead and Downed Woody Debris**

Dead and downed woody debris include logs, stumps, and upturned root masses (Koehler 1990, Koehler and Brittell 1990). Potential lynx dens generally consist of inter-tangled woody material with interstitial spaces large enough to provide lynx cover. Lynx dens have been described as having a high density (40 pieces per 50 m) of downed woody debris that were vertically structured 0.3-1.2 m above the ground (Brittell et al. 1989, Koehler 1990). These types of structures are often dependent on micro-site characteristics (e.g., areas susceptible to wind throw; drainages) and are often uncommon across entire forest stands, thus, it was deemed impractical to systematically sample this attribute within a stand and base the estimate on a mean value. As an alternative approach, walk-through inventories of stands identified as potential denning habitat should be conducted to ensure micro-sites exist. Walk-through inventories conducted in northeast Washington revealed that a high percentage of stands (≈70%) contained suitable micro-sites (Brian Gilbert, Wildlife Biologist, Plum Creek Timber Company, Spokane, WA, pers. comm.).

Within lynx home ranges, multiple denning sites are important. Females may move kittens to better foraging areas or to avoid disturbance (Koehler and Brittell 1990). In low quality habitat, the inability of females to move kittens may increase kitten mortality (Koehler and Aubry 1994). Assuming that the majority of denning stands contain suitable micro-sites (verified by walk-through inventories), the quantity and spatial extent of denning stands is used to index denning habitat quality. An optimal home range is assumed to contain a mosaic of vegetation types that include foraging and denning habitat (Koehler and Aubry 1994). In the lynx habitat model, the denning score in a home range is based on the average distance in a home range to denning sites. The variable $HSI_{lynx,den}$ is calculated on the premise that multiple, interspersed denning sites in a home range is of better habitat quality than a home range containing few, blocked sites. To assess each home range area, a 100x100 m grid of points is overlaid and the average nearest distance to a suitable denning site from all points is calculated. Optimum denning habitat is provided when the average distance to denning sites is 400 m and denning habitat is deemed unsuitable if average distance is 1,750 m (Fig. 9). Under these parameters, optimum conditions roughly correspond to a denning site every 16 ha.

## Lynx Interspersion Component

The interspersion component is designed to address the "travel corridor" needs of lynx (Washington Department of Wildlife 1993). Lynx travel through and on a variety of vegetation cover types and landscape features including thinned and un-thinned forested stands, regeneration, open meadows (≤ 100 m in width), ridges just above timberline, roads, and forest trails (Taylor and Shaw 1927, Parker 1981, Brittell et al. 1989, Koehler 1990). This model assumes lynx will traverse most cover types except open or sparsely-stocked stands >100 m in width. The interspersion component of this model uses a 2 step process: 1) identify areas of "non-lynx" cover. and 2) index the amount and spatial distribution of "non-lynx" habitat in the home range. "Non-lynx" habitat is defined as map polygons (or portions of map polygons) with a summer foraging or denning HSI of 0.00 that are:

a) permanent "openings" >91 m in width (e.g., meadows),

b) map polygons with perennial vegetation <2 m tall and >91 m in width, and

c) map polygons with <72 trees/ha having a 2-3 m understory providing <50% visual obstruction.

It is important to note that some map polygons may be split during this process, (i.e., a portion of the polygon is >91 m in width and the other portion is <91 m in width). These portions need to be segregated during the analysis to reduce assessment error. Suitable travel cover is subsequently delineated as map polygons not identified as forage, denning, or "non-lynx" habitat.

The interspersion index ($HSI_{lynx,inter}$) is based on the average nearest distance within a home range to "non-lynx" polygons. A systematic grid (100 x 100 m) is used to estimate the average distance to "non-lynx" polygons (Fig. 10). The $HSI_{lynx,inter}$ is based on the premise that a lower average distance to "non-lynx" polygons equates to a more interspersed configuration of habitats, and thus, to a greater probability of lynx encountering travel barriers (Fig. 11). The 100 x 100 m grid corresponds to the maximum hypothetical distance lynx will traverse without sufficient cover. Model simulations conducted on ≈5,000 ha's in potential lynx habitat in northeast Washington demonstrated that the size of the sample grid had negligible impact on the average nearest distance to "non-lynx" habitats (Table 2). Of more importance is the relationship depicted in figure 11. Lynx will traverse long distances to fulfill their life requisites. For example, Brand et al. (1976) and Nellis and Keith (1968) found that lynx traveled 8.8 km hunting during hare population lows and 4.7 km when hares were plentiful. Parker et al. (1983) calculated daily cruising distances of 6.5-8.8 km in winter and 7.3-10.1 km during summer in Nova Scotia. Koehler (1990) documented females foraging 6-7 km from their den sites. The habitat model for lynx penalizes landscapes that restrict these movements. Figure 11 attempts to quantify the effects of barriers to movement on habitat quality (i.e., how often can lynx encounter movement barriers without detracting from habitat quality?). A low average nearest distance to "non-lynx" habitat in a home range (i.e., the chances of encountering a "non-lynx" polygon are high) equates to a poor habitat quality rating (Fig. 11). As with all of the relationships depicted in this model, the distances in figure 11 are believed to be conservative approximations and should be refined with empirical data.

## Computation of Overall Lynx HSI

The 3 primary components of the lynx habitat model; foraging ($HSI_{lynx,food}$), denning ($HSI_{lynx,den}$), and interspersion ($HSI_{lynx,inter}$) are combined into one index value ($HSI_{lynx}$) depicting overall habitat suitability for lynx in the 250 ha area. All components of the lynx model are weighted equally and deemed critical for a functional home range, therefore, a geometric mean was used to represent the final HSI (Equation 8). The geometric mean causes the final HSI to equal 0.00 if any of the components equal 0.00. These 250 ha areas can be subsequently aggregated into home ranges of differing functionality and used for resource planning and modeling (Roloff and Haufler 1997).

