Habitat selection of fishers (*Martes pennanti*) in an untrapped refugium: Algonquin Provincial Park

A Thesis Submitted to the Committee on Graduate Studies in Partial Fulfillment of the
Requirements for the Degree of Master of Science in the Faculty of Arts and Science.

Trent University
Peterborough, Ontario, Canada

Watershed Ecosystems Graduate Program

June 2006

Abstract

Habitat selection of fishers (*Martes pennanti*) in an untrapped refugium:
Algonquin Provincial Park.

Susan M. Tully

Historically, fisher (*Martes pennanti*) populations throughout Ontario experienced marked declines, and were extirpated from many areas. These declines may be attributed to the pressures from overtrapping and habitat loss from logging. A population persisted in Algonquin Park and served as a source for reintroduction and natural recolonization into areas that were formally occupied. The ecology of this remnant population has not been studied. I present our findings from a live-trapping and radio-telemetry study within Algonquin Provincial Park.

I examined live-trapping results of fishers and martens with reference to season and weather. Based on our trapping experiences, I developed models to predict capture success. Using Akaike Information Criterion for model selection I found that the most predictive model for fisher trapping included the effects of the year and maximum daily air temperature. The best season for trapping fishers was spring and the best individual month was March. For the closely related American marten (*Martes americana*), the best trapping model included the combined effects of year and maximum daily air temperature. I found that the best marten trapping season was winter and the best individual month was March. Information from a rabies trap-vaccinate-release program in the St. Lawrence area suggested higher capture rates in fall than in summer.

I used radio-tracking data to assess female home range sizes and habitat selection at two hierarchical levels. Locations from aerial and ground telemetry were combined to estimate home range areas using 95% Minimum Convex Polygons. Analysis of data collected on 9 females indicated the average home range size was 29.9 ± 5.19 km^2. These estimates are at the high end of the range of previously reported home range sizes. Through analysis of habitat selection, I tested the validity of a habitat suitability model (Allen 1983) at two hierarchical levels: among home ranges and within home ranges. Using standardized resource selection functions I evaluated habitat selection by fishers based on 3 habitat variables from Allen's model: mean overstory tree diameter at breast height (dbh), percent of tree canopy diversity, and percent of overstory composed of deciduous species. There was a significant positive correlation between our RSFs and Allen's HSI among home ranges, but not within home ranges. Further analysis within home ranges indicated that female fishers selected old tolerant hardwood stands.

Keywords: Fisher (*Martes pennanti)*, Algonquin Provincial Park, Akaike's Information Criterion, Minimum Convex Polygon, Habitat Suitability Index, Resource Selection Functions

Acknowledgements

I would foremost like to express my appreciation to Dr. Jeff Bowman for providing me with this amazing opportunity. Thank you for your patience, time and guidance. I would like to thank the other members of my committee: Dr. Dennis Murray, Dr. Brent Patterson, and Dr. Paul Wilson for their valuable insights along the way. Thank you also to Dr. Martyn Obbard and Dr. Jean-Francois Robitaille for your time and suggestions.

There are many people who invested time and patience into the fieldwork for this study. I am very grateful to Richard Shalla for passing on his trapping knowledge; thank you for teaching me how to be a successful trapper. I was lucky to have been able to work with capable and enthusiastic field assistants. Thank you to Amelia Argue for bearing with me while I figured things out and for your patience and persistence during dry times. Also to Shaun "would you rather" McIntosh for your sense of humor and hard work, nighttime field work would not have been the same without you. Thank you to Carrie Sadowski and Derek Potter for your help tracking, trapping, and retrieving collars. There are many people from Harkness who were always interested and ready to help: Doug Brown, Dea Chute, Caleb Sylvester, and Mark Rose. Thank you for your help and providing a distraction from the insanity of the blackflies! Thank you to my pilot Geoff Bohart for your patience while I learned the ropes, and to Derek Meier who taught me the art of aerial telemetry.

Thank you also to those who gave me technical support: Kevin Middel, your ArcView expertise amazes me, and Bruce Pond your help with the FRI was greatly appreciated.

I would like to thank those who provided funding for this project: Ontario Ministry of Natural Resources Wildlife Research and Development section, NSERC, and Ontario's Living Legacy Trust Fund. Thank you also to Algonquin Provincial Park and Ontario Ministry of Natural Resources Rabies Research Unit for their co-operation and contributions.

There are two people I would like to thank, not only for their help in the field but also for their friendship and camaraderie. I would like to thank Ken Mills for always looking out for me, lending a helping hand when I needed it, and of course all of those adventures to Mordor. To Jaime Mills, the ice cream guru, thank you for your help trapping and all the little adventures and good times we shared.

To my parents for always being supportive and taking an active interest in this project. Thank you for being my most willing volunteers; up for anything no matter how long the paddle, or how thick the bush. Thank you to my best friend Michelle, for always having open ear and time to talk. Finally to Ryan, your understanding, patience and encouragement were steadfast. Your gentle push allowed me to see the light at the end of the tunnel, thank you.

Table of Contents

List of Tables

List of Figures

General Introduction

The Fisher

The fisher is a medium-sized solitary carnivore that is the largest member of the genus *Martes* (Powell and Zielinski 1994). This opportunistic predator eats a variety of small- to medium-sized mammals and birds as well as carrion and fruit. Preferred prey species include snowshoe hare *(Lepus americanus)* and porcupine *(Erethizon dorsatum)*. Other readily consumed prey are mice and voles, shrews and moles, squirrels, and flying squirrels (Powell and Zielinski 1994). Their diet has not show major changes in response to changing seasons (Clem 1977). Fishers are crepuscular hunters but are active at other times as well (Douglas and Strickland 1987). Though intraspecific strife and competition with other predators (e.g., red fox (*Vulpes vulpes*), lynx (*Lynx lynx*)) do occur, the fisher has no known natural enemies (Brander and Brooks 1973, Douglas and Strickland 1987). Their greatest cause of mortality is from fur harvesting (Strickland 1994).

Habitat has been defined as the location that has the ability to support a population (Farmer *et al.* 1982). Habitat characteristics include space, food, cover, and other animals (Farmer *et al.* 1982). Fishers have been associated with late-successional, conifer-dominated forests (Powell and Zielinski 1994). Physical structure is a common attribute of these forest types, especially near the ground (Powell *et al.* 2003). Most recent habitat studies have suggested that tree species composition is of less importance than the amount of forest structure (Buskirk and Powell 1994). Structure includes vertical and horizontal complexity, tree size, tree shape diversity, dead and downed woody debris. The degree of structure influences the abundance and vulnerability of prey as well as the provision of den and rest sites (Powell and Zielinski 1994). Snow accumulation is also affected by the physical structure of an area; fishers are restricted to areas with low accumulation (Arthur *et al.* 1989, Krohn *et al.* 1995, Powell and Zielinski 1994).

Distribution and Population Fluctuations

The fisher is endemic to North America (Powell *et al.* 2003). During pre-settlement times, fisher populations ranged from the forests of Canada and the northern United States along the Appalachian and Pacific coast mountain ranges in the south (Kyle *et al.* 2001, Powell and Zielinski 1994). Their historical northern limit west of Hudson Bay was 60° N, in the east it extended to the southern tip of James Bay (Powell and Zielinski 1994). As human populations increased and land was cleared for development there was a sharp decline in fisher populations. This decline occurred between the late 1800s and 1940s and was attributed to overharvesting and habitat destruction via logging and human settlement (Kyle *et al.* 2001, Powell and Zielinski 1994). Populations in Canada exhibited these significant declines but they were somewhat obscured by the 10-year population cycles in which the fisher responds to the cycles of the snowshoe hare (Powell and Zielinski 1994). Closure of the trapping season in eastern North America in the 1930s and the end of the logging boom around the same time allowed the fisher populations to begin to recover (Powell and Zielinski 1994). Fishers now occupy much of their former range as a result of reintroduction initiatives and forest regeneration (Powell 1993).

The Fisher in Ontario

Historically in Ontario the fisher harvest has been of economic importance and still is today, although to a lesser extent (Powell 1993). In the Algonquin Region alone fisher harvest sold for over $79,000 during the 1973 – 1974 season (Strickland and Douglas

1975). Harvests went unregulated until the mid 1920s and during these times pelt prices were high (Powell and Zielinski 1994, Strickland 1994). Populations declined severely due probably to over-harvest and habitat degradation through timber harvest. A quota system was devised for the fisher in Ontario and applied at the start of the 1949 – 1950 trapping season (deVos 1951). By this time fishers were nearly extirpated. There were however, a couple of remnant populations including one in Algonquin Park (Strickland and Douglas 1975). Through reintroductions and natural migration of these animals, populations in surrounding areas began to increase. Although the harvests were regulated, fisher populations in the Algonquin region began to decline again in the early 1970s due to excessive trapping pressure. Decreasing the length of the season in 1975 resulted in an increased breeding rate and the population began to increase once again (Powell 1993). Through harvest regulation and habitat improvement, fishers now occupy much of their natural range (Powell 1993) and populations in Ontario continue to expand (Carr 2005, Koen 2005).

Forest Management

The forests of Algonquin Park have structural and compositional differences compared to the forests that existed there in pre-settlement times (Quinn 2004). The fluctuations of some of Algonquin's wildlife populations have been attributed to historical changes in the structure and composition of the Park's forests (Quinn In press). Though some of these changes have occurred naturally (lightning strikes caused an average of 12.8 fires a year in the park between 1921 - 1973 [Runge and Theberge 1974]), most changes have been anthropogenic in origin. Prior to the establishment of the park in 1893 aggressive logging took place in the Algonquin region and much pine and hardwood timber was removed.

By the time the park was established almost the whole eastern third of the park had been cut and burned (Runge and Theberge 1974). It was not until the 1950s that logging became regulated, and harvesting methods shifted from clear-cutting to the less aggressive methods of single-tree selection and uniform shelterwood that are still used today (Quinn 2004). Selection was based on a minimum tree diameter that harvestable trees had to meet (Runge and Theberge 1974). Today, logging is allowed in 78% of the Park in what is called the Recreation-Utilization zone (Quinn In press). Forest fires also played a role in shaping the structure of Algonquin’s forests. Fires were fuelled by the debris left from clear-cuts and large areas burned in an uncontrolled way. In the 1920s loss of timber to fire became a concern so airplanes and watch towers were introduced as a means of fire suppression (Runge and Theberge 1974). By 1936 < 2 square miles of forest was lost to fire each year (Runge and Theberge 1974). Today, Algonquin’s forests are in a mature state due to the loss of forest renewal stimuli (Quinn 2004, Runge and Theberge 1974).

Importance of Refuges

Wildlife refuges provide a unique opportunity to study wildlife population dynamics in an environment largely free from anthropogenic influences. To date, almost all fisher populations that have been studied have been from harvested populations (Powell 1994). It is critical to study the dynamics of unharvested fisher populations as the responses to harvesting can effectively mask the response of a population to natural events (Powell 1994). Harvesting pressures on a population not only affect its size, but also population dynamics, age structure, sex ratio, spacing patterns, and probably mating patterns and foraging costs (Powell 1994). The removal of animals by harvesting affects the population size which will have indirect effects on the social structure and spacing patterns of the remaining animals (Garant and Crete 1997). Habitat selection in trapped areas is thought to result more from high trapping pressure in high quality habitats than from optimal habitat selection (Bowman Unpubl.). Habitat selection within refuges will reflect uninfluenced natural habitat selection. Refuges are considered to be the most effective measure in preventing the decline and possible extirpation of fisher populations (deVos 1952). Algonquin Park was Ontario's first provincial park, and is one of the largest refugia in Ontario (Ontario Ministry of Natural Resources 1998). Thus, it is an optimal study site to look at the natural movements of this dynamic furbearer.

The objectives of this study were to examine the movements of fishers in an untrapped refugium. Specifically, I wanted to look at habitat selection by the Algonquin fishers at two hierarchical levels: among home ranges and within home ranges. Using these results I aimed to test the validity of a fisher habitat suitability model (Allen 1983). Allen's model had not been tested like this before.

Literature Cited

Allen, A.W. 1983. Habitat suitability index models: Fisher. U.S. Dept. Int. Fish Wildl. Serv. FWS/OBS-82/10.45. Pp 19.

Arthur, S. M., W.B. Krohn, and J.R. Gilbert. 1989. Habitat use and diet of fishers. J. Wildl. Manage. 53(3): 680-688.

Bowman, J. Unpublished manuscript. Source-sink dynamics of hunted populations.

Brander, R.B. and D.J. Brooks. 1973. Return of the fisher. Natural History. 82(1): 52 – 57.

Buskirk, S.W. and R.A. Powell. 1994. Habitat ecology of fishers and American martens. In: Buskirk, S.W., A. Harestad, M. Raphael, comps. Eds. Biology and conservation of martens, sables and fishers. Ithica, NY. Cornell Univesity Press: 283 – 296.

Carr, D. 2005. Genetic structure of a recolonizing population of fishers *(Martes pennanti)*. M.Sc. Thesis. Trent University. Peterborough, Ontario.

Clem, M.K. 1977. Interspecific relationships of fishers and martens in Ontario during winter. In: R.L. Phillips and C. Jonkel, eds. Proc. 1975 Predator Symp., Mont. For. Conserv. Exp. Stn., Univ. of Montana, Missoula. Pp 165 - 182.

deVos, A. 1951. Recent findings in fisher and marten ecology and management. Trans. N. Am. Wildl. Nat. Res. Conf. Pp 498 – 507.

deVos, A. 1952. The ecology and management of fisher and marten in Ontario. Ont. Dep. Lands For., Tech. Bull., Wildl. Ser. 1. Pp 90.

Douglas, C.W. and M.A. Strickland. 1987. Fisher. In: M. Novak, J.A. Baker, M.E. Obbard, and B. Malloch eds. Wild Mammals of North America. Ontario Ministry of Natural Resources. Toronto, Ontario. Pp 511 – 529.

Farmer. A.H., M.J. Armbruster, J.W. Terrell, and R.L. Schroeder. 1982. Habitat models for land – use planning: assumptions and strategies for development. Trans. N. Am. Wildl. Nat. Res. Conf. Pp 47 – 56.

Garant, Y. and M. Crête. 1997. Fisher, *Martes pennanti*, home range characteristics in a high density untrapped population in southern Québec. Can. Field Nat. 111(3): 359 – 364.

Koen, E. 2005. Home range, population density, habitat preference, and survival of fishers *(Martes pennanti)* in eastern Ontario. M.Sc Thesis. Ottawa – Carleton Institute of Biology. Ottawa, Ontario.

Krohn, W.B., K.D. Elowe, and R.B. Boone. 1995. Relations among fishers, snow, and martens: Development and evaluation of two hypotheses. For. Chron. 71(1): 97 – 105.

Kyle, C.J., J.F. Robitaille, and C. Strobeck. 2001. Genetic variation and structure of fisher (*Martes pennanti)* populations across North America. Molecular Ecology. 10: 2341 – 2347.

Ontario Ministry of Natural Resources. 1998. Algonquin Provincial Park Management Plan. Queen's Printer for Ontario. Ontario, Canada. Pp 83.

Powell, R.A. 1993. The fisher: life history, ecology, and behavior. University of Minnesota Press. Minneapolis, MN. Pp 237.

Powell, R.A. 1994. Structure and spacing of Martes populations. In: S.W Buskirk, A. Harestad, M. Raphael comps. Eds. Martens, sables and fishers: biology and conservation. Cornell University Press. Ithica, NY. Pp 101 – 121

Powell, R.A., S.W. Buskirk, W.L. Zielinski. 2003. Fisher and Marten. In: G.A. Feldhamer, B.C. Thompson, and J.A. Chapmand. Wild Mammals of North America. The Johns Hopskins University Press

Powell, R.A. and W.J. Zielinski. 1994. Fisher. In: L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon and W.J. Zielinski eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the western United States. US Dept. Ag. Pp 38 – 73.

Quinn. N.W.S. 2004. The presettlement hardwood forests and wildlife of Algonquin Provincial Park: a synthesis of historic evidence and recent research. For. Chron. 80(6): 705 – 717.

Quinn, N.W.S. In press. Reconstructing changes in abundance of White-tailed deer, *Odocoileus virginianus,* Moose, *Alces alces,* and Beaver, *Castor Canadensis,* in Algonquin Park, Ontario, 1860 – 2004.

Runge, R.A. and J.B. Theberge. 1974. Algonquin decline: of the deer. Ontario Naturalist. 14(2): 7 – 10.

Strickland 1994. Structure and spacing of Martes populations. In: S.W Buskirk, A. Harestad, M. Raphael comps. Eds. Martens, sables and fishers: biology and conservation. Cornell University Press. Ithica, NY. Pp 149 – 164.

Strickland, M.A. and C.W. Douglas. 1975. Advance in fisher and marten management. Canadian Trapper. 4(1): 8 – 11.

Techniques for live capturing fishers (*Martes pennanti*) and marten (*Martes americana*) with reference to season and weather effects

Abstract

A field study involving the live-trapping of fisher (*Martes pennanti)* and American marten (*Martes americana)* was conducted in Algonquin Provincial Park, Ontario. We examined the influence of season and weather on our capture success. We conducted live trapping from August 2003 through December 2004 for a total of 10,375 trap nights. This trapping effort resulted in 56 fisher captures (0.54 captures/100 TN), 646 marten captures (6.23 captures/100 TN), and 233 captures of other species. Twenty-six individual fishers (10 males: 16 females) were caught in single-door live traps, immobilized and radio-collared. Fifty-eight marten (52 males: 6 females) were immobilized and collared using the same method. Age was determined through sagittal crest palpitation and tooth wear. Weather data were obtained from two sources and used to develop models of trapping success for each species. The best model for fisher trapping included the effects of the year and maximum daily air temperature ($w_i = 0.643$). The second best model included the main effects of the amount of precipitation and maximum daily air temperature ($\Delta_i = 3.55$, $w_i = 0.109$. The main effect that was evident was the difference in capture rates between years ($H = 8.345$, $P = 0.004$). The best season for trapping fishers was spring (March – May), and the best individual month was March. The best marten trapping model included the combined effects of year and maximum daily air temperature ($w_i = 0.568$). The second best model, included the effects of year and minimum daily air

temperature, and was similar in rank ($\Delta_i = 0.58$, $w_i = 0.425$). These models, when combined, accounted for > 0.99 of the Akaike weight. We found the best trapping season for marten was winter and the best individual month was March, which agrees with previous studies. Additional fisher capture data from a rabies trap-vaccinate-release program in the St. Lawrence area of eastern Ontario suggested higher capture rates in fall than in summer. Our results suggest that fisher trapping is most successful during spring and fall; however, summer was more successful than expected based on previous studies.