*Equation 8:*

$$(HSI_{lynx,food} \times HSI_{lynx,den} \times HSI_{lynx,inter})^{0.33} = HSI_{lynx}$$

### Literature Review

Adams, L. 1959. An analysis of a population of snowshoe hares in northwestern Montana. Ecol. Monogr. 29:141-170.

Bailey, T.N., E.E. Bangs, M.F. Portner, J.C. Malloy, and R.J. McAvinchey. 1986. An apparent overexploited lynx population on the Kenai Peninsula, Alaska. J. Wildl. Manage. 50:279-290.

Behrend, D.F. 1962. An analysis of snowshoe hare habitat on marginal range. Proc. N.E. Sect. Wildl. Soc., Monticello, NY. 7pp.

Bookhout, T.A. 1965a. The snowshoe hare in upper Michigan-its biology and feeding co-actions with white-tailed deer. Michigan Dept. Conserv., Resour. Dev. Rep. 38. 191pp.

______. 1965b. Feeding co-actions between snowshoe hares and white-tailed deer in northern Michigan. Trans. N. Amer. Wildl. Nat. Resour. Conf. 30:321-335.

Boutin, S. 1984. Effect of late winter food addition on numbers and movements of snowshoe hares. Oecologia 62:393-400.

______, C.J. Krebs, A.R.E. Sinclair, and J.N.M. Smith. 1986. Proximate causes of losses in a snowshoe hare population. Can. J. Zool. 64:606- 610.

Brand, C.J., L.B. Keith, and C.A. Fischer. 1976. Lynx responses to changing snowshoe hare densities in central Alberta. J. Wildl. Manage. 40:416- 428.

______, and _______. 1979. Lynx demography during a snowshoe hare decline in Alberta. J. Wildl. Manage. 43:827-849.

Brittell, J.D., R.J. Poelker, S.J. Sweeney, and G.M. Koehler. 1989. Native cats of Washington. Unpubl. Rep., Wash. Dept. Wildl., Olympia.

Brocke, R.H. 1975. Preliminary guidelines for managing snowshoe hare habitat in the Adirondacks. Trans. Northeast Sec., The Wildl. Soc. Fish Wildl. Conf. 32:46-66.

Buehler, D.A., and L.B. Keith. 1982. Snowshoe hare distribution and habitat use in Wisconsin. Can. Field-Nat. 96:19-29.

Cary, J.R., and L.B. Keith. 1979. Reproductive change in the 10-year cycle of snowshoe hares. Can J. Zool. 57:375-390.

Carreker, R.G. 1985. Habitat suitability index models: snowshoe hare. U.S. Fish Wildl. Serv., Biol. Rep. 82(10.101). 21pp.

Conroy, M.J., L.W. Gysel, and G.R. Dudderar. 1979. Habitat components of clear-cut areas for snowshoe hares in Michigan. J. Wildl. Manage. 43:680-690.

deVos, A. 1962. Changes in the distribution of the snowshoe hare in southern Ontario. Can. Field-Nat. 76:183-189.

Dolbeer, R.A. 1972. Population dynamics of the snowshoe hare in Colorado. Ph.D. Diss., Colorado State Univ., Fort Collins. 210pp.

______, and W.R. Clark. 1975. Population ecology of snowshoe hares in the central Rocky Mountains. J. Wild. Manage. 39:535-549.

Ferron, J., and J.-P. Ouellet. 1992. Daily partitioning of summer habitat and use of space by the snowshoe hare in southern boreal forest. Can. J. Zool. 70:2178-2183.

Gysel, L.W., and L.J. Lyon. 1980. Habitat analysis and evaluation. Pages 305-327 in S.D. Schemnitz, ed. Wildlife management techniques manual. The Wildlife Soc., Washington, D.C.

Harestad, A.S., and F.L. Bunnell. 1979. Home range and body weight-A reevaluation. Ecology 60:389-402.

Hart, J.S., H. Pohl, and J.S. Tener. 1965. Seasonal acclimatization in varying hare (*Lepus americanus*). Can. J. Zool. 43:731-744.

Hoover, R.L., and D.L. Wills. (eds.). 1987. Managing forested lands for wildlife. Colo. Div. Wildl., Denver. 459pp.

Hulbert, I.A.R., G.R. Iason, D.A. Elston, and P.A. Racey. 1996. Home-range sizes in a stratified upland landscape of two lagomorphs with different feeding strategies. J. Appl. Ecol. 33:1479-1488.

Jones, K.J., R.S. Hoffman, D.L. Rice, et al. 1992. Revised checklist of North American mammals north of Mexico, 1991. Occ. Papers of Texas Tech Univ. 146:1-23.

Keith, L.B. 1974. Some feature of population dynamics in mammals. Trans. Intern. Congress Game Biol. 11:17-67.

_______, and L.A. Windberg. 1978. A demographic analysis of the snowshoe hare cycle. Wildl. Monogr. 58. 79pp.

_______, J.R. Cary, O.J. Rongstad, and M.C. Brittingham. 1984. Demography and ecology of a declining snowshoe hare population. Wildl. Monogr. 90. 43pp.

_______, S.E.M. Bloomer, and T. Willebrand. 1993. Dynamics of a snowshoe hare population in fragmented habitat. Can. J. Zool. 71:1385-1392.

Koehler, G.M. 1990. Population and habitat characteristics of lynx and snowshoe hares. J. For. 88:10-14.

_______, and J.D. Brittell. 1990. Managing spruce-fir habitat for lynx and snowshoe hares. J. For. 88:10-14.

_______, and K.B. Aubry. 1994. Lynx. Pages 74-98 *in* L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon, and W.J. Zielinski, tech. eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine. USDA For. Serv. Gen. Tech. Rep. RM-254.

Litvaitis, J.A., J.A. Sherburne, and J.A. Bissonette. 1985. Influence of understory characteristics on snowshoe hare habitat use and density. J. Wildl. Manage. 49:866-873.

Macfarlane, D. 1994. Appendix C: National Forest system status information. Pages 176-184 *in* L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon, and W.J. Zielinski, tech. eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine. USDA For. Serv. Gen. Tech. Rep. RM-254.

Martin, A.C., H.S. Zim, and A.L. Nelson. 1951. American wildlife and plants. A guide to wildlife food habits. Dover Publ., Inc., New York. 500pp.

McCord, C.M., and J.E. Cardoza. 1982. Bobcat and lynx. Pages 728-766 *in* J.A. Chapman and G.A. Feldhamer, eds. Wild mammals of North America. John Hopkins Univ. Press, Baltimore, MD.

Monthey, R.W. 1986. Responses of snowshoe hares, *Lepus americanus*, to timber harvesting in northern Maine. Can. Field-Nat. 100:568-570.

Murray, D.L., S. Boutin, and M. O'Donoghue. 1994. Winter habitat selection by lynx and coyotes in relation to snowshoe hare abundance. Can. J. Zool. 72:1444-1451.

Nellis, C.H., and L.B. Keith. 1968. Hunting activities and success of lynxes in Alberta. J. Wildl. Manage. 32:718-722.

O'Donoghue, M., and C.J. Krebs. 1992. Effects of supplemental food on snowshoe hare reproduction and juvenile growth at a cyclic population peak. J. Anim. Ecol. 61:631-641.

Orr, C.D., and D.G. Dodds. 1982. Snowshoe hare habitat preferences in Nova Scotia spruce-fir forests. Wildl. Soc. Bull. 10:147-150.

Paquet, P., and A. Hackman. 1995. Large carnivore conservation in the Rocky Mountains. A long-term strategy for maintaining free-ranging and self-sustaining populations of carnivores. World Wildl. Fund Canada, Toronto. 52pp.

Parker, G.R. 1981. Winter habitat use and hunting activities of lynx (*Lynx canadensis* ) on Cape Breton Island. Can. J. Zool. 61:770-786.

_______, J.W. Maxwell, L.D. Morton, and G.E.J. Smith. 1983. The ecology of the lynx (*Lynx canadensis*) on Cape Breton Island. Can J. Zool. 61:770-786.

_______. 1986. The importance of cover on use of conifer plantations by snowshoe hares in northern New Brunswick. The For. Chronicle 62:159-163.

Pease, J.L., R.H. Vowles, and L.B. Keith. 1979. Interaction of snowshoe hares and woody vegetation. J. Wildl. Manage. 43:43-60.

Rogowitz, G.L. 1988. Forage quality and use of reforested habitats by snowshoe hares. Can. J. Zool. 66:2080-2083.

Roloff, G.J., and J.B. Haufler. 1997. Establishing population viability planning objectives based on habitat potentials. Wildl. Soc. Bull. 25:

Scott, D.P., and R.H. Yahner. 1989. Winter habitat and browse use by snowshoe hares, *Lepus americanus*, in a marginal habitat in Pennsylvania. Can. Field-Nat. 103:560-563.

Severaid, J.H. 1942. The snowshoe hare, its life history and artificial propagation. Maine Dept. Inland Fish Game. 95pp.

Sievert, P.R., and L.B. Keith. 1985. Survival of snowshoe hares at a geographic range boundary. J. Wildl. Manage. 49:854-866.

Sinclair, A.R.E., C.J. Krebs, J.N.M. Smith, and S. Boutin. 1988. Population biology of snowshoe hares. III. Nutrition, plant secondary compounds and food limitation. J. Anim. Ecol. 57:787-806.

Sullivan, T.P., and R.A. Moses. 1986. Demographic and feeding responses of a snowshoe hare population to habitat alteration. J. Appl. Ecol. 23:53-63.

_______, and D.S. Sullivan. 1988. Influence of stand thinning on snowshoe hare population dynamics and feeding damage in lodgepole pine forest. J. Appl. Ecol. 25:791-805.

Taylor, W.P., and W.T. Shaw. 1927. Mammals and birds of Mount Rainier National Park. U.S. Government Printing Office, Washington, D.C.

Telfer, E.S. 1972. Browse selection by deer and hares. J. Wildl. Manage. 36:1344-1349.

Thomas, J.A., J.G. Hallett, and M.A. O'Connell. 1997. Habitat use by snowshoe hares in managed landscapes or northeastern Washington. Washington Dept. Fish and Wildl., Unpubl. Rep. 58pp.

U.S. Fish and Wildlife Service. 1981. Standards for the development of habitat suitability index models. U.S. Fish Wildl. Serv. ESM 103. n.p.

Vaughan, M.R., and L.B. Keith. 1981. Demographic response of experimental snowshoe hare populations to over-winter food shortage. J. Wildl. Manage. 45:354-380.

Ward, R., and C.J. Krebs. 1985. Behavioral responses of lynx to declining snowshoe hare abundance. Can. J. Zool. 63:2817-2824.

Washington Department of Natural Resources. 1996. Lynx habitat management plan. Washington Dept. Nat. Resour., Olympia. 180pp.