Introduction

Marten (*Martes americana*) and fisher (*M. pennanti*) are two solitary, elusive, forest-dwelling carnivores. When we began our study in 2003, we had some sources of information to guide us in live-trapping fishers (e.g., Loucks 1957, Catton 1958, Arthur 1988, Kohn *et al.* 1993) or marten (Archibald and Jessup 1984, deVos 1952, Francis and Stephenson 1972, Graf 1994); however, useful technical details were sparse. The few documents making recommendations regarding optimal trapping season indicated that the most successful times to trap are during the months of October, March, and April for fisher (Arthur 1988, Kohn *et al.* 1993) and during July and August for marten (Stanfield 1956). Fishers mate during March or April, which results in males making large extraterritorial movements, increasing their susceptibility to capture (Arthur and Krohn 1991). Marten mate during late summer, usually July or August (Strickland *et al.* 1982), which explains the difference in optimal trapping season suggested by Stanfield (1956). Juveniles of both species disperse in autumn, suggesting that this should also be a relatively easier time to capture these species.

The most common bait used for fisher is beaver (*Castor canadensis)*, this been associated with many fisher captures (Loucks 1957, Catton 1958). Other baits that have been used frequently for fishers are meat scraps and carrion (Arthur 1988, Gilbert *et al.* 1997, Leonard 1986, Zielinski *et al.* 2004). Baits such as sardines, fruit jam, meat scraps and beaver appear to be attractive to marten (Archibald and Jessup 1984, Francis and Stephenson 1972, Naylor and Novak 1994). All studies used wire cage traps, but only Loucks (1957) and Arthur (1988) included descriptions of the set that they found to be most effective for fishers. Loucks found the best set to be one where the bait was placed in a hole under the trap with dead wood and evergreen brush covering the trap. Arthur's traps were anchored firmly, baited with meat scraps and commercial lure and then covered with conifer boughs. Literature for marten live capture was marginally more descriptive regarding successful trap sets (Archibald and Jessup 1984, Francis and Stephenson 1972, Graf 1994) indicating that concealment of the trap was pertinent for both capture and safety of the marten. To our knowledge, no published studies for either species have considered the effects of weather on capture success.

We hypothesized that fisher trapping would be most successful during the fall and winter seasons when daily temperatures are low and the amount of precipitation, in the form of snow, increases. There are higher energy demands associated with lower temperatures and movement through snow making fishers more susceptible to traps. Dispersal also occurs in the fall. Increased movement and associated energy demands would increase susceptibility to traps. Spring also could have higher capture rates as movement is increased during the mating season. We expected lower capture success to occur during the summer when daily temperatures are moderate to high and precipitation is in the form of rain. During this season natural food sources will be abundant and necessary travel will be at a minimum. Rain could dampen the scent of the bait and possibly minimize foraging time. We predicted higher capture success for marten during the summer, fall, and winter compared to spring. Marten movement should increase during summer and fall as mating and juvenile dispersal occur increasing the probability of encountering a trap. Capture rates in winter should be higher due to the increased energetic requirements of martens combined with reduced foraging opportunities. We report our observations on the capturing of fisher and marten using live traps and test for variability in catch success caused by season and weather.

Study Area

Algonquin Provincial Park

This study was conducted in Algonquin Provincial Park (48^{o}N, 78^{o}W) from August 2003 – December 2004 (Figure 1.1). Algonquin Park is one of the largest untrapped refugiums in Ontario with 7,725 km^2 of land and water (Strickland and Rutter 2002). The park was established in 1893 for the purpose of forest conservation and the protection of birds and animals (Ontario Ministry of Natural Resources 1998). It is located between Georgian Bay and the Ottawa River in south-central Ontario and lies in the transition zone between the northern boreal forests and the southern temperate forests. Algonquin experiences warm summers and cool winters. It lies on a dome of Canadian Shield bedrock with elevations reaching 587 meters above sea level in the west and 150 meters above sea level in the east. The higher elevations in the west produced a cooler, wetter environment while the east experienced a dryer, warmer climate. The majority of precipitation fell as snow in the winter (Ontario Ministry of Natural Resources 1998). Algonquin Park was made up of a mosaic of habitat types. These habitat types were primarily comprised of hardwood stands but there were also mixedwood and coniferous stands. The west was comprised of tolerant hardwoods such as sugar maple (*Acer saccharum)*, yellow birch *(Betula alleghaniensis)*, and American beech (*Fagus grandifolia)* along with some softwood species such as hemlock *(Tsuga canadensis)* and white pine (*Pinus strobus)*. These made up the mature climax forests of the west. To the east, forests were dominated by pine/intolerant hardwood: white pine, red pine *(Pinus resinosa)*, jack pine *(Pinus banksiana)*, white birch *(Betula papyrifera)*, trembling aspen *(Populus tremuloides)*, and red oak *(Quercus rubra)*. In the lowlands there was mostly balsam fir *(Abies balsamea)* and black spruce *(Picea mariana)* (Strickland 1996).

The areas used for live-trapping consisted of both unharvested and harvested sections of forest. The main Highway 60 corridor was used as the primary travel route as it provided 63 linear kilometers of highly accessible trapping areas. Interior access roads off the main highway were also used.

Eastern Ontario (St. Lawrence)

Fisher capture data were also available from the Ontario Ministry of Natural Resources Rabies Research Unit. A trap-vaccinate-release (TVR) program was conducted in Eastern Ontario in the St. Lawrence River area during 1995 to 2003 (Figure 1.1). This study area of approximately 950 km^2 was delimited to the north by 45°00’, to the south by the St. Lawrence River, to the west by 79°10’, and to the east by 75°15’. The western portion of this study area, which lies on top of a section of the Canadian Shield, was comprised of deciduous and mixed forests that included sugar maple, American beech, bass wood (*Tilia americana*), red maple (*Acer rubrum*), white ash (*Fraxinus americana*), white pine, white spruce (*Picea glauca*), hemlock, eastern white cedar (*Thuja occidentalis*) and balsam fir. The eastern portion of this study area was off the Canadian Shield. In this area human settlement and subsequent development had occurred more heavily. Here agricultural developments such as crops of corn and soy beans as well as pasture dominated the land (Bowman *et al.* in press).

Materials and Methods

Algonquin Provincial Park

Animal capture

Live trapping was conducted in Algonquin Provincial Park from August 2003 through March 2004 and from May – December 2004. Initial trapping locations were selected based on observations of fisher tracks found while snow tracking in winter 2002 / 2003. In addition, other sites were selected that we considered suitable foraging habitat for fishers based on Kohn *et al.* (1993). These were usually sites preferred by fisher prey species, such as snowshoe hares (*Lepus americanus)*, and were usually in conifer stands that had sufficient understory to provide protective cover for snowshoe hares as well as low lying browse. Fishers were targeted because they were considered to be more difficult to catch than marten, and for most of the field project, they comprised our primary study species.

Initially traps were set in clusters of 5-10 traps depending on the microhabitat in each location. Traps were placed generally in a transect or loop using low branches or downed logs to help conceal the trap. In the winter of 2003 / 2004 our approach changed so that traps were set at a lower density, approximately one every kilometer and back approximately 10 meters in the bush (Catton 1958, Loucks 1957). This allowed a greater coverage of area per number of set traps, with the intention of maximizing the number of fisher home ranges covered each trapping night. In the fall of 2003 traps were baited with sardines and a commercial fisher/marten trapping lure was placed on a branch above the trap (Arthur 1988, Graf 1994, Kohn *et al.* 1993). As the temperature dropped beaver meat was used as this did not freeze as easily as did the fish. When venison from the deer hunting season became available, bait bags were made and hung in the tree above or next to the trap. These bags were sometimes used to prebait a site for 2 to 4 days before a trap was set (Catton 1958, Loucks 1957). Beaver or other meat scraps were used for the remainder of the study (Gilbert *et al.* 1997, Kohn *et al.* 1993, Leonard 1996). Bait was tied inside the trap at the back and replaced every 4-5 days or when the bait was missing (Loucks 1957).

Fisher and marten were captured in single-door live traps (Models No. 106 and 108, Tomahawk Live Trap Co., Tomahawk, WI) as these traps were considered to be the most effective in previous studies (Arthur 1988, Coffin 1997, deVos 1952, Gilbert *et al.* 1997, Kohn *et al.* 1993). Dry leaf material was spread on the floor of the trap as well as on the treadle, taking care not to obstruct the trip mechanism in any way, in order to hide the appearance and feel of metal from the animal. The exposed sides of the traps were covered in bark and then soft needled conifer boughs, preferably balsam fir *(Abies balsamea)* were placed over the trap so that each animal was protected from the elements until the traps were checked (Arthur 1988, Francis and Stephenson 1972). Traps were checked at least once daily. If the weather was extreme (hot, cold, or rainy), traps were checked twice daily: early in the morning and early in the evening. In order to minimize human scent, leather gloves were worn when handling the traps. Traps were not treated (i.e., cleaned or waxed) before or after use. Traps were not anchored to the ground.

All fishers were immobilized, weighed, aged, sexed, ear tagged, and radio-collared and subsequently released. All martens caught in November 2003 and from July through December 2004 were processed as described above for fisher. All martens caught prior to this period were simply released.

Capture data

Trapping data from Algonquin were analyzed as captures/ 100 trap nights (TN) by day, month, season, and year to test for variation in capture rates of fishers and martens. Tripped or missing traps were not corrected for. The normality of these data sets was tested using the Shapiro-Wilk test.

A non-parametric Kruskal-Wallis analysis of variance was used to test for differences in monthly capture rates among years. Due to small sample sizes for analysis at smaller intervals Algonquin's capture data were used mostly for descriptive purposes. We looked for differences between capture rates among months and seasons: winter (December, January, February), spring (March, April, May), summer (June, July, August), and fall (September, October, November). Upon rejection of the null hypothesis for the Kruskal-Wallis test, a post-hoc analysis was done to identify significant differences using multiple comparisons on mean ranks for the groups.

Weather data

Weather data obtained from the Algonquin Park Wildlife Research Station's weather station was supplemented with daily precipitation data from Environment Canada's weather station in Dwight, approximately 25 km southwest of the park. The data used coincided with the time of the trapping portion of the Algonquin Study and included readings every half hour for: daily maximum air temperature, daily minimum air temperature, as well as daily precipitation. For our purposes maximum air temperature and minimum air temperature were summarized into daily readings for comparison with daily capture success (1 = capture, 0 = no capture) in an attempt to locate differences in capture occurrences as the environmental conditions fluctuated. Using these variables, models were developed using single variables and combinations of variables. Variable combinations evaluated the main effects and interactions between the variables. Environmental and non-environmental variables were used. Each variable or combination of variables was chosen because we considered them to have a potential effect on capture success. Precipitation measurements (mm) did not discriminate between rain and snow. In the models that we developed, we accounted for this difference by including precipitation with a variable that would describe the season that it was occurring in. The variables that we considered to describe the state of precipitation were season and daily minimum air temperature. Based on the associated category (i.e. winter, spring, summer, fall) or measurement of these two variables we can infer what state of precipitation was recorded.

Normality of the weather data was tested and logistic regression was conducted with the daily binomial capture records as the dependent variable and hypothesized model variables as potential explicative variables. This process served as an initial evaluation for the potential capture success models. The models were assessed by comparing the log likelihood value for each model with that of the constants-only model.

Log likelihood values of all models were used to calculate Akaike's Information Criterion (AIC) for model selection. We evaluated the quality of model performance by looking at the relative distance of model *i* (AIC_i) from the model with the lowest AIC value (AIC_{min}). We calculated the Akaike weights (w_i) for each model and used this normalized relative likelihood value to compare the data (e.g. Anderson and Burnham 2002).

St. Lawrence

Animal Capture

The TVR program used similar trapping methods as the Algonquin study. Live trapping was conducted from June through to October for the years 1995 to 2003 inclusive. The main objective of this intensive program was to vaccinate potential carriers of raccoon rabies to inhibit the spread of rabies in Ontario. Trapping areas were determined by sectioning the study area into cells, which averaged 12 km^2. One hundred traps were set each night within these cells. Wire cage traps were used (Tomahawk 106 and 108, Tomahawk Live Trap Co., Tomahawk, Wisconsin, USA) and baited with sardines (Rosatte *et al.* 2001). Traps were checked daily for captures. Raccoons (*Procyon lotor*) and skunks (*Mephitis mephitis*) were the target species in this study; therefore, fisher captures were noted by workers and the animal was released. Fishers were only marked during the later years of the study. Animal handling procedures for both studies were approved by the OMNR Animal Care Committee.

Capture Data

We summarized the TVR data into monthly capture rates (captures/ 100 TN) for fishers and did not correct for tripped or missing traps. A Shapiro-Wilk test for normality was used to determine if this data set was normally distributed ($P < 0.05$).

The Kruskal-Wallis and median test were used to test for differences in capture rates among months, seasons (winter, spring, summer, and fall), and years. If the null hypothesis was rejected for the Kruskal-Wallis test, a multiple comparisons test done to determine where the differences in capture rates were. Capture rates from this long running program were compared with the capture rates from the current Algonquin Park study in an attempt to draw generalizations about the seasonal variability in fisher capture success.

Results

Algonquin Provincial Park

Fisher Capture

We accumulated 10,375 trap nights (TN) with 56 fisher captures (0.54 captures/ 100 TN), which accounted for 6% of the total captures in our Algonquin study. These 56 fisher captures, included 26 individual radio-collared fishers (10 males: 16 females), the rest of the captures were recaptured collared animals. Fishers made up the third highest percentage of captures after martens and raccoons (Table 1.1). Monthly capture rates for fishers ranged between 0.08 fishers/ 100 TN (October 2003) and 8.40 fishers/ 100 TN (December 2004) (Table 1.2) with an overall mean of 1.39 captures per 100 trap nights.

Marten Capture

We captured a total of 646 marten within the same trapping period resulting in a capture rate of 6.23 captures/ 100 TN. During the entire study, the ratio of martens to fishers captured was approximately 12 : 1 with marten accounting for 69% of all captures (Table 1.1). Martens were captured and radio-collared in November 2003 and from July – December 2005 which resulted in 58 (52 males: 6 females) individuals being collared. The monthly capture rates for marten were considerably higher than for fisher, and ranged between 2.42 marten/ 100 TN (September 2004) and 21.80 martens/ 100 TN (March 2004) (Table 1.2). The overall mean for marten captures was 8.86 captures per 100 trap nights.

Season and Year Effects on Fisher Capture

Due to small sample sizes and for ease of comparability between the two study sites, fisher capture data were pooled between sexes. The fisher data set was not normally distributed (W = 0.64, $P < 0.0001$). There were significant differences in monthly capture rates between years (H = 8.35, $P = 0.004$). The Algonquin data showed a significant difference existed between 2003 and 2004; 2003 had lower monthly capture rates for fisher (mean = 0.16, minimum = 0.08, maximum = 0.25) than did 2004 (mean = 1.94, minimum = 0.14, maximum = 8.40) (Table 1.3). Spring (1.78 fishers / 100 trap nights) had the highest capture rate and winter (1.61 fishers / 100 TN) had the second highest capture rate, regardless of year (Table 1.4). When each year was considered separately, highest capture rates both occurred in winter. March (3.01 fishers / 100 TN) had the overall highest capture rate, and February (2.38 fishers / 100 TN) had the second highest regardless of year (Table 1.4). When we considered individual years the highest capture rate occurred in November (0.25 fishers / 100 TN) in the first year, and in December (8.40 fishers / 100 TN) during the second year.

Season and Year Effects on Marten Capture

Marten capture data were pooled between sexes. The marten data was not normally distributed ($w_i = 0.86$, $P = 0.018$). Capture rates did not differ significantly among years (Table 1.5). The season with the highest capture rate was winter (12.9 martens / 100 TN); each individual year also had the highest capture rate occur in winter (Table 1.4). Among months, the overall highest capture rate for martens occurred in March (21.8 martens / 100 TN). During the first year the capture rate was highest in December (10.0 martens / 100 TN), and during the second year it was highest in March (21.8 martens / 100 TN).

Model of Fisher Trapping Success

Male and female fisher capture data were pooled for these analyses. None of the weather variables were normally distributed (Table 1.6).

We developed 26 logistic regression models using different combinations of weather variables, year, month, and season (Table 1.7). The highest ranked model based on Akaike's Information Criterion ($w_i = 0.64$), included the main effects of year (*t*-ratio = 2.22, $P = 0.026$) and the maximum temperature (*t*-ratio = -3.01, $P = 0.002$,). This was our best fisher trapping model and it accounted for over > 64% of the Akaike weight (Table 1.7). This function indicated that fisher capture rate was positively associated with the year and negatively associated with the maximum temperature. The second ranking model ($\Delta_i = 3.55$, $w_i = 0.11$) incorporated the main effects of precipitation (*t*-ratio = 1.48, $P = 0.140$) and maximum temperature (*t*-ratio = -2.87, $P = 0.004$). This accounted for

10.9% of the Akaike weight. This model suggested that the daily fisher capture rate was positively dependent on the daily amount of precipitation and negatively dependent on the daily maximum temperature. The third ranking AIC model included the single predicting variable maximum temperature ($\Delta_i = 3.57$, $w_i = 0.11$) and accounted for 10.8% of the Akaike weight. The regression coefficient suggested that daily capture rate was negatively dependent on the daily maximum temperature. These three models accounted for > 85% of the Akaike weight, and therefore were considered to be the most influential models for predicting the trapping success of fishers.

Model of Marten Trapping Success

We used the same combinations of environmental and non-environmental variables as were used for the fisher capture analyses (Table 1.8.). Logistic regression and Akaike's Information Criterion showed the highest ranked model ($w_i = 0.57$) included the effects of year (*t*-ratio = -2.94, $P = 0.003$) and daily maximum temperature (*t*-ratio = -2.96, $P = 0.003$). This model accounted for > 56% of the Akaike weight and indicated that the daily marten capture rate was negatively related to year and daily maximum temperature. The second ranked model ($\Delta_i = 0.58$, $w_i = 0.43$) also accounted for a high percentage of the Akaike weight. This involved the effects of the year (*t*-ratio = -2.98, $P = 0.005$), and the daily minimum temperature (*t*-ratio = -2.82, $P = 0.005$). The regression coefficients suggested that the daily marten capture rate was negatively associated with the year and the daily minimum temperature. The McFadden's *rho*-squared for these two models were 0.122 and 0.119 respectively indicating that these particular predictors each had approximately 12% association with the capture of an animal. Together these two models accounted for over 99% of the Akaike weights, making these the two most likely predictor models.

St. Lawrence

Fisher Capture

The trapping periods conducted during 1995 to 2003 for the TVR program resulted in 432,077 trap nights, and a mean effort of 9002 TN/ month (range: 300 – 30224). Although the target species of this program were raccoons and skunks, fishers were also caught. The monthly incidental capture rates of fishers ranged from a low of 0 which was recorded in >1 months and years to a high of 0.466 which occurred in October of 2003 (Figure 1.2).

Season and Year Effects on Fisher Capture

The trappers in the St. Lawrence area did not make a distinction between male and female fisher captures; therefore data was pooled for both sexes. These data were not normally distributed (W = 0.68, P = 0.000).