Washington Department of Wildlife. 1993. Status of the North American lynx (*Lynx canadensis* ) in Washington. Unpubl. Rep., Wash. Dept. Wildl., Olympia.

Windberg, L.A.. and L.B. Keith. 1976. Snowshoe hare population response to artificial high densities. J. Mammal. 57:523-553.

Wolfe, M.L., N.V. DeByle, C.S. Winchell, and T.R. McCabe. 1982. Snowshoe hare cover relationships in northern Utah. J. Wildl. Manage. 46:662-670.

Wolff, J.O. 1978. Food habits of snowshoe hares in interior Alaska. J. Wildl. Manage. 46:662-670.

_______. 1980. The role of habitat patchiness in the population dynamics of snowshoe hares. Ecol. Monogr. 50:111-130.

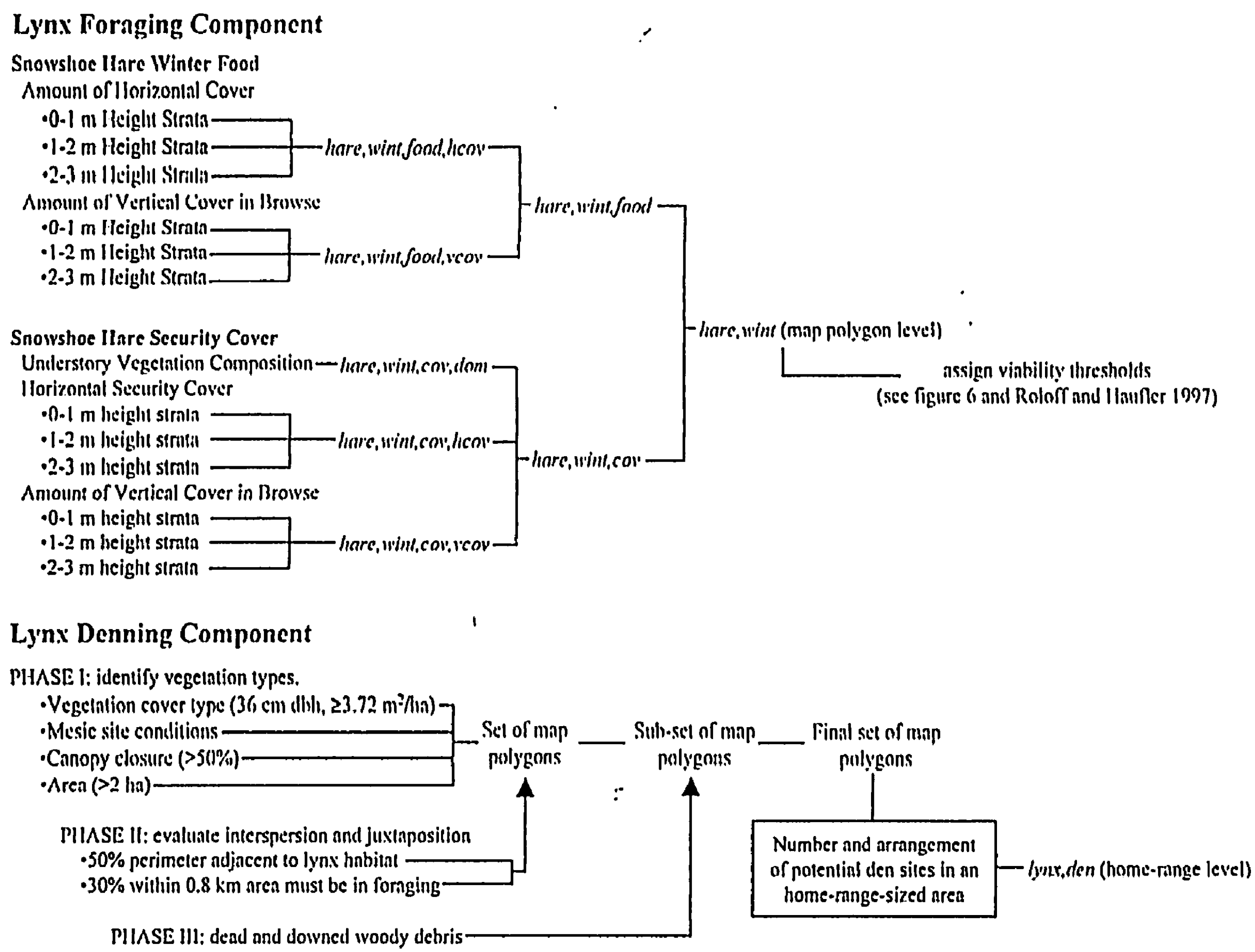



Figure 1. Schematic of the lynx habitat potential model.

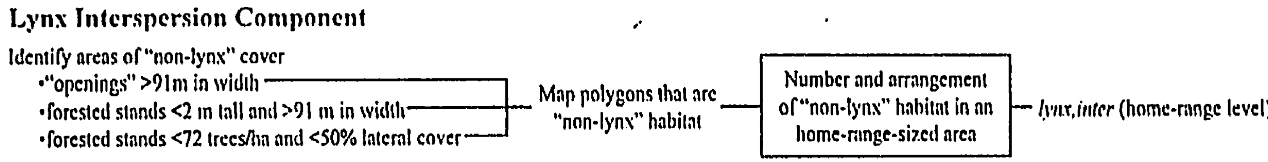



Figure 1 (cont.). Schematic of the lynx habitat potential model.

Figure 2. Relationship between horizontal cover of browse and HSI score for the 0-1, 1-2, and 2-3 m height strata. Line equation between 10 and 35% is $y=0.04x-0.4$.

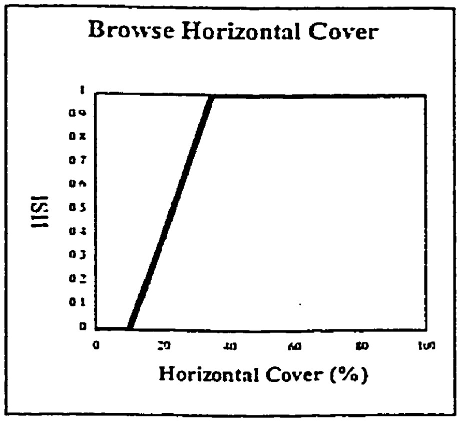



Figure 3. Relationship between vertical cover of browse and HSI score for the 0-1, 1-2, and 2-3 m height strata. Line equation between 20 and 80% is $y=0.01666x-0.332$.

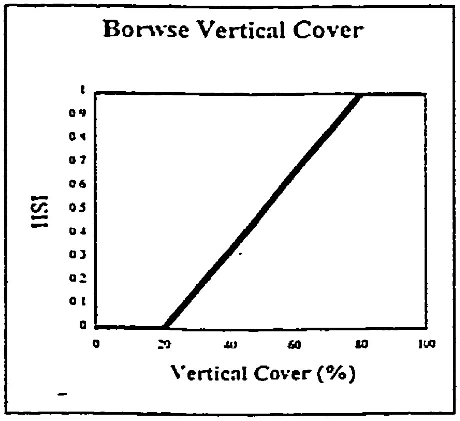



Figure 4. Relationship between horizontal security cover and HSI score for the 0-1, 1-2, and 2-3 m height strata. Line equation between 40 and 90% is $y=0.02x-0.8$.

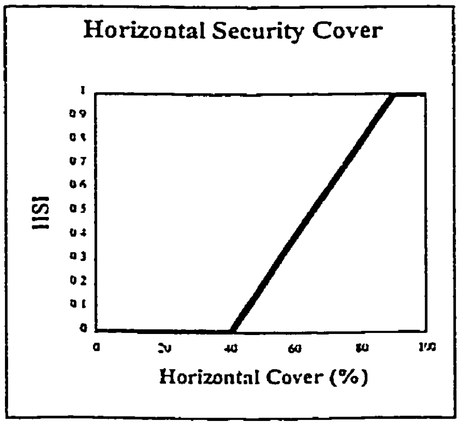
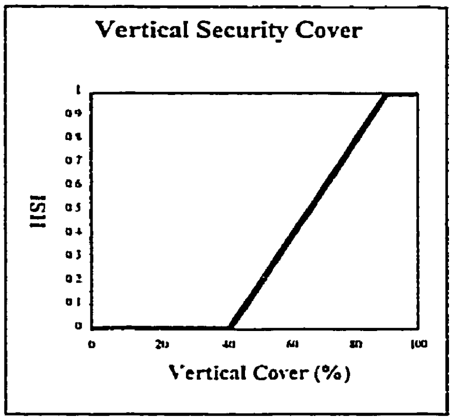
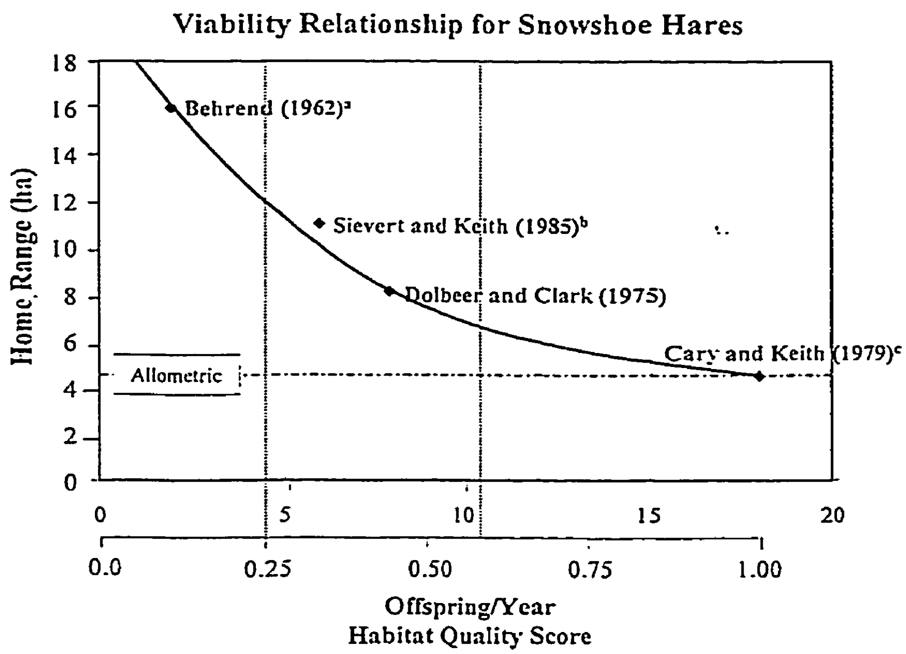


Figure 5. Relationship between horizontal cover of browse and HSI score for the 0-1, 1-2, and 2-3 m height strata. Line equation between 40 and 90% is $y=0.02x-0.8$.




[a]Annual productivity assumed to be 2 in that females are only replacing themselves.
[b]Home range presented as >10 ha in literature. Here assumed as 11 ha.
[c]Inferred home range size based on allometric theory.

Figure 6. Viability relationship for snowshoe hares developed for the lynx habitat model. Viability threshold was established at 12 young/year (0.60 score). The marginal threshold was established at 5 young/year (0.25 score).