Throughout the St. Lawrence TVR program, there were significant differences in monthly capture rates among years (H = 15.724, P = 0.047). There were no differences in the capture rates among trapping seasons (spring, summer, fall), or individual months (Table 1.3). The median test indicated that there were differences in capture rate at all three grouping levels (Table 1.3). There were more captures below the overall median (median = 0.044) in July and more captures above the median in October (χ^2 = 14.44, P = 0.044) when comparing among months. There were more capture rates below the median during the summer and more capture rates above the median during fall (χ^2 = 9.50, P =

0.009). In 1997 there were more capture rates below the median and in 2000 and 2003 there were more capture rates above the median ($\chi^2 = 8.00$, p = 0.045) when comparing among years.

The greatest difference (*P* = 0.032) in monthly capture rates (captures/ 100 TN) from the St. Lawrence area came between the years 2003 (mean = 0.218, minimum = 0.091, maximum = 0.466) and 1997 (mean = 0.015, minimum = 0.000, maximum = 0.037).

Discussion

Fisher Capture

Algonquin Provincial Park

Our overall capture rates for fishers were 0.16 / 100 TN for 2003 and 0.98 / 100 TN for 2004. This indicated either that fisher abundance or trappability increased during 2004, or that our trapping methods improved. We suggest that the latter explanation was most likely. Clusters of traps, which we used in 2003, were less effective for fisher capture compared to placing traps at lower densities. Our effort was increased by the high density of traps but our capture rate was lower. Based on observations we found the most effective set to be one that was baited with beaver and heavily covered with bark and conifer boughs; we used this set more often during 2004, after discussions with local fur trappers. Our overall total capture rate of 0.54 fishers / 100 TN was considerably lower than that of Loucks (1957) and Catton (1958). Although they too found beaver bait to be successful in and around Algonquin Park, they had much higher capture rates of 3.50 / 100 TN and 10.9 / 100 TN respectively. Our lower capture rates could have been a result of factors such as greater prey availability, making fishers less susceptible to being trapped or because of lower fisher abundance since the 1950s. Recent genetic research has suggested that Algonquin Park is not productive habitat for fishers (Carr 2005).

The variance of capture rates between years potentially overshadowed the effects of season and month on capture rates. We demonstrated the ability to trap fishers in all months and seasons through out the year. Previous studies have demonstrated the best seasons for fisher capture to be fall and spring. Although our experience partially agrees with this – our highest overall seasonal capture rate was in the spring – we have also shown that fishers may also be successfully trapped outside of these seasons. Our highest monthly capture rate occurred in December (8.4 captures per 100 TN), which supported our hypothesis that snow and low temperatures would increase our capture success. This was most likely overshadowed by the low capture rates of the first winter in our overall monthly capture rate. Our low summer capture rates also supported with our prediction that higher temperatures would reduce capture success. This too was likely overshadowed by the overall lower capture rates in the first year.

St. Lawrence

The TVR program was aimed at trapping rabies vectors for vaccination. The target species for this program were raccoons and skunks so sardines were used as bait. In Algonquin Park, we found sardines to be less effective bait for fishers than beaver, moose, or venison, and the use of sardines in the TVR program likely contributed to the relatively low capture rates of fishers in the program.

The large difference in capture rates is not indicative of the density of fishers in these two areas. The average capture rate for fishers in the TVR program was 0.062

captures / 100 TN, whereas in the Algonquin study it was much higher at 1.385 captures / 100 TN. However, a recent study has suggested that fisher densities in the St. Lawrence area are high at 32.6 / 100 km² assuming occupancy of suitable habitat (Koen 2005). Furthermore, we suspect that the fishers in Algonquin are present at low densities especially compared to the 1950s. Low capture rates in St. Lawrence can be explained by the fact that fishers were incidental captures.

In 1997 there were two months out of five where the trappers for the TVR program did not catch any fishers. This explains the low annual capture rate in 1997; other months had < 0.04 captures / 100 TN. In 2003, when fisher abundance was higher (Bowman *et al.* in press) capture success was considerably higher: all months had capture rates > 0.09 captures / 100 TN. There were no differences in capture rate among seasons or months within this data set. This could be because the effects of inter-year variation overrode any seasonal or monthly effect that otherwise may have been detected. Seasonal and monthly differences may not have been evident because of the monthly capture rates themselves. The St. Lawrence data had two seasons with > 3 monthly capture rates: summer (24), and fall (21). Winter and spring had 4 monthly capture rates each. The less sensitive median test however, did indicate that significantly more captures occurred in fall than in summer. This was consistent with our hypothesis that more fishers would be caught as juveniles dispersed. As they left their mother's home range in search of their own, their chances of encountering a trap increased. Food availability would also be high in the summer lowering the chance that they would be attracted to the bait. This hypothesis was supported by the median test. This test resulted in October having higher capture rates than July. The TVR program is not conducted in the winter, and rarely in the spring, therefore there were insufficient data to assess these seasons.

Model of Fisher Trapping success

The best logistic regression model for fishers included the variables year and daily maximum air temperature. The effects of these two variables described the higher capture rates of our second trapping year as well as the negative relationship between capture success and temperature. Temperature is an important variable as it is directly related to the energy requirements of the fisher. More energy is required to maintain a stable body temperature as the temperature decreases, which means increasing food intake. Trap bait provides an alternative food source. Increased movement of prey as well as the fisher as the temperatures decrease would also increase the likelihood of the fisher encountering a trap.

The second best model included the effects of precipitation and daily maximum air temperature. The regression indicated a positive relationship between the daily precipitation and capture success and an inverse relationship between daily maximum air temperature and capture success; this too supported our hypothesis of increased capture success with the occurrence of snow and low temperatures. As the recorded amount of precipitation increased and the daily maximum temperature decreases, we can infer that this reflects the change of rain to snow. Therefore as the rain changed to snow and then as the snow persisted we saw increased capture success. There are high energetic demands associated with winter and cold conditions as fishers have to use more energy to move through snow and to keep their body temperature stable (Krohn *et al.* 1995). This increased need for energy intake could make the fisher more attracted to bait.

The third ranked model included only daily maximum air temperature. The top three models combined had a McFadden's *rho*-value of < 0.07, demonstrating the high variability in catch success.

Marten Capture

Our capture rates for marten were considerably higher than for fisher. Our overall yearly capture rate for marten in 2003 was 5.74 / 100 TN and in 2004 it was 6.78 / 100 TN. There was no significant difference between these two rates. High capture rates were most likely due to a high density of marten within the study area. This abundance of marten could also have led to a decrease in food availability making them more attracted to trap bait. Martens were caught on a daily basis regardless of the trap set that was used. Our capture rates were much higher than those of Loucks (1957) or Francis and Stephenson (1972). Loucks (1957) also used beaver and other meats but had a capture rate of only 1.21 / 100 trap nights. Francis and Stephenson (1972) had a capture rate of 1.60 / 100 trap nights while using jam and fish oil for bait. The differences in capture rates between our study and the previous studies may have been due to an increase in marten abundance compared to previous investigations in the same area. Marten abundance could have increased as fisher numbers decreased in the area. Powell and Zielinski (1983) suggested that coexistence was only temporary for 2 or more *Mustela* species, and by inference, *Martes* species. They proposed that the species will fluctuate inversely with a series of extinctions and reoccupations perhaps initiated by declining prey numbers. Factors determining which population persists are predation abilities, population sizes, and reproductive adaptations (Powell and Zielinski 1983).

There were no significant differences in capture rates between years. Winter produced the highest seasonal capture rates for both years suggesting that this season is best for catching martens. This concurs with our prediction that winter would bring increased trapping success most likely due to higher energetic needs as temperatures dropped. Even though March held the highest capture rate among months, it was overshadowed by the low capture rate in May (3.4 marten / 100 TN) that resulted in spring having the second highest seasonal capture rate. Spring had higher capture success (12.1/ 100 TN) than we anticipated. This could have been a result of low temperatures and increased amounts of precipitation, which would result in a high energy need and may have made marten more likely to take the bait. The capture rate of marten did not vary significantly when either trap set or bait was altered. This suggests that martens were much easier to catch than fishers, or simply more abundant. Our results do not support previous recommendations of trapping in July and August (Stanfield 1956).

Model of Marten Trapping Success

The best model of marten trapping success combined the effects of the year and the daily maximum air temperature (Table 1.8). Although there was no significant difference in the marten capture rate between years, there was negative association between years and capture success. There was also a negative association between daily maximum air temperature and capture success. This relationship with lower temperatures supported our hypothesis of increased capture rates during colder temperatures. As overall temperatures decrease, marten energy demands increase, which in turn increases their need to actively search for prey. Increasing movement would render them more likely to encounter a trap. Higher maximum temperatures can be associated with primary productivity which supplies an additional food source for marten decreasing their vulnerability to bait causing capture success to decline.

The second ranked model for martens included the effects of the year and the minimum temperature. This again suggested that there were higher capture rates in the first year. Lower temperatures were again associated with higher capture rates, supporting our hypothesis, and emphasizing marten's increase in energy demands as the temperature drops. Low temperatures in 2003 appear to be responsible for our marten trapping success in this study. These two models accounted for over 98% of the Akaike weight; however the *rho*-value was approximately 0.12 for each of the models. The weak association in this case was most likely due to the consistently high capture rates over all years, months and seasons regardless of environmental conditions. Martens were caught with relative ease throughout the study.

Overall our results supported our hypotheses regarding the optimal trapping conditions for both fishers and martens. The most important variable for both species was temperature. Fisher and marten captures were associated with low temperatures. From our results we were able to infer the best season(s) to trap fishers and martens; however, we have also demonstrated that it is possible to successfully trap either species in any season under a variety of environmental variables.

Recommendations

Fisher

We had a total of 56 fisher captures recorded during our field work in Algonquin Provincial Park using Tomahawk cage traps. We had the most success when beaver meat was used for bait and the trap was thoroughly concealed. In our study we found spring to be the best season overall for trapping fisher, and in particular March to be the best overall month. The St. Lawrence study, which was conducted mainly in summer and fall, revealed that fall was better than summer, and more specifically that October was significantly better than July. Models including weather data further suggested that cold weather and high precipitation amounts were positively associated with fisher captures. Taken together, our results suggest that fisher trapping should be most successful during spring and fall. However, trapping during winter was quite successful and summer was more successful than expected based on previous studies.

Marten

There were a total of 646 marten captures in 2003-2004 using both models (No. 106 and 108) of Tomahawk traps. Martens did not seem to demonstrate a preference towards a particular bait or trap set. They were captured almost every day using either sardines or meat scraps. We found winter to be the best overall season for trapping, while March was the best individual trapping month. Caution should be applied when trapping in extreme cold – heavy insulation of the trap is necessary in these cases. Our trapping models suggested that lower temperatures caused higher capture rates, therefore we would suggest that live trapping efforts be focused during the months in which daily maximum temperatures are low.

Acknowledgements

Thank you to following people for your contributions and persistence: Richard Shalla, Amelia Argue, Shaun McIntosh, Ken Mills, Derek Potter, Carrie Sadowski, Richard and

Jane Tully, Jaime Mills, and Ryan Collis. Our trapping and handling protocol was approved by the Ontario Ministry of Natural Resources Animal Care Committee.

Literature Cited

Anderson, D. and C. Burnham. 2002. Avoiding pitfalls when using information-theoretic methods. J. Wildl. Manage. 66: 912 – 918.

Archibald, W.R., and R.H. Jessup. 1984. Population dynamics of the pine marten (Martes Americana) in the Yukon Territory. In: R. Olsen, R. Hastings and F. Geddes, eds. Northern ecology and resource management. University of Alberta Press. Edmonton, Alberta. Pp 81 – 97.

Arthur, S.M. 1988. An Evaluation of techniques for capturing and radio collaring fishers. Wildl. Soc. Bull. 16: 417 – 421.

Arthur, S.M., and W.B. Krohn. 1991. Activity patterns, movements, and reproductive ecology of fishers in southcentral Maine. J. Mamm. 72(2): 379 – 385.

Bowman, J., Donovan, D., and Rosatte, R. C. In press. Numerical response of fishers to synchronous prey dynamics. J. Mamm. 0: 000 – 000.

Catton, W.R. 1958. Live Fisher Trapping, Pembroke District, 1958. Fish and Wildlife Management Report 43. Dep. Lands and forests.

Carr, D. 2005. Genetic structure of a recolonizing population of fishers (*Martes pennanti*). MSc Thesis. Trent University, Peterborough, ON.

Coffin, K.W. 1997. Interactions among marten prey availability, vulnerability, and habitat structure. In: G. Proulx, H.N. Bryant, and Paul M. Woodard, eds. Martes: taxonomy, ecology, techniques, and management: proceedings of the Second International Martes Symposium. Edmonton, Provincial Museum of Alberta. Pp 199 – 210.

deVos, A. 1952. Preliminary live-trapping studies of marten. J. Wildl. Manage. 16(2): 207 – 214.

Francis, G.R., and A.B. Stephenson. 1972. Marten ranges and food habits in Algonquin Provincial Park, Ontario. Wildlife Report 91.

Gilbert, J.H., Wright, J.L., D.J. Lauten, J.R. Probst. 1997. Den and rest-site characteristics of American marten and fisher in northern Wisconsin. In: G. Proulx, H.N. Bryant, and Paul M. Woodard, eds. Martes: taxonomy, ecology, techniques, and management: proceedings of the Second International Martes Symposium.Edmonton, Provincial Museum of Alberta. Pp 135 – 145.

Graf, R.P. 1994. Experimental overharvest of martens, (*Martes americana*), in Northwest Territories, Canada. XXI IUWB.

Koen, E. 2005. Home range, population density, habitat preference, and survival of fishers *(Martes pennanti)* in eastern Ontario. M.Sc Thesis. Ottawa – Carleton Institute of Biology. Ottawa, Ontario.

Kohn, B.E., N.F. Payne, J.E. Ashbrenner, and W.A. Creed. 1993. The fisher in Wisconsin. Technical Bulletin No. 183. Dept. of Natural Resources.

Krohn, W.B., K.D. Elowe, and R.B. Boone. 1995. Relations among fishers, snow, and martens: Development and evaluation of two hypotheses. For. Chron. 71(1): 97 – 105.

Leonard, R.D. 1986. Aspects of reproduction of the fisher, *Martes pennanti*, in Manitoba. Can. Field Nat. 100: 32 – 44.

Loucks, M.G. 1957. Live trapping of fisher in Algonquin Park, winter of 1957.

Naylor, B.J., and M. Novak. 1994. Catch Efficiency and Selectivity of Various Traps and Sets Used for Capturing American Martens. Wildl. Soc. Bull. 22: 489 – 496.

Ontario Ministry of Natural Resources. 1998. Algonquin Provincial Park Management Plan. Queen's Printer for Ontario. Ontario, Canada. Pp 83.

Powell, R.A. and W.J. Zielinski. 1983. Competition and coexistence in mustelid communities. Acta. Zool. Fennica. 174: 223 – 227.

Rosatte, R., D. Donovan, M. Allen, L.A. Howes, A. Silver, K. Bennett, C. MacInnes, C. Davies, A. Wandeler, and B. Radford. 2001. Emergency response to raccoon rabies introduction into Ontario. J. Wildl. Dis. 37(2): 265 -279.

Standfield, R.O. 1956. Trapping, handling, and marking live marten. Ontario Department of Lands and Forests Division of Research. Section Report (Wildlife) No.9.

Strickland, M.A., C.W. Douglas, M. Novak, N.P. Hunziger. 1982. Marten. In: J.A. Chapman, and G.A. Feldhamer, eds. Wild mammals of North America biology, management, and economics. Johns Hopkins University Press. USA. Pp 599 – 612.

Strickland, D. 1996. Trees of Algonquin Provincial Park. The Friends of Algonquin Provincial Park and the Ontario Ministry of Natural Resources. Whitney, ON.

Strickland, D. and R.J. Rutter. 2002. Mammals of Algonquin Provincial Park. The Friends of Algonquin Provincial Park. Whitney, ON.

Table 1.1 Species composition of the total captures during the 2003/2004 field seasons in Algonquin Provincial Park. Capture rate is in captures/ 100 trap nights (TN)

Species	**Common Name**	**Total Captures**	**Capture Rate**
Martes pennanti	Fisher	56	0.54
Martes americana	American Marten	646	6.23
Procyon lotor	Raccoon	179	1.73
Vulpes vulpes	Red Fox	17	0.16
Ursus americanus	Black Bear Cub	9	0.09
Lepus americanus	Snowshoe Hare	9	0.09
Mustela vison	American Mink	5	0.05
Cyanocitta cristata	Blue Jay	4	0.04
Perisoreus canadensis	Grey Jay	3	0.03
Corvus corax	Common Raven	3	0.03
Tamiasciurus hudsonicus	Red Squirrel	3	0.03
Bonasa umbellus	Ruffed Grouse	1	0.01

Table 1.2 Monthly capture rates (animals / 100 trap nights) in Algonquin for marten and fisher. Separate monthly rates for 2003 and 2004 as well as an overall monthly rate for combined years.

Month	**Marten 2003**	**Marten 2004**	**Marten Total**	**Fisher 2003**	**Fisher 2004**	**Fisher Total**
January	——	13.08	13.08	——	1.54	1.54
February	——	18.25	18.25	——	2.38	2.38
March	——	21.8	21.8	——	3.01	3.01
April	——	——	——	——	——	——
May	——	3.38	3.38	——	0.68	0.68
June	——	3.67	3.67	——	0.46	0.46
July	——	2.96	2.96	——	0.49	0.49
August	3.05	4.22	3.56	0.11	0.14	0.12
September	6.69	2.42	5.26	0.09	0.35	0.17
October	4.82	9.11	5.95	0.08	0.54	0.28
November	5.46	19.32	7.25	0.25	3.41	0.66
December	10.00	13.45	10.48	0.14	8.40	1.28

Table 1.3 Descriptive statistics showing the mean monthly fisher capture rate (fisher/100TN), standard error of the mean, minimum and maximum capture rate (fisher/100TN), and the standard deviation. Shapiro-Wilk's (W) test for normality describes the distribution of each data set. Kruskal-Wallis (H) and Median test (χ^2) test for significant differences in ranked data and Multiple comparisons show where this difference occurs.