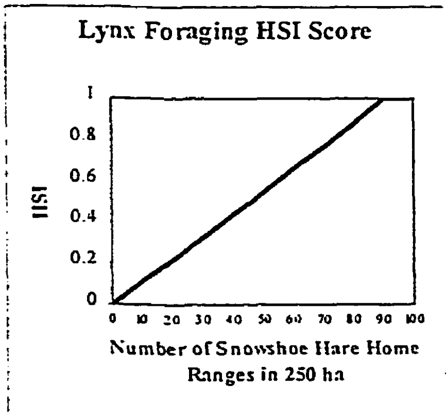



Figure 7. Relationship between the number of snowshoe hare home ranges in a lynx allometric home range and the lynx foraging HSI score. Line equation between 0 and 90 home ranges is $y=0.011x$.

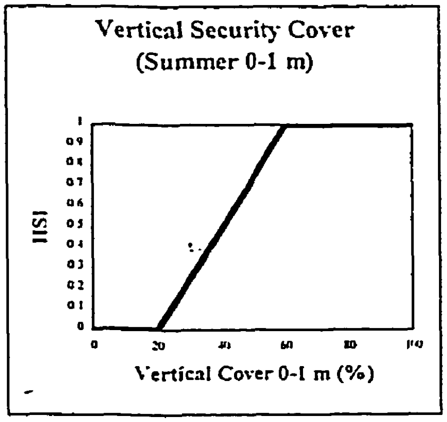



Figure 8. Relationship between summer security cover and HSI score for the 0-1 m height strata. Line equation between 20 and 60% is $y=0.025x-0.5$.

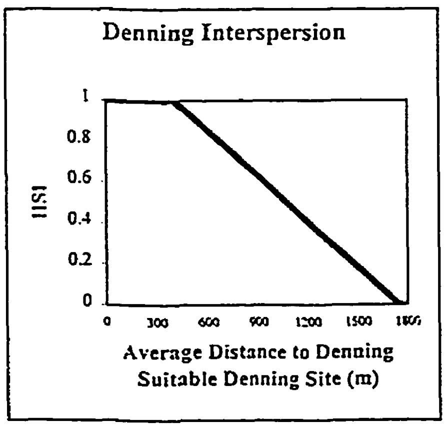



Figure 9. Relationship between the average distance to suitable denning sites in a 250 ha lynx home range and HSI score. Line equation between 400 and 1,750 m is $y=-0.000741x+1.2964$.

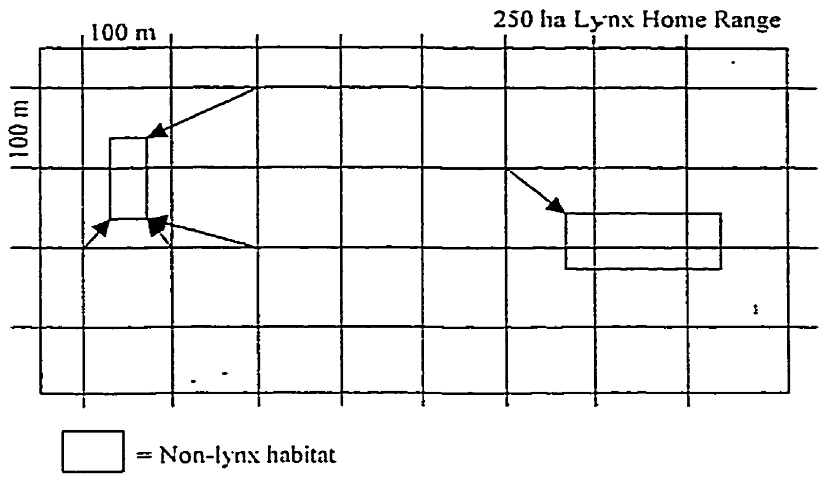
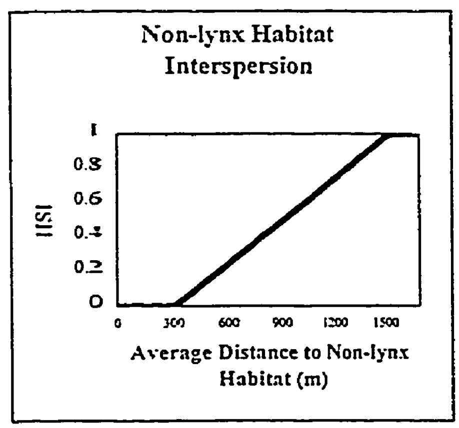



Figure 10. Calculating the average distance to non-lynx habitat using a 100 x 100 m sample grid. Distance from each grid intersection to the nearest non-lynx habitat is measured.

Figure 11. Relationship between the average distance to "non-lynx" habitat in a 250 ha lynx home range and HSI score. Line equation between 300 and 1500 m is $y=0.0008333x-0.25$.



# APPENDIX B

DATA SHEET

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| OBSERVER NAME | | | | | | | | |
| DATE | | | | | | | | |
| WEATHER CONDITIONS | sun | cloud | mixed | snow | | | | |
| TEMPERATURE | | | | | | | | |
| CLOUD COVER LAST NIGHT | none | partial | complete | | | | | |
| DATE OF LAST SNOWFALL | | | | | | | | |
| TRAILS FOLLOWED TODAY | | | | | | | | |
| ADDITIONAL COMMENTS | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| **TRACKS** | **TRACKS** | **DIRECTION** | **ESTIMATED** | **HABITAT** | **VISUAL** | **VISUAL** | **VISUAL** | **HARE SIGN** |
| **# OF** | **# OF** | **OF** | **TRACK** | **CLASS** | **CANOPY COVER** | **VEG. TYPE** | **STAND AGE** | **BY LYNX TRACK** |
| **ADULTS** | **KITS** | **TRAVEL** | **AGE** | (GPS) | NONE, LOW, MED., H | CONIF., DECID., MIXE | YNG, MED., OLD | NONE, LOW, MED. HI |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |
| | | | | | | | | |

1997-1998 LYNX TRACK COUNTS IN RIDING MOUNTAIN NATIONAL PARK

# APPENDIX C

# GUIDELINES FOR MONITORING LYNX POPULATIONS AND HABITAT USE IN RIDING MOUNTAIN NATIONAL PARK

The first step in developing any monitoring program is to assign one person or group the responsibility of over-seeing the project. This person should be responsible for initiating and coordinating the monitoring program by distributing the data sheets (found at the end of Appendix C) and instructions, ensuring the return of the data, making arrangements to monitor any uncovered areas, analyzing the data, and ensuring the integrity of the data.

The months of February and March are conducive to snow-tracking lynx in RMNP. Young lynx tend to stay with their mother throughout their first winter until they are about 9-10 months of age (Carbyn and Patriquin 1983, Bailey et al. 1986, Koehler 1990, Parker et al. 1983, Slough and Mowat 1996). They disperse at the onset of the breeding season (Quinn and Parker 1987) which occurs at the end of March to the beginning of April of each year (McCord and Cardoza 1982, Quinn and Parker 1987, Slough and Mowat 1996). Tracking in February increases the likelihood that groups of tracks are from females with kittens, rather than some other combination of individuals. Parker et al. (1983) stated that groups of three or four tracks typically are a female with kittens. Two tracks together may be two yearlings, or perhaps, during the breeding season, a male and female. It was also noted that during periods of abundant hare, single lynx tracks are often those of yearlings, however, during a hare low, single tracks are more likely an adult. It has also been reported that two females occasionally travel together with their kittens although lynx are typically known as solitary (Koehler and Aubry 1994). Additionally, by late winter, it is difficult if not impossible to distinguish adult tracks from young-of-year tracks. Thus, for February and early March track surveys, groups of tracks are most likely to be that of a female with kittens from the previous summer. Another advantage to tracking in February is that tracking conditions are typically good in RMNP. At this time, there is sufficient snow to use snow-machines in otherwise inaccessible areas and the temperature is cool enough to ensure tracks have not melted out. Tracking during this time period also reduces costs and personnel hours since this time period also corresponds to the annual wolf monitoring program. It is recommended that the tracking period be conducted from February 15 to March 15 of each year.

Guidelines for snow-tracking in RMNP suggest that tracking should begin approximately 48 hours after a fresh snowfall (Stelfox 1976). This time period provides for enough time for lynx movement but not too much time lapse for the tracks to become obscured by other animals or by tracking over from the same lynx. It is recommended that tracking be performed as close to this time interval as possible. Ideally, the data should be gathered within 3-10 days following a snowfall (Stephenson 1986). Koehler (1990) suggested that monitoring lynx population trends should include snow-tracking more than three times per winter to account for variations in snow and lighting conditions and experience of personnel. This suggestion is recommended.

During the lynx tracking survey of 1997-1998 and 1999, the majority of the Park wardens assisted in collecting track information. This method appeared to work well, allowing personnel with detailed knowledge of their district to travel areas unfamiliar to others. As such, a large data set was developed. Additionally, more area could be covered in a shorter period of time. Observer bias is one disadvantage of using several trackers, however, by providing clear and detailed instructions, a detailed data sheet, knowledge of track appearance, and stressing the importance of accurate information, the probability of error or oversight of important information can be minimized.

The routes established in the preliminary monitoring program are mapped in Figure 3.6. These routes were traversed over a 4-month period. The tracking routes followed in the 1999 tracking season (February 15-March 15) should form the basis of the annual monitoring route protocol since the time frame is the same as what is recommended for future monitoring. These routes can be found in Figure C.1. In order to ensure consistency in data collection, the same routes should be travelled on an annual basis. It is very important to ensure that data from only one trip down a route is used in the analysis to ensure a discrete data set. In other words, if two observers travel one trail during the survey, or if one observers traverses one trail more than once, only use the data from the first observation in the analysis. Information gathered from the other observer is