		St. Lawrence			**Algonquin**
		Month	**Season**	**Year**	**Year**
Descriptive Statistics	**N**	48.00			16.00
	Mean	0.06			1.39
	Std. Err	0.01			0.54
	Min	0.00			0.08
	Max	0.47			8.40
	SD	0.08			2.16
Normality	**Shapiro-Wilk**	0.69			0.64
	p	0.00			0.00
Kruskal-Wallis	**H**	10.50	5.72	15.72	8.35
	k	8.00	3.00	9.00	2.00
	N	48.00	48.00	48.00	16.00
	p	0.16	0.06	0.05	0.00
Median Test	**Overall Med.**	0.044	0.044	0.04	0.48
	Chi-square	14.444	9.500	15.81	7.27
	df	7.000	2.000	8.00	1.00
	p	0.044	0.009	0.05	0.01
	<= Med	July	Summer	1997	First
	> Med	October	Fall	2000/2003	Second
Multiple Comparison P-Values	**Sig Diff**	No Sig	No Sig	2003/1997	Second/First
	P			0.03	0.00

Table 1.4 Seasonal capture rates (animals / 100 trap nights) in Algonquin for marten and fisher. Seasonal rates for 2003 and 2004 as well as an overall seasonal rate for combined years. Seasons include winter (December, January, February), spring (March, April, May), summer (June, July, August), and fall (September, October, November).

Season	Marten 2003	Marten 2004	Marten Total	Fisher 2003	Fisher 2004	Fisher Total
Winter	12.82	13.45	12.87	0.96	8.4	1.61
Spring	—	12.1	12.10	—	1.78	1.78
Summer	3.05	3.54	3.41	0.11	0.38	0.31
Fall	5.57	7.53	6.12	0.18	0.84	0.35

Table 1.5 Descriptive statistics for Algonquin data showing the mean monthly marten capture rate (marten/100TN), standard error of the mean, minimum and maximum capture rate (marten/100TN), and the standard deviation. Shapiro-Wilk's (W) test for normality describes the distribution of the data set. Kruskal-Wallis (H) and Median test (χ^2) test for significant differences in ranked data and multiple comparisons show where this difference occurs.

		Algonquin Year
Descriptive Statistics	**N**	16.00
	Mean	8.86
	Min	2.42
	Max	21.80
	SD	6.46
Normality	**Shapiro-Wilk**	0.86
	p	0.02
Kruskal-Wallis	**H**	0.26
	k	1.00
	N	16.00
	p	0.61
Median Test	**Overall Med.**	6.08
	Chi-square	0.30
	df	1.00
	p	0.59
	<= Med	
	> Med	
Multiple Comparisons p-values	**Sig Diff**	No Sig
	P	

Table 1.6 Shapiro-Wilk's (W) test for normality of Algonquin weather data. Weather variables included minimum and maximum temperature (°C), amount of precipitation (mm), and the number of lapsed days since the last precipitation occurred.

		Min. Temp.	**Max. Temp.**	**Precip.**	**Lapsed Days**
Normality	**Shapiro-Wilk (W)**	0.736	0.786	0.731	0.648
	p	0.000	0.000	0.000	0.000

Table 1.7 Logistic Regression (LL) and Akaike's Information Criterion results (AIC) for the candidate fisher trapping models listed in order of fit. K is the number of parameters in model, $\triangle_i$ is the distance between model i and the best fitting model, and w_i is the calculated AIC weight of the model. Models with an asterisk (*) show interaction between two variables and an addition sign (+) indicates main effects of two variables.

Model	**LL**	**K**	**AIC**	$\triangle_i$	w_i
Year + Max	-99.87	4	207.74	0.00	0.643
Precip + Max	-101.65	4	211.29	3.55	0.109
Max	-102.66	3	211.31	3.57	0.108
Year + Min	-102.09	4	212.18	4.44	0.070
Precip + Min	-102.89	4	213.78	6.04	0.031
Min	-104.40	3	214.80	7.06	0.019
Precip * Winter	-105.36	3	216.73	8.99	0.007
Min * Winter	-106.19	3	218.38	10.64	0.000
Max * Summer	-106.26	3	218.53	10.78	0.000
Precip * Min	-106.34	3	218.68	10.93	0.000
Precip * Max	-107.14	3	220.27	12.53	0.000
Precip + Winter	-106.77	4	221.55	13.81	0.000
Winter	-108.03	3	222.06	14.32	0.000
Year + Precip	-107.04	4	222.08	14.34	0.000
Year	-108.55	3	223.09	15.35	0.000
Days + Winter	-107.67	4	223.34	15.59	0.000
Year * Precip	-108.92	3	223.84	16.10	0.000
Month	-109.65	3	225.29	17.55	0.000
Summer	-109.86	3	225.72	17.98	0.000
Days * Winter	-110.06	3	226.12	18.38	0.000
Fall	-110.39	3	226.77	19.03	0.000
Spring	-110.47	3	226.93	19.19	0.000
Precip	-117.01	3	240.02	32.27	0.000
Days * Precip	-117.42	3	240.84	33.10	0.000
Precip + Days	-116.70	4	241.41	33.66	0.000
Days	-117.76	3	241.51	33.77	0.000

Table 1.8 Logistic Regression (LL) and Akaike's Information Criterion results (AIC) for the candidate marten trapping models listed in order of fit. K is the number of parameters in model, $\triangle_i$ is the distance between model i and the best fitting model, and w_i is the calculated AIC weight of the model. Models with an asterisk (*) show interaction between two variables and an addition sign (+) indicates main effects of two variables.

Model	**LL**	**K**	**AIC**	$\triangle_i$	w_i
Year + Max	-88.09	4	184.17	0.00	0.568
Year + Min	-88.38	4	184.76	0.58	0.425
Precip + Max	-93.87	4	195.73	11.56	0.002
Max	-94.50	3	195.00	10.83	0.003
Precip + Min	-94.79	4	197.57	13.40	0.001
Min	-95.05	3	196.09	11.92	0.001
Min * Winter	-97.26	3	200.52	16.35	0.000
Precip * Min	-98.50	3	203.01	18.83	0.000
Precip * Max	-98.68	3	203.35	19.18	0.000
Max * Summer	-99.94	3	205.87	21.70	0.000
Year + Precip	-102.09	4	212.18	28.01	0.000
Year	-102.71	3	211.42	27.25	0.000
Days + Winter	-106.15	4	220.29	36.12	0.000
Precip + Winter	-106.91	4	221.83	37.65	0.000
Winter	-107.13	3	220.25	36.08	0.000
Precip + Days	-107.35	4	222.70	38.53	0.000
Spring	-107.39	3	220.77	36.60	0.000
Days * Winter	-107.57	3	221.13	36.96	0.000
Days	-107.71	3	221.41	37.24	0.000
Year * Precip	-107.90	3	221.79	37.62	0.000
Summer	-108.50	3	223.01	38.83	0.000
Days * Precip	-108.64	3	223.28	39.11	0.000
Precip	-108.68	3	223.36	39.18	0.000
Fall	-108.82	3	223.66	39.48	0.000
Precip * Winter	-108.90	3	223.79	39.61	0.000
Month	-108.90	3	223.80	39.62	0.000

Figure 1.1 Map of study areas. Area A: Algonquin Provincial Park, Ontario, Canada. Area B: Ontario Ministry of Natural Resources rabies trap-vaccinate-release study area, St. Lawrence region, Ontario, Canada.

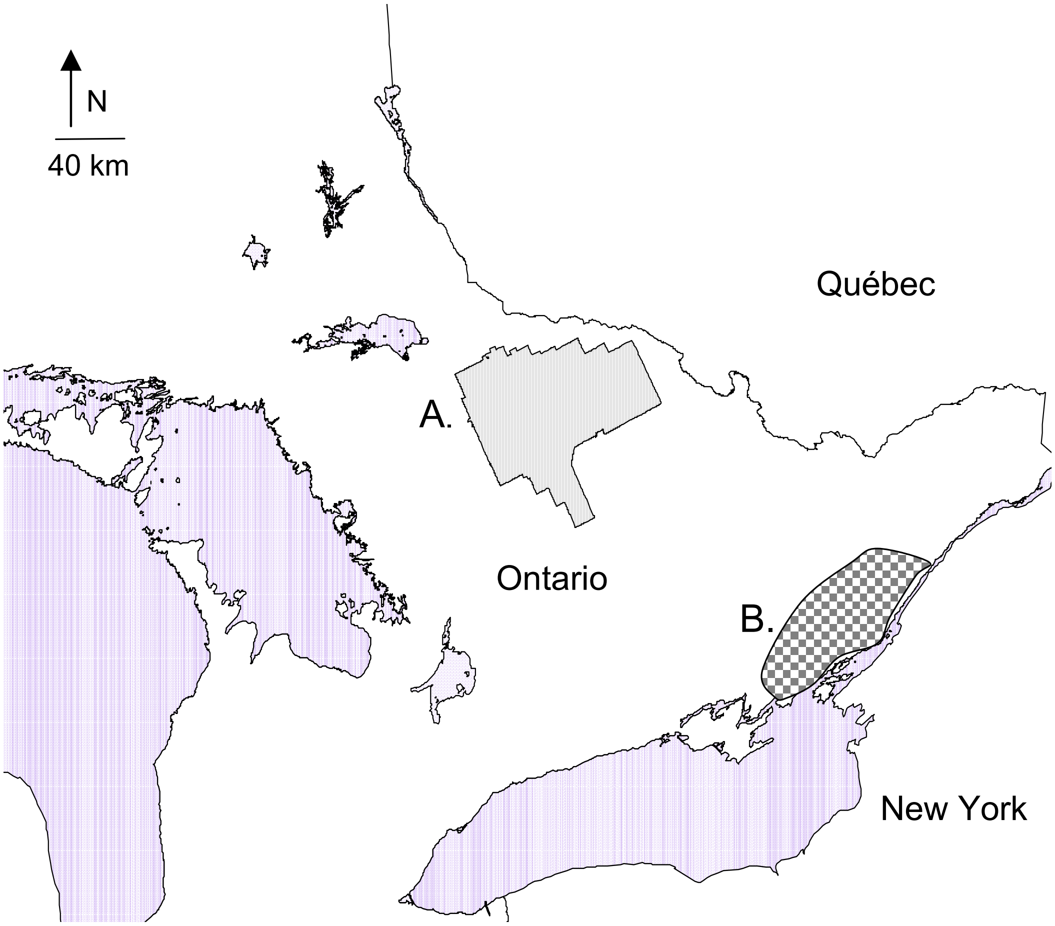
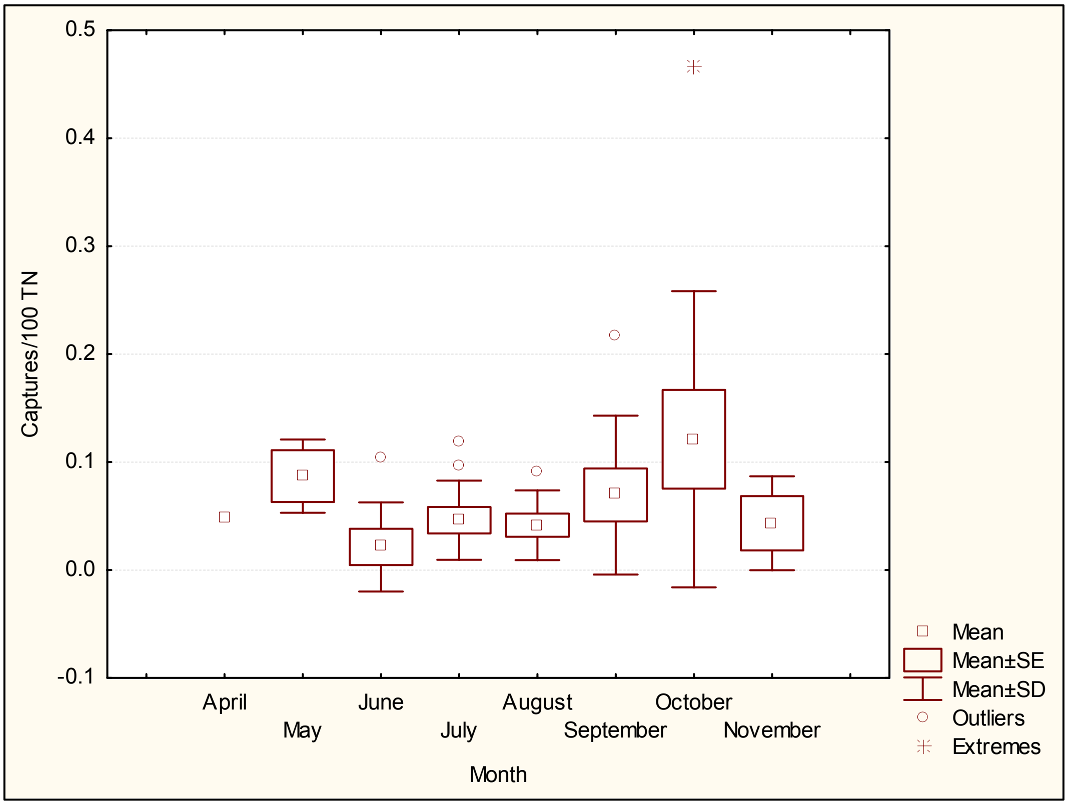

N
40 km
Québec
A.
Ontario
B.
New York

Figure 1.2 Mean (± SE mean) monthly fisher capture rates from the trap-vaccinate-release rabies program in the St. Lawrence River area, Ontario.


0.5
0.4
0.3
0.2
0.1
0.0
-0.1
Captures/100 TN
April
May
June
July
August
September
October
November
Month
Mean
Mean±SE
Mean±SD
Outliers
Extremes

Hierarchical Habitat Selection of fishers in an untrapped refugium; Algonquin Provincial Park, Ontario.

Abstract

We evaluated habitat selection and suitability of fishers (*Martes pennanti*) in Algonquin Provincial Park, Ontario, Canada. We used ground and aerial telemetry to estimate home range size for 9 adult females (mean 29.9 ± 5.2 km^2) and 2 adult males (30.06 and 39.13 km^2). We then tested the validity of a fisher habitat suitability model (Allen 1983) using telemetry data. We compared habitat use and availability data at two hierarchical levels: among and within home ranges. A standardized resource selection function (RSF) was used to evaluate fishers' selection of 3 different habitat variables as defined by Allen: mean overstory tree diameter at breast height, tree canopy diversity, and percent of overstory composed of deciduous species. Our RSFs were correlated with Allen's HSI scores among home ranges; larger RSFs were associated with higher HSI scores (r_s = 0.414, p = 0.005). There was no correlation however, between the RSFs and HSI scores within home ranges. Further analysis at this level demonstrated selection by fishers for old tolerant hardwood stands. Average HSI scores for the overall study area, home ranges and all telemetry locations pooled were low (0.254, 0.273, and 0.286 respectively) indicating that optimal habitat as rated by Allen for fishers was not abundant at either level.

Introduction

Fishers (*Martes pennanti*) have long been associated with dense coniferous and mixed forests throughout Canada and parts of the United States (Clem 1977, Coulter 1966, deVos 1952, Powell 1979), although they are not restricted to these forest types. Prior to European settlement their distribution was extensive ranging from the continuous forests of Canada and the Northern U.S. south through the Appalachians and the Pacific coast mountain ranges (Gibilisco 1994). Throughout their range fisher populations declined sharply throughout the early 1900s due to over harvesting and habitat destruction resulting from logging and human settlement. Although they faced extinction throughout most of their range, their populations began to recover as the logging boom curtailed and harvests were restricted (Powell and Zielinski 1994). As abandoned farmlands began returning to forest and remnant populations recovered, successful reintroductions occurred and viable populations returned to much of the extensively forested areas of Eastern North America as well as areas in the west (Powell and Zielinski 1994).

Fisher's distribution is thought to be opportunistic. Their numbers are thought to be governed predominantly by prey abundance (Clem 1977). Though forest cover seems to be an undisputed habitat requirement (Arthur 1989b, Jones and Garton 1994, Kelly 1977, Powell 1994b, Thomasma *et al.* 1991), there is debate over suitable forest composition. Douglas and Strickland (1987) suggested that fishers can inhabit any forested area given a suitable prey base. Similarly, Arthur *et al.* (1989b) hypothesized that habitat use was variable because of the diversity in the fishers' diet. Buskirk and Powell (1994) suggested that it is not only prey availability but the physical structure of the forest that explains fishers' habitat use. Mature to climax stage forest is thought to provide the highest habitat quality as it provides the best cover and denning sites (deVos 1951). Fishers have been identified, for the purpose of forest management, as an umbrella species for those species requiring late successional forest habitats. If coniferous forests are managed on the basis of the fishers' large habitat requirements, protection of habitat for many other vertebrates will also be addressed (Buskirk 1992).

Habitat Selection

Habitat is defined as the resources and conditions that are necessary for an area to be used by a given species (Garshelis 2000). Animals make hierarchical selection decisions about the ability of a habitat to provide the necessary life requisites (Johnson 1980). These decisions ultimately define a settlement location (George and Zack 2001). Rettie and Messier (2000) suggested that there was a direct relationship between limiting factors of a population and the hierarchical level (e.g., forest and stand) at which an animal demonstrates selection in order to reduce the effects of such factors. They suggested that the most influential limiting factors occurred at higher levels and less influential factors occurred at lower levels. Therefore, suitable habitat for a species is best described by the environmental features that affect their fitness, a relative measure of survival and number of surviving offspring (Fisher 1930). It is assumed that animals are able to choose and utilize the most suitable habitats within the available range in order to maximize their fitness (Farmer *et al.* 1982, Garshelis 2000, Powell 2004). According to Rettie and Messier (2000) individual fitness in Caribou is most affected by decisions made at coarse spatial and temporal scales. Caribou are thought to avoid factors that have the highest potential to decrease fitness at these scales.

Habitat suitability models inferentially model fitness. The purpose of the model is to demonstrate how the critical resources (i.e., rest sites, food and cover), and costs (i.e., travel and predation risk) associated with a given area will affect the fitness of an animal. The main assumption behind the suitability model is that the model output is correlated with population fitness (Brennan *et al.* 1986). As habitat suitability values increase, so do the corresponding potential fitness levels.

Fisher Habitat Suitability Indices

Habitat suitability has been defined as the potential of an area to support a particular species and has been represented by an index ranging from 0 (completely unsuitable) to 1 (suitable) (Brennan *et al.* 1986), suitability corresponds with the fitness of an animal. For fishers the suitability of a forest stand has been described by stand characteristics that have been or are able to be measured and modeled. Model variables are assessed either through literature review or direct observation. Fisher suitability indices are then assigned to stand characteristics in a way that putatively reflects contribution towards individual fitness.

Allen (1983) first developed the HSI model for the fisher for the U.S. Fish and Wildlife Service for the purpose of future impact assessment and habitat management throughout the fisher's range. This model was constructed using hypotheses regarding optimal fisher habitat. The model was intended for year round use throughout the fishers' geographic range. Allen assumed habitat selection for fishers was governed primarily by prey availability and foraging strategies. He also assumed that winter cover was the most restrictive habitat requirement for fishers, therefore winter cover was the life requisite of the model. The model assumed the preferred stand-level habitat for fishers was mature to climax dense coniferous and coniferous/deciduous forests, which would have den site selection and adequate cover for fishers as well as prey. He described poor fisher habitat as hardwood stands, open areas and areas with low amounts of canopy cover such as recently burned or logged forest. Allen included stand-level habitat variables: percent tree canopy closure, average diameter at breast height (dbh) of overstory trees, tree canopy diversity, and the percent of the overstory canopy comprised of deciduous species. He developed suitability indices for each of these variables (Fig. 2.1) and combined these suitability index values to calculate a life requisite value for the fisher. The equation reflected each variable's perceived contribution towards overall habitat suitability. Individual plots/stands were meant to be assigned an HSI value. A weighted HSI (average) was calculated that was applicable to the extent of the study area. HIS models have been criticized because their validity is rarely tested (Cook and Irwin 1985).

Validity Tests

Thomasma *et al.* (1991) tested Allen's model in Michigan, U.S., and it was not invalidated by their data. They used track observations as an indicator of fisher presence. Habitat measurements for all of Allen's model variables were made where fisher tracks were found (used habitat) as well as randomly selected plots (available habitat). Allen's selection index values were applied to the measured variables and HSI scores were calculated. They found that overall there was a greater number of high HSI scores in the used plots versus the available plots, and reported that the mean HSI score for the used plots was greater than that of the available plots. They did not invalidate the HSI model since fishers did use habitats with higher HSI values more often than expected relative to availability; therefore Thomasma *et al.* (1991) recommended use of the existing HSI model for fishers. Powell (1994a) also tested Allen's model in Michigan, U.S., using snow tracking and habitat data. He found that fishers spent more time and traveled farther distances in habitats with higher suitability. Badry *et al.* (1997) evaluated Allen's model variables for applicability in the Aspen parkland of Alberta, Canada. They found that habitat selection by radio collared fishers did not did follow Allen's predictions. Deciduous forest was used more than its relative availability, and coniferous forest was used less than its availability. However, they did find that fishers used stands that had a high diameter at breast height (dbh) and a diverse canopy.

Recognizing the importance of observational scale in the analysis of habitat selection, we chose to test the suitability of Allen's model for different orders of selection (Johnson 1980). This model has not been tested in this way. The previous tests were all done at the third-order selection level using individual locations. We hypothesized that examination of habitat selection at two levels; among home ranges (Johnson second-order selection) and within home ranges (Johnson third-order selection), will demonstrate the overall usage of mature to old growth coniferous and mixed forests (predominantly coniferous) in a proportion significantly greater than their availability. We expect this because higher amounts of canopy cover and prey availability are associated with this type of forest compared to younger forest stands dominated by hardwood species. Accepting Allen's hypothesis that winter cover is the most limiting habitat variable for the fisher, fishers should demonstrate preference for mature to old growth conifer dominated stands (high HSI values) at the coarse scale analysis (among home ranges). At a finer scale analysis (within home ranges) I expected that habitat use will be more variable than at the among home range scale. At this third order selection level decisions regarding the most important limiting factors have been made and the next most important limiting factors (e.g., prey availability), should dictate selection processes.

Study Area

Algonquin Provincial Park

This study took place in Algonquin Provincial Park (48^{o}N, 78^{o}W; 7,725 km^2) in south-central Ontario from August 2003 through December 2004 (refer to Fig. 1.1). The park provided a mosaic of stand types which is typical of the Great Lakes – St. Lawrence forest region of Canada, a transition zone between the northern boreal forest and the southern temperate forest. The topography and vegetation in the park differed from the west to the east. The western side of the park was primarily composed of hardwood forest. This portion of the park experienced a wetter, colder environment as it lay on top of a dome of Canadian Shield bedrock that rose to 587 meters above sea level. The dominant forest types that made up the mature climax forests of the west were tolerant hardwoods such as sugar maple (*Acer saccharum)*, yellow birch *(Betula alleghaniensis)*, and American beech (*Fagus grandifolia)* along with some softwood species such as hemlock *(Tsuga canadensis)* and white pine (*Pinus strobus)*. The eastern portion of the park experienced a warmer, dryer climate than the west as it sat only 150 meters above sea level. This area was mainly composed of pine and intolerant hardwood forest such as white pine, red pine *(Pinus resinosa)*, jack pine *(Pinus banksiana)*, white birch *(Betula papyrifera)*, trembling aspen *(Populus tremuloides)*, and red oak *(Quercus rubra)*. In the lowlands there was mostly balsam fir *(Abies balsamea)* and black spruce *(Picea mariana)* (Strickland 1996).

Live-trapping occurred mostly along the highway corridor that runs through the south-central portion of the park, as well as along interior access roads that were accessible from the highway. Our study was in a section of the park where fur trapping was prohibited; however there were portions of the park where fur harvesting was permitted. Trapping was also permitted outside the boundaries of the park. Timber harvesting occurred in parts of the study area. The main forest management system employed in the park (~ 56% of tree harvesting operations in the park) encouraged uneven aged forest through the use of a selection cut system. The second most intensely used system was a uniform shelterwood system. Both of these harvesting systems maintained forest cover as opposed to clear cutting which made up ~ 2% of Algonquin harvesting operations. Of the total forested area in the park approximately 1.5% of it has active timber harvesting in a given year (Algonquin Forestry Authority 2004).

Methods

Trapping and Radio collaring

Fishers were caught in single door wire mesh live traps (models 106 and 108 Tomahawk Live Trap Co., Tomahawk, WI). Traps were baited with beaver meat or other meat scraps and checked at least once daily depending on weather conditions. Bait was wired to the back of the trap and replaced as needed.

Captured fishers were processed on site. If temperatures were very low fishers were processed in the cab of the field truck to avoid hypothermia (Kohn *et al.* 1993). Fishers were immobilized in the trap with the use of a trap divider and a custom made plunger that was constructed from a square piece of wood with a handle. An intramuscular injection of ketamine/ xylazine (10:1 mixture) at 20 mg/kg body mass was given using a handheld syringe (Gilbert *et al.* 1997, Mitcheltree *et al.* 1999, Weir and Harestad 1997). Once anesthetized, fishers were weighed, sexed and aged. Age determination was based on palpatation of the sagittal crest and a visual assessment of tooth wear (Francis and Stephenson 1972, Leonard 1986, Mitcheltree *et al.* 1999, Weir and Harestad 1997). Fishers were placed in age categories of either juvenile (<1 year), subadult (1-2 years), or adult (>2 years). To reduce the amount of stress on the animals blindfolds were used throughout the handling process and a minimum number of personnel were on site as recommended by the Canadian Council on Animal Care (2003). Mortality sensing radio collars (model SMRC-3, Lotek Wireless, Newmarket, ON) were fitted to each fisher. The minimum battery life of these collars was 11 months. A numbered metal ear tag (National Band and Tag Co., Newport, KY) was attached to one ear for identification in the case of a dropped collar (Mitcheltree *et al.* 1999) and an antibacterial ointment was applied to the ear. Throughout the handling process the animal's vital signs (temperature, respiratory rate, and heart rate) were monitored (Mitcheltree *et al.* 1999). An intramuscular injection of yohimbine was administered at 0.1 mg/kg body mass when handling was complete. Animals were placed back in the trap and the trap was covered until a full recovery had been made. Recovery was evaluated based on behavioural observations that were characteristic of the fisher previous to being immobilized (Coffin 1997). The fisher was then released at the site of capture.

Telemetry

Animals were located using standard aerial and ground telemetry procedures (White and Garrott 1990). Animals were located opportunistically via ground telemetry. Small hand-held receivers (model R-1000, Communications Specialists Inc., Orange, CA, USA) and either a 2 or 3 element antenna (Telonics Inc., Mesa, AZ, USA) were used on the ground. At least 3 bearings were attained, to reduce the effects of signal reflection and increase accuracy (Garrott *et al.* 1986), with at least 150 meters between each. All three bearings were recorded within 15 minutes to minimize the effect of a moving animal. Locations were determined using LOAS 2.10.1 (Ecological Software Solutions™ 2003) with a maximum likelihood estimator. This estimator minimized the angular error between the set of bearings and the estimated signal location. Data was omitted from analyses if a location could not be estimated (i.e., if error prohibited estimation). Aerial telemetry was conducted approximately once a month from a fixed wing aircraft. Two types of aircraft were used: a Cessna 172 in the summer and fall and a Cessna 182 in the winter and spring. Each plane had an H-antenna fixed to each wing strut. Telemetry error for both ground and aerial telemetry was assessed via blind tests comparing the known location of a stationary transmitter and the estimated location from either the air or the ground (White and Garrot 1990). The known and estimated positions were plotted and the distance separating them was measured. These evaluations were done multiple times (n = 5 and n = 6 respectively).

Home range

We adopted the original definition of home range (Burt 1943) as the area around a home site, over which the animal normally travels in search of food. We calculated home range sizes for both males and females with > 15 locations. Area-observations curves showed 7 of these 9 home ranges reached an asymptote. These animals did not disperse and did not show extraterritorial excursions during the mating season or clustering during the denning season. Only locations separated by > 16 hours were used to ensure only independent locations were used for analyses, since autocorrelation may lead to underestimated home range sizes (Arthur *et al.* 1989b, Hansteen *et al.* 1997, Harris *et al.* 1990). Home ranges were calculated using the *home range* extension for ArcView (Rodgers and Carr 1998). We used the 95% minimum convex polygon (MCP) home range estimator for ease of comparison with other studies (Fuller *et al.* 2001). Although many studies have used MCP for home range estimation some studies have estimated home ranges using other methods such as snow tracking and live trapping (Hamilton and Cook 1955, DeVos 1952). Each technique resulted in different determinations of size (Table 2.1). These differences can be falsely interpreted as variations in home range size, but are more likely due to different calculation methods. For this study the arithmetic mean of all the longitude (x) and latitude (y) coordinates was calculated and then 95% of the points closest to that arithmetic mean point were selected.

Habitat Selection

Habitat Maps

Forest Resource Inventory (FRI) maps from the Ontario Ministry of Natural Resources were used to describe the habitat in our study area. These maps originated from aerial photographs that were manually interpreted, field checked, and assembled in a Geographic Information System (GIS). Forest stands were classified into discrete categorical polygons on the basis of forest unit (FU) and development stage (DS). A forest unit is an aggregation of stand types that have similar succession pathways and species composition (Bowman and Robitaille 2005). Development stage is a categorization of stand age. Twenty five forest units, plus 8 non-forest units and 5 development stages (presapling, sapling, immature, mature, old) were present in the FRI (Appendix I). The FRI was reinventoried every 7 years and was current as of 2001. The output scale of the maps was 1: 10000 (Ontario Ministry of Natural Resources 2001).

Variables for Allen's model were developed from these discrete polygons by estimating percent deciduous forest and tree species diversity for each FU using data from Ontario ecoregions 4E and 5E. Ontario ecoregions are classified based on geologic, climatic, vegetation, soil, and landform features (Ontario Ministry of Natural Resources 2001). For each discrete polygon within our study area we assigned suitability scores for Allen's habitat categories through interpretation of his suitability index graphs (Table 2.3). The mean dbh of overstory trees (related to development stage) was inferred from the age of the stand. One of four suitability index (SI) levels was assigned to each discrete age category: pre-sapling, sapling and non-forest dominated areas were given a suitability score if 0 (SI = 0.0), for immature stands SI = 0.4, for mature stands SI = 0.8 and for old growth stands SI = 1.0. Tree canopy diversity was categorized into three levels based on the number of tree species present within a stand (presence was defined as ≥ 0.1% of the stand composition). These categories were defined by tallying the number of tree species in each forest unit and then dividing this range into three categories (low, medium, high). Boundaries were made to give each category a similar number of forest unit. For our situation low diversity stands were those consisting of 1 – 13 species. These were given a SI score of 0.2. Stands showing medium diversity were those with 14 – 18 species. These stands were given SI scores of 0.8. Highly diverse stands with > 18 species were considered optimal and given SI scores of 1.0. The percent of the overstory comprised of deciduous species was evaluated in 10% increments and given SI scores according to Allen's SI graph. This variable was determined *a priori* in our FRI dataset as a representative percentage for each of the 25 identified forest units. These percentages were then applied to each discrete polygon based on its forest unit (Table 2.2). We recognized the close association of canopy cover in a stand with the dbh and percent of deciduous species comprising that stand. Therefore we chose to omit the variable involving canopy closure as it was not available in the FRI. Thomasma *et al.* (1991) also found that percent tree canopy closure did not enter into their logistic discriminant analysis model when they evaluated the importance of each variable. We condensed the number of forest units from 25 to 11 (plus one non-forest unit) based on similar SI scores for the variables tree canopy diversity and the percent of the overstory comprised of deciduous species. We also reduced the number of development stages from 5 to 4. Both procedures were done to minimize the number of categories in the analyses (Table 2.4). Combinations were based on suitability index (SI) scores; those forest units that had the same SI score for the tree canopy diversity and the amount of overstory canopy composed of deciduous species represented one category.

Among Home Range Analysis

Only female fishers were considered for habitat selection analysis, as male home range selection was thought to be a response of female home range placement (Phillips *et al.* 1998). The dispersal capabilities of a species govern the spatial scale of selection (George and Zack 2001). Therefore, the extent for among home range analysis was delimited non-arbitrarily. Using only the female home ranges, we calculated the expected dispersal distance (which represented available habitat) for each individual (Bowman *et al.* 2002) as:

dispersal distance = 7 x (square root of the home range area) (1)

Available habitat was not thought to be equal among females in our study due to the variation in home range size (13.17 km^2 - 56.51 km^2), therefore space-use was analyzed separately for each individual. This allocation of available habitat reduced the sensitivity of selection patterns to an arbitrarily defined extent (Johnson 1980).

Error estimates were applied to all telemetry locations for among and within home range analysis. These buffered areas were included in the calculation of home range size for this analysis. There were two buffer sizes used for error estimates: one for locations derived via ground telemetry and another for aerial locations. Both buffer sizes were the largest distance between the known and estimated signal locations as determined through accuracy tests.

Calculated dispersal areas were displayed in ArcView (ArcView GIS 3.2a, Environmental Research Institute Inc., U.S.) and categorized by the FRI for the area. Individual polygons within the study site were classified according to forest unit (FU) and development stage (DS). The number of discrete polygon categories (FU * DS) was determined for the whole study area. The “available” habitat was calculated by determining the proportion of each type (FU * DS) in the study area. The “used” habitat was calculated by determining the proportion of each habitat type in the individual home ranges. The used and available habitat proportions were then used to calculate a resource selection probability function for each of the forest types (FU * DS) within each home range. The selection function for each forest type was first calculated as:

$$W_{\mathrm{i}} = \frac{\text{proportion used}_{\mathrm{i}}}{\text{proportion available}_{\mathrm{i}}} \qquad (2)$$

These values were then standardized using the equation that follows;

$$b_{\mathrm{i}} = \frac{\mathrm{W_i}}{\sum \mathrm{W_i}} \qquad (3)$$

As described by Rettie and Messier (2000). These resource selection function values demonstrated the selection probability for each habitat category. If available habitat categories were unused, standardized resource selection functions were optimal as they were insensitive to such occurrences. The habitat suitability index was then calculated for each forest type (FU * DS). To calculate the HSI score for each polygon we used the equation given by Allen;

$$\mathrm{HSI} = (\mathrm{SV_1{*}SIV_2{*}SIV_3})^{1/3}\mathrm{SIV_4} = \qquad (4)$$

Where V_1 = percent tree canopy closure; V_2 = mean overstory tree dbh; V_3 = tree canopy diversity; and V_4 = percent overstory canopy composed of deciduous species. Omitting V_1 (percent tree canopy closure) gave us the equation;

$$\mathrm{HSI} = (\mathrm{SIV_2{*}SIV_3})^{1/2}\mathrm{SIV_4} = \qquad (5)$$

High quality habitat to received high scores (0.7-1.0) and low quality habitat received low scores. HSI scores needed to span this range in order to describe differences among sites (Brooks 1997). The HSI developed by Allen (1983) makes 3 assumptions: (1) that the study area is within the range of the fisher, (2) that fishers' habitat use was not obstructed, and (3) that the fisher population was not harvested.

The standardized resource selection functions (equation 3) for each forest type were averaged among all females and were then plotted against the corresponding HSI scores resulting from Allen's equation. The relationship between these two variables was tested using a Spearman Rank correlation. Analysis of ranked data was recommended by Johnson (1980) to avoid assumptions about measured data and avoid absolute statements about preference.

Within Home Range Analysis

To evaluate Allen's model within home ranges, we first analyzed females individually using their home ranges and telemetry locations. Error buffers for ground and aerial telemetry were calculated as previously described. Home ranges and telemetry locations were displayed in ArcView and categorized following the methods for among home range analysis. Available habitat was calculated as the proportion of each forest type (FU * DS) in the female's home range. The amount of used habitat was calculated as the proportion of each type (FU * DS) within the error buffer around individual telemetry locations. The resource selection functions were calculated by standardizing W_i (see equation 2) into b_i (see equation 3). Standardized resource selection functions (b_i) were calculated for all habitat categories within each telemetry location. These were then summarized for each animal, as individual fishers were the replicated datum. The standardized resource selection functions were then averaged among animals to determine the overall resource selection function for each habitat category. The HSI scores for all habitat types were calculated using equation 5. The three assumptions that were made for the among home range level applied to this level. For each category type (FU * DS) we compared the standardized resource selection function (b_i) and the corresponding HSI value with the Spearman Rank correlation.

If there was no significant statistical correlation between HSI and b_i, we reclassified the forest units into broader categories (N = 10, plus a non-forest category) to explain the selection patterns of the female fishers (Table 2.5). We also reclassified the development stage to be one of two types: mature or immature (Table 2.5). Each discrete polygon was classified according to the revised FU * DS. The proportions of used and available habitat categories were determined for each animal as before. For each animal the RSFs (W_i) were calculated for all habitat types and then standardized following the previous method (equations 2 and 3). The standardized RSFs were then summarized as before providing a mean resource selection function for each animal. These selection functions were rank transformed prior to further analyses. We used analysis of variance to compare selection between habitat types. We determined significant differences between habitat types by using a Tukey test with $P = 0.90$ (McLoughlin *et al.* 2002).

Results

Trapping and Radio collaring

We radio-collared 26 individual fishers (10 males : 16 females). All fishers were successfully immobilized and radio-collared with the exception of one large male that escaped before immobilization could take place.

Age determination identified all of our study animals as consisting of 4 juveniles (<1 year), 5 subadults (1-2 years), and 17 adults (>2 years).

The recovery period for each animal varied. Generally, within two hours the animal's behaviour appeared normal. All animals were then successfully released.

Telemetry and Home range

Between August 2003 and June 2005 we located the 26 fishers a total of 442 times (range = 1 – 99) through both ground telemetry and aerial telemetry. Tracking of radio-collared female fishers resulted in 9 females having sufficient locations for home range estimation (range = 17-99). The average female home range (n = 9) was estimated to be 29.9 ± 5.2 km² (mean ± standard error) (Table 2.5). Of the ten radio-collared males, only two had more than 15 data points to estimate home range size. These two adult males had individual home ranges estimates of 38.1 and 39.3 km² (Table 2.5). These estimations did not include error calculations for the telemetry locations. Our accuracy tests for ground telemetry resulted in a buffer of 848 meters radius around all ground telemetry locations. Aerial telemetry was more accurate, and a buffer of 310 meters was used for all of those locations. These errors represent the largest from our blind accuracy tests.

Habitat Suitability

Among Home Range

There were 45 discrete polygons (FU*DS) in the study area that resulted from 11 forest units and 4 development stages and a non forest category (Table 2.6). There was a significant correlation between the ranks of the calculated standardized resource selection function (b_i) and the corresponding HSI score ($r_s = 0.414$, $P = 0.005$) (Figure 2.3). Thus, Allen's model could not be invalidated for use at this scale. The highest standardized resource selection function ($b = 0.112$) was for a mature lowland mixedwood forest unit, with an HSI of 0.580. All variables of this forest unit had moderately high suitability index scores (Table 2.4). The unit was mainly comprised of Eastern white cedar (*Thuja occidentalis)*, black ash *(Fraxius nigra)* and trembling aspen (Appendix I). The second highest resource selection function (*b* = 0.090) was for forest units with an overall HSI score of 0.325. This was a result of high amounts of deciduous species in the overstory canopy ($V_4 = 0.325$), in a unit of old growth development stage dominated by tolerant hardwoods. The resource selection functions with the lowest values (*b* = 0.000) were for units that were immature and younger (sapling, presapling) in age. Though not all units of these age categories had a resource selection function of 0.000, many of them did especially immature units. The species composition of these least preferred units did not indicate an avoidance of a particular species. The common descriptor for these units was age.

Within Home Range

The extent of the study area for within home range analyses had 33 discrete polygon types (FU*DS), 12 less than the among home range level (Table 2.8). There was no significant correlation ($r_s = 0.052$, $P = 0.770$) between the standardized resource selection functions (b_i) and the corresponding HSI score. The forest unit with the highest standardized resource function (*b* = 0.177) had a low HSI score. This unit was comprised of deciduous species such as sugar maple, yellow birch, red oak, and American beech as well as hemlock and pine, in an old development stage (Table 2.8, Appendix I). Overstory tree dbh (SI = 1.000) and canopy diversity (SI = 1.000) were highly ranked for this forest unit, but a low score for deciduous composition (SI = 0.325) decreased the HSI score. The lowest selection functions (*b* = 0.000) were for stands that were mostly classified as immature or younger (Table 2.8). There were a variety of tree species associated with these stands. The age of the stand was the most common variable for these least preferred stands. The mean HSI score for the available habitat at this scale was 0.273, which was similar to the mean score for the used habitat (0.286). Highly suitable habitat was not abundant within home ranges.

Reclassification of forest units into broader classes (N = 10; including a non-forest category) revealed selection for mature tolerant hardwoods at the within home range level (Figure 2.3). The ANOVA results demonstrated selection by fishers at this level for some of the 10 classes ($F_{10, 64} = 9.73$, $p < 0.001$). The comparison of ranked selection functions showed that there were 5 homogenous subsets of habitat types (Figure 2.3). The group of habitats with the highest selection included mature tolerant hardwood, non-forested habitat, mature mixedwood, mature pine, and mature intolerant hardwood. The habitats with the lowest selection were immature pine, immature softwood, immature tolerant hardwood, and immature intolerant hardwood.

Discussion

Telemetry and Home range

Our average female home range estimates greatly exceeded most of the previous estimates throughout the fisher's range. DeVos' (1952) estimate via snow tracking is an exception. He estimated female home ranges to be 100 km^2. The largest comparable difference among home ranges for females calculated using 95% MCP was between our study and a contemporary one done in eastern Ontario. Koen (2005) studied fishers in an agriculturally developed area and she documented an average female home range size of

2.1 km² ± 1.1 (SD). The home range sizes for males in our study also exceeded previously documented sizes obtained using the same method. Badry *et al.* (1997) and Fuller *et al.* (2001) conducted studies in comparable habitat: Alberta, Canada and Massachusetts, U.S. respectively, but found adult male home ranges to be smaller than those of the two Algonquin males. Koen (2005) reported small home range size for males (6.4 km² ± 2.3 SD), which is considerably smaller than our findings. These large home ranges suggest a low density of fishers in Algonquin Park during our study. We hypothesize that the following are contributing variables. Algonquin Park is predominantly hardwood forest which provides little protection from accumulating winter snow. Based on genetics, Carr (2005) suggested that Algonquin Park was low quality habitat (i.e., was a net receiver of immigrants) possibly as a result of snow conditions. Extensive hardwood forests could also lead to larger home ranges being necessary to encompass suitable prey habitat (Stickel 1954), as many prey species of the fisher are commonly associated with coniferous dominated forests. Low prey density could cause an increase in fisher home range size (Powell 1994b).

Most of the previous studies of home range size were estimated for harvested populations. The harvest of resident animals affected both spacing and social aspects of a population which influenced the home range selection and size (Garant and Crete 1997, Krohn *et al.* 1995). This makes unharvested populations important to study because they will reveal more natural population structure and spacing (Garant and Crete 1997, Krohn *et al.* 1995). Male home range size has an additional dependent factor: mate access. Prey availability as well as accessibility to reproductive females could lead to either the smaller home ranges found in the previous studies (abundance of prey and mature females) or to the larger home ranges found in our study (lesser amounts of prey and receptive females).

We chose to use the MCP method to determine home range sizes for ease of comparison with other studies. With this approach there are no assumptions about the distributions of the locations that are used to estimate size (Gallerani *et al.* 1997). We acknowledge the limitations that lie within the use of the MCP method such as underestimating home range when the sample size is small and assuming that all area is used by the animal (Andruskiv 2003, Arthur *et al.* 1989a). The latter assumption is addressed in our habitat suitability analyses.

Habitat Suitability

Among Home Range

Fishers in Algonquin primarily placed their home ranges in habitat which Allen (1983) suggested to be of moderately high value to their overall fitness. The highest ranked forest types generally had large dbhs, high species diversity, and inferentially high canopy diversity. The lowest ranked forest types were generally immature and younger in development, with small dbhs. The largest selection function for home ranges was for habitat with an HSI score of 0.580. This forest unit had a diverse canopy that we interpreted to be structurally comparable to a two-storied stand by Allen's definition. This habitat was mainly hardwood dominated by species such as black ash, trembling aspen, sugar maple, and yellow birch, but was also associated with cedar/hardwood mixed stands (Appendix I). It had 31 – 40% coniferous species in the overstory canopy, and it also had a large average dbh. Allen thought that this type of habitat has potential to provide protection in the winter as well as den sites, resting sites and foraging areas among the often abundant downed woody debris. The habitats with the lowest selection for among home ranges were all of an immature development stage or younger. This indicated that fishers placed their home ranges among more developed forest units more often than young forest units. This could again be interpreted as selecting areas with less snow accumulation, as well as higher prey, rest site and den site availability. The average HSI within the study site and among the home ranges was low, which concurs with Carr's (2005) suggestion that Algonquin Park is of low quality habitat for fisher. Therefore the top probability selection functions, which are of higher HSI values than the average, suggest that fishers are placing their home ranges in more suitable habitat than average within our study area, but that the amount of suitable habitat is low.

Within Home Range

Fishers used habitats within their home ranges that did not correspond to Allen's hypothesized optimal habitat. Fishers were most often found in habitats with a moderate to low suitability index. These forest units were hardwood dominated old-growth stands with at least 60% of the canopy made up of deciduous species. The species diversity, and inferentially canopy diversity, was also high (> 19 species) for these habitats which agreed with Allen's evaluation of suitability. It was the abundance of deciduous species within these preferred habitats that resulted in low HSI scores. Although predominantly hardwood, hemlock, white pine, and white spruce were also associated with these types of habitat. Coniferous stands provide optimal winter cover and the fishers' preferred prey species are associated with these stands. Perhaps deciduous habitats provide better foraging and increased fitness potential than previously thought. Arthur *et al.* (1989b) suggested that lack of demonstrated habitat selection could be attributed to the diversity of the fishers' diet. Fishers were also commonly found among areas without canopy cover, although these areas were rated the poorest by Allen. Within the area bounded by all telemetry locations, polygons representing water accounted for 70% of the area classified as non-forest. This could have been a result of home range placement in close proximity to water. Telemetry positioning did not normally result in the placement of an animal in the water or on the ice but with the application of the error buffer, locations commonly included water in their habitat composition. This behaviour could be attributed to microclimate as coniferous species are most often found to dominate the species composition of riparian habitat in Algonquin Park (Quinn 2004). Although fishers are not commonly associated with open habitat, they have traveled along the edges of such habitat (Powell 1979).

Although fishers' habitat selection within home ranges did not reflect Allen's evaluation of suitable habitat, it is clear that selective processes occurred at this hierarchical level. Fishers preferred stands of a mature age and older while seemingly having the least preference for immature stands as suggested by Allen (1983). Mature forests offer greater structure than younger forests due to increasing prey habitat, rest and den site availability, and cover (Powell and Zielinski 1994). However, their choice habitats consisted of primarily hardwood species. This may reflect optimal habitat selection for den sites, as Paragi *et al.* (1996) found that 94% of the cavities for natal dens were in hardwood trees and 50% of these were in snags. If snowshoe hare populations are low in the park, this could cause fishers to hunt in habitat other than conifer and dense lowland vegetation, which is preferred by hares. To our knowledge porcupine numbers were low within our study area; no sign of porcupine was seen over the extent of the study. Therefore we do not think that hunting for porcupines contributed to their selection of hardwood habitat. Bowman *et al*. (2005) noted that southern flying squirrels (*Glaucomys volans)*, chipmunks *(Tamias striatus)*, and other hardwood specialists such as deer mice (*Peromyscus maniculatus)* were abundant during our study perhaps providing adequate food sources within hardwood forests. Although fishers appeared to use coniferous stands less than expected, these types of habitats are still of importance to the fisher as it is the primary habitat associated with the snowshoe hare. Buskirk and Powell (1994) have suggested that forest structure and available prey within the forest are indicative of habitat use for fishers and not the actual forest types.

Our results support the concept of hierarchical habitat selection. Johnson (1980) suggested that the selection of habitat within a home range was of a higher order than the selection of an actual home range because the availability of habitats within the home range is dependent on the selection of the home range itself. Rettie and Messier (2000) proposed that through the nature of hierarchical habitat selection, there is a direct relationship between the importance of limiting factors and the level at which selection is occurring to minimize these factors. The results of our habitat selection analyses support Rettie and Messier's (2000) proposal. We found a correlation between the resource selection functions and Allen's HSI for analyses among home ranges. Allen's HSI assumed that winter cover was the most limiting factor for the fisher. We found that there was no correlation at a higher order of selection: within home ranges. At this level animals are thought to be selecting habitat based on the next most limiting factor, which for fishers is most likely prey availability and den site selection (Zielinski *et al.* 2004).

The demonstrated use of hardwood forests in our study may be reflective of the year-round analysis of our data. Although Allen's model was developed for year round use it evaluates areas based on the provision of winter cover. Arthur *et al.* (1989b) demonstrated that fisher's habitat use was variable among seasons. It is possible that our data reflects the use of hardwood habitats in seasons without snow.

We acknowledge that resource selection functions require several assumptions. The primary assumption (Manly *et al.* 1993) is that all of the collared fishers had free and equal access to all of the habitats. With territorial animals, such as the fisher, there may be spatial constraints imposed due to intraspecific strife. These constraints might be limited in our study and assumptions met because of the large home ranges and inferentially low density of the fishers in the park. However, we do acknowledge that this assumption is hard to meet, as changes in an animal's perception of available habitat as a result of social constraint is hard to detect (Garshelis 2000). We also assume that the availability of habitat was constant during the study period (Arthur *et al.* 1996).

Recommendations and Management Implications

We did not invalidate Allen's (1983) model for use at a coarse scale, equivalent to the among home range level. However, caution should be used when applying this model to finer scale analysis, as our data did not support the predictions of the model for movements within home ranges. Our results demonstrate the importance of scale selection when directly assessing habitat suitability of the fisher. As Powell (1994a) suggested, knowing the forest composition used by a population of fishers is different from knowing the individual preferences for certain habitats within the forest. Management for the provision of fisher habitat is best suited to a landscape-level approach. A landscape-level approach should view the forest as a working unit and not individual stands. Jones and Garton (1994) suggested applying management initiatives to systems of mature forests was more effective in promoting suitable fisher habitat than aiming goals to individual stands. Management at the landscape-level should incorporate a variety of succession stages which would encourage high prey diversity for fishers. Key habitat elements within home ranges: prey diversity and resting/denning habitat will be addressed within landscape level management plan as well as the most limiting factor: winter cover.

Guidelines for the management of fisher habitat in Ontario with regards to timber harvesting were developed by the Ministry of Natural Resources (1986) and tend to follow the concepts set out in Allen's model. Though attention has been given to mature age-class forests with dense coniferous cover and the retention of snags, it is also important to acknowledge the other types of habitats that contribute to the fishers' overall fitness. Mature hardwood-dominated sites appeared to be important for Algonquin Park fishers and this should be recognized in management efforts.

Acknowledgements

Thank you to following people: Richard Shalla, Amelia Argue, Shaun McIntosh, Ken Mills, Geoff Bohart, Derek Potter, Carrie Sadowski, Richard and Jane Tully, Jaime Mills, Ryan Collis. Our trapping and handling protocol was approved by the Ministry of Natural Resources Animal Care Committee.

Literature Cited

Algonquin Forestry Authority. 2004. Forest Management. www.algonquinforestry.on.ca/management.html#Anchor-Forest-17304

Allen, A.W. 1983. Habitat suitability index models: Fisher. U.S. Dept. Int. Fish Wildl. Serv. FWS/OBS-82/10.45. Pp 19.

Andruskiv. M. 2003. Prey abundance, availability, and anxiety in structured environments. MSc Thesis. University of Guelph, Guelph, Ontario, Canada.

Arthur, S.M., W.B. Krohn, and J.R. Gilbert. 1989a. Home range characteristics of adult fishers. J. Wildl. Manage. 53(3):674-679.

Arthur, S. M., W.B. Krohn, and J.R. Gilbert. 1989b. Habitat use and diet of fishers. J. Wildl. Manage. 53(3): 680-688.

Arthur, S.M., B.F.J. Manly, L.L. McDonald, and G.W. Garner. 1996. Assessing habitat selection when availability changes. Ecology. 77(1): 215 – 227.

Badry, M.J., G. Proulx, P.M. Woodard. 1997. Home – range and habitat use by fishers translocated to the Aspen parkland of Alberta. In: In: G. Proulx, H.N. Bryant, and Paul M. Woodard, eds. Martes: taxonomy, ecology, techniques, and management: proceedings of the Second International Martes Symposium. Edmonton, Provincial Museum of Alberta.

Bowman, J., J.A.G. Jaeger, and L. Fahrig. 2002. Dispersal distance of mammals is proportional to home range size. Ecology 83(7): 2049-2055.

Bowman, J. and J-F. Robitaille. 2005. An assessment of expert-based marten habitat models used for forest management in Ontario. For. Chron. 81(6): 801 – 807.

Bowman, J., G.L. Holloway, J.R. Malcolm, K.R. Middel, and P.J. Wilson. 2005. Northern range boundary dynamics of southern flying squirrels: evidence of an energetic bottleneck. Can. J. Zool. 83: 1486 – 1494.

Brennan, L.A., W.M. Block, and R.J. Gutierrez. 1986. The use of multivariate statistics for developing HSI. In: J. Verner, M.L. Morrison and C.J. Ralph eds. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press.

Brooks, R.P. 1997. Improving habitat suitability models. Wildl. Soc. Bull. 25:163-167.

Buck, S,., C. Mullis, A. Mossman,. 1983. Final report: Corral Bottom-Hayfork Bally fisher study. [Unpublished report]. U.S. Department of Agriculture. In cooperation with Arcata, CA and Humbolt State University.

Burt, W.H. 1943. Territoriality and home range concepts as applied to mammals. J. Mammal. 94:364-352.

Buskirk, S.W. 1992. Conserving circumboreal forests for martens and fishers. Conser. Biol. 6(3): 318 – 321.

Buskirk, S.W. and R.A. Powell. 1994. Habitat ecology of fishers and American martens. In: Buskirk, S.W., A. Harestad, M. Raphael, comps. Eds. Biology and conservation of martens, sables and fishers. Ithica, NY. Cornell Univesity Press: 283 – 296.

Canadian Council on Animal Care. 2003. Guidelines on: the care and use of wildlife. Pp 25 – 47.

Carr, D. 2005. Genetic structure of a recolonizing population of fishers *(Martes pennanti)*. M.Sc. Thesis. Trent University. Peterborough, Ontario.

Clem, M.K. 1977. Interspecific relationships of fishers and martens in Ontario during winter. In: R.L. Phillips and C. Jonkel, eds. Proc. 1975 Predator Symp., Mont. For. Conserv. Exp. Stn., Univ. of Montana, Missoula. Pp 165 – 182.

Coffin, K.W. 1997. Interactions among marten prey availability, vulnerability, and habitat structure. In: G. Proulx, H.N. Bryant, and Paul M. Woodard, eds. Martes: taxonomy, ecology, techniques, and management: proceedings of the Second International Martes Symposium. Edmonton, Provincial Museum of Alberta. Pp 199 – 210.

Cook, J.G. and L.L. Irwin. 1985. Validation and modification of a habitat suitability model for proghorns. Wildl. Soc. Bull. 13: 440 – 448.

Coulter, M.W. 1960. The Status and distribution of fisher in Maine. J. Mamm. 41(1): 1 – 9.

deVos, A. 1951. Recent findings in fisher and marten ecology and management. Trans. N. Am. Wildl. Conf. 16:498 – 507.

deVos, A. 1952. The ecology and management of fisher and marten in Ontario. Ont. Dep. Lands For., Tech. Bull., Wildl. Ser. 1. Pp 90.

Douglas, C.W. and M.A. Strickland. 1987. Fisher. In: M. Novak, J.A. Baker, M.E. Obbard, and B. Malloch eds. Wild Mammals of North America. Ontario Ministry of Natural Resources, Toronto, Ontario. Pp 511 – 529.

Ecological Software Solutions. 2003. LOAS 2.10.1. Sacramento, CA. U.S.

Farmer, A.H., M.J. Armbruster, J.W. Terrell, R.L. Schroeder. 1982. Habitat models for land-use planning: Assumptions and strategies for development. Trans. N. Amer. Wildl. Conf. 47: 47 – 56.

Fisher, R.A. 1930. The genetical theory of natural selection. Clarendon Press, Oxford, England.

Fuller, T.K., E.C. York, S.M. Powell, T.A. Decker, and R.M. DeGraaf. 2001. An evaluation of territory mapping to estimate fisher density. Can. J. Zool. 79: 1691 – 1696.

Francis, G.R., and A.B. Stephenson. 1972. Marten ranges and food habits in Algonquin Provincial Park, Ontario. Wildlife Report 91.

Gallerani, E.J., and A.R. Rodgers. 1997. Differences in home-range size computed in commonly used software programs. Wildl. Soc. Bull. 25(3):721-729.

Garant, Y., and M. Crete. 1997. Fisher, Martes pennanti, home range characteristics in a high density untrapped population in southern Quebec. Can. Field Nat. 111(3):359 – 364.

Garrott, R.A., G.C. White, R.M. Bartmann, and D.L. Weybright. 1986. Reflected signal bias in biotelemetry triangulation systems. J. Wildl. Manage. 50(4): 747 – 752.

Garshelis, D.L. 2000. Delusions in habitat evaluation: Measuring use, selection, and importance. In: L. Boitani and T.K. Fuller eds. Research techniques in animal ecology: Controversies and consequences. Columbia University Press. New York, USA.

George, T.L. and S. Zack. 2001. Spatial and temporal considerations in restoring habitat for wildlife. Restoration Ecology. 9(3): 272 – 279.

Gibilisco, C.J. 1994. Distributional dynamics of modern *Martes* in North America. In: S.W. Buskirk, A.S. Harestad, M.G. Raphael, and R.A. Powell, eds. Martens, sables, and fishers: biology and conservation. Cornell University Press. Ithica, NY. Pp 59 – 70.

Gilbert, J.H., Wright, J.L., D.J. Lauten, J.R. Probst. 1997. Den and rest-site characteristics of American marten and fisher in northern Wisconsin. In: G. Proulx, H.N. Bryant, and Paul M. Woodard, eds. Martes: taxonomy, ecology, techniques, and management: proceedings of the Second International Martes Symposium.Edmonton, Provincial Museum of Alberta. Pp 135 – 145.

Hansteen, T.L., H.P. Andreassen, R.A. Ims. 1997. Effects of spatiotemporal scale on autocorrelation and home range estimators. J. Wildl. Manage. 61(2):280-290.

Harris, S., W.J. Cresswell, P.G. Forde, W.J. Trewhella, T. Woollard, and S. Wray. 1990. Home-range analysis using radio-tracking data- a review of problems and techniques particularly as applied to the study of mammals. Mammel. Rev. 20(2/3):97-123.

Heinemeyer, K.S. 1993. Temporal dynamics in the movements, habitat use, activity, and spacing of reintroduced fishers in northwestern Montana. M.S. thesis. University of Montana. Missoula, MT.

Johnson, D.H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology. 61(1): 65 – 71.

Johnson, S.A. 1984. Home range, movements, and habitat use of fishers in Wisconsin. M.S. Thesis, University of Wisconsin, Stevens Point, USA.

Jones, J.L. and E.O. Garton. 1994. Selection of successional stages by fishers in northcentral Idaho. In: Buskirk, S.W., A. Harestad, M. Raphael, comps. Eds. Martens, sables and fishers: biology and conservation. Ithica, NY. Cornell University Press. Pp 377 – 387.

Kelly, G.M. 1977. Fisher (Martes pennanti) biology in the White Mountain National Forest and adjacent areas. Ph.D. Thesis, University of Massachusetts, Amherst, Mass..

Koen, E. 2005. Home range, population density, habitat preference, and survival of fishers *(Martes pennanti)* in eastern Ontario. M.Sc Thesis. Ottawa – Carleton Institute of Biology. Ottawa, Ontario.

Kohn, B.E., N.F. Payne, J.E. Ashbrenner, and W.A. Creed. 1993. The fisher in Wisconsin. Technical Bulletin No. 183. Dept. of Natural Resources.

Krohn, W.B., K.D. Elowe, and R.B. Boone. 1995. Relations among fishers, snow, and martens: Development and evaluation of two hypotheses. For. Chron. 71(1): 97 – 105.

Leonard, R.D. 1986. Aspects of reproduction of the fisher, *Martes pennanti*, in Manitoba. Can. Field Nat. 100: 32 – 44.

Manly, B.F.J., McDonald, L.L. and Thomas, D.L. 1993. Resource selection by animals: statistical design and analysis for field studies. Chapman and Hall, London.

McLoughlin, P.D., R.L. Case, R.J. Gau, H.D. Cluff, R. Mulders, F. Messier. 2002. Hierarchial habitat selection by barren-ground grizzly bears in the central Canadian Arctic. Oecologia 132:102 – 108.

Mitcheltree, D.H., T.L. Serfass, W.M. Tzilkowski, R.L. Peper, M.T. Whary, and R.P. Brooks. 1999. Physiological response of fishers to immobilization with ketamine, ketamine-xylazine, or Telazol®. Wildl. Soc. Bull. 27(3): 582 – 591.

Ontario Ministry of Natural Resources. 1986. Guidelines for providing furbearer habitat in timber management. OMNR. Pp 29.

Ontario Ministry of Natural Resources. 2001. State of the forest report 2001: Forest information series. Queen’s Printer for Ontario. Ontario, Canada.

Paragi, T.F., S.M. Arthur, and W.B. Krohn. 1996. Importance of tree cavities as natal dens for fishers. NJAF. 13(2): 79 – 83.

Phillips, D.M., D.J. Harrison, D.C. Payer. 1998. Seasonal changes in home-range area and fidelity of martens. J Mamm. 79(1):180-190.

Powell, R.A. 1977. Hunting behaviour, ecological energetics and predator-prey community stability of the fisher *(Martes pennanti).* Ph.D. thesis. University of Chicago. Chicago, IL.
Powell, R.A. 1979. Ecological energetics and foraging strategies of the fisher *(Martes pennanti)*. J. Anim. Ecol. 48: 195 – 212.

Powell, R.A. 1982. The fisher: life history, ecology, and behavior. University of Minnesota Press. Minneapolis, MN. Pp 237.

Powell, R.A. 1994b. Effects of scale on habitat selection and foraging behaviour of fishes in winter. J. Mamm. 75(2): 349 – 356.

Powell, R.A. 1994b. Structure and spacing of Martes populations. In: S.W Buskirk, A. Harestad, M. Raphael comps. Eds. Martens, sables and fishers: biology and conservation. Cornell University Press. Ithica, NY. Pp 101 – 121.

Powell, R.A. 2004. Home ranges, cognitive maps, habitat models, and fitness landscapes for Martes. In: D.J. Harrison, A.K. Fuller, and G. Proulx eds. Martens and fishers (Martes) in human – altered environments: an international perspective. Springer Science and Business media. NY, USA.

Powell, R.A. and W.J. Zielinski. 1994. Fisher. In: L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon and W.J. Zielinski eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the western United States. US Dept. Ag.: 38 – 73.

Quinn, N.W.S. 2002. Algonquin wildlife: Lessons in survival. Natural Heritage Books. Toronto, Ontario.

Quinn, N.W.S. 2004. The presettlement hardwood forests and wildlife of Algonquin Provincial Park: A synthesis of historic evidence and recent research. For. Cron. 80(6): 705 – 717.

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

Rodgers, A.R. and A.P. Carr. 1998. HRE: the home range extension for ArcView. Centre for Northern Forest Ecosystem Research. OMNR.

Self, S.E. and S.J. Kerns. 1992. Pine marten – Pacific fisher study, phase II report, 10 June 1002. Redding, CA: Wildland Resource Managers. Pp 34.

Stickel, L.F. 1954. A comparison of certain methods of measuring ranges of small mammals. J. Mamm. 35(1):1-15.

Strickland, D. 1996. Trees of Algonquin Provincial Park. The Friends of Algonquin Provincial Park and the Ontario Ministry of Natural Resources. Whitney, ON.

Thomasma, L.E., T. Drummer, and R.O. Peterson. 1991. Testing the habitat suitability index model for the fisher. Wild. Soc. Bull.

Weir, R.D. and A.S. Harestad. 1997. Landscape-level selectivity by fishes in south-central British Columbia. In: G. Proulx, H.N. Bryant, and P.M. Woodward, eds. *Martes*: taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Weir, R.D. and A.S. Harestad. 2003. Scale-dependent habitat selectivity by fishers in south – central British Columbia. J. Wildl. Manage. 67(1): 73 – 82.

White, G.C. and R.A. Garrott. 1990. Analysis of wildlife radio-tracking data. Academic Press Inc. San Diego, CA.

Zielinski,W.J., R.L. Truex, G.A. Schmidt, F.V. Schlexer, K.N. Schmidt, R.H. Barrett. 2004. Resting habitat selection by fishers in California. J. Wild. Manage. 68(3): 475 – 492.

Table 2.1 Compilation from literature of previous fisher home range calculations (km²) using various techniques.

Year	Method	Male		Female		Author
		Mean (SE)	Range	Mean (SE)	Range	
2005	95% MCP	6.4 (2.3)*		2.1 (1.1)*		Koen
2005	100% MCP	14 (5.8)*		3.6 (1.7)*		Koen
2004	100% MCP	30.0 (7.8)		5.3 (0.7)		Zielinski *et al.*
2004	100% MCP	58.1 (29.6)		15.0 (2.2)		Zielinski *et al.*
2002	90% Kernel	73.9		25.0		Weir & Harestad
2001	95% MCP	10.0	6.5 – 16.6	7.6	2.9 – 11.1	Fuller *et al.*
1997	100% MCP	9.2 (1.8)	6.0 - 12.3	5.4 (0.9)	2.2 - 9.6	Garant & Crete
1997	95% MCP	24.3 (11.1)		14.9 (3.5)		Badry *et al.*
1993	Adap. Kern.	85		17		Heinemeyer
1993	100% MCP	39.63 (7.7)*		8.23 (1.6)*		Kohn *et al.*
1992	Conv Poly**	16 (6)				Self & Kerns
1989	100% MCP	31.0 (9.3)	12.3 - 83.2	16.0 (4.7)	11.6 - 43.7	Arthur *et al.*
1989	90% HM	27 (24)*		16 (12)*		Arthur *et al.*
1984	Conv Poly**	49 (37)*		8 (4)*		Johnson
1983	Conv Poly**	20.0 (12.0)		4.2		Buck *et al.*
1982	Min. area	NA			15.0 & 20.5	Raine
1977	Conv Poly**	35		15		Powell
1977	Conv Poly**	19 (17)*		15 (3)*		Kelly
1955	Snow Track	25			NA	Hamilton & Cook
1952	Snow Track	NA			100	DeVos

* Standard deviation

** Type of convex polygon was not indicated

Table 2.2 Quantification of habitat variables used for evaluation of Allen's HSI model derived from a forest resource inventory (FRI). Percent deciduous represents the amount of deciduous tree species in the canopy for a given forest unit, predetermined in the FRI. The number of tree species present within a forest unit is a tally of present species (>0.1%) based on a generalized description of the forest unit within the FRI.

Forest Unit	Percent Deciduous	No. Tree Species Present
BW	81.26	16
BY	72.5	16
CE	8.32	17
HDSL1	87.56	19
HDSL2	92.56	18
HDUS	86.86	20
HE	37.67	19
LC	3.71	11
LWMW	68.23	18
MWD	53.37	15
MWR	63.63	21
MWUS	61.71	22
OAK	87.78	19
PJ1	6.25	9
PJ2	20.28	12
PO	86.26	16
PR	3.49	12
PWOR	47.53	19
PWST	37.84	19
PWUS4	22.25	18
PWUSC	13.51	16
PWUSH	43.99	19
SB	1.96	9
SF	17.62	18
SP1	18.68	13

Table 2.3 Assignment of suitability scores, for Algonquin forest units, to habitat variables developed by Allen (1983). The percent of overstory canopy composed of deciduous species was determined *a priori* in the FRI. Stands were placed into one of the 10 percentiles determining their suitability (SDEC). The suitability score for mean diameter at breast height (dbh) of overstory trees (SDEV) was inferred from the development stage (DS) of each stand. Canopy diversity was determined by forest unit characteristics. Species composition for each forest unit was determined *a priori* in the FRI. Species were tallied (those representing ≥0.1% canopy composition), assigned to a group indicating either low, medium, or high diversity, and given a suitability score (SDIV).

Overstory Deciduous		Mean dbh		Canopy Diversity	
%	SDEC	DS	SDEV	No. Species	SDIV
0 - 9	0.800	N	0.000	≤ 13	0.200
10 - 19	1.000	P	0.000	14 - 18	0.800
20 - 29	1.000	S	0.000	≥ 19	1.000
30 - 39	1.000	I	0.400		
40 - 49	1.000	M	0.800		
50 - 59	1.000	O	1.000		
60 - 69	0.725				
70 - 79	0.400				
80 - 89	0.325				
90 - 99	0.275				
100	0.200				

Table 2.4 Forest stand classification. Original classification (FRI FU) of forest units was by the Ontario Ministry of Natural Resources for the forest resource inventory database. New forest units (NEW FU) were identified based on suitability indices (V_3 and V_4). V_3 is the suitability index score for the amount of tree canopy diversity in a stand. V_4 is the suitability index score for the percent of overstory canopy that is composed of deciduous species.

FRI FU	NEW FU	V_3	V_4
BSH OMS DAL OTH GRS RCK TMS UCL WAT	NF	0.000	0.000
HDSL2	HDSL2	0.800	0.275
BW PO BY	BP	0.800	0.325
LWMW	LWMW	0.800	0.725
HDSL1 HDUS OAK	HDO	1.000	0.325
MWR MWUS	MW	1.000	0.725
CE	CE	0.800	0.800
LC PJ1 PR SB	MCON	0.200	0.800
HE	HE	1.000	1.000
SF PWUS4 MWD PWUSC	PSMW	0.800	1.000
PJ2 SP1	PJSP	0.200	1.000
PWOR PWST PWUSH	PWPR	1.000	1.000

Table 2.5 Estimated home range sizes (km²) for female (F) and male (M) fishers. Independent locations (N) were used to calculate a 95% minimum convex polygon. Mean and standard error of the mean for each sex was also calculated.

Sex	N	95% MCP (km^2)	Mean	SE
F	19	56.51		
F	85	15.30		
F	17	30.73		
F	38	48.72		
F	28	39.27		
F	30	23.79		
F	45	13.98		
F	18	27.90		
F	30	13.17		
			29.93	15.57
M	16	38.06		
M	23	39.31		
			38.69	0.63

Table 2.6 Habitat suitability (HSI) calculations were done for discrete polygons (FU*DS). Standardized resource selection scores (b_i) were also calculated for each discrete polygon. Data in the table is ranked according to b_i score for among home range analysis.

FU*DS	HSI	Bi
LWMW_M	0.580	0.112
HDO_O	0.325	0.090
MW_M	0.648	0.064
BP_M	0.260	0.060
PWPR_M	0.894	0.059
PSMW_PS	0.000	0.053
BP_O	0.291	0.053
PSMW_M	0.800	0.047
MW_O	0.725	0.047
LWMW_O	0.648	0.047
NF_PS	0.000	0.046
CE_O	0.716	0.044
HE_O	1.000	0.042
HDSL2_O	0.246	0.036
PJSP_O	0.447	0.028
MW_PS	0.000	0.025
HDO_M	0.291	0.024
PSMW_O	0.894	0.021
HE_M	0.894	0.016
HE_I	0.632	0.015
CE_PS	0.000	0.011
PWPR_O	1.000	0.011
MCON_O	0.358	0.010
MW_I	0.459	0.006
PJSP_M	0.400	0.005
PSMW_I	0.566	0.005
BP_PS	0.000	0.005
BP_I	0.184	0.004
HDO_PS	0.000	0.004
HDSL2_M	0.220	0.003
CE_M	0.640	0.002
MCON_PS	0.000	0.002
MCON_M	0.320	0.001
HDO_I	0.206	0.000
CE_I	0.453	0.000
HDSL2_I	0.156	0.000
HDSL2_PS	0.000	0.000
HE_PS	0.000	0.000
LWMW_I	0.410	0.000
LWMW_PS	0.000	0.000
MCON_I	0.226	0.000
PJSP_I	0.283	0.000
PJSP_PS	0.000	0.000
PWPR_I	0.632	0.000
PWPR_PS	0.000	0.000

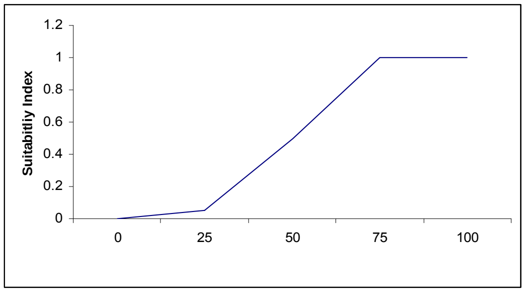
Table 2.7 Habitat suitability (HSI) calculations were done for discrete polygons (FU*DS; FU combinations based on suitability index scores for V_3 and V_4). Standardized resource selection scores (b_i) were also calculated for each discrete polygon (FU*DS). Data in the table is ranked according to bi score for within home range analysis.

FU*DS	HIS	Avg Bi
HDO_O	0.325	0.177
MW_M	0.648	0.110
NF_PS	0.000	0.109
BP_M	0.260	0.087
HDSL2_O	0.246	0.073
BP_O	0.291	0.067
HDO_M	0.291	0.066
HE_O	1.000	0.066
PWPR_M	0.894	0.062
HE_M	0.894	0.059
PSMW_M	0.800	0.051
PSMW_PS	0.000	0.043
CE_PS	0.000	0.043
PSMW_O	0.894	0.039
MCON_O	0.358	0.037
MW_O	0.725	0.035
MCON_M	0.320	0.032
MCON_PS	0.000	0.031
PSMW_I	0.566	0.022
HDSL2_M	0.220	0.016
CE_O	0.716	0.013
HDO_PS	0.000	0.012
PJSP_M	0.400	0.011
PJSP_O	0.447	0.011
MW_I	0.459	0.009
PWPR_O	1.000	0.006
LWMW_M	0.580	0.002
BP_PS	0.000	0.000
BP_I	0.184	0.000
CE_M	0.640	0.000
HDO_I	0.206	0.000
HE_I	0.632	0.000
LWMW_O	0.648	0.000
MW_PS	0.000	0.000

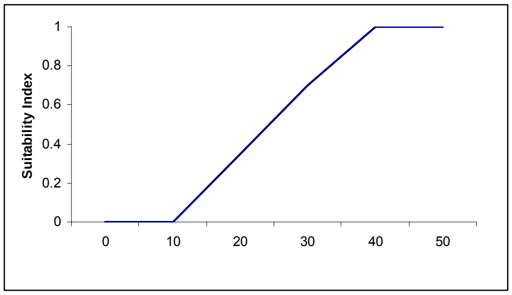
Table 2.8 Reclassification of forest units into broader categories. New forest units (New FU) and new development stages (New DS) are described. The mature development stage includes previous classifications of mature and old-growth forest, and immature includes immature, sapling, and presapling.

New FU	Description	New DS	Description
TH	Tolerant hardwoods	O	Mature
MW	Mixedwood	I	Immature
P	Pine		
IH	Intolerant hardwood		
SW	Softwood		
NF	Non-forest		

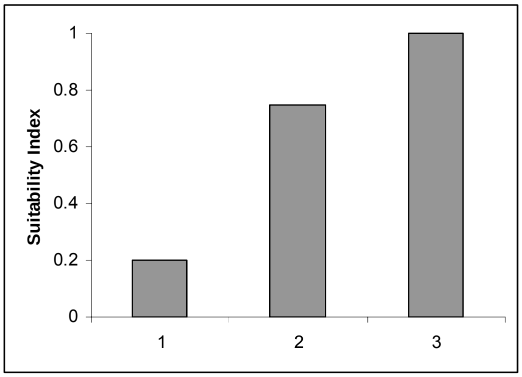
Figure 2.1 a) Percent tree canopy closure (V_1), b) Average dbh of overstory trees (V_2) of species ≥ 80% of the tallest tree, c) Tree canopy diversity (V_3); 1 = single-storied stand, 2 = two-storied stand, 3 = multistoried stand, d) Percent of overstory canopy comprised of deciduous species (V_4). Redrawn from Allen's (1983) habitat suitability variables for the fisher habitat suitability index.


Suitabitliy Index
1.2
1
0.8
0.6
0.4
0.2
0
0
25
50
75
100

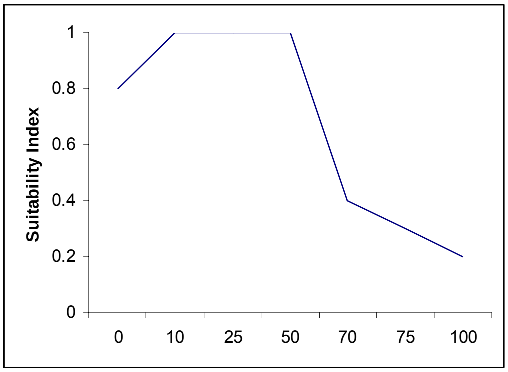
a)


Suitability Index
1
0.8
0.6
0.4
0.2
0
0
10
20
30
40
50

b)

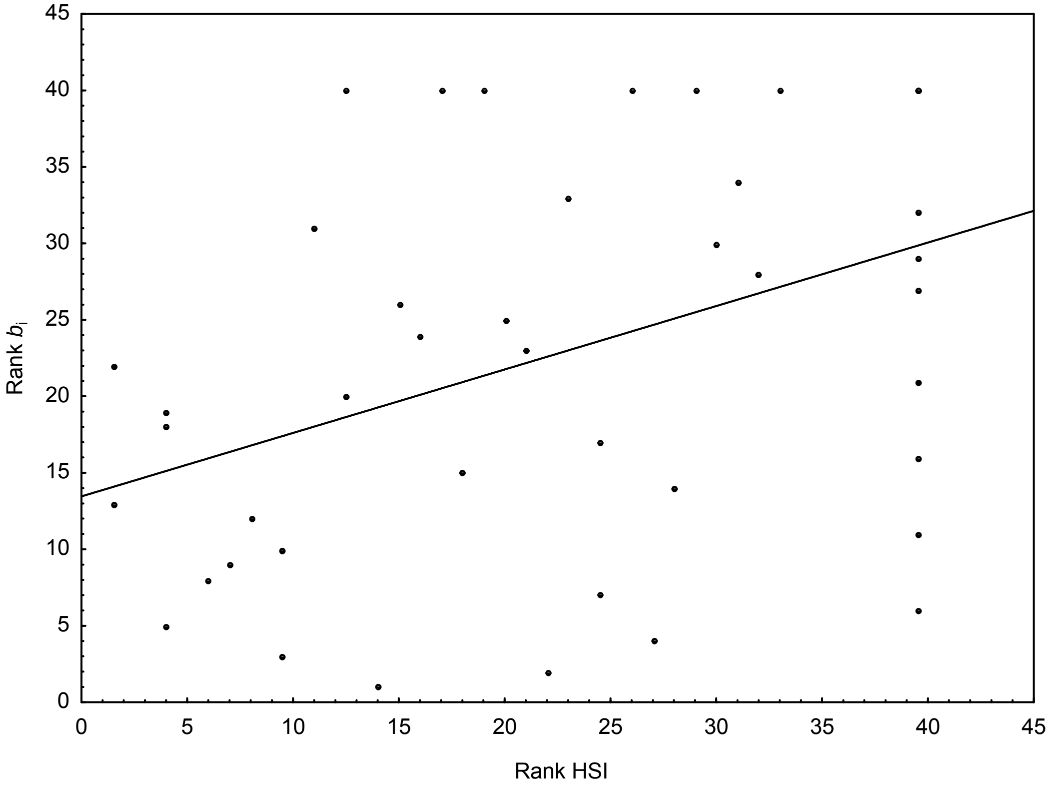

Suitability Index
1
0.8
0.6
0.4
0.2
0
1
2
3

c)


Suitability Index
1
0.8
0.6
0.4
0.2
0
0
10
25
50
70
75
100

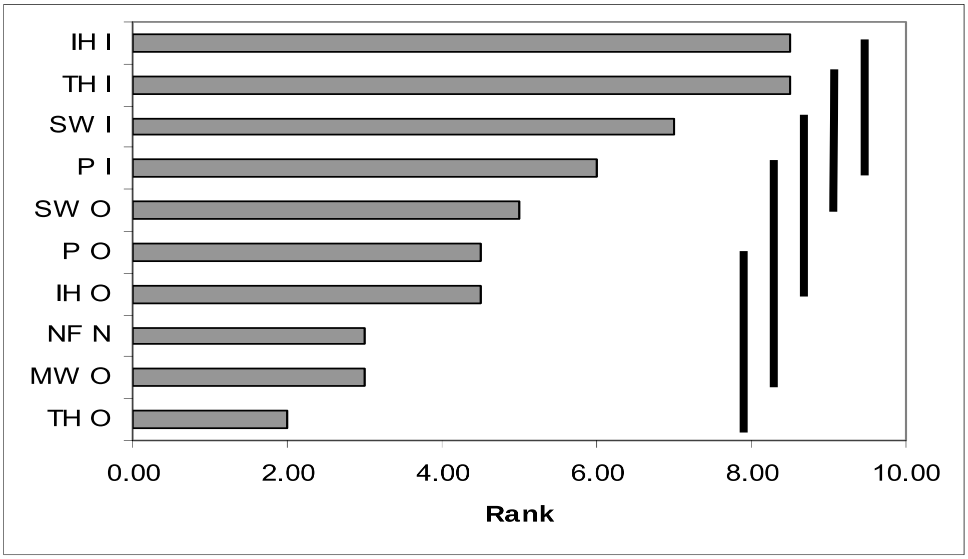
d)

Figure 2.2 Linear fit scatterplot depicting the correlation between ranks of the average standardized resource selection functions (b_i) of the Algonquin forest units, averaged among animals, and the corresponding habitat suitability index (HSI) scores of Allen's (1983) expert-based model. Analyses were done at the among home range level.



Rank b_i
Rank HSI
0
5
10
15
20
25
30
35
40
45

Figure 2.3 Mean ranks of standardized resource selection functions (b_i) for reclassified forest units. Ranks are averaged among animals. Lower ranks indicate stronger preference and higher ranks indicate weaker preference. Habitat specific standardized resource selection functions were calculated as the mean b_i among telemetry location for individual animals. This analysis was done at the within home range scale. Vertical lines indicate homogenous subsets of habitats as classified by a multiple comparisons test.


IH I
TH I
SW I
P I
SW O
P O
IH O
NF N
MW O
TH O
0.00
2.00
4.00
6.00
8.00
10.00
Rank

General Discussion

Marten and Fisher interactions

The occurrence of competitive interactions and competitive exclusion between fishers and martens has been the subject of an ongoing debate among members of the scientific community. DeVos (1952) suggested that there could be competition for food and den sites between the two species. He also thought that fishers might prey upon martens. Clem (1977) found a significant overlap in the winter diets of martens and fishers. He felt that this made it possible for competitive interaction to occur if prey abundance or availability was reduced for either predator. He did not think however, that this could induce competitive exclusion. He felt that their diets, in seasons other than winter, had enough differences to facilitate coexistence. Raine (1983) also suggested that the two species could coexist. He proposed that two factors regulated their coexistence; their different use of habitat and their inherent size difference. Differing responses to snow cover allows for temporal differences in habitat use by the two species fisher and marten. Fishers are more restricted by snow accumulation than marten. The large body size difference between the two species encourages different hunting and prey preferences. Douglas and Strickland (1987) analyzed fisher and marten harvest data for Ontario and reported an inverse relation. Krohn *et al.* (1995) thought that an inverse relationship could result from competitive displacement of marten by fishers, and the avoidance of areas with deep and frequent snowfalls by fishers but not martens. This echoed Douglas and Strickland's original hypothesis. Krohn *et al.* (1995) concluded that fishers and martens can and do live sympatrically, and were not convinced that competitive exclusion existed.

They suggested that at a certain density of fishers, interference competition with martens could become high enough that marten populations would not be sustainable. We found that in Algonquin Park, fishers and martens do coexist. Both of these species were caught in the same areas, more specifically on the same transect, and even in the same trap, within 24 h. Our marten capture rates (6.23/ 100 TN) greatly outnumbered fisher capture rates (0.54/ 100 TN). This suggests that martens were present at a higher density than fishers. These implied large differences in population densities could be explained by the influence of competitive interactions. Martens are considered to be more aboreal and efficient at exploiting small mammals than fishers (Raine 1983). Therefore, they could have been exploiting high densities of flying squirrels, chipmunks, and mice, that were in existence due to a peak in mast crops in 2002 (Bowman *et al.* 2005). Alternatively, habitat requirements could be driving these differences. Necessary habitat requirements for fishers, discussed in chapter three, include provision of winter cover. Deep snow accumulation is energetically costly for fishers so they are found more often in areas of low snow accumulation, whereas marten are less restricted by snow cover (Raine 1983). High amounts of snow accumulation in the park could favour marten populations and cause fisher populations to decline. The ratio of fishers to martens in our capture sample is opposite to capture ratios found in the park during the late 1950s. In these years fishers were being caught for the purpose of reintroduction into western and northwestern Ontario. In 1957 and 1958, trappers caught fishers at a rate of 2.53/ 100 TN and 10.86/ 100 TN respectively, and martens at a rate of 0.87/ 100 TN and 0.57/ 100 TN respectively (Loucks 1957, Catton 1958). The apparent reversal in the rates of capture of fisher and marten from the late 1950s to the early 2000s suggest an alteration within this time of the limiting factors for both species.

Populations in Algonquin

Habitat has changed significantly in Algonquin Park since the 1950s. Prior to the establishment of the Park in 1893 aggressive timber harvesting cut-over large portions of the Park, encouraging forest fires (Quinn In press). Upon regeneration, these processes created significant amounts of forage biomass for species such as deer, moose, and snowshoe hare. Deer populations grew under park protection as new habitat was produced by the continued clear cutting and forest fires (Runge and Theberge 1974). Trends continued this way until the 1930s when harvest methods changed and forest fire suppression was introduced. Selection–cut methods replaced clear–cuts and large openings in the canopy were no longer created, reducing the production of shrubs and small trees (Runge and Theberge 1974). It was around this time that the deer population crashed due to a decline in food and habitat production (Runge and Theberge 1974, Quinn 2004). Reduced stress on the habitat allowed for improvement and deer numbers began to increase, until they crashed again during the severe winters of 1958-59 and 1959-60 (Runge and Theberge 1974). It was during these winters that researcher Douglas Pimlott found wolves killing deer in excess and leaving carcasses behind (Quinn 2004). Fishers consume deer carrion as part of their diet (deVos 1952, Kuehn 1989, Powell 1981). Perhaps the deer population, until its crash, presented an available food source that allowed fisher populations to be more numerous than they are today.

The selection-cut system is still used today (Quinn 2004) and limits habitat suitability for snowshoe hares (*Lepus americanus)*, a preferred prey species of the fisher (deVos 1952). The snowshoe hare is a species that thrives in early successional habitats (Litviatis *et al.*1985). Today, due to fire suppression and selective logging, the Park's forests are in an essentially mature state (Quinn In press). The most critical habitat attribute affecting hare population levels is understory cover (Hoover *et al.* 1999) which provides protection from predators in the winter and thereby directly influences survival rates (Litviatis *et al.* 1985). Studies have indicated that it is the density of the understory and not composition that affect habitat use; however it is the composition that affects hare density (Litviatis *et al.* 1985). Dense softwood provides superior thermal cover and predator cover than hardwoods (Hoover *et al.* 1999, Litviatis *et al.* 1985). It is clear-cut logging and forest fires that create early successional stages that are rich with shrub and sapling growth providing optimal cover and food for snowshoe hares (Hoover *et al.* 1999, Litviatis *et al.* 1985). Through the removal of a vast majority of the coniferous cover in the first half of the 1900s (Quinn In press) and the gradual maturation of the Algonquin forests, it seems possible that the amount of optimal hare habitat has also declined. Snowshoe hare populations can be limited by the density of understory cover (Litviatis *et al.* 1985). Although the selection-cut system used today does temporarily open up the canopy, the scale of these openings and therefore the amount of plant biomass produced is not as great as with clear-cutting (Quinn In press). Potentially lower snowshoe hare populations in the park might be causing fisher population numbers to be low. Putatively, low fisher numbers could be responsible for the apparent increase in marten numbers, indicating that they are not fully symaptric. Current timber harvesting methods however, could also be the cause of marten population increases. These methods produce mature forests, which are ideal marten habitat (Buskirk and Powell 1994).

Sustainability in Algonquin

Sustainable forest management has been described as the maintenance and enhancement of the long-term health of a forest ecosystem while providing ecological, economic, social and cultural opportunities to benefit present and future generations (Corbett 2001). In the early stages of the Park's establishment sustainability of the forest was questionable. However, since the change in harvesting methods and the introduction of an overriding authority (Algonquin Forest Authority) there are now guidelines in place for the provision of a healthy ecosystem. For the fisher, these guidelines provide some critical habitat requirements. Tree markers, who mark trees for the selection-cut, protect essential habitat for species such as the fisher, by leaving den trees, mast trees, and browse in the harvested areas (Corbett 2001). Snags play a major role in the persistence and viability of a fisher population through the protection of young from weather and predators.

There are, however, still questions regarding the optimal management strategy for the Park's wildlife. Whether it is best returned to a natural state with no logging or fire suppression, to reinitiate the use of clear-cuts as the predominant harvesting method, or to continue with current management practices remains to be decided. Encouragement of fisher populations seems to coincide with the encouragement of deer and snowshoe hare populations. Development of optimal habitat for these populations would require increased disturbances to promote regeneration. Fishers in Algonquin Park may be limited by prey availability therefore provision of prey habitat should encourage the continued persistence of the fisher.

Literature Cited

Bowman, J., G.L. Holloway, J.R. Malcolm, K.R. Middel, and P.J. Wilson. 2005. Northern range boundary dynamics of southern flying squirrels: evidence of an energetic bottleneck. Can. J. Zool. 83: 1486 – 1494.

Buskirk, S.W. and R.A. Powell. 1994. Habitat ecology of fishers and American martens. In: Buskirk, S.W., A. Harestad, M. Raphael, comps. Eds. Biology and conservation of martens, sables and fishers. Ithica, NY. Cornell Univesity Press: 283 – 296.

Catton, W.R. 1958. Live fisher trapping, Pembroke District, 1958. Fish Wildl. Manage. Rep. 43.

Clem, M.K. 1977. Interspecific relationships of fishers and martens in Ontario during winter. In: R.L. Phillips and C. Jonkel, eds. Proc. 1975 Predator Symp., Mont. For. Conserv. Exp. Stn., Univ. of Montana, Missoula. Pp 165 – 182.

Corbett, C. 2001. Sustainable forest management in Algonquin Provincial Park. For. Chron. 77(5): 836 – 838.

deVos, A. 1952. The ecology and management of fisher and marten in Ontario. Ont. Dep. Lands For., Tech. Bull., Wildl. Ser. 1. 90 pp.

Douglas, C.W. and M.A. Strickland. 1987. Fisher. In: M. Novak, J.A. Baker, M.E. Obbard, and B. Malloch eds. Wild Mammals of North America. Ontario Ministry of Natural Resources, Toronto, Ontario. Pages 511 – 529.

Hoover,A., J. Watson, B. Beck, J. Beck, M. Todd, R. Bonar. 1999. Snowshoe hare winter range: Habitat suitability index model version 4. Foothills model forest. Alberta, Canada. www. fmf.ca

Krohn, W.B., K.D. Elowe, and R.B. Boone. 1995. Relations among fishers, snow, and martens: development and evaluation of two hypotheses. For. Chron. 71(1): 97 – 105.

Kuehn, D.W. 1989. Winter foods of fishers during a snowshoe hare decline. J. Wildl. Manage. 53(3): 688 – 692.

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

Loucks, M.G. 1957. Live trapping of fisher in Algonquin Park, winter of 1957.

Powell, R.A. 1981. Mammalian species: *Martes pennanti*. Am. Soc. Mamm. 156: 1 – 6.

Raine, R.M. 1983. Winter habitat use and response to snow cover of fisher (*Martes pennanti)* and marten *(Martes americana)* in southeastern Manitoba. Can. J. Zool. 61: 25 – 34.

Runge, R.A. and J.B. Theberge. 1974. Algonquin decline: of the deer. Ontario Naturalist. 14(2): 7 – 10.

Quinn, N.W.S. 2004. The presettlement hardwood forests and wildlife of Algonquin Provincial Park: A synthesis of historic evidence and recent research. For. Chron. 80(6): 705 – 717.

Quinn, N.W.S. In press. Reconstructing changes in abundance of White-tailed deer, *Odocoileus virginianus,* Moose, *Alces alces,* and Beaver, *Castor Canadensis,* in Algonquin Park, Ontario, 1860 – 2004.

Appendix I.

Description of forest units used in the forest resource inventory.

Forest Unit	Description
BSH	brush and alder
BW	white birch
BY	yellow birch
CE	cedar
DAL	developed agriculture
GRS	grass
HDSL1	hardwood selection (north)
HDSL2	hardwood selection (south)
HDUS	tolerant hardwood shelterwood
HE	hemlock
LC	mixed lowland conifer
LWMW	lowland mixedwood
MWD	mixedwood dry
MWR	mixedwood rich
MWUS	mixedwood shelterwood
OAK	oak
OMS	open wetlands
OTH	other
PJ1	pure jack pine
PJ2	jack pine - black spruce mix
PO	poplar
PR	red pine
PWOR	white pine, oak
PWST	white pine - red pine seed tree
PWUS4	white pine 4-cut shelterwood
PWUSC	white pine - red pine 2-cut shelterwood
PWUSH	white pine - red pine 2-cut shelterwood, low basal area
RCK	rock
SB	pure or lowland black spruce
SF	spruce fir
SP1	upland black spruce
TMS	treed wetlands
UCL	unclassified/ unsurveyed
WAT	water