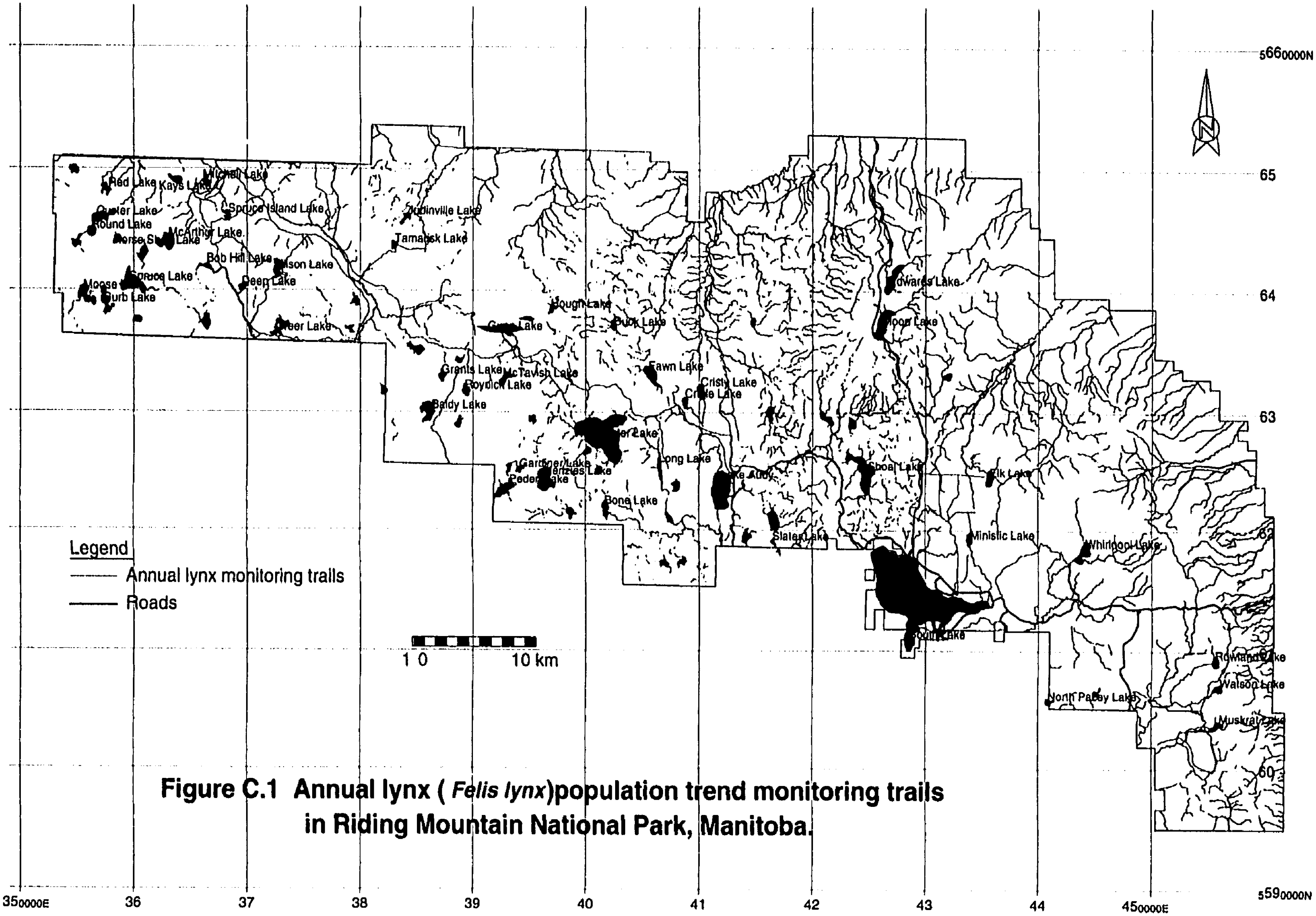



**Figure C.1 Annual lynx (*Felis lynx*) population trend monitoring trails in Riding Mountain National Park, Manitoba.**

useful as supplemental information, but cannot be used for the monitoring calculations.

All lynx tracks are to be recorded using a Geographical Position System (GPS) using the map datum NAD 83. Ensure that all lynx track crossings are counted provided that they can not be visually connected to one another (O'Donoghue et al. 1997). If a lynx follows the trail, count it as only one animal.

Once all the data have been collected and returned, the analysis begins by plotting all the data points (i.e., lynx tracks) onto the lynx home-range-level HSI map. Since the same trails will be traversed each year, and the time frame will be the same (one month), then the number of tracks found per tracking period will be directly comparable each year. Trends in the lynx population should become apparent on a direct comparison basis.

Quinn and Parker (1987) stated that lynx populations rise and fall approximately one year behind the snowshoe hare. Therefore, lynx population trends can be estimated by observing the population trends of hare. This observation will only give the observer an index, not density, but it will allow the observer to estimate the status of the lynx population relative to the natural population cycle.

The annual monitoring program should be conducted for a minimum of ten years or until a complete lynx population cycle has occurred. This time period will provide a baseline index of the RMNP lynx population at all stages of the population cycle. Each year, the statistical tests (G-test and Bonferonni z) performed in chapter 3 of this study should be re-run to verify the performance of the HSI model. The ten years correspond to the full lynx population cycle documented in the literature. Ten years should also provide sufficient testing of the habitat model to determine its full usefulness over a range of lynx population levels.

Habitat modelling is used to estimate population trends. As such, if Roloff's (1998) model appears to work well (i.e., a significant difference is found in habitat use versus availability each year as predicted by the model) after the ten-year period, then managers could rely more heavily on the model to estimate the status of lynx populations within RMNP. After the ten-year period, the monitoring program should continue on a tri-annual basis as a periodic check of the model. During these monitoring years, the same tracking routes established from this study should be followed to ensure consistency.

LYNX TRACK COUNT DATA SHEET FOR THE
RIDING MOUNTAIN NATIONAL PARK ANNUAL LYNX SURVEY

| | | | | | |
|---|---|---|---|---|---|
| OBSERVER NAME | | | | | |
| DATE | | | | | |
| WEATHER CONDITIONS | sun | cloud | mixed | snow | |
| TEMPERATURE | | | | | |
| DATE OF LAST SNOWFALL | | | | | |
| | | | | | |
| **EXACT** DESCRIPTION OF | | | | | |
| YOUR ROUTE TAKEN | | | | | |
| TODAY | | | | | |
| | | | | | |
| COMMENTS | | | | | |
| | | | | | |
| | | | | | |
| TRACK INFORMATION | | | | | |
| | | | | | |
| **# OF ADULT TRACKS** | **# OF KITTEN TRACKS** | **ESTIMATED TRACK AGE (DAYS)** | **UTM NORTHING** | **UTM EASTING** | **NAD 27, 83, OR OTHER** |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |