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Quantification and Assessment of Caribou Habitat Fragmentation: An Integrated Remote Sensing, GIS, and Landscape Ecology Method

by

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ABSTRACT

Habitat fragmentation is a landscape evolution process characterized by the conversion of natural vegetation to other land cover types as a result of both natural and human disturbance agents. The most obvious effects of this process are the loss and isolation of natural habitat over time, a trend that has been recognized as a major risk to biodiversity at both a global and regional scale. The region to the north of Mount Revelstoke and Glacier National Parks, near Revelstoke, British Columbia, is an example of a landscape under change. Fragmentation, characterized by changes in the composition and spatial configuration of critical early winter habitat for woodland caribou (*Rangifer tarandus caribou*), may be occurring as a result of timber harvesting and natural disturbances such as wildfires. The development of an appropriate method is required to address this uncertainty and facilitate landscape-scale monitoring of caribou habitat supply, connectivity, and fragmentation over time. The end result will be an increased understanding of the impacts and processes related to caribou habitat availability and quality.

The main objective of this research is to determine the spatial effects of timber harvesting and wildfires on caribou habitat composition and configuration in the Revelstoke region for the period from 1975 to 1997. The research hypothesis tested is that fragmentation of critical early winter habitat is occurring in the study area over time due to these specific disturbance factors. The methodology employed to test this hypothesis is based on the integration of remote sensing, Geographical Information Systems (GIS), and landscape models. First, habitat suitability models were created to represent the landscape for the study area in a past (1975) and more recent condition (1997). Elevation, slope and forest stand age data for each time period were used to develop the habitat models. Habitat unit information used to build the habitat suitability models was generated using Landsat Multispectral Scanner (MSS) imagery for 1975 and Thematic Mapper (TM) imagery for 1997, and an Hybrid Decision Tree Classifier which combined maximum likelihood decision rules, a brightness differencing technique, and a spatial/contextual rule base. The

accuracy of the resulting habitat unit maps was 91.8% overall in the 1997 TM classification, and 89.5% overall in the 1975 MSS classification.

Based on a comparative analysis of selected spatial landscape metrics calculated for each time period, changes in the composition and spatial configuration of early winter habitat were quantified. Overall, the amount of suitable winter habitat available in 1997 represented a decrease of approximately 832.88 hectares (8.53%) from 1975 levels. As a result of the observed disturbance pattern, early winter habitat patches in 1997 were smaller and more uniform in size than in 1975, as suggested by a decrease in mean patch size and patch size coefficient of variation indices between time periods.

Spatial pattern indices calculated for habitat patches in 1997 also indicated a reduction in geometric complexity, interior core area, and mean proximity, while patch abundance and density, edge density, and juxtaposition had increased since 1975. The habitat fragmentation results are consistent with those reported in the literature in other regions, and indicate that fragmentation of early winter caribou habitat is occurring in the study area.

In conclusion, two main contributions of this research can be identified. First, this research has developed and tested a rigorous methodology for use in environmental monitoring which can be adapted for other species and regions. Second, it has provided strong evidence to support the original hypothesis that habitat fragmentation in relation to critical early winter habitat for caribou has occurred in the study area.

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DEDICATION

This thesis is dedicated to my grandfather, Bill Deuling and grandmother Noreen Delange. I thank both of you for always encouraging and supporting my academic achievements. From my first report card in Grade One to my university convocation, I was always excited to show you my grades and awards in return for your kind words (and money!). I dedicate this thesis to both of you for the pride, love, and inspiration that you provided to me over the years.

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LIST OF ACRONYMS AND ABBREVIATIONS

ANN	Artificial Neural Network
ANOVA	ANalysis Of VAriance
AT	Alpine Tundra
AVHRR	Advanced Very High Resolution Radiometer
BCMOF	British Columbia Minstry Of Forests
BGCZ	BioGeoClimatic Zone
DEM	Digital Elevation Model
ESSFvc	Engelmann Spruce Subalpine Fir Very wet Cold zone
ESSFvcp	Engelmann Spruce Subalpine Fir Very wet Cold Parkland zone
GIS	Geographic Information System
GPS	Global Positioning System
HDT	Hybrid Decision Tree
HSI	Habitat Suitability Index
ICHwk	Interior Cedar Hemlock Wet Cool zone
IGDS	Interactive Graphics Design Software
IJI	Interspersion and Juxtaposition Index
I/R/R/L	Ice / Rock / Road / Lake
MDT	Multivariate Decision Tree
MLC	Maximum Likelihood Classifier
MNN	Mean Nearest Neighbour
MODIS	MODerate resolution Imaging Spectroradiometer
MPI	Mean Proximity Index
MSS	MultiSpectral Scanner
NAD 83	North American Datum 1983
NDT1	Natural Disturbance Type One
NDT2	Natural Disturbance Type Two
NDVI	Normalized Difference Vegetation Index

RMS	Root Mean Square
RS	Remote Sensing
SEAS	Spatial Ecology Analysis System
SIGSEP	SIGnature SEParability
SPC1	SPeCies One (most abundant species type in stand)
SPC2	SPeCies Two (second most abundant species type in stand)
SPC3	SPeCies Three (third most abundant species type in stand)
TBC	Tree Based Classification
TIN	Triangular Irregular Network
TM	Thematic Mapper
TRIM	Terrain Resource Information Mapping
UDT	Univariate Decision Tree
UTM	Universal Transverse Mercator

1.0 INTRODUCTION

Monitoring landscapes over time enables resource managers to analyze and understand the effects of environmental practices and processes on natural ecosystem functions and interactions. These observations may then be incorporated into management plans that strive to maintain a balance between potential economic growth and environmental sustainability.

The region to the north of Mount Revelstoke and Glacier National Parks, near Revelstoke, British Columbia, is an area where this balance is in question. The balance needed is between logging and the preservation of viable wildlife habitat. The influence of extensive timber harvesting and natural disturbances (such as fire and avalanches) in the region has resulted in the replacement of large areas of the natural forest cover with an extensive pattern of clear cuts, burns, and avalanche paths. As a result, suitable habitat patches for local woodland caribou (*Rangifer tarandus caribou*) and natural corridors allowing connectivity between these sites are possibly being fragmented.

Habitat fragmentation is a landscape evolution process characterized by both the reduction in the total amount of suitable habitat available (habitat loss) and the spatial isolation of the remnant habitat patches over time (Saunders *et al.* 1991; McGarigal and Marks 1995). A major cause of habitat fragmentation is the removal of natural vegetation for human land use purposes including agriculture, urbanization, and extraction of timber and mineral resources. Biological conservation theory recognizes that habitat fragmentation may eventually result in the separation of vertebrate populations into small island subpopulations. Due to stochastic variability in demographics, small populations are more prone to local extinction (Schonewald-Cox *et al.* 1983; Harris 1984). Therefore, to ensure the long-term survival of woodland caribou in the Revelstoke region, extractive resource activities and other environmental disturbance factors should be managed to maintain regional habitat connectivity.

Environmental disturbance factors include any human or natural event that has affected landscape structure and composition. In the study area, the main human disturbance agent is timber harvesting. The primary natural disturbance is fire, but also include avalanches, insects, disease, and wind. The development of an appropriate method is required to quantify and assess the spatial characteristics of such disturbances and facilitate landscape-scale monitoring of caribou habitat connectivity and fragmentation over time. The end result will be an increased understanding of the environmental processes and potential consequences resulting from landscape fragmentation in relation to caribou habitat availability and quality. This thesis research also represents a contribution to resource managers for facilitation of appropriate practices in environmental monitoring, fragmentation analysis, and sustainable use of resources.

1.1 Research Objectives

The main goal of this research is to determine the spatial effects of timber harvesting and wildfires on caribou habitat composition and configuration in the Revelstoke forest region for the period from 1975 to 1997. The first research hypothesis (question) being tested is that fragmentation of critical early winter caribou habitat is occurring in the study area. The second question is to determine if the integration of remote sensing, Geographic Information Systems (GIS), and landscape ecology will provide an effective methodology by which to quantify these types of landscape changes. Two interconnected objectives are required to address these research questions:

1. The development of an integrated GIS/remote sensing classification procedure to map and classify habitat variables required to produce a habitat suitability model for the past (1975) and more recent (1997) time periods.

2. Quantification and assessment of fragmentation of early winter caribou habitat in the study area. Habitat fragmentation will be evaluated in terms of :

   a) Habitat loss as reflected by changes in overall habitat composition, and

   b) Changes in the spatial characteristics and configuration of early winter habitat occurring between 1975 and 1997.

The first objective is required to map caribou habitat and disturbance features and assign early winter habitat suitability ratings from multiple data sources including a forest inventory database, Landsat MSS imagery acquired in 1975 and Landsat TM imagery from 1997. The end result is two Habitat Suitability Index (HSI) models describing early winter habitat suitability in the study area.

The maps produced as a result of the application of each HSI model can then be compared and evaluated in order to identify, document, and analyze changes in land cover and habitat suitability that have occurred between 1975 and 1997. This enabled the quantification of changes in the total amount of habitat available over the 22 year period of study.

Finally, the computation and comparative analysis of spatial pattern indices for each HSI model was undertaken to detect changes in the spatial characteristics and configuration of suitable habitat areas that indicate a process of landscape fragmentation.

1.2 Thesis Organization

This thesis is organized into six chapters. Chapter 1 has introduced the concept of habitat fragmentation and suggested the importance of monitoring and quantifying landscape changes over time. An integrated remote sensing, GIS, and landscape ecology approach has been proposed to facilitate landscape-level monitoring of caribou habitat fragmentation in the study area.

Chapter 2 reviews previous and current research on the habitat needs of mountain caribou in western Canada and the United States. This is followed by a discussion of the use of spatial methodologies such as remote sensing, GIS, and landscape ecology for the assessment of caribou habitat. Current research concerned specifically with remote sensing / GIS integration and landscape ecology is then examined.

Chapter 3 describes the study area and summarizes the steps involved to create, process, integrate and determine the accuracy of the various digital data sources required for HSI model development. Chapter 4 outlines 1) the methodology employed for HSI modeling and classification accuracy assessment of the model variables, 2) temporal change analysis of land cover and habitat suitability, and 3) a comparison of habitat spatial pattern and structure between time periods.

Chapter 5 provides an analysis and discussion of overall map accuracy and individual class accuracy levels to determine the success of the image classification algorithm. The habitat fragmentation results and analysis are then reported in terms of the amount of habitat lost and observed changes in habitat spatial characteristics and configuration.

Chapter 6 presents the thesis summary, conclusions and recommendations for further research related to long-term monitoring of landscape disturbance and dynamics.

2.0 REVIEW OF RELATED RESEARCH

2.1 Introduction

The Revelstoke forest region is home to a population of woodland caribou (*Rangifer tarandus caribou*) numbering approximately 400 animals. Special adaptations to the high snow levels in the mountains of west-central Alberta, southeastern B.C. and northern Washington and Idaho has warranted the recognition of a separate ecotype of the species referred to as mountain caribou (Bradshaw *et al.* 1995). During the winter, mountain caribou rely almost exclusively on arboreal lichens occurring in old growth forest communities. As such, mountain caribou are considered an "umbrella species" in that protection of winter caribou habitat may serve to preserve and enhance the habitat requirements of other old-growth dependent species (Parks Canada 1999; McLellan *et al.* 1995)

The North American range of both woodland and mountain caribou has decreased significantly since the early 1900s as a result of human settlement (Edmonds 1991). Hunting, road and railway deaths, loss of habitat due to resource industries, and an alteration of the predator-prey system as a result of human disturbances have all been cited as leading causes for this decline (Cumming 1992). There is therefore a need to understand several important issues concerning caribou habitat in order to reverse this trend and maintain viable populations into the future. Specifically, research has been initiated to determine:

- caribou habitat requirements
- the status and spatial distribution of critical habitat, and
- the effects of anthropogenic and natural disturbances on important habitat areas

For the purposes of this literature review, each of these general themes is discussed in the following sections.

The first section of this chapter focuses on research addressing the specific habitat needs of mountain caribou in western Canada and the United States. This is followed by a review of studies on the spatial assessment of caribou habitat status using remote sensing, GIS, and landscape ecology techniques. A discussion of current research dedicated to these spatial methodologies, specifically remote sensing/GIS integration and landscape ecology, is then provided.

2.2 Caribou Habitat Research

An extensive body of literature has been compiled on the observed movements and habitat use of mountain caribou in Canada and the United States. Results from different studies conducted across the region are consistent in revealing a distinct migration of animals across elevation gradients throughout the year. The transition between different elevations seems to be driven primarily be animal response to snowfall and other seasonal parameters. Edwards and Ritcey (1959) first documented this behaviour among the caribou of Wells Gray Provincial Park in British Columbia. Their findings were later supported by telemetry analysis carried out on caribou herds in the Selkirk Mountains of Idaho, Washington and southeastern British Columbia (Scott and Servheen 1985; Simpson *et al.* 1987; Servheen and Lyon 1989; Warren *et al.* 1996). This work has allowed the development of a seasonal behaviour profile of mountain caribou.

During the fall and early winter seasons, as snow begins to accumulate at higher elevations, caribou move into mature and old growth cedar-hemlock forests at lower elevations. The closed canopy of this community provides protection from early winter storms, and intercepts snowfall which allows the caribou to continue to forage on understory vegetation such as falsebox (*Pachistima*) and wintergreens (*Pyrola* sp.) An important characteristic of old growth cedar-hemlock habitat is a large occurrence of windfall trees and branches that facilitates forage on arboreal lichens (*Alectoria sarmentosa*, *Bryoria* spp.) to supplement the early winter diet (Rominger and Oldemeyer 1989). As winter progresses and the snow depth increases, caribou move upslope into the

subalpine parkland. Open alpine meadows and dispersed stands of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) characterize the subalpine parkland zone. By late winter the snow pack in this zone has settled allowing easy movement between areas and access to arboreal lichens on the lower branches of standing Engelmann spruce and subalpine fir trees. During spring green-up the animals migrate back to lower elevations to forage on vascular plants, and then disperse across variable elevations in summer and fall. Several researchers have stressed the critical importance of early winter habitat, as it allows caribou to adapt to snow-forage dynamics and gradually shift from a summer diet of vascular green plants to a winter diet of arboreal lichens (Scott and Servheen 1985; Simpson *et al.* 1985; Rominger and Oldemeyer 1989).

The British Columbia Ministry of Forests (BCMOF) and Parks Canada have initiated research similar to the studies cited above in order to determine the habitat needs of mountain caribou in the Revelstoke region. Caribou telemetry locations have been collected for 60 animals on a weekly basis since 1992. Important objectives of the Revelstoke caribou project are the assessment of seasonal habitat use (McLellan *et al.* 1994) and the development of habitat suitability coefficients to facilitate habitat mapping (McLellan *et al.* 1995).

Results from a project report released in 1994 (McLellan *et al.* 1994) indicate that local caribou populations exhibit seasonal shifts in elevation consistent with subpopulations monitored in the Selkirk Mountains (Scott and Servheen 1985; Simpson *et al.* 1987; Servheen and Lyon 1989; Warren *et al.* 1996). Average elevations of caribou locations were plotted against the Julian week to identify specific division points for seasonal changes exhibited by Revelstoke region caribou. The following seasonal observations were reported:

1. Early winter – occurring between October 29 to December 23. The major habitat component used was mature cedar-hemlock stands

2. Late winter (December 24 - April 22). Telemetry observations were limited to the subalpine parkland zone
3. Spring (April 23 - May 27) Observations were in mature cedar-hemlock stands
4. Summer / Fall (May 28 - October 28). A range of habitats was used including subalpine parkland, spruce-balsam, and cedar-hemlock habitats.

It was argued that the emphasis on habitat protection should be placed on early winter requirements (mature cedar-hemlock) as "there is greater potential for conflict between traditional forest management and caribou habitat in the area" (McLellan *et al.* 1994: p. 12). Also, the report recognized the need to maintain regional connectivity throughout the range of mountain caribou to ensure that populations are not fragmented.

McLellan *et al.* (1995) developed habitat suitability indices based on the proportion of use by caribou to the proportion of total habitats available. A range of habitat variables was analyzed to develop a final HSI model. Elevation, slope, vegetation species composition, stand age, and stand height were identified as the most important contributing factors in habitat selection. During the early winter season, caribou were observed most frequently within a range of slopes between 20 and 50 per cent occurring in cedar-hemlock stands greater than 250 years of age at lower elevations (between elevations of 850 and 1300 metres). The observed seasonal habitat requirements and the HSI habitat variables (elevation, slope etc.) form the input variables for the HSI models developed within this research agenda. Also, the habitat unit classification scheme developed for this thesis is based on the caribou habitat units suggested by McLellan *et al.* (1994, 1995) (see Table 2.1).

Table 2.1 Caribou habitat units in the Revelstoke area (after McLellan *et al.* 1994)

Habitat Unit	Field Description
Alpine	Above 2300 metres elevation – land cover is predominantly tundra plant species such as mountain heather
Subalpine Parkland	**ESSFvcp** – Very Wet Cold Parkland Engelmann Spruce-Subalpine Fir biogeoclimatic ecosystem zone (Meidinger and Pojar 1993). Elevation range of 2300 – 1800 m. ASL – characterized by open alpine meadows and discontinuous stands of subalpine fir
Spruce-Subalpine Fir mature forest	**ESSFvc** – Very Wet Cold Engelmann Spruce-Subalpine Fir biogeoclimatic zone. Elevation range of 1800 – 1300 m. ASL – dominated by Engelmann spruce and subalpine fir stands greater than 80 years of age. Common understory shrubs include white-flowered rhododendron, black huckleberry, oval-leaved blueberry, and false azalea
Cedar-Hemlock mature forest	**ICHwk** – Wet-Cool Interior Cedar-Hemlock biogeoclimatic zone. Elevation range of 1300 – 500 m. ASL - dominated by western red cedar and western hemlock stands greater than 80 years of age. Common understory shrubs include falsebox, Devil's club, and oval-leaved blueberry
Immature forests	Stands in the early seral stage of succession (20 to 80 years of age)
Young burns	Areas subject to wildfire in the past 20 years. Previously forested habitat has been temporarily replaced by shrub and herb cover.
Recent cutting units	Areas subject to recent timber harvesting within last 20 years
Avalanche Paths	Areas of recent and/or frequent avalanche activity Dominated by shrub and herbaceous cover
Riparian	Large areas of sedges, grasses, willow and other shrubs immediately adjacent to streams. Tend to have gentle slopes and are subject to flooding
Ice/Rock/Road/Reservoir	Snow fields and glaciers at higher elevations. Bedrock, talus slopes, gravel and asphalt roads and lakes at various elevations

McLellan *et al.* (1994, 1995) have provided important information on specific habitat needs of caribou in the Revelstoke region and included a spatial component of analysis by examining elevation differences between seasons. However, the locations of important habitat units were not identified or mapped and the spatial configuration of habitat units relative to one another was beyond the scope of the project. McLellan *et al.* (1995) recognized that a GIS could be useful in analyzing the distribution of important habitat areas.

Other research has utilized spatial technologies such as GIS and remote sensing to analyze caribou habitat. For example, Bradshaw *et al.* (1995) and Ouellet *et al.* (1996) applied GIS data and methods to determine habitat availability within home ranges. Observed habitat use was then quantified by superimposing telemetry locations on digitized habitat unit maps. Similar to McLellan *et al.* (1995), habitat selection analysis was executed by comparing habitat use in relation to habitat availability. Statistical analyses suggested which habitat types were selected more or less frequently than random observations. Bradshaw *et al.* (1997) also employed GIS techniques to determine the effects of geophysical exploration noise disturbance on the movements of woodland caribou in northeastern Alberta. Animal positions before, during, and after a simulated seismic blast were juxtaposed with digital peatland habitat maps. The observed habitat patch changes were then tested for significance using analysis of variance (ANOVA). The results indicated that caribou exposed to the simulated disturbance made more habitat patch changes during the test period than did unaffected animals, suggesting movement away from the disturbance source. Each of these projects helps to illustrate the utility of using GIS for habitat analysis. As noted by Bradshaw *et al.* (1995: p. 1567):

> *"...the advent of 'geographic information systems' and their application to wildlife ecology has enabled the habitat selection analysis of vast areas over which large mammals often range."*

Other studies have integrated remote sensing image analysis and GIS for caribou habitat analysis. For example, Chubbs *et al.* (1993) demonstrated the effects of clearcutting on woodland caribou movements in east central Newfoundland through the integration of digital image classification and a Spatial Ecology Analysis System (SEAS). First, a Landsat TM image was classified into seven different terrestrial habitat types. Cut locations were then digitized, superimposed on the classification results, and analyzed using the SEAS software package. Habitat availability, habitat use, and the movements of caribou affected by clearcut activity were then analyzed. Results indicated that caribou, in particular cow-calf pairs, avoided clearcut activities during ongoing operations. It was determined that clearcutting activities in mature forests in summer range may affect the movement and distribution of woodland caribou.

Arseneault *et al.* (1997) used Landsat Thematic Mapper imagery, fire history data, and field observations to determine the effects of caribou grazing on ground lichen biomass in northern Quebec. The Landsat imagery was used to map postfire successional stages occurring between 1984 and 1989. The area of these units was determined using a GIS. Specific classification accuracy of this method was not addressed in this report, but the authors suggested that the combination of remotely sensed imagery, GIS analysis, and the use of fire history data may provide a useful tool for the management of wild caribou herds.

A more recent trend in caribou habitat analysis is a landscape ecology approach, which addresses the influence of the spatial configuration of habitat patches on population dynamics. For example, Stuart-Smith *et al.* (1997) used GIS software in conjunction with a spatial pattern analysis program called FRAGSTATS (McGarigal and Marks 1995) to evaluate the influence of landscape configuration on the population characteristics of woodland caribou in northeastern Alberta. Results indicated that lower calf survival and smaller home ranges occurred in more fragmented landscapes. Additionally, the researchers suggested that there was a strong pressure for the caribou to avoid matrix habitat (a highly fragmented landscape). The strategy employed was useful for the

description of landscape characteristics influencing caribou survival, but similar to Chubbs *et al.* (1993) and Arseneault *et al.* (1997), in that an accuracy assessment of data sources was not provided.

2.2.1 Summary

The caribou habitat studies cited thus far provide a conceptual context on the specific habitat needs of mountain caribou in the study area and validate the use of remote sensing, GIS and landscape ecology technologies for the analysis of habitat composition and spatial configuration. The intent of this thesis is to integrate the conceptual and technical components found in previous literature, while at the same time offering an innovative direction for the analysis of mountain caribou habitat. In particular the approach followed here incorporates two issues that have not been addressed within the body of caribou habitat research: 1) an accuracy assessment of habitat map sources and 2) an historical temporal analysis of habitat suitability. None of the studies discussed mentioned the accuracy of digital images or digitized map sources. To provide a more comprehensive spatial analysis of caribou habitat, a mapping methodology should incorporate an assessment of its accuracy. In terms of temporal analysis, Chubbs *et al.* (1993) and Bradshaw *et al.* (1997) compared caribou movements before and after a disturbance event, but long term changes in habitat composition and configuration were not analyzed. This is an important component of the methodology in this thesis, which employs a combination of remote sensing, GIS, and spatial pattern analyses over a long-term temporal scale. The effectiveness of this methodology is also assessed.

2.3 Remote Sensing / Geographic Information System Integration

Recent developments in the integration of GIS and remotely sensed image analysis have indicated that a combination of data sources and techniques may provide more information about environmental change than any method used in isolation. A review by Wilkinson (1996) identifies three ways in which remote sensing and GIS technologies are complementary:

1. remote sensing techniques can be used to acquire GIS data sets
2. GIS data can be incorporated as ancillary information to improve remote sensing products and
3. remote sensing data and GIS data can be used in conjunction for environmental modeling and analysis.

Wilkinson (1996) also recognizes a close interrelationship between these three types of integration.

Important sources of GIS data are thematic maps derived from remotely sensed image classification techniques, as suggested by Wilkinson's first type of GIS / RS integration. Traditional classification methods have involved the use of a parametric classifier such as a minimum Euclidian distance or maximum likelihood approach which assign the class membership of a pixel based on the statistical characteristics of its spectral values (Lillesand and Kiefer 1994). Limitations of this approach recognized within the scientific literature include the common misclassification of mixed pixels in which more than one land cover category occurs (Wang and Civco 1994), the difficulties of incorporating data sources that do not conform to a Gaussian distribution (Srinivasan and Richards 1990; Wang and Civco 1994; Peddle 1995) or sources having different data levels (nominal, ordinal, interval, ratio) (Hutchinson 1982; Srinivasan and Richards 1990; Peddle 1995).

2.3.1 Knowledge and Rule-Based Classification

To overcome the limitations of traditional image classification algorithms and improve accuracy levels, new methods have been proposed that follow Wilkinson's second type of GIS/remote sensing integration - the incorporation of GIS data as ancillary information to improve remote sensing products. One such approach is referred to as knowledge-based image classification (also known as rule-based or expert systems) which concerns the combined use of digital imagery, ancillary data, heuristics, and context rules. An expert system is a computer algorithm that has the ability to learn, reason, and apply a knowledge base on a specific subject to provide advice or solve a given problem, thereby imitating the logic and reasoning applied by an expert human analyst. At the core of an expert system is the knowledge base, which usually consists of a set of facts or rules (Huang *et al.* 1995). Several different methods for the construction of a rule set can be found within the literature on computing and artificial intelligence and more recently, in remote sensing applications. These knowledge-based methods tend to fall under one of two general categories: 1) those based upon heuristic or qualitative assumptions or 2) those based upon quantitative information.

The heuristic approach provides interpretations and rules based on expert knowledge or experience concerning a problem. Thus, informally stated assumptions about classification criteria would be considered "heuristic" in nature. In contrast, a quantitative methodology uses training samples to create decision boundaries (Møller-Jensen 1990). In this manner, the algorithm "learns" about the nature of classes from the statistical information contained within random samples from each category. Several knowledge-based classification studies have either employed a heuristic knowledge base (Running *et al.* 1995), a quantitative approach (Janssen and Middelkoop 1992; Peddle 1995; Brown *et al.* 1993), or a combination of both (Møller-Jensen 1990).

Within the quantitative approach, different classification methods have been developed. These include boolean or binary decision tests, evidential reasoning, and modified Bayes' classification.

Boolean decision tests or rules divide a data set into classes based upon a specified threshold such that $X_i > b$, where X_i is a feature in data space and b is a threshold in the range of X_i (Friedl and Brodley 1997). For example, a boolean test could divide an image spectral class into upland or lowland shrub communities based on an elevation boundary of 1800 metres. Boolean decision tests were used by Cibula and Nyquist (1987) to divide 9 initial spectral classes into 21 vegetation/landcover classes, thereby providing a better identification and stratification of forest types within Olympic National Park, Washington, USA. Hansen *et al.* (1996) classified Advanced Very High Resolution Radiometer (AVHRR) imagery into 13 broad vegetation types at a global scale. Important thresholds for derived metrics such as maximum Normalized Difference Vegetation Index (NDVI), total length of growing season, and rate of greenup were then applied to assign class membership.

Another variation of quantitative knowledge-based classification is evidential reasoning (Srinivasan and Richards 1990; Peddle 1995). This procedure relies on the Dempster-Shafer theory of evidence (Shafer 1976). This theory has provided a framework for computing evidence or belief values used to determine the probability of the class membership of a pixel. Evidence derived from several data sources is used to assign basic probability assignment values. Dempster's Rule of Combination is then used to merge bodies of evidence, resolve conflicts between possible pixel classifications and commits a pixel to a particular class label based on the combined probability values.

Janssen and Middelkoop (1992) took a different knowledge-based approach by constructing transition matrices to formalize knowledge about crop transitions. Spectral image data, GIS attributes and transition matrices were then entered into a modified Bayes' classification. In this manner, a conditional probability value was calculated for

each pixel based on the mean and covariance matrix of training areas. The observation or pixel is then assigned to the class having the highest probability value. This type of classification provided a 6 to 20 percent improvement over a maximum likelihood classifier, depending on spectral class discrimination.

Other researchers applying evidential reasoning and similar knowledge-based techniques have adopted a post-classification approach (Kontoes *et al.* 1993; Wang and Civco 1994). This involves combining the output from a statistically based image classifier with ancillary data sources that provide a body of knowledge or rules. The ancillary knowledge base is used to refine the designation of pixels that could not be correctly classified from image information alone. Examples of ancillary data sources include topographic variables such as elevation, slope, and aspect, and contextual rules such as the distance from roads or streams. This method has been proven to significantly improve classification accuracy compared to using a simple parametric classifier. For example, Kontoes *et al.* (1993) achieved an accuracy improvement of 12.8% (77.3% knowledge-based vs. 64.5% parametric) by incorporating image and geographical context rules to refine estimates of crop acreages. Wang and Civco (1994) first applied a supervised maximum likelihood classification to map eighteen different land cover categories in Connecticut. The initial accuracy of 76.4% was improved to 88.6% by incorporating slope, hydrography, and transportation data in an evidential reasoning model.

In this research, a variety of ancillary data sources are available for the study area. This information, in combination with Landsat satellite imagery, provides a multi-source database appropriate for the application of a knowledge-based classification. The literature reviewed above indicates that a post-classification approach may provide improved classification accuracy compared to conventional statistical algorithms. Important caribou habitat knowledge and rules established by research conducted in the study area (McLellan *et al.* 1994, 1995) also were incorporated into a knowledge-based classification for the purposes of mapping critical habitat locations.

2.3.2 *Decision Tree Classification*

Another image classification approach is decision tree or tree-based classification (TBC) (Brieman *et al.* 1984). A decision tree is defined as a classification that recursively partitions a data set into smaller subdivisions (classes) with a statistical test or splitting rule being applied at each split within the tree structure. The TBC algorithm represents a specialized example of a knowledge-based classification approach, as decisions or rules are employed to split the data into classes. Decision tree algorithms offer an advantage over conventional statistical classifiers such as the Maximum Likelihood Classifier (MLC) due to their nonparametric nature and the flexibility to handle different data types and nonlinear relationships (Hansen *et al.* 1996; Friedl and Brodley 1997).

Friedl and Brodley (1997) provide a comprehensive discussion on the different types of decision tree classification algorithms. Univariate Decision Trees (UDT) are the same as the boolean decision test employed in knowledge-based classification. Figure 2.1(a) provides an example of how the authors employed boolean decisions to split an AVHRR data set based on Normalized Difference Vegetation Index (NDVI) and surface temperature. Multivariate Decision Trees (MDT) differ somewhat from UDTs as the splitting test may be based on more than one feature of the data set. Figure 2.1(b) illustrates how two different land cover features could be entered into a linear discriminant function within the MDT. Finally, a Hybrid Decision Tree (HDT) employs the use of multiple decision algorithms or statistical tests to divide the data into classification levels. Figure 2.1(c) illustrates how a linear discriminant function, K-Means classifier, and univariate decisions may be incorporated within the tree structure. An advantage of this approach over that of the UDT or MDT is the flexibility to choose the most appropriate algorithm to map a particular class within the tree structure, rather than relying on a single type of decision test. Brodley (1993) referred to this property as "selective superiority". The decision tree approach in general, and hybrid decision trees in particular, provided higher classification accuracies than a linear discriminant function or maximum likelihood classification. For example, on imagery acquired from the

AVHRR satellite at a spatial resolution of one kilometre, the HDT algorithm produced an overall classification accuracy that was 9% higher than that produced by the UDT and MDT approaches (Friedl and Brodley 1997).

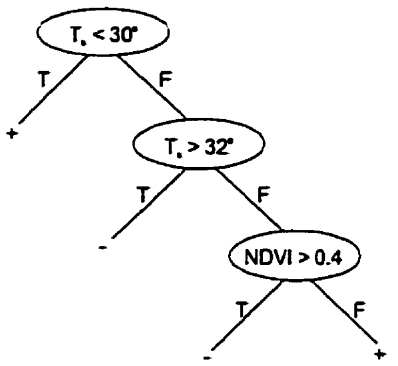


Figure 2.1 (a)
Example of an Univariate Decision Tree (UDT) used to separate two different land cover classes based on surface temperature and NDVI thresholds.

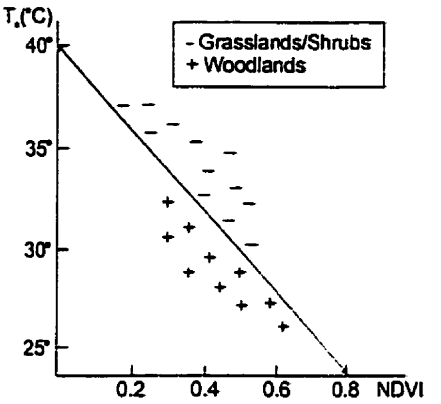


Figure 2.1 (b)
Example of a Multivariate Decision Tree (MDT) in which a linear discriminant function test is used to derive an equation for separating the two land cover classes. $T_s + 20^\circ$ NDVI < 40

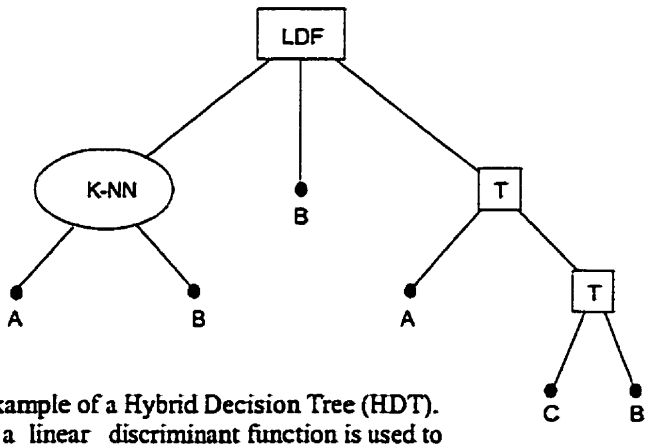


Figure 2.1 (c) Example of a Hybrid Decision Tree (HDT). At the root node a linear discriminant function is used to split the data set. The left subtree is split using a K-Means algorithm, while the right tree is a univariate decision tree.

(Friedl and Brodley 1997)

In a similar study, Hansen *et al.* (1996) compared a decision tree algorithm to the performance of the maximum likelihood classifier on an AVHRR land cover data set. Results indicated that each algorithm produced similar output, as the classification

accuracies were found to be comparable at 82%. The authors also illustrated that decision trees could also be used to reduce the dimensionality of input data sets by choosing the most appropriate metrics for cover type discrimination, thereby simplifying the classification procedure and optimizing computational efficiency.

Borak and Strahler (1999) developed a tree-based feature selection model for a Moderate-resolution Imaging Spectroradiometer (MODIS) image of Cochise County, Arizona, USA. The performance of the decision tree classifier was then compared with that of a maximum likelihood algorithm and an Artificial Neural Network classifier (ANN). The ANN achieved the highest accuracy of 78.6% compared to 73.5% for the decision tree and 71.1% accuracy for the MLC. The authors supported the findings of Hansen *et al.* (1996) by illustrating the ability of the decision tree to reduce a high-dimension data set for classification without degrading the information content of the input features. Despite the better performance of the ANN classifier, several advantages of tree based classifiers over the ANN approach were outlined. These included less computational time, and the systematic optimization of the tree structure to balance parsimony and classification accuracy. Also, by not incorporating hidden layers, as with some forms of ANNs, a source of ambiguity is avoided in a tree-based classifier. Therefore, the relative simplicity of a tree-based classifier may make this approach easier to use and comprehend (Friedl and Brodley 1997).

The results from the research cited above suggest that a Hybrid Decision Tree algorithm might be an appropriate method by which to incorporate TM and MSS satellite imagery, forest inventory data, and terrain variables into a single classification approach. This premise is based upon the improved performance of a decision tree classification over traditional statistical classifiers, such as the MLC. The flexibility to incorporate different data types and classification algorithms in a comprehensive methodology also supports the use of a Hybrid Decision Tree over a conventional classification approach.

2.4 Landscape Ecology

Landscape ecology concepts, which rely on GIS and remote sensing technologies for data sources and analysis algorithms, may be classified under Wilkinson's (1996) third type of remote sensing/GIS integration. For example, a landscape-scale investigation of forest extent might involve the use of digital imagery and GIS data together in a spatially-explicit environmental model. The analysis might focus on the explanation of certain stand age distributions or the determination of edge/interior ratios relative to linear developments such as a road network or seismic pattern.

A relatively new research discipline, landscape ecology focuses on the study of landscape patterns, the influences of natural and human environmental interactions on a landscape mosaic, and changes in landscape pattern and environmental processes over time (McGarigal and Marks 1995). A long and thoughtful discussion has occurred in the natural sciences on the meaning and measurement of landscape and landscape structure or pattern (e.g. Forman and Godron 1986; Urban *et al.* 1986; Forman 1995). From this body of literature, several key concepts have been developed and are now summarized briefly.

Various definitions of the term *landscape* have evolved from landscape ecology research For example, Diaz (1996: p. 12) defined *Landscapes* as:

> *"...aggregates of homogeneous patches of vegetation or landuse types and landforms that come into being through climatic influences, geomorphic processes, natural disturbances, human activities, and plant succession"*

Forman and Godron (1986) defined a *landscape* as heterogeneous land area consisting of a cluster of interacting ecosystems repeated in similar form throughout. McGarigal and Marks (1995) have synthesized the variable interpretations of a *landscape* into a simple definition: an area of land containing a mosaic of patches or landscape elements. The authors point out that how the area of land (landscape boundary) or patch elements within

a landscape are characterized depends upon the scientific context or nature of the investigation. For example, from a timber management perspective, a *landscape* could be defined as a regional management unit (such as the Golden/Revelstoke Forest Region) while the landscape elements or *patches* may correspond to stand boundaries delineated in a forest inventory survey. However from a wildlife management perspective, a landscape and habitat patches could be defined relative to a particular organism's utilization or perception of the same environment. Following this interpretation, a *landscape* might be defined as a mosaic of habitat patches, within which a target or more suitable habitat patch is embedded (Dunning *et al.* 1992), and the habitat *patches* themselves could be conceptualized as discrete environmental units possessing different habitat quality or fitness ratings (Wiens 1976). It is evident from these examples and the various definitions of landscape and patches within the literature, that an important step of any landscape ecology approach is to define each term in an appropriate manner that is relative to the phenomenon under investigation (McGarigal and Marks 1995).

Based on this concept of patch, McGarigal and Marks (1995) argue that an important underlying notion of landscape ecology research is that the spatial configuration or structure of landscape patches influences ecological characteristics, such as vertebrate populations. *Landscape structure* refers to the spatial relationships among the patches or elements, and can be characterized by both the composition and configuration of a landscape. *Landscape composition* relates to the abundance and variety of patch types while *landscape configuration* is the physical distribution and spatial character of patches within a landscape mosaic (McGarigal and Marks 1995). The consequences of fragmentation are evident through changes in both composition and configuration. That is, landscape fragmentation is characterized by a reduction in total amount of habitat (*habitat loss*) and a change in the spatial characteristics and configuration of remaining patches (*habitat isolation*) (Saunders *et al.* 1991; Forman 1995).

As part of a growing new research thrust to study the effects of fragmentation and other changes in landscape structure, an extensive set of indices or metrics has been developed

for quantifying composition and configuration at the landscape and patch levels of analysis (Olsen *et al.* 1993; Johnsson 1995; Riitters *et al.* 1995; Haines-Young and Chopping 1996). For example, Hargis *et al.* (1998) used a series of nine maps from a fragmented landscape to analyze the behavior of a few key metrics such as contagion, perimeter-area fractal dimension, and mass fractal dimension. In a second study, Hargis *et al.* (1999) calculated a suite of landscape metrics to determine the effects of forest fragmentation on American marten (*Martes americana*) population counts in the Uinta Mountains of northern Utah. Marten capture rates were found to have a significant negative correlation with the loss of forest habitat. Martens also showed significant responses to changes in landscape pattern between 18 different study sites, with capture rates being lowest in areas with extensive, closely spaced non-forested patches. Statistical analysis also indicated that the mass fractal dimension landscape index had the highest correlation with the amount of habitat loss ($r = -0.97$). The results suggested that martens were sensitive to changes in landscape pattern resulting from natural openings and timber harvesting and that a reduction in forest interior area may jeopardize the survival of future populations.

Landscape metrics have been successfully applied to a variety of applications including the measurement of environmental change caused by beaver activity (Townsend and Butler 1996), the analysis of landscape patterns resulting from logging clearcuts (Ripple *et al.* 1991), and the examination of long-term urban and rural landscape changes from human activities and policies (Turner 1990; Medley *et al.* 1995; Diaz 1996). Many GIS and image analysis software packages (e.g. IDRISI, GRASS) include landscape metric algorithms. Other landscape metric software packages such as FRAGSTATS (McGarigal and Marks 1995) and Patch Analyst (Elkie *et al.* 1999) have been developed to operate in conjunction with a GIS. The integration of GIS, remote sensing, and landscape metric software provides the opportunity for advanced analysis of landscape changes and disturbances.

2.4.1 Forest Fragmentation Analysis

The research focus of landscape ecology includes not only the application of landscape metrics, but also the development of new metrics and their application for the development of predictive models of landscape processes and disturbances (Frohn 1998). Rather than simply quantifying patterns of change, landscape ecologists also attempt to explain or predict the relationships between a landscape pattern (as quantified by landscape metrics) and underlying environmental processes (Diaz 1996). Several studies analyzing the fragmentation of forested habitat have been conducted in this manner. For example, Luque *et al.* (1994) documented the extent of landscape fragmentation in the New Jersey Pine Barrens region by illustrating significant changes in selected spatial metrics between a forested landscape in 1972 and 1988 landscape. As a result of human disturbances such as suburban / exurban development and logging activities over the 16 year period, a range of landscape metrics including Fractal Dimension, Diversity, and Contagion decreased while Dominance, Disturbance and Edge indices increased at the landscape level. At the patch level of analysis, the Mean Size of forested patches decreased significantly. These results indicated a trend to a more dissected or fragmented landscape over time as a result of human disturbances.

Tinker *et al.* (1998) used FRAGSTATS to calculate spatial pattern metrics in order to compare the effects of clearcutting and road building on the landscape pattern of the Bighorn National Forest, Wyoming. A principal components analysis was conducted to group the FRAGSTATS metrics into three uncorrelated components. The first component explained 72.4 % of the variation in the original set of metrics and was highly correlated to Patch Core Area, Edge, Patch Size and Density, and Patch Diversity. The second component was highly correlated to the Landscape Shape Index, Mean Core Area Index, Patch Richness, and Patch Richness Diversity. The Total Edge Contrast Index and Mean Shape Index were highly correlated to the third component. Based on the results, the authors suggested the effects of timber harvesting and road construction may be easily monitored by the analysis of a few key metrics: Patch Core Area, Patch Size and Number

of Patches, Edge Density, and Patch Shape. Each of these metrics reported the highest loadings or scores across the three different principal components. Measures of inter-patch distance, such as Mean Nearest Neighbour distance (MNN) and the Mean Proximity Index (MPI), were also included in the Tinker *et al* (1998) study. Results indicated that either the MNN or MPI inter-patch distance measures could contain unique information about landscape structure based on low correlations with the other metrics.

2.4.2 Appropriate Metrics for Quantification of Fragmentation

The recommendations made by Tinker *et al.* (1998) for a reduced metric set have touched on a question of paramount importance for any landscape ecology project. *Which metrics should be chosen in order to quantify and assess the effects of habitat fragmentation?* It is obvious from the literature that a confusing array of interrelated and commutative metrics are available and with little effort could be applied post-hoc to any digital map or GIS database. However, a more appropriate scientific approach may be to examine the findings and recommendations of the available studies of forest fragmentation (e.g. Luque *et al.* 1994; Reed *et al.* 1996; Zheng *et al.* 1997; Sachs *et al.* 1998; Hargis *et al.* 1998). The most obvious spatial effects of forest fragmentation recognized in the literature include:

- habitat loss or a reduction in the area of some habitat classes,
- decreased patch size,
- increased distance between patches (increased patch isolation),
- increased edge density between non-forest and forest habitat,
- increased patch density or patch abundance,
- reduced interior or core habitat area, and
- reduced geometric complexity or patch shape

Based on the apparent complex nature of the fragmentation process, it has been suggested that a single landscape metric is insufficient to capture all aspects of fragmentation. Instead a set of metrics should be chosen that quantify the different types of landscape changes listed above (Ripple *et al.* 1991; Davidson 1998). Table 2.2 summarizes the metrics chosen by similar forest fragmentation studies and indicates that the most common metrics applied at the class or patch type level include Number of Patches, Mean Patch Size, Patch Shape, Edge Density / Total Edge distance, Mean Core Area, Patch Density, Fractal Dimension, Inter-patch Distance, and Interspersion and Juxtaposition. While this sample of forest fragmentation studies is by no means exhaustive, it suggests that a common approach of past fragmentation studies is to calculate a set of metrics that capture a range of fragmentation landscape changes. The approach of this research will be to rely on the previous research to select an appropriate set of metrics proven to capture the changes in habitat patch composition and configuration as a result of fragmentation. Table 2.3 lists the set of metrics chosen along with a brief description and rationale for each spatial metric that was calculated in this thesis to quantify habitat fragmentation levels in 1975 and 1997. Based on the literature reviewed, each of these metrics appear to capture the essential changes that are hypothesized to have occurred in the Revelstoke area in the past few decades as a result of habitat fragmentation. The use of a well-documented set of metrics also allows the results of this research to be compared with similar investigations on the quantification of forest fragmentation in other regions.

Table 2.2 Summary of Class-level Metrics used in fragmentation studies

Reference	Application	Metrics used
Ripple *et al.* 1991	Forest fragmentation as a result of timber harvest	Patch abundance (# patches) Mean patch size Total edge distance Patch density Patch shape Inter-patch distance Mean core area
Luque *et al.* 1994	Monitoring spatial and temporal changes in a forested environment	Patch abundance (# patches) Mean patch size Total edge distance Patch shape
Sader 1995	Spatial characteristics of forest clearings	Patch abundance (# patches) Mean patch size Patch shape
Reed *et al.* 1996	Forest fragmentation as a result of timber harvest	Patch abundance (# patches) Mean patch size Total edge distance
Zheng *et al.* 1997	Quantification of rates and patterns of change in forested areas as a result of logging	Mean patch size Patch density Edge density
Stuart-Smith *et al.* 1997	Fragmentation of winter woodland caribou habitat areas	Mean patch size Patch size variance Largest patch index Interspersion / Juxtaposition
Sachs *et al.* 1998	Forest fragmentation as a result of timber harvest, wildfire and avalanches	Mean patch size Patch shape Patch fractal dimension Mean core area Interspersion / Juxtaposition
Hargis *et al.* 1998	Modeling forest disturbances and fragmentation to test behavior of common fragmentation metrics	Edge density Contagion / Interspersion Inter-patch distance Fractal dimension
Kushla & Ripple 1998	Quantification of effects of wildfire on succession stage patterns and wildlife habitat	Patch abundance Class area Mean patch size Patch size standard deviation Edge density Inter-patch distance Mean core area Fractal dimension

Table 2.3 Class-level Metrics calculated for 1997 and 1975 Habitat Patches

Spatial Metric	Description / Responses to Fragmentation
Class Area (Ha)	Total area of all patches per class The total amount of suitable habitat is expected to decrease as a result of fragmentation due to timber harvest and wildfires
Number of Patches	Total number of patches within each individual class To measure patch abundance and determine if suitable patches are becoming more or less numerous over time. (McGarigal and Marks 1995)
Patch Density	Number of class patches relative to total landscape area As the number of patches increases on the landscape, the density of suitable patches is expected to be positively correlated and also increase with habitat fragmentation (McGarigal and Marks 1995)
Mean Patch Size (Ha)	Average size of patches is expected to decrease with increasing fragmentation (McGarigal and Marks 1995)
Patch Size Coefficient of Variation	The average relative variability about the mean for each class (variability as a percentage of the mean) is generally preferable to Patch Size Standard Deviation for comparing patch size variability between landscapes. Patch Size Coefficient of Variation is expected to decrease as patches become less variable or more similar in patch size over time (McGarigal and Marks 1995)
Edge Density	Amount of edge relative to the landscape area is expected to increase in the initial stages of habitat fragmentation (McGarigal and Marks 1995)
Mean Shape Index	Average shape index of all class patches Patch shape equals the perimeter divided by the square root of the patch area, adjusted to a square standard. Patches are expected to become less geometrically complex in a managed landscape (McGarigal and Marks 1995)
Mean Proximity Index	Average proximity index for all patches in a class Proximity index is calculated as the sum of the ratio of patch size to nearest neighbour edge-to-edge distance for all patches within a specified search radius. Measures the isolation of patches in a landscape and is expected to decrease over time as there are fewer patches of the same class in close proximity to each patch type (Gustafson and Parker 1992)
Mean Core Area	Average size of disjunct core patches or interior patch areas remaining after specifying an "edge effect" buffer area One of the main effects of forest fragmentation is the conversion of interior habitat to edge habitat (Tinker *et al.* 1998). It is expected that the amount of interior habitat will decrease as a result of fragmentation (McGarigal and Marks 1995)
Interspersion/Juxtaposition	The juxtapositioning of a patch type or class with all others of the same class The Interspersion / Juxtapostion Index (IJI) is not affected by patch dispersion (Mean Proximity Index) and is a relative index that measures the observed level of patch interspersion as a percentage of the maximum possible given the total number of patch types. A fragmented landscape with many small isolated patches is expected to have a higher IJI value than a landscape with larger more contiguous habitat patches (McGarigal and Marks 1995)

2.5 Summary

The decline of woodland and mountain caribou populations in the last century as a result of human settlement has initiated scientific research on the habitat requirements of the species, the spatial distribution of critical habitat areas, and the effect of human and natural disturbances on caribou habitat distribution and connectivity. Telemetry analysis carried out within the Selkirk Mountains of Idaho, Washington, and southeastern British Columbia has revealed an observed elevational migration in response to the seasonal habitat preferences of mountain caribou. Several researchers have identified that early winter habitat of mature and old growth cedar-hemlock forests may constitute the most critical component for continued survival. McLellan *et al.* (1994) argued that emphasis should be placed on conserving early winter habitat based on potential conflict with timber harvesting activities within the Revelstoke forest region and suggested that Geographic Information Systems may provide a valuable tool for analyzing habitat distribution and connectivity within the study area. Several studies concerning woodland caribou in Quebec, Newfoundland and northeastern Alberta were then cited as examples of the utility of remote sensing, GIS, and landscape ecology techniques for caribou habitat analysis.

Recent research on remote sensing/GIS integration supports a combination of data sources and techniques to provide a more comprehensive analysis of environmental change. Recent classification algorithms applied in remote sensing applications include knowledge-based techniques. Such procedures have been shown to produce higher classification accuracies than statistical classifiers such as a maximum likelihood approach. Decision tree classifiers represent another form of knowledge-based classification that provide important advantages over conventional methods. For example, the Hybrid Decision Tree algorithm described by Friedl and Brodley (1997) allows the incorporation of different data types and classification algorithms in a single approach. These characteristics seem to support a Hybrid Decision Tree as an appropriate

method to incorporate various data sources acquired for the study area into a procedure for mapping critical habitat locations.

Research in the field of landscape ecology supports the application of spatial pattern metrics for quantifying environmental change. Several studies investigating the effects of human disturbances such as timber harvesting on forested environments have shown significant relationships between changes in landscape metrics and underlying ecological processes. A sample of landscape metrics commonly used to quantify the effects of forest fragmentation is evident within the literature. Specifically, these include measures quantifying mean patch size, interior core area, inter-patch distance, edge density, mean patch shape and interspersion and juxtaposition. A similar approach was taken in this research, as a select set of landscape indices was calculated and analyzed to measure the amount of change and fragmentation in suitable early winter caribou habitat patches between 1975 and 1997. This approach represents the first attempt to execute a long-term temporal analysis of caribou habitat change in the study area using a combination of spatial methodologies.

3.0 STUDY AREA AND DATA ACQUISITION AND PROCESSING

3.1 Introduction

This chapter provides a description of the study area and summarizes the methodology employed to collect and process the digital data sources required to achieve the first objective of this thesis: mapping and classification of habitat variables for the development of Habitat Suitability Index (HSI) models. Secondary sources of data include Landsat satellite imagery, a Digital Elevation Model, and various ancillary information. Primary data sources include important forest stand parameters and ground truth information collected in the field during the summer of 1998. The final section of this chapter provides a brief evaluation and comparison of the accuracy and utility of the forest inventory database provided by the British Columbia Ministry of Forests (BCMOF) for habitat unit classification.

3.2 Study Area

Figure 3.1 illustrates the study area to the north of Mount Revelstoke and Glacier National Parks, near Revelstoke, British Columbia (51° 24' N, 117° 44' W). The study area boundary corresponds to the home range of a herd of mountain caribou numbering approximately 140 animals (a small subpopulation of the larger Revelstoke herd of 400 animals). The region exhibits high relief as elevations vary from 610 to 2700 metres with treeline located at approximately 1980 m (McLellan *et al.* 1995). Forest cover is predominantly coniferous. According to the British Columbia Biogeoclimatic Ecosystem Classification System (Meidinger and Pojar 1993), the area falls within three biogeoclimatic zones depending on elevation. Lower slopes are in the wet-cool Interior-Cedar-Hemlock (ICHwk) zone and are dominated by western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and to a lesser extent Douglas fir (*Pseudotsuga menziesii*), western white pine (*Pinus monticola*), and white birch (*Betula papyrifera*). Mid elevations fall within the very wet cold Engelmann Spruce-Subalpine Fir zone (ESSFvc) and are dominated by Engelmann spruce (*Picea engelmannii*),

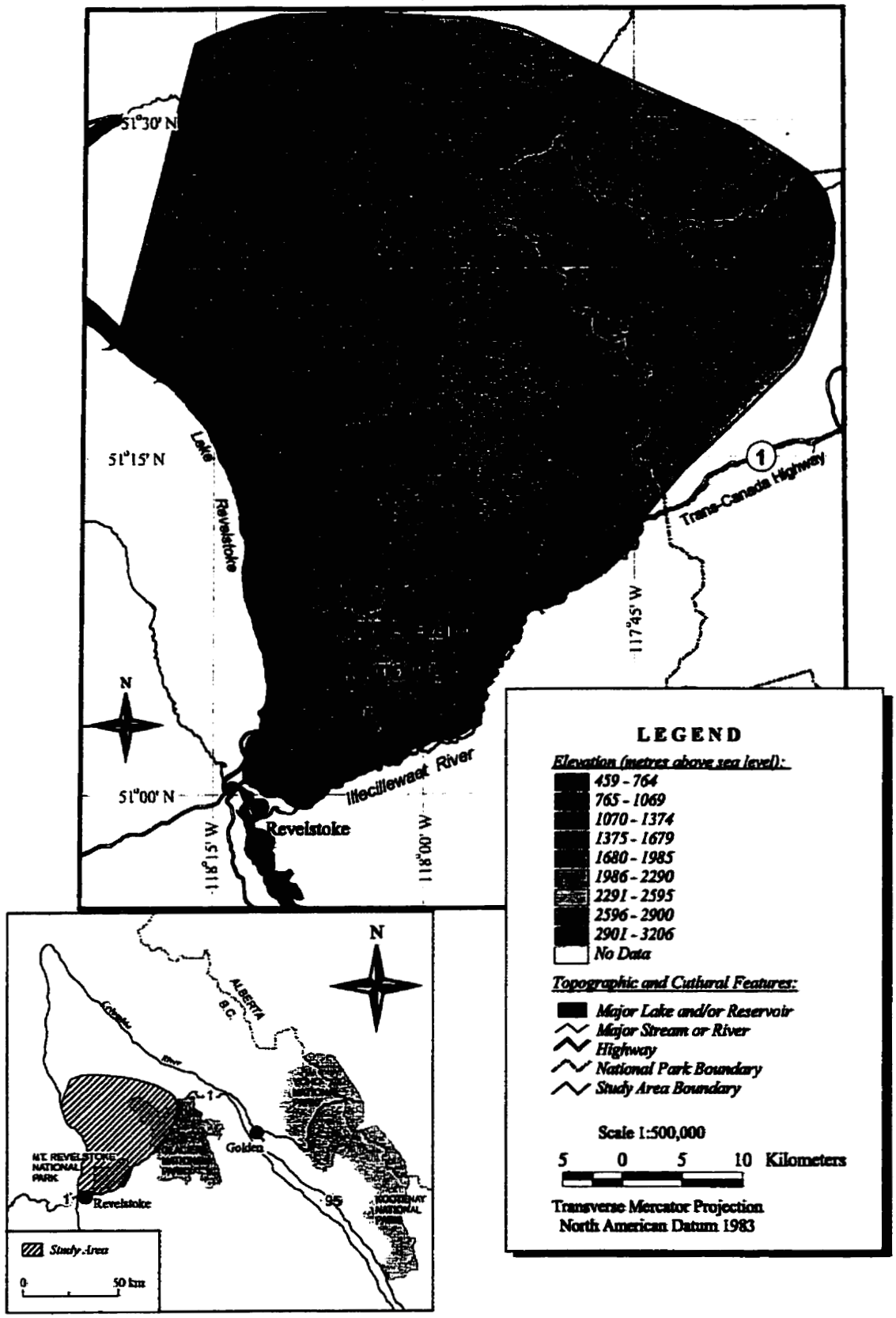


Figure 3.1 Study area map

subalpine or balsam fir (*Abies lasiocarpa*), and some mountain hemlock (*Tsuga mertensiana*). Higher elevations consist of open alpine parkland (very wet cold parkland Engelmann Spruce-Subalpine Fir zone – ESSFvcp) and are characterized by discontinuous stands of subalpine fir and some Engelmann spruce separated by open alpine meadows, rock, and glaciers. Avalanches are common in the study area and have established numerous paths or tracks upon the landscape (McLellan *et al.* 1995).

Two different natural disturbance regimes exist within the study area: NDT1 and NDT2 according to the B.C. Natural Disturbance classification (Government of British Columbia 1995b). NDT1 refers to Natural Disturbance Type 1 - Ecosystems with rare stand-initiating events, while NDT2 (Natural Disturbance Type 2) represents ecosystems with infrequent stand-initiating events. Contributing disturbance agents resulting in stand initiation are predominantly wildfires and windstorms and to a lesser degree avalanches, insects, and landslides. Active suppression of these agents, wildfires in particular, has been carried out for over 50 years to protect timber resources and communication and transportation routes (Peterson 1997).

The structural attributes of old growth stands in the study area seem to provide important habitat requirements for the local caribou population. Forest stands ranging from 180 to 200 years and older produce an environment conducive to the production of arboreal lichens. This vegetation component is the most important source of winter forage for mountain caribou populations (McLellan *et al.* 1995). Old growth stands are also the most economically valuable for timber harvesting. Thus, a conflict exists between economic and environmental values in these areas.

3.3 Digital Image Processing

3.3.1 Satellite Imagery

Two Landsat scenes were acquired to provide both a past and more recent view of the study area at the landscape level of analysis. A Landsat Multispectral Scanner (MSS) image captured the forested landscape in a near-baseline condition prior to intensive timber harvesting in the area. The MSS image was acquired on September 13, 1975 with a sun elevation of 38.84 degrees and azimuth of 148.70 degrees. A Landsat Thematic Mapper (TM) image with a sun elevation of 49.52 degrees and azimuth of 142.85 degrees was acquired for August 10, 1997 to represent the present landscape composition. Both images were acquired for the late summer season to minimize phenological differences in vegetation.

A subscene from each image roughly corresponding to the boundaries of the study area was extracted to reduce data storage and processing requirements. To ensure the greatest precision and accuracy when comparing the satellite imagery to existing GIS and ancillary data sources, pixel locations for the 1997 TM image were georeferenced or matched to the corresponding UTM coordinates of digital vector topographic features such as roads, streams, and lakes. The TM subscene was geocorrected to a NAD83 UTM projection using 52 ground control points. A second-order nearest neighbour resampling algorithm was employed yielding a Root Mean Square (RMS) value of 0.375 and a pixel ground resolution of 25 metres. An image-to-image correction was then applied to register the MSS subscene to the geocorrected TM imagery. A RMS value of 0.66 was achieved using 58 ground control points in a second-order nearest neighbour algorithm. The MSS imagery was resampled to match the 25 metre pixel resolution of the TM scene. It should be noted that resampling the original 80 metre pixel resolution of the MSS imagery to 25 metres did not actually improve the effective resolution of this data source. As will be explained in section 4.3.3 of this thesis, the coarser resolution MSS imagery was resampled to match the 25 metre TM imagery to more accurately map smaller objects

such as alpine tundra and avalanche communities that have remained unchanged over the 22 year period of study.

Related research on mapping of forested environments suggested that the Tasseled Cap components of Brightness and Greenness were useful for the classification of land cover and landscape changes between Landsat TM and MSS data (Zheng *et al.* 1997; Cohen *et al.* 1998). Based on these recommendations, a Tasseled Cap transformation was applied to the geocorrected radiance values for the four MSS spectral channels to produce a Brightness and Greenness component (Kauth and Thomas 1976). The TM bands 1 through 5 and band 7 were transformed to the Brightness, Greenness, and Wetness indices (Crist and Cicone 1984). The spectral radiance data for each image was also converted to reflectance channels using the PCI atmospheric correction algorithm (Richter 1990; PCI Inc. 1997) to correct for the effects of atmospheric scattering, attenuation, and ground scattering.

3.3.2 Digital Elevation Model

Previous caribou habitat models created for the study area (McLellan *et al.* 1994, 1995) have identified elevation and slope as important determining variables. This information may be extracted from a Digital Elevation Model (DEM) by using a surface analysis software package such as PCI or Arc/Info. B.C. Ministry of Environment, Lands, and Parks 1:20,000 scale TRIM (Terrain Resource Information Mapping) maps were used to create a 25 metre resolution raster DEM. According to the *British Columbia Specification & Guidelines for Geomatics* (Government of British Columbia 1992), 90 percent of all discrete spot elevations in a TRIM file are accurate within 5 metres of their true elevation. Similarly, 90 percent of all interpolated contour features are within 10 metres of their true elevation.

The Arc/Info GIS software package was used to interpolate a triangulated irregular network (TIN) surface from contour elevation data, an irregular grid of elevation points,

and hard and soft elevation breaklines. Each TIN was then converted to a GRID or raster elevation file. This procedure was repeated for each of the approximately 30 TRIM map sheets covering the study area. All 30 grid files were then edge-matched to create one large DEM for the research area. The study area boundary was then used to extract or "clip" out the region of interest. Derivative variables of slope and aspect were calculated for each 25 metre pixel from the DEM using the PCI image analysis software package.

3.3.3 Hydrological and Cultural Features

The 1:20,000 scale TRIM planimetric files were converted to Arc/Info format for each map sheet. Hydrological features including streams and lakes, natural features such as glaciers and cultural features including primary and secondary roads were extracted from each TRIM map sheet. Individual planimetric Arc/Info files were joined to create one large tile of the study area for each theme of interest. This data set was useful for geocorrecting the satellite imagery to the NAD 83 UTM projection of the TRIM database and for logistical planning of field sampling.

3.3.4 Forest Cover Maps

The B.C. Ministry of Forests (Revelstoke) provided 27 forest cover maps at 1:20,000 scale in digital form and associated attribute database files for the study area. Forest stand polygons and access roads were extracted from each file and converted from Microstation IGDS format to Arc/Info format using the Arc IGDSARC conversion tool. The BCMOF forest attribute database files were converted to INFO format files and linked to the forest stand polygon features. Each forest stand could then be queried for such attributes as species composition, stand height, stand age, crown closure, and disturbance history. All 27 forest coverages were then joined to create one large tile for the study area. This particular data source is referred to as the BCMOF forest stand inventory database.

3.4 Field Data Collection

The main objectives of the field campaign were three-fold: 1) to verify the accuracy and determine the utility of the BCMOF forest stand inventory database for the development of training areas for image classification and caribou habitat suitability modeling, 2) to record tree species and biogeoclimatic ecosystem zone designation for image post-classification accuracy assessment, and 3) to collect information on forest stand parameters for structural classification. The first two objectives deal specifically with the content of this thesis, while the third objective was initiated as part of a related research project focusing on empirical relations between forest structural parameters and satellite imagery. Specifically, the focus was to investigate the potential of the Landsat TM Wetness component as a predictive measure of forest structural complexity in the study area. The reader is referred to Deuling *et al.* (1999) for a complete discussion of the forest structural classification project, and for a more detailed description of the field methodology. For the purposes of this thesis, only relevant sections of the field sampling protocol are discussed.

3.4.1 Measurement of Forest Stand Parameters

A stratified random sampling strategy was used to select 100 suitable stands for estimating forest stand parameters from the sample of all forest stands contained within the BCMOF forest stand inventory database. Due to logistical and time limitations, only 37 of the 100 randomly selected stands were located in the field using a Trimble GPS unit. Within each stand, three variable size plots were established through a basal area prism method (Lukai and Lukai 1984). Data for 111 field plots in total was collected (3 plots per 37 forest stands). Once a stand selected for sampling was located in the field, the first plot was established 75 metres in from the stand boundary with the second and third plots located 150 metres apart on a constant arbitrary compass bearing (Figure 3.2). This protocol was designed to avoid a situation in which image values would be erroneously related to adjacent stand parameters when carrying out statistical analysis

with field measurements. A differential GPS location was recorded at each plot centre along with an estimate of elevation (measured from an altimeter), aspect, and slope (calculated using a Bunton field compass). From the plot centre, a basal area prism sweep determined which trees to include in the sample. Appropriate prism factors for the study area varied between 4, 6, and 9. A fixed plot of 25 metres by 25 metres in area was established in regenerating or parkland sites where basal area prisms returned a sample size less than six trees. Each tree selected was numbered in sequence counter-clockwise from 180° south. The species was recorded for each tree in the sample. From the complete plot sample, three trees were chosen at random and tree height was measured to the nearest metre for each using an inclinometer. One age sample core was taken per plot from a tree that appeared to be typical of other trees in the stand. Averaging all tree height and age measures per site then derived a plot level database, and an average of plot level variables (three per stand) produced a stand level sample. The most abundant or primary species, secondary and tertiary species per plot and per stand were recorded. Information describing the composition of the understory vegetation was also noted for each plot location to aid in biogeoclimatic zone classification. The biogeoclimatic zone per field plot was determined by comparing the elevation, aspect, tree species and understory species composition to those outlined in the *B. C. Guide for Site Identification and Interpretation* (Braumandl and Curran 1992). A stand level biogeoclimatic zone represented the majority designation of all three plots within a stand. For example, if two of three plots for a stand were considered to fall within the cedar-hemlock (ICHwk) zone and one plot was designated as ESSFvc, then the stand would be considered to be characteristic of the ICHwk zone. Table 3.1 lists and describes the individual tree, plot, and stand level measurements.

Table 3.1. Tree, plot and stand-level field measurement methodology

Individual Tree Measures	Description / Method of Measurement
Species	Recorded per tree
Age	One tree per plot – collected with increment bore Successful collection dependent on tree diameter relative to bore size
Height	Three trees per plot measured by inclinometer

Plot Level Variables	Description / Method of Measurement
GPS location	UTM Easting and Northing in differential mode
Elevation	Measured by altimeter
Slope	Measured by compass
Aspect	Measured by compass
AGE_{GIS}	Stand age extracted from BCMOF forest stand inventory
HGT_{MN}	Average height of three trees per plot
HGT_{SD}	Standard deviation of three height values per plot
HGT_{U}	Average height of upper layer trees per plot Measured by inclinometer
Shrub	% Shrub cover - 25 m. radius around plot centre
Grass	% Grass/Forbes cover - 25 m. radius around plot centre
Moss	% Moss cover - 25 m. radius around plot centre

Stand Level Variables	Description / Method of Measurement
SPC1, SPC2, SPC3	Primary, secondary, and tertiary species type per stand
BGCZ	Biogeoclimatic zone designation based on elevation, aspect, tree species and understory composition
AGE_{GIS}	Stand age extracted from BCMOF forest stand inventory
HGT_{MN}	Average height of nine trees per stand (3 per plot)
HGT_{SD}	Standard deviation of nine height values per plot
HGT_{U}	Average height of upper layer trees per three plots
Shrub	Average % Shrub cover for three plots
Grass	Average % Grass/Forb cover for three plots
Moss	Average % Moss cover for three plots

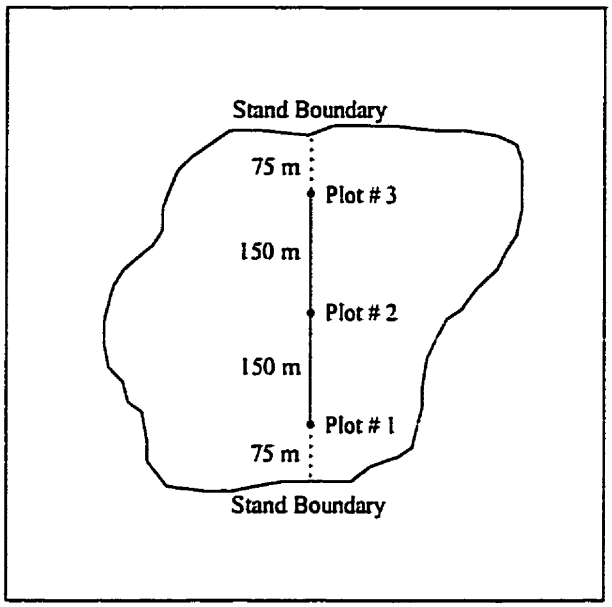


Figure 3.2 Field plot locations within a forest stand

3.4.2 Comparison of Field and BCMOF Forest Stand Parameters

The BCMOF forest stand inventory database described in section 3.3.4 contained information on tree species composition, stand age, stand height, and disturbance classifications for the Revelstoke forest region. This data source could be particularly helpful for developing training areas for the image classification stage and assignment of caribou habitat suitability ratings. However, the attribute accuracy of features within the forest inventory database was unknown. The forest cover data are based on an inventory completed in 1976. Although annual updates were made regarding road construction and harvesting operations and 1997 projections for stand age and height have been included, most stand parameters represent conditions that existed over twenty years ago. To confidently incorporate species composition, height, and age information from the BCMOF database into habitat suitability modeling and image classification training, a comparison was made between stand level variables measured in the field and the BCMOF stand parameters.

Executing a point-in-polygon overlay between the field plot locations and the BCMOF forest inventory stand attributes created a stand level database for comparison of the two data sources. Stand age, height, primary, secondary, and tertiary species information was extracted from the BCMOF inventory database for each of the stands visited in the field. Unfortunately, the BCMOF database did not contain understory vegetation information and the biogeoclimatic zone designation was estimated based on the elevation and aspect range and tree species composition of the stand. For example, if the primary and secondary species of a stand polygon were western red cedar and hemlock, then the stand was considered to be characteristic of the ICHwk zone.

Table 3.2 compares the field data for this thesis and BCMOF forest inventory database measurements for stand age, and stand height for 37 sample stands. The RMS value for stand age and height represents the root-mean-square error and provides a measure of the variability of the BCMOF forest stand inventory derived values about the field-measured height and age data. Table 3.3 compares the species composition and biogeoclimatic zone designation between the field and BCMOF forest inventory database. The percent agreement between primary, secondary, and tertiary species and the biogeoclimatic zone labels based on the species composition is also provided to indicate the correspondence between forest cover type.

A comparative analysis between the field and BCMOF forest inventory stand parameters is now provided in order to validate the use of the BCMOF forest inventory database in the subsequent image classification methodology discussed in Chapter 4. The RMS value for stand age reported in Table 3.2 is 68.27 years and illustrates the problematic collection of age core samples during the field campaign. Subalpine fir trees cored in the field were most often soft or rotten providing inaccurate and uncertain age values. Immature western red cedar and western hemlock trees were easily sampled, however mature cedar and hemlock specimens were either too large in diameter for the bore device, or the interior sapwood was hollow. Thus, it is possible that the high level of error is more an indication of the complications encountered obtaining valid age core

Table 3.2 Comparison of field and BCMOF measures of stand age and height

	Stand Age (yr.)				*Stand Height (m)*			
Stand ID	**Field**	**BCMOF**	**Residual**	**Residual²**	**Field**	**BCMOF**	**Residual**	**Residual²**
1	71	87	16	256	23.08	24.8	1.72	2.95
2	144	152	8	64	22.11	37.0	14.89	221.68
3	133	152	19	348	27.56	37.0	9.44	89.20
4	69	182	113	12769	27.61	47.3	19.69	387.65
5	57	77	20	413	19.11	16.8	2.31	5.34
6	118	45	73	5256	18.17	13.2	4.97	24.67
7	99	161	62	3885	15.83	16.5	0.67	0.44
8	26	32	7	42	10.34	11.9	1.56	2.42
9	-	287	-	-	30.11	30.2	0.09	0.01
10	-	257	-	-	26.72	29.3	2.58	6.64
11	-	307	-	-	38.83	36.2	2.63	6.95
12	84	97	13	169	25.76	31.2	5.44	29.64
13	113	267	155	23870	31.22	30.2	1.02	1.04
14	-	327	-	-	29.33	40.2	10.87	118.08
15	212	237	25	625	35.44	35.2	0.24	0.06
16	85	67	18	324	26.22	23.7	2.52	6.36
17	-	267	-	-	30.17	27.2	2.97	8.80
18	184	307	123	15129	34.83	32.2	2.63	6.93
19	-	307	-	-	31.06	37.2	6.14	37.75
20	-	307	-	-	29.22	35.2	5.98	35.73
21	130	140	10	100	15.92	26.1	10.18	103.70
22	125	224	100	10000	20.89	25.7	4.81	23.15
23	-	224	-	-	30.56	34.4	3.84	14.78
24	70	180	110	12100	15.33	23.7	8.37	70.00
25	-	224	-	-	32.67	34.4	1.73	3.00
26	180	224	44	1936	38.83	34.4	4.43	19.65
27	-	324	-	-	36.06	41.9	5.84	34.16
28	-	224	-	-	26.78	25.1	1.68	2.81
29	-	324	-	-	33.44	33.8	0.36	0.13
30	-	324	-	-	32.00	31.7	0.30	0.09
31	-	324	-	-	35.06	33.8	1.26	1.58
32	108	57	51	2601	22.56	15.5	7.06	49.78
33	67	97	31	930	23.33	25.0	1.67	2.78
34	-	-	-	-	37.67	36.7	0.97	0.93
35	-	89	-	-	20.56	31.0	10.44	109.09
36	208	224	16	245	16.78	28.6	11.82	139.76
37	138	220	83	6806	20.78	25.1	4.32	18.68
			Sum	97871.03			Sum	1586.43
			Mean	4660.53			Mean	42.88
			RMS Age	68.27			RMS HGT.	6.55

RMS Error:
Stand age 68.27 yr.
Height 6.55 m

Table 3.3 Comparison of field and BCMOF tree species composition and biogeoclimatic zone designation

	SPECIES						*BGCZ*	
	Field			BCMOF			Field	BCMOF
Stand ID	SPC1	SPC2	SPC3	SPC1	SPC2	SPC3		
1	PW	FD	BL	AT	FD	PW	ICHwk	ICHwk
2	BL	HM	SE	BL	SE	-	ESSFvc	ESSFvc
3	BL	SE	FD	BL	SE	-	ESSFvc	ESSFvc
4	CW	HW	PW	HW	CW	-	ICHwk	ICHwk
5	PW	AT	FD	AT	FD	PW	ICHwk	ICHwk
6	BL	HW	SE	BL	SE	HM	ESSFvc	ESSFvc
7	BL	SE	HW	BL	-	-	ESSFvc	ESSFvc
8	HW	PW	CW	HW	CW	-	ICHwk	ICHwk
9	HW	B	SE	HW	BL	SE	ICHwk	ICHwk
10	BL	SE	-	SE	BL	-	ESSFvc	ESSFvc
11	CW	HW	-	CW	HW	-	ICHwk	ICHwk
12	FD	HW	PW	HW	FD	CW	ICHwk	ICHwk
13	CW	HW	-	HW	CW	-	ICHwk	ICHwk
14	HW	CW	-	HW	CW	SE	ICHwk	ICHwk
15	HW	SE	BL	SE	BL	HM	ICHwk	ESSFvc
16	FD	CW	HW	FD	HW	CW	ICHwk	ICHwk
17	HW	B	SE	HM	-	-	ICHwk	ESSFvc
18	HW	CW	-	HW	CW	-	ICHwk	ICHwk
19	HW	CW	-	HW	CW	-	ICHwk	ICHwk
20	HW	CW	-	HW	CW	-	ICHwk	ICHwk
21	BL	-	-	BL	SE	-	ESSFvc	ESSFvc
22	BL	SE	-	SE	BL	-	ESSFvc	ESSFvc
23	HW	CW	FD	HW	CW	-	ICHwk	ICHwk
24	BL	MH	SE	BL	-	-	ESSFvc	ESSFvc
25	HW	SE	-	HW	-	-	ICHwk	ICHwk
26	HW	CW	SE	HW	CW	-	ICHwk	ICHwk
27	CW	HW	-	CW	HW	-	ICHwk	ICHwk
28	HW	CW	-	HW	CW	-	ICHwk	ICHwk
29	CW	HW	FD	CW	-	-	ICHwk	ICHwk
30	CW	HW	-	CW	SE	-	ICHwk	ICHwk
31	CW	HW	SE	CW	HW	-	ICHwk	ICHwk
32	FD	HW	PW	FD	HW	SE	ICHwk	ICHwk
33	FD	SE	CW	FD	PW	AT	ICHwk	ICHwk
34	HW	CW	SE	HW	CW	-	ICHwk	ICHwk
35	FD	SE	CW	FD	SE	PW	ICHwk	ICHwk
36	BL	SE	-	BL	SE	-	ESSFvc	ESSFvc
37	BL	-	-	BL	SE	-	ESSFvc	ESSFvc

Species Accuracy:

Primary Species (SP1)	66.66%
Secondary Species (SP2)	40.00 %
Tertiary Species (SP3)	4.55 %

BGCZ Accuracy: 94.87%

BGCZ = BioGeoClimatic Zone designation

Species codes:

AT = Trembling Aspen (*Populus tremuloides*)
CW = Western Red Cedar (*Thuja plicata*)
HW = Western Hemlock (*Tsuga heterophylla*)
PW = Western White Pine (*Pinus Monticola*)
BL = Subalpine Fir (*Abies lasiocarpa*)
FD = Douglas Fir (*Pseudotsuga menziesii*
HM = Mountain Hemlock (*Tsuga mertensiana*)
SE = Engelmann Spruce (*Picea engelmanii*)

samples in the field than the BCMOF forest inventory database itself. Because valid age samples of old growth stands could not be collected (due to the large tree bole diameter compared to the bore device), visual comparison between the stand age attribute on the BCMOF forest inventory map product and the structural characteristics of the stand in the field served as the only verification of the accuracy of the BCMOF data products. Following this alternative method, old growth stands generally seemed to conform to the age class assigned in the BCMOF forest inventory. Unfortunately, due to the complications outlined above, the early winter caribou habitat suitability model would have to be based on the BCMOF stand age information without full comprehension of the accuracy of this data source.

The RMS value reported in Table 3.2 for BCMOF forest inventory stand height is 6.55 metres. This level of error is relatively high considering the height range of all stands measured in the field was only 28.49 metres. One possible source for the low correspondence between the BCMOF forest inventory and field height values might be the differences in sampling protocol between the BCMOF forest inventory and the field methodology of this project. Stand height for the BCMOF forest inventory is estimated based on the average height of the dominant and codominant trees of the primary species of a particular forest stand (Gillis and Leckie 1993). Stand height measurements collected in the field for the forest structural classification were not limited to the primary species of the stand. For example, if a field plot was located in a mixed stand consisting of 60% Engelmann spruce and 40% subalpine fir, then the height values for both species would be averaged to provide a stand height value. Mature subalpine fir trees were found to be generally shorter than Engelmann spruce trees of the same age (Deuling *et al.* 1999), thus the average stand height would be lower than if only the leading species (Engelmann spruce) had been included in the stand height calculation (as is the case in the BCMOF forest inventory database).

The accuracy level of the BCMOF primary, secondary, and tertiary species type in Table 3.3 is 66.66%, 40.00%, and 4.55% respectively. It is possible that the low correspondence between field and BCMOF inventory species cover type may be a result of the small number of sample plots established in each stand. By only collecting species data for three plots per stand, the variability of species composition for the whole stand may not have been estimated effectively. However, by comparing the field and BCMOF primary and secondary species type listed for each stand in Table 3.3, it seems that the field sampling protocol corresponds reasonably well to the general species composition reported in the BCMOF forest stand attributes. Even though there is not a direct correspondence between primary and secondary species designation in many cases, there is a general agreement between the types of species found within each stand. For example, stands 4, 10, 12, and 22 in Table 3.3 report a disagreement between primary and secondary field and BCMOF forest inventory species type; however, the same two species types (SPC1 and SPC2) are listed for each stand, there is only confusion over which species is dominant. This indicates that the field sampling may have been more successful at capturing the general species composition characteristics of each stand (the types of species present) rather than the exact percentage of each species type.

By focusing on the agreement between the types of species reported in the BCMOF database and in the field, the BCMOF forest inventory stand data seems to provide a good indication of the type of vegetation community that exists on the ground. Based on this interpretation, the BCMOF forest stand data may be more suitable for collecting training areas at the forest community level of analysis, such as a Cedar-Hemlock versus a Spruce-Balsam forest community, rather than for discriminating between primary and secondary species type per stand. The high level of agreement between the field and BCMOF biogeoclimatic zone (94.87%) supports this recommendation and validates the decision (discussed in section 4.2) to map caribou habitat units according to biogeoclimatic zone rather than by leading tree species.

3.5 Summary

The acquisition of primary and secondary data sources was required to facilitate the mapping and classification of habitat variables and the development of habitat suitability models for the 1975 and 1997 time periods. Secondary digital data sources collected for the study area include Landsat satellite imagery for 1975 and 1997, a digital elevation model, digital vector files of hydrological and cultural features, and BCMOF forest stand inventory information. Field data were collected in order to verify the accuracy of the BCMOF forest inventory data and collect ground truth information for image classification accuracy assessment.

One important objective of the field campaign was to determine the BCMOF forest inventory database attribute error of important habitat suitability variables (stand age, height, and tree species). Each data source could then be more confidently incorporated into the habitat suitability models and image classification training stage outlined in the methodology section that follows. Several logistical complications were encountered during the collection of tree age samples and prevented the calculation of a reliable and quantifiable measure of stand age uncertainty. However, a visual comparison between field structural characteristics and the BCMOF stand age class indicated a general correspondence between the data sources. Based on the importance of stand age in determining early winter habitat suitability (Scott and Servheen 1985; Rominger and Oldemeyer 1989; McLellan *et al.* 1994, 1995), stand age should be included in a habitat suitability model. Therefore, the BCMOF forest inventory stand age information was incorporated into the HSI models without an empirical measure of attribute uncertainty. The observed root-mean-square error (RMS) value for stand height was also large, indicating high variability between stand height measurements collected in the field and the BCMOF inventory stand height attribute values for the same locations. A partial explanation for these differences may be the field sampling protocol. Based on the high attribute uncertainty for the BCMOF inventory stand height (and other reasons discussed

in section 4.2), the inclusion of a stand height variable within a habitat suitability model may introduce unacceptable levels of error into the analysis.

Finally, a high level of agreement between general species composition and biogeoclimatic zone was reported for the field and BCMOF forest inventory data sources. This suggests that the BCMOF forest stand database may be applied more confidently as image training data at the biogeoclimatic ecosystem level of analysis, rather than at a more detailed species classification level.

4.0 LANDSCAPE ANALYSIS METHODOLOGY

4.1 Introduction

This chapter outlines the methodology developed to address two research objectives. First, to address research objective 1, an integrated GIS/remote sensing classification procedure for mapping habitat variables and the development of a Habitat Suitability Index (HSI) model for 1975 and 1997 is described. The procedures employed for the calculation and comparative analysis of caribou habitat fragmentation levels for each time period, as related to research objective 2, are then summarized. Figure 4.1 provides an overview of the research methodology and illustrates the relationships between the different components.

First, the suitability or value in terms of caribou habitat quality was assessed for each spatial feature within the study area for each time period. One common approach for quantitative evaluation of habitat is the development of a Habitat Suitability Index (HSI) model (U.S. Fish and Wildlife Service 1981; Rickers *et al.* 1995). Such habitat models are developed to predict the value of a particular habitat based on a species' observed preference or use of different land cover types. Section 2.2 of this thesis summarized an HSI model developed for mountain caribou in the study area by McLellan *et al.* (1995). Elevation, slope, land cover, stand age, and height were determined to be important habitat variables. Step 1 in Figure 4.1 illustrates the extraction of each of these habitat variables from multiple digital data sources including the BCMOF forest stand inventory database, the digital elevation model, and Landsat MSS and TM imagery. Each variable was then used to calculate 1997 habitat suitability for caribou during the early winter months of October through December, following the HSI model developed by McLellan *et al.* (1995). In Step 2 of Figure 4.1, a change detection procedure, referred to as "backcasting" for the purposes of this research, was applied to project backward in time and produce images representing 1975 habitat units and forest stand age.

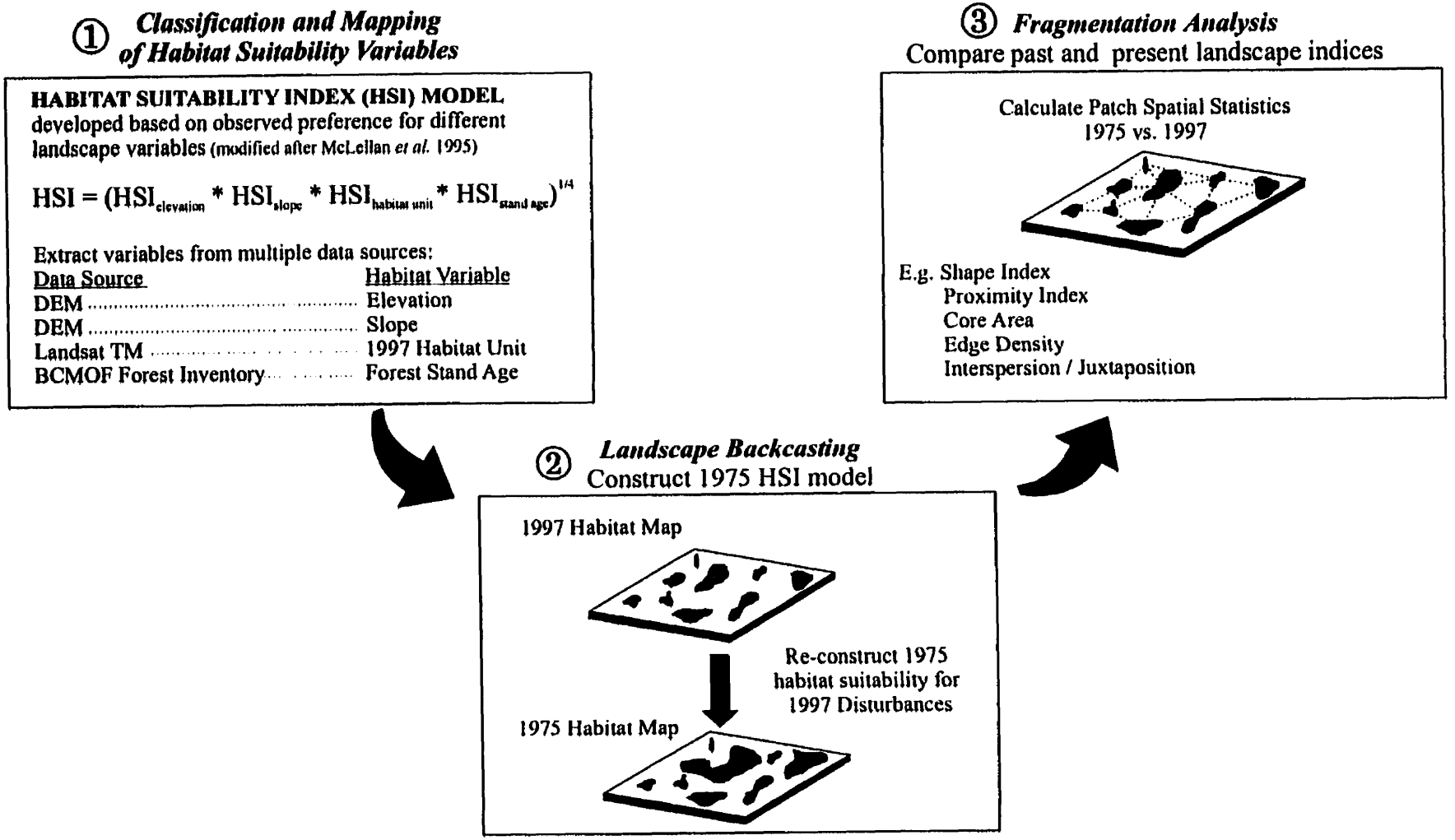


Figure 4.1 Overview: Landscape analysis methodology

This allowed the direct comparison of habitat suitability class areas and fragmentation levels between time intervals. The "backcast" 1975 habitat unit and stand ages images, along with elevation and slope data, were used to develop the 1975 HSI model. Finally, in Step 3 of Figure 4.1, habitat fragmentation was quantified by calculating a selection of spatial metrics for each HSI model. The metrics were then used to compare the composition and spatial configuration of early winter caribou habitat in 1975 and 1997 in order determine if habitat fragmentation has occurred in the study area.

4.2 Classification and Mapping of Habitat Variables (1997 Landscape)

Based on the findings and HSI coefficients developed by McLellan *et al.* (1995), the following equation was used to assign habitat suitability indices to spatial features for each temporal landscape (1975 and 1997):

$$\mathbf{HSI_{total} = (HSI_{elevation} * HSI_{slope} * HSI_{habitat\ unit} * HSI_{stand\ age})^{1/4}} \qquad \text{Eq. 1}$$

The total or final HSI value for any location within the study area represents the geometric mean of HSI values for each habitat variable. The equation listed above has been modified from the original model by replacing the HSI variable for the primary tree species ($HSI_{species}$) with the HSI coefficients for the habitat unit ($HSI_{habitat\ unit}$) and by removing the stand height variable (HSI_{height}) of a forest stand.

Habitat units classified from Landsat MSS and TM imagery replaced primary tree species due to the spectral resolution of the satellite sensors. Primary tree species of forest stands could not be mapped within acceptable accuracy levels from the digital imagery and the BCMOF forest inventory database only provides species composition information for stands with potential commercial value. Mapping the suitability of a range of habitat units with and without commercial value would seem to provide a more complete analysis of habitat distribution within the study area. Habitat classification of digital

satellite imagery may also provide other advantages over the use of the BCMOF forest inventory database. Some of these include:

1. Low cost of satellite image acquisition relative to aerial photo and digital imagery. The price of one Landsat TM image is substantially lower than the cost of flight planning and purchase of hundreds of aerial photos or several airborne spectral images required to cover the study area and update the BCMOF forest inventory.

2. Temporal coverage of Landsat satellite imagery for historical studies of landscape change. The longevity of the Landsat satellite program, which was initiated with the launch of Landsat-1 in 1972, and an orbit rotation of 16 days (for MSS) and 18 days (for TM) has served to establish a substantial historical archive of digital imagery for every point on the globe. The BCMOF forest inventory database has not maintained the same historical records. For example, many of the harvest and burn areas occurring in the study area do not have pre-disturbance information such as the previous land cover type, stand age etc. included in the database. For the purposes of this thesis, this shortcoming of the BCMOF database (and any traditional forest inventory) required the acquisition of the 1975 MSS satellite imagery to determine the land cover type of disturbance areas occurring between 1975 and 1997.

3. Uniform standards across the study area. The development of the HSI models required two similar data sets characterizing the land cover or habitat units of the study area at two different time periods (1975 and 1997). The comparison of the 1975 MSS satellite imagery (a raster data source) and the BCMOF forest inventory database for 1997 (a vector data source) raises concerns over the calculation of HSI class areas and the sensitivity of landscape metrics derived from different data sources. Several researchers have addressed the issue of metric sensitivity to a raster or vector data format (Haines-Young and Chopping 1996; Wickham *et al.* 1996) and recommendations have been made to compare landscape metrics derived from the same data format (either raster or vector) (McGarigal and Marks 1995; Franklin *et al.*

1999). The decision was made to include raster format habitat classifications derived in a similar manner for each time period. The use of Landsat satellite imagery facilitated comparable habitat classification between time periods.

Based on these considerations, the primary tree species variable included in the HSI model developed by McLellan *et al.* (1995) was replaced by a habitat unit variable that could be mapped for each time period in a manner that was consistent and provided complete coverage of the study area. The use of satellite imagery could represent a low cost solution for mapping habitat units consistently across the study area at different temporal intervals.

The height of a forest stand was eliminated from the original HSI model developed by McLellan *et al.* (1995) due to BCMOF forest inventory attribute uncertainty issues (see section 3.4.2) and other limitations of the BCMOF inventory database and the limitations of the satellite data. Stand height values for 1997 could be extracted from the BCMOF database but were not included for 1975. Linear regression equations were developed based on stand field variables to predict height based on the age of a forest stand in 1975 (Table 4.1). However, the Standard Error of the Estimate (SEE) for the model was high (4.676 metres) and the regression equation tended to overestimate the height of young regenerating stands. Such inaccuracies supported the exclusion of this variable from the model.

Table 4.1 Linear regression model for 1975 stand height

Regression Model Summary	
Sample size (n)	= 37
Dependent Variable	= Stand Height (HGT75)
Predictors (Constant)	= Stand Age (AGE75)
Pearson's Coefficient (R)	= 0.792 *
Regression Coefficient (R^2)	= 0.628
Adjusted R^2	= 0.616
Standard Error of Estimate	= 4.676
Regression Equation	HGT75 = AGE75(0.0611) + 15.923

Each of the HSI variables used to calculate the 1997 HSI model (elevation, slope, habitat unit, and stand age) were extracted from the project database discussed in section 3. A separate raster Arc/Info GRID image was created for each habitat variable. Elevation and slope images were derived from the DEM for the study area. Stand ages for 1997 were projected from the BCMOF forest inventory "established year" attribute which represents the year of stand initiation. The remaining HSI variable (Habitat Unit) for 1997 was derived from a decision tree classification algorithm.

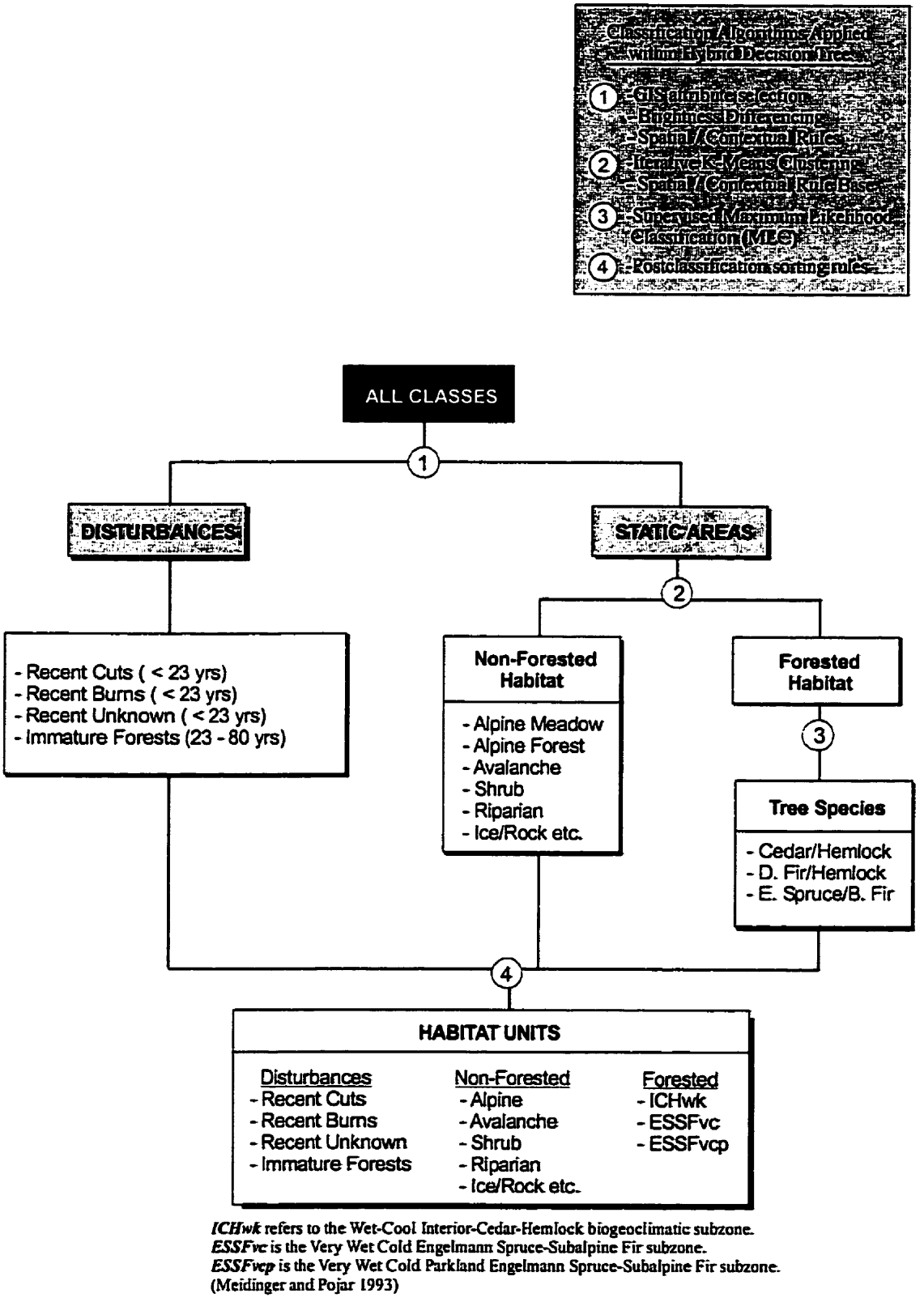
4.2.1 Hybrid Decision Tree Algorithm for Habitat Unit Classification

The classification scheme used to map habitat units for 1997 and 1975 is outlined in Figure 4.2. The classification procedure involved the integration of GIS and remote sensing analysis in a Hybrid Decision Tree (HDT) algorithm as outlined by Friedl and Brodley (1997).

Figure 4.2 illustrates the application of four general components within the decision tree structure to separate spatial features (contiguous pixels of the same cover type) into the appropriate class designation.

1. Disturbance classes were identified using a combination of GIS attribute selection and a brightness differencing technique. This separated the study area into two classes, disturbed and static areas. Disturbances reflected a change in land cover between 1975 and 1997 while static areas were assumed to have remained unchanged.

2. Static areas were then entered into a K-Means unsupervised classification algorithm. The output pixels from this algorithm were then assigned a class value representing different land cover types based on a spatial/contextual rule base.



ICHwk refers to the Wet-Cool Interior-Cedar-Hemlock biogeoclimatic subzone.
ESSFvc is the Very Wet Cold Engelmann Spruce-Subalpine Fir subzone.
ESSFvcp is the Very Wet Cold Parkland Engelmann Spruce-Subalpine Fir subzone.
(Meidinger and Pojar 1993)

Figure 4.2 Hybrid Decision Tree classification scheme

3. Features identified as forested areas in the above procedure were then split into tree species composition classes by a Maximum Likelihood Classification (MLC) algorithm.

4. All three components (disturbances, non-forested and forested habitat units) were merged and classified pixel values were passed through a final algorithm that applied aspect and elevation rules to distinguish between biogeoclimatic zones and assign a final habitat unit class designation.

The sections that follow discuss each component of the Hybrid Decision Tree algorithm in greater detail. The end result of this procedure is a map representing habitat units in 1997, which is assessed for accuracy in section 4.2.1.5.

4.2.1.1 Disturbance Classification

Disturbance features existing in 1997 were identified using a combination of GIS attribute selection and image-based change detection methods. The forest cover maps provided by the B. C. Ministry of Forests contained disturbance features that had been initiated or created before 1995. Disturbances occurring between 1995 and 1997 had not been included in the database and had to be mapped using the satellite imagery in a separate procedure.

Disturbances occurring before 1995 were extracted from the BCMOF forest cover maps and classified as recent cuts, recent burns, recent disturbances with an unknown agent, or immature forests. Recent cuts were identified by selecting those stands having an ACTIVITY code of "L" indicating logging and a projected 1997 age less than 23 years of age (occurring since 1975). Recent burns had an ACTIVITY code of "B" for burn and a 1997 stand age of less than 23 years. Recent disturbances with an unknown disturbance agent had no ACTIVITY code within the database, but had a stand age of less than 23 years. These areas were relatively few and visual assessment of a sample of features

suggests that most represent burns not coded as such in the BCMOF inventory database. The immature forest class was represented by any stand having a 1997 age of 23 to 80 years.

Disturbances (cuts and burns) occurring between 1995 and 1997 were isolated by applying a TM brightness differencing technique (Muchoney and Haack 1994; Cohen *et al.* 1998) using Arc/Info GRID. The Brightness component of the TM Tasseled Cap is a linear combination of all six TM bands and responds primarily to changes in soil reflectance (Lillesand and Kiefer 1994). Forested stands in the study area tend to have low brightness values relative to recent harvest areas in which a large amount of soil is exposed. Therefore, it is expected that a stand harvested between 1975 and 1997 would have a low Brightness value in 1975 as a forested area and a high Brightness value in 1997 as a recent clear cut. To isolate such areas, a "Brightness Difference" image was created by subtracting the 1975 MSS Brightness image from the 1997 TM Brightness image (BD97 = B97 – B75). Pixels having a high difference value were isolated as being potential cuts or burns.

Two sources of error complicated the selection of disturbances using a simple difference threshold:

- errors due to misregistration between the TM and MSS image and
- errors due to illumination differences

These errors resulted in several pixels that did not represent cuts or burns having high difference values. For example, if a pixel was cast in shadow in 1975 due to the sun angle the Brightness value would be low, while if the same location was illuminated in 1997, a high brightness value would be recorded. Consequently, the Brightness difference for the pixel would also be high. A distinction was made between these erroneous regions of change and "true" change features by applying a contextual rule base (summarized in Table 4.2) to areas having high Brightness difference values.

Cuts and burns in the study area tended to have a unique spatial or contextual signature that was different from areas of misregistration or illumination differences. They were larger in areal extent, had a distinct elevation range and pattern or shape. Recent cuts were found in close proximity to forestry roads and other harvest areas within the study area. These expected characteristics were used to construct a contextual rule set for isolating distinct areas of change. The rule set was constructed following a quantitative approach in that decision rules were constructed based on statistical evaluation of training areas derived from the digital BCMOF forest inventory database. A random sample of known cuts and burns occurring between 1975 and 1997 was extracted from the database as training areas. Brightness differences, minimum elevation, maximum elevation, elevation ranges, maximum slope, distance from roads, shape indices and area to perimeter ratios were extracted for the training features. Areas not conforming to the disturbance rule base were eliminated from the analysis, while features selected by the algorithm were merged with disturbances extracted from the BCMOF forest inventory database to create a 1997 Disturbance image. Examples of contextual rules used to isolate cuts and burns are provided in Table 4.2, while Appendix A outlines the classification algorithm in greater detail.

4.2.1.2 Classifying Non-Forested Habitat Units

Disturbance classes identified by the disturbance mapping algorithm were masked out of the 1997 TM imagery. Remaining pixels were passed to a K-Means unsupervised classification using the PCI image analyst software package (PCI Inc. 1997). Unsupervised classification algorithms aggregate image pixels into a number of classes based on the natural breaks or groupings within the image values. The user does not provide training data and, therefore, the identity of the classes with reference to the natural environmental features being represented is initially unknown. The image analyst must compare the output from an unsupervised classification with field data, reference maps and air photos of the study area to determine the information that is being represented by each class (Lillesand and Kiefer 1994).

Table 4.2 Examples of spatial/contextual rules used for disturbance detection

Minimum Area Rule: Cuts and Burns

RULE: The minimum area for sampled cuts and burns is 1.06 Hectares.
ACTION: Select any feature having an area greater than 1.06 Hectares.
GRID Code: IF ((AREA / 10000) >= 1.06) THEN RULE1 = 1
ENDIF

Elevation Range Rule: Cuts and Burns

RULE: Cuts and burns occur between 540 metres and 1990 metres elevation.
ACTION: Select features having an elevation between 540 and 1990 metres
GRID Code: IF (ELEV > 540 & ELEV < 1990) THEN RULE2 = 1
ENDIF

Maximum Slope Rule: Cuts and Burns

RULE: The maximum slope recorded for cuts and burns is 55 degrees.
ACTION: Select any feature with a maximum slope of less then 55 degrees.
GRID Code: IF (SLOPE <= 55) THEN RULE3 = 1
ENDIF

Area to Ratio Perimeter Rule: Cuts and Burns

RULE: The minimum area to perimeter ratio for cuts and burns is 25.
ACTION: Select any feature with an area to perimeter ratio greater than 25.
GRID Code: IF (RATIO > 25) THEN RULE4 = 1
ENDIF

Feature Shape Rule: Cuts and Burns

RULE: The maximum shape index for cuts and burns is 5.
ACTION: Select any feature with a shape index less than 5.
GRID Code: SHAPE = (0.25 * PERIMETER) / (SQRT (AREA))
IF (SHAPE < 5) THEN RULE5 = 1
ENDIF

Distance From Existing Roads Rule: Cuts

RULE: Cuts are within 875 metres of an existing road .
ACTION: Select any feature within 875 metres of a road and code feature attribute as CUT.
GRID Code: IF (RDMAX <= 875) THEN CUTS = 1
ENDIF

IF RULE 1, 2, 3, 4, 5 = 1 AND CUTS = 1 THEN FEATURE = CUT
IF RULE 1, 2, 3, 4, 5 = 1 AND CUTS = 0 THEN FEATURE = BURN

An iterative unsupervised procedure was used to classify the TM Brightness, Greenness, and Wetness Tasseled Cap components into land cover classes. This process involved re-classifying output spectral clusters that are confused among two or more land cover classes (Cohen *et al.* 1995; Sachs *et al.* 1998). Seven output classes resulted from the unsupervised classification. The information content of each K-Means spectral class (the type of land cover that each class represented) was interpreted by comparison with the TM imagery, forest cover maps and personal knowledge of the study area. Spatial and contextual rules derived from a random sample of spatial features of a known cover type were then applied to separate the K-Means spectral classes into the appropriate land cover type (see Figure 4.3). For example, pixels classified into K-Means class 4 were coded as one of the following classes - alpine meadow, low elevation shrub, avalanche path, or riparian areas - based on the minimum elevation, maximum elevation and elevation range of each spatial feature.

The spatial/contextual rule base separated the initial seven K-Means spectral classes into twelve different land cover classes: eleven non-forested land cover categories and one forested class. All non-forested cover classes were merged into a single image representing non-forested habitat units. All pixels within the forest class were written to a separate forested habitat image. Figure 4.3 summarizes the classification procedure used to map non-forested habitat units while Appendix B provides the spatial/contextual rule base algorithm.

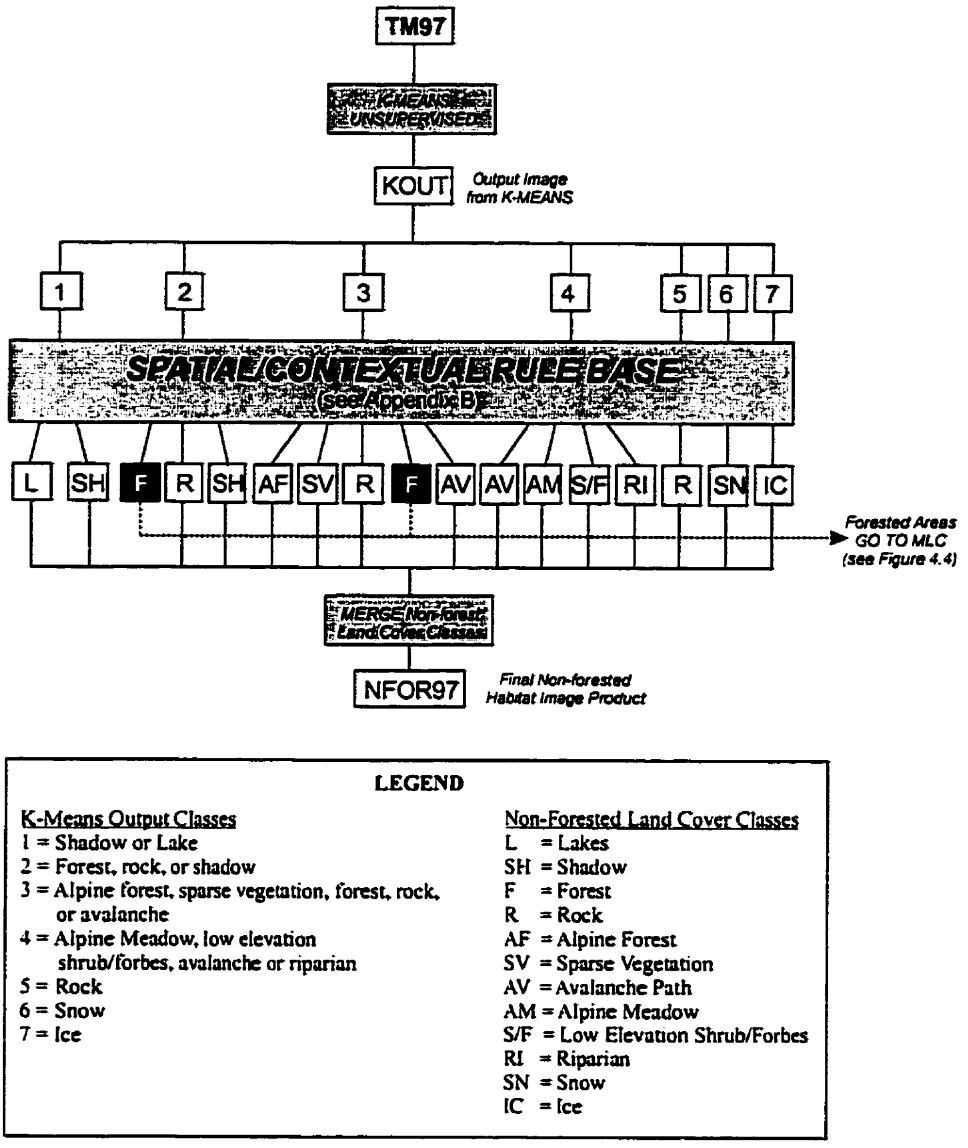


Figure 4.3 Classification of non-forested habitat

4.2.1.3 Classifying Forested Habitat Units

Forested habitat units were first classified into tree species composition classes using a supervised Maximum Likelihood Classification algorithm. This procedure was carried out on the forest class identified in the previous section while disturbance and non-forested habitat units were not included in the MLC analysis.

Input training classes for the MLC procedure were based on primary and secondary species attribute information extracted from the forest inventory database. A training coverage was created in Arc/Info by aggregating the Primary and Secondary species attribute values into a single species value. For example, a forest stand dominated by western red cedar (Primary Species) with a secondary species of western hemlock would be assigned a combined species code of CWHW. The vector species polygons were then converted to a raster grid.

One problem encountered during training set delineation was a high variance of pixel values within training classes due to shadowing in the imagery. This phenomenon is referred to as the "topographic effect" (Cohen and Spies 1992; Fiorella and Ripple 1993) and can result in training areas of the same class having different reflectance values based on slope position relative to the sun. Slopes facing away from the sun during image acquisition (shaded or in shadow) have lower reflectance values than areas of the same species class located on sunlit slopes. To address this issue, a shaded relief image was created from the DEM in Arc/Info. Shaded relief values ranged from 0 to 255 and quantified the direct illumination or sun incidence angle. The species training image was then stratified by sun incidence angle (Eby 1987; Hutchison 1982). Empirically, species classes were divided into three incidence classes: shadow (from 0 to 50 shaded relief), shaded (50 to 130), and normal illumination (greater than 130). Pixels falling within the shadow class (less than 50 shaded relief) appeared very dark in the TM imagery. Class designation could not be determined by spectral information in such areas and was assigned to a habitat unit based on ancillary data sources such as elevation and aspect. Thus, shadow areas were not included in the MLC algorithm.

Each unique species/illumination class was extracted from the grid file into a separate bitmap image for import into PCI. In PCI, the RANDBIT task was used to select a random set of pixels from each training class to generate the MLC signature for use in the classification. Table 4.3 lists the training classes used in the MLC of forested areas within the TM imagery.

Input channels for the Maximum Likelihood Classification included TM Brightness, Greenness, Wetness, elevation, and shaded relief. Related research on mapping of forested environments supports the application of the Tasseled Cap components for land cover classification and change detection (Zheng *et al.* 1997; Cohen *et al.* 1998). Similar studies indicate that TM Wetness may be related to important forest structural components (Cohen and Spies 1992; Fiorella and Ripple 1993; Cohen *et al.* 1995). The choice of channels was decided based on these recommendations and by comparing Bhattacharrya Distance signature separability measures between different band combinations using the PCI task SIGSEP. The chosen band combination provided the highest average separability between the training class signatures.

4.2.1.4 Post-classification Aggregation into Habitat Units

The classification output channel from the Maximum Likelihood Classification was then passed on to the final phase of the decision tree algorithm, which performed post-classification aggregation or sorting of classes. Disturbance, non-forested, and forested features were all combined into a final image and assigned to the appropriate habitat unit class. Four of the final habitat units (Alpine Tundra, ESSFvcp, ESSFvc, and ICHwk) are based on the biogeoclimatic zone classification system after Meidinger and Pojar (1993). Each habitat class was identified based on the expected tree species composition, non-forested land cover type, elevation, and aspect ranges outlined in Braumandl and Curran (1992). A series of binary decision rules were applied to assign each pixel to the appropriate habitat class. After habitat unit assignment, a three by three pixel majority or modal filter was applied to the image to merge small isolated habitat features with neighbouring classes. The post-classification algorithm and the specific decision rules used to aggregate each input class into the appropriate habitat unit are outlined in Figure 4.4. The Arc/Info GRID algorithm for post-classification aggregation is provided in Appendix C.

Table 4.3 Training class scheme for classification of forested habitat

Normal Illumination Classes	Shaded Classes
(> 130 shaded relief)	(50 – 130 shaded relief)
Species Code	Species Code
AT / HW	BL / CW
BL / CW	CW / HW
CW / FD	CW / SE
CW / HW	HW / BL
CW / SE	HW / CW
FD / AT	HW / FD
FD / CW	HW / SE
FD / HW	SE / CW
FD / PW	BL
HW / BL	BL / HW
HW / CW	BL / HM
HW / FD	BL / SE
HW / SE	HM / BL
SE	HM / SE
SE / CW	SE / BL
BL	SE / HW
BL / HW	
BL / HM	
BL / SE	
HM / BL	
HM / SE	
SE / BL	
SE / HW	
SE / HM	

Species codes:

AT = Trembling Aspen (*Populus tremuloides*)
CW = Western Red Cedar (*Thuja plicata*)
HW = Western Hemlock (*Tsuga heterophylla*)
SE = Engelmann Spruce (*Picea engelmanii*)
BL = Subalpine Fir (*Abies lasiocarpa*)
FD = Douglas Fir (*Pseudotsuga menziesii*)
HM = Mountain Hemlock (*Tsuga mertensiana*)

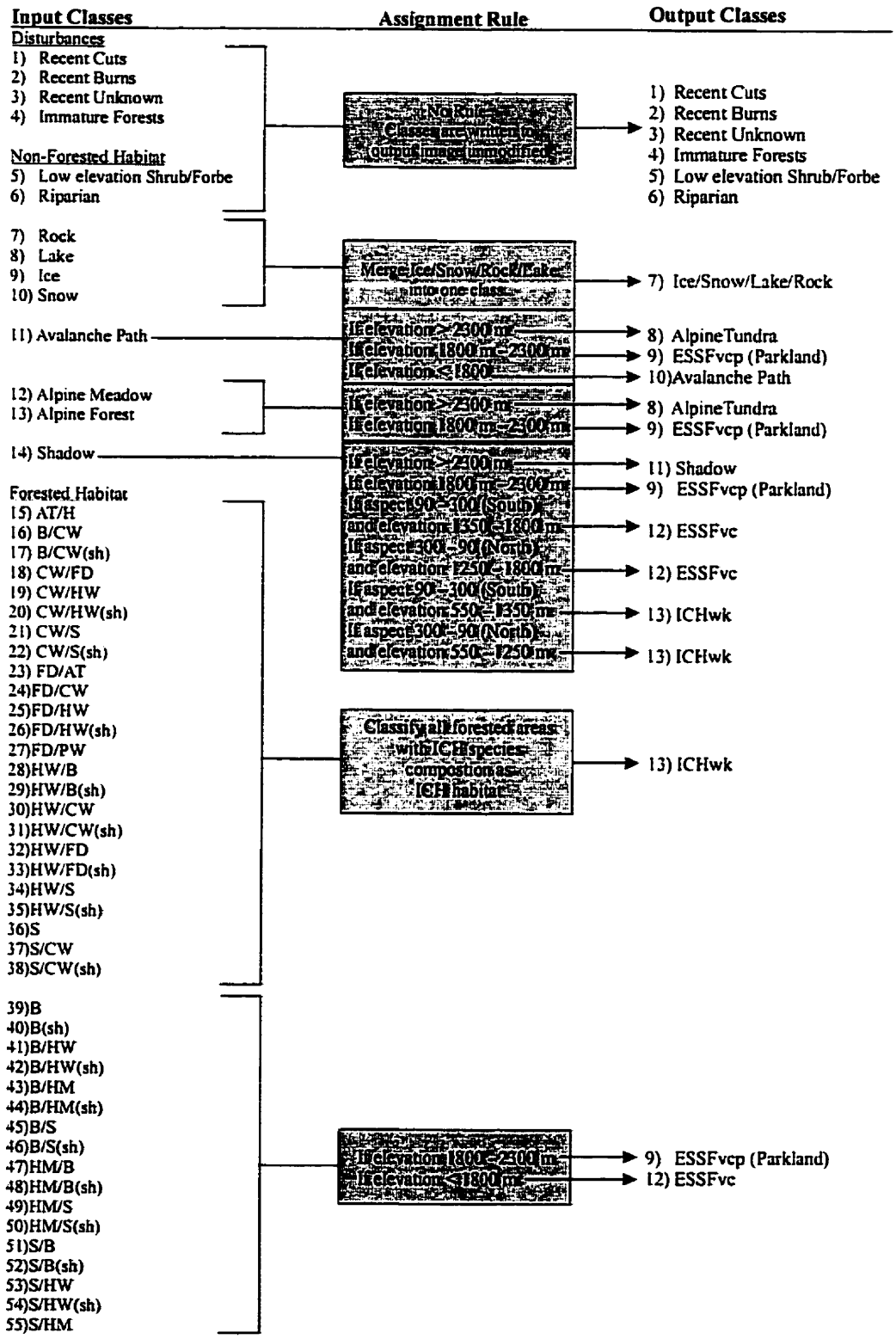


Figure 4.4 Post-classification class aggregation scheme for final 1997 habitat image

4.2.1.5 Classification Accuracy Assessment

The classification accuracy of the 1997 habitat unit classification was assessed using a combination of field verification and aerial photointerpretation. First, a stratified random sampling procedure was used to select 30 to 50 points per habitat class. A sample of 575 pixels was located on the classified map and satellite image products, and these pixels were either verified in the field (forest classes) or interpreted on aerial photographs (non-forest and disturbance classes) to determine the agreement between what a pixel was classified as a result of the HDT algorithm compared to the actual pixel designation in the field.

Following the methodology outlined by Lillesand and Kiefer (1994), the results of the accuracy assessment are reported in the form of contingency tables or classification confusion matrices (see Table 5.1: page 86). Pixel counts along the diagonal of the table have been correctly classified as the appropriate field class, while pixel counts above and below the diagonal represent classification errors. The overall map accuracy was calculated by summing the diagonal pixel counts and dividing by the total number of reference pixels. Likewise, each individual class accuracy (reported in rows and columns) can be determined by dividing the total number of pixels correctly classified in each category by the total number of class sample pixels. The class *user's accuracy* is reported in the table rows and tallied along the right side of the table. This value represents the probability that a pixel classification actually represents that category on the ground. User's accuracy can be considered a measure of the *error of omission*, in which a pixel was omitted from the correct habitat class and grouped into another. The table columns represent *errors of commission*; a pixel is grouped or committed to the wrong habitat class. The *producer's accuracy* indicates the overall accuracy or success with which each reference cover type has been mapped by the classification algorithm. Both types of classification errors (omission and commission) are addressed in section 5.0 in order to interpret the individual class and overall map accuracy.

4.2.2 Calculation of a 1997 Habitat Suitability Model

A GRID image for each of the habitat variables required for calculating a final HSI model had now been created for the 1997 landscape (elevation, slope, habitat unit, and stand age). Each image was then reclassified into the appropriate early winter habitat suitability indices as outlined by McLellan *et al.* (1995). Table 4.4 lists the habitat suitability indices (HSI) for each variable included in the HSI model equation. The geometric mean of the four individual HSI variable images (elevation, slope, habitat unit, and stand age) was then calculated to create a final image of early winter habitat suitability for the 1997 time interval.

4.3 Developing a Habitat Suitability Model for 1975

Determining changes in caribou habitat suitability that have occurred between 1975 and 1997 required the development of a habitat suitability model for the 1975 time interval using the same equation developed for the 1997 HSI model:

$$\mathbf{HSI_{1975} = (HSI_{elevation} * HSI_{slope} * HSI_{habitat\ unit75} * HSI_{stand\ age75})^{1/4}} \qquad \text{Eq. 2}$$

Since the elevation and slope of a pixel were expected to remain stable between time intervals, only the habitat unit and stand age in 1975 had to be determined for the 1975 model calculation.

Table 4.4 Early winter habitat suitability indices

Elevation (m)	HSI	Slope (%)	HSI
< 700	0.20	< 10	0.05
701 – 850	0.50	11 – 20	0.60
851 – 1000	0.80	21 – 30	0.70
1001 – 1150	1.00	31 – 40	0.80
1151 – 1300	0.80	41 – 50	0.60
1301 – 1450	0.50	51 – 70	0.20
1451 – 1600	0.50	> 71	0.10
1601 – 1750	0.20		
1751 – 1900	0.10		
1901 – 2050	0.10		
2051 – 2200	0.05		
> 2201	0.05		

Habitat Unit	HSI	Stand Age (yr.)	HSI
Alpine	0.05	< 40	0.20
Avalanche Path	0.05	41 - 100	0.50
Cedar/Hemlock	1.00	101 - 140	0.50
Recent Cut	0.20	141 - 250	0.40
Immature Forest	0.50	> 250	1.00
Riparian	0.50		
Rock/Ice/Lake	0.20		
Spruce/Balsam	0.50		
Subalpine Parkland	0.20		
Recent Burn	0.50		

(McLellan *et al.* 1995)

4.3.1 Habitat Unit Classification for 1975

The Hybrid Decision Tree algorithm discussed in Section 4.2.1 was also executed on the MSS imagery to map disturbance and habitat features existing in 1975. The only modification to the 1997 algorithm was a choice of different MSS input channels for the K-Means unsupervised classification of non-forested habitat and the Maximum Likelihood Classification of species composition.

Table 4.5 compares the input channel set for the 1997 and 1975 classification of non-forested and forested habitat. It was important that the classification algorithm identify habitat units in 1997 and 1975 in a similar manner to allow a direct comparison of class

features between time periods. It would have been preferable to use the same input channels for each classification; however, a direct equivalent of the TM Wetness component is not calculated in the MSS Tasseled Cap transformation because of the lack of mid-infrared reflectance bands in the MSS satellite sensor. To determine the MSS input channel set that best replicated the TM classification results, different combinations of channels were entered into a K-Means unsupervised classification. The output image from each classification run was compared visually to the TM output. The combination of MSS Brightness, Greenness, MSS band 6 and MSS band 7 appeared to provide the closest match with the results of the TM classification in areas of known cover type that had not changed significantly in the intervening years. The SIGSEP task in PCI was used to determine whether high Bhattacharrya Distance separability measures between the MSS training class signatures were available in these data before executing the Maximum Likelihood Classification algorithm. Similar to the 1997 Hybrid Decision Tree algorithm, disturbance, non-forested and forested habitat classes were entered into the post-classification aggregation procedure, resulting in a map of habitat units for the 1975 image data.

Table 4.5 Comparison of 1997 TM and 1975 MSS image band selection for image classification

Sensor:	1997 Landsat TM		1975 Landsat MSS	
Algorithm:	**K-Means**	**MLC**	**K-Means**	**MLC**
Input Bands:				
	TM Brightness	TM Brightness	MSS Brightness	MSS Brightness
	TM Greenness	TM Greenness	MSS Greenness	MSS Greenness
	TM Wetness	TM Wetness	MSS Band 6	MSS Band 6
		Elevation	MSS Band 7	MSS Band 7
		Shaded Relief		Elevation
				Shaded Relief

4.3.2 *Classification Accuracy Assessment (1975 Habitat Units)*

Classification accuracy assessment of the 1975 habitat unit map was conducted in a similar manner to that of the 1997 product. Pixels that had been verified as mature forested areas in the field and in the 1997 air photos were assumed to have belonged to the same habitat unit class in 1975, and therefore, were not confirmed by interpretation of 1975 air photos. A new random sample of points was extracted for each of the 1975 disturbance and non-forested classes and the image classification was confirmed against a set of air photos covering a portion of the study area in 1974 at 1:19,000 scale. The 1975 classification accuracy is summarized in Table 5.2 and is interpreted in section 5.2.

4.3.3 *"Backcasting" 1975 Habitat Units from the 1997 Database*

A visual comparison of the 1997 and 1975 habitat unit images revealed key variations between feature boundaries and habitat patch sizes resulting from the differing spatial resolution of the Landsat TM and MSS imagery. Specifically, the 80 metre pixel size of the MSS sensor seemed to limit the delineation of small or narrow landscape patches. Figure 4.5 illustrates how narrow avalanche paths (less than 80 metres in width) were detected in the Landsat TM imagery but omitted from the MSS habitat classification. As a result of these differences, a direct comparison of class areas between each habitat unit image may result in an overestimation of the amount of suitable forested habitat in 1975. In turn, any change statistics and spatial pattern indices calculated for the 1997 and 1975 landscapes could give a false indication of the fragmentation levels between time periods. An innovative technique, similar to one used by Reed *et al.* (1996) and Sachs *et al.* (1998), was applied to reconstruct a 1975 map product that could be more easily compared to the 1997 habitat unit image.

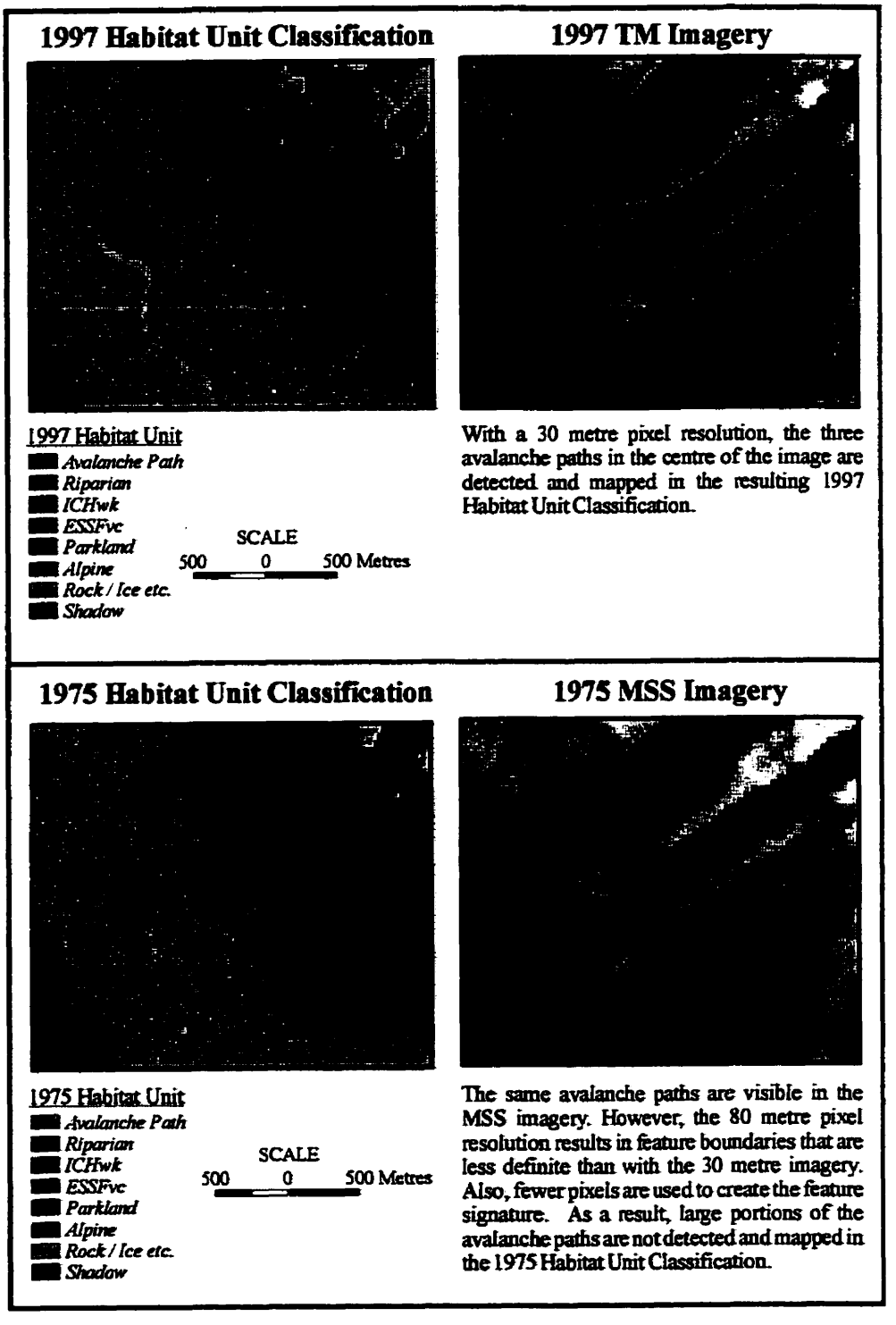


Figure 4.5 Comparison of 1997 and 1975 habitat classification results due to differences in pixel resolution

Figure 4.6 provides a graphic representation of this procedure. The intent was to use the 1997 habitat unit map as a base image for a form of image change detection that worked backward in time from 1997 (T_0) to 1975 (T_{0-1}). For the purposes of this research, the term "backcasting" was applied to illustrate the concept of projecting changes in habitat type backward in time from the present (1997) to the past (1975). First the Arc/Info COMBINE function was used to overlay the 1997 and 1975 habitat images. This created a value attribute table that compared the cover type designation of each pixel in 1997 and 1975. A series of rules about expected land cover transitions between 1975 and 1997 were then applied to reclassify each pixel from the combined 1997/1975 habitat map into the appropriate 1975 habitat unit. The specific rules used to assign a new 1975 habitat type are listed in Table 4.6.

Based on these landscape change rules, pixels for the same location in each time period that were classified as a habitat category in both 1997 and 1975 were considered to have remained constant over time. These pixels retained the 1997 habitat category and were written to the new 1975 "backcast" image. Pixels classified as habitat in 1997 and Immature Forest in 1975 represented forested areas that were less than 80 years of age in 1975 but greater than 80 years of age in 1997. This marked a transition from an immature (disturbance feature) to a mature forest stand (habitat feature) within the classification scheme. In this specific case, the 1997 habitat pixels were assigned a new 1975 disturbance code of Immature Forest and written to the backcast 1975 image. For 1997 disturbance features, such as recent cuts and burns etc., a binary mask image was created in which all disturbances were given a pixel value of 1 and all habitat features a value of 0. The mask image was then multiplied by the original 1975 habitat image. This resulted in each 1997 disturbance pixel being replaced by the 1975 pixel classification and allowed the past (1975) habitat types within the disturbance areas to be determined. The end product of this procedure was a reconstructed 1975 habitat unit image in which the 1997 pixel classifications were maintained where appropriate and only the habitat class designation for 1997 disturbance pixels changed between time periods (as derived from the 1975 pixel classifications). A comparison of the habitat spatial pattern indices

calculated for the backcast 1975 and 1997 habitat images gave a more realistic indication of fragmentation levels resulting from observed changes in land cover rather than from image misregistration or differences in pixel resolution between landscapes.

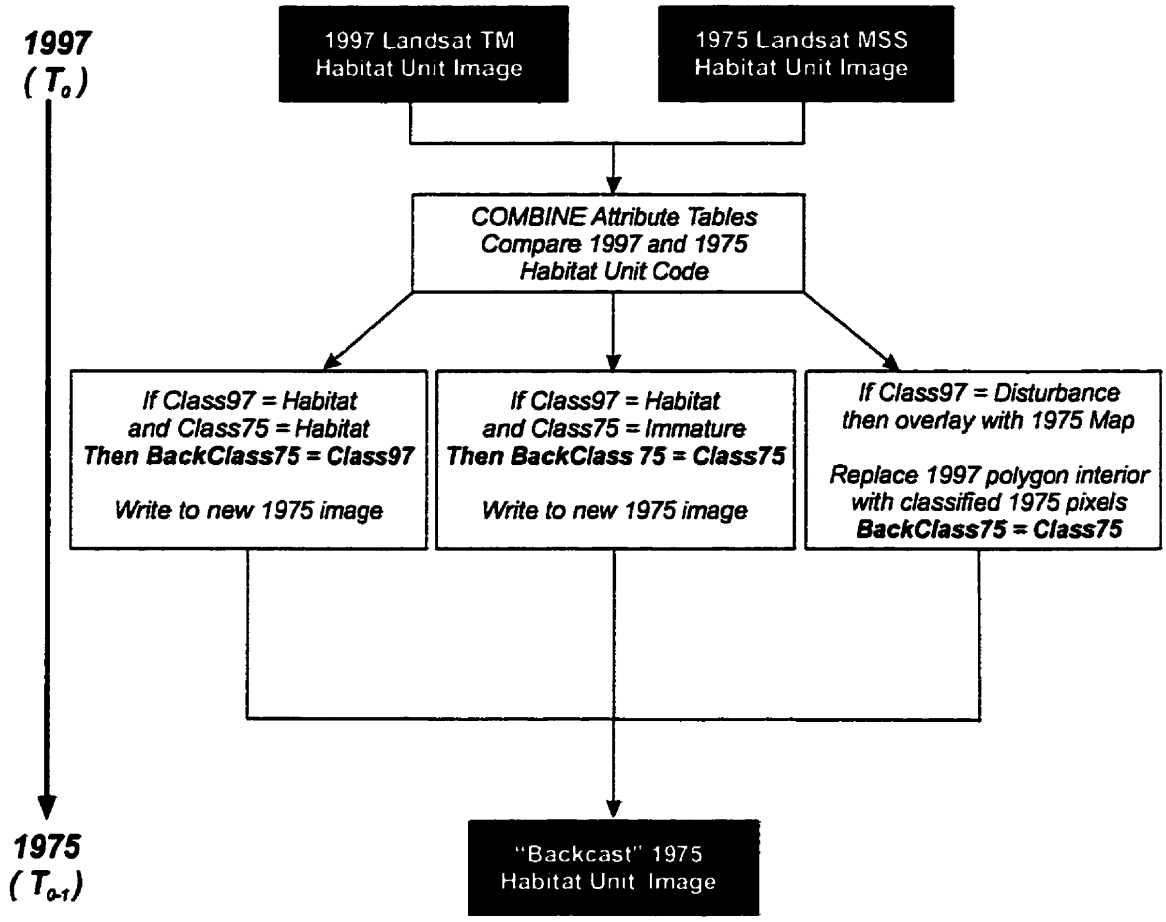


Figure 4.6 "Backcasting" technique used to reconstruct a new 1975 habitat image that could be directly compared to the 1997 classification

Table 4.6 Rules and procedures for assigning "Backcast" 1975 habitat units

Habitat Areas

RULES:

1) All pixels classified as a habitat category in 1997 and in 1975 will maintain the 1997 habitat code.
2) All pixels classified as habitat in 1997 and Immature forest in 1975 will maintain the 1975 code.

RATIONALE:

1) Non-forested habitat units (e.g. shrub, rock etc.) not classified as a disturbance area and forested habitat units (e.g. ICHwk, ESSFvc, Parkland) are held constant and do not change between 1975 and 1997 so that the spatial effects of clear cuts and burned areas on the landscape structure over time may be determined.
2) Areas that have changed from immature forest less than 80 years of age in 1975 to mature forested habitat (greater than 80 years of age) in 1997 will be assigned the 1975 Immature Forest code to represent the past disturbance features in the backcast 1975 image.

ACTION:

1) For each pixel, determine 1997 and 1975 habitat code value. If the 1975 class is not immature forest, then assign 1997 habitat code and write to new backcast image (BACK75).
2) If 1975 class is Immature Forest, then assign pixel code for Immature Forest and write to BACK75.

GRID CODE:

1997 Class Code		1975 Class Code		New 1975 Class Code
IF HAB97 = AVALANCHE (5)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = AVALANCHE (5)
IF HAB97 = SHRUB (6)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = SHRUB (6)
IF HAB97 = RIPARIAN (7)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = RIPARIAN (7)
IF HAB97 = ICHwk (8)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = ICHwk (8)
IF HAB97 = ESSFvc (9)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = ESSFvc (9)
IF HAB97 = PARKLAND (10)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = PARKLAND (10)
IF HAB97 = ALPINE (11)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = ALPINE (11)
IF HAB97 = ROCK/ICE (12)	*and*	HAB75 NE IMMATURE	*then*	BACK75 = ROCK ETC. (12)
IF HAB97 = SHADOW	*and*	HAB75 NE IMMATURE	*then*	BACK75 = HAB75
IF HAB97 = (5 – 13)	*and*	HAB75 = IMMATURE	*then*	BACK75 = IMMATURE (4)

Table 4.6 (cont'd) Rules and Procedures for "Backcasting"

Disturbance Areas

ACTION:
Create mask in which 1997 disturbance features = 1 and 1997 habitat areas = 0

GRID CODE:
IF HAB97 = CUT (1), BURN (2), UNKNOWN (3) OR IMMATURE (4)
then DISTMASK = 1
ELSE DISTMASK = 0

ACTION:
Multiply Disturbance Mask (DISTMASK) by original 1975 habitat map (HAB75)
This replaces 1997 Immature pixels with past 1975 habitat unit code

GRID CODE:
DISTHAB = DISTMASK * HAB75

Create Final Backcast Habitat Map for 1975

ACTION:
Merge backcast disturbance map (DISTHAB) with backcast habitat map (BACK75)
by superimposing one map OVER the other

GRID CODE:
BACKMAP75 = DISTHAB OVER HABBACK

4.3.4 Estimation of Forest Stand Age for 1975

Forest stand age was the final variable required for the 1975 Habitat Suitability Index (HSI) model. For forested areas existing in both time periods, 1975 stand age was calculated by simply subtracting the time interval between 1975 and 1997 (22 years) from the 1997 stand age attribute. However, pre-disturbance stand age information for features that had been harvested or burned between 1975 and 1997 was not included in the BCMOF forest inventory database. Ancillary data sources such as pre-disturbance cruising records were unavailable for many of the areas harvested or burned since 1975. An interpolation routine was therefore developed to estimate the 1975 stand age of 1997 disturbance features based on the average age of surrounding forest stands.

Figure 4.7 illustrates the estimation of 1975 stand age for disturbance features. First, each disturbance feature was extracted to a separate GRID file. A 100 metre buffer was delineated around each feature. A distance of 100 metres (four 25 metre pixels) was chosen to isolate areas immediately adjacent to each disturbance feature. Given an average RMS of 0.375 between the TM imagery and the TRIM vector data and an RMS of 0.66 between the TM and MSS imagery, a distance of four pixels appeared to be adequate to isolate adjacent forest stands while accounting for possible misregistration between these data sources. The average age of the surrounding forested stands was then calculated within each buffered region using the Arc/Info ZONALMEAN grid command. This task provides an average value of all pixels within a specific mask area or zone.

By calculating the average value within the buffer area, the procedure assumed that a disturbance feature had an equal probability of belonging to any of the surrounding stands rather than trying to predict the most probable stand membership. Without the ancillary data sources required to accurately predict the 1975 forest stand that a disturbance feature occurred within, the mean value of surrounding stands was considered the best estimate of stand age in 1975 and assigned to the associated disturbance feature. The individual GRID files representing the estimated stand age for each disturbance feature were merged into a single image. Finally, the image representing estimated stand ages in 1975 for the disturbance features was combined with the 1975 stand age image for forested areas to produce the final 1975 stand age image.

Each of the 1975 habitat model images (elevation, slope, habitat unit and stand age) were reclassified into the appropriate habitat suitability indices from Table 4.4. The geometric mean of the four individual HSI variable images was then calculated to create a final image of early winter habitat suitability for the 1975 time interval.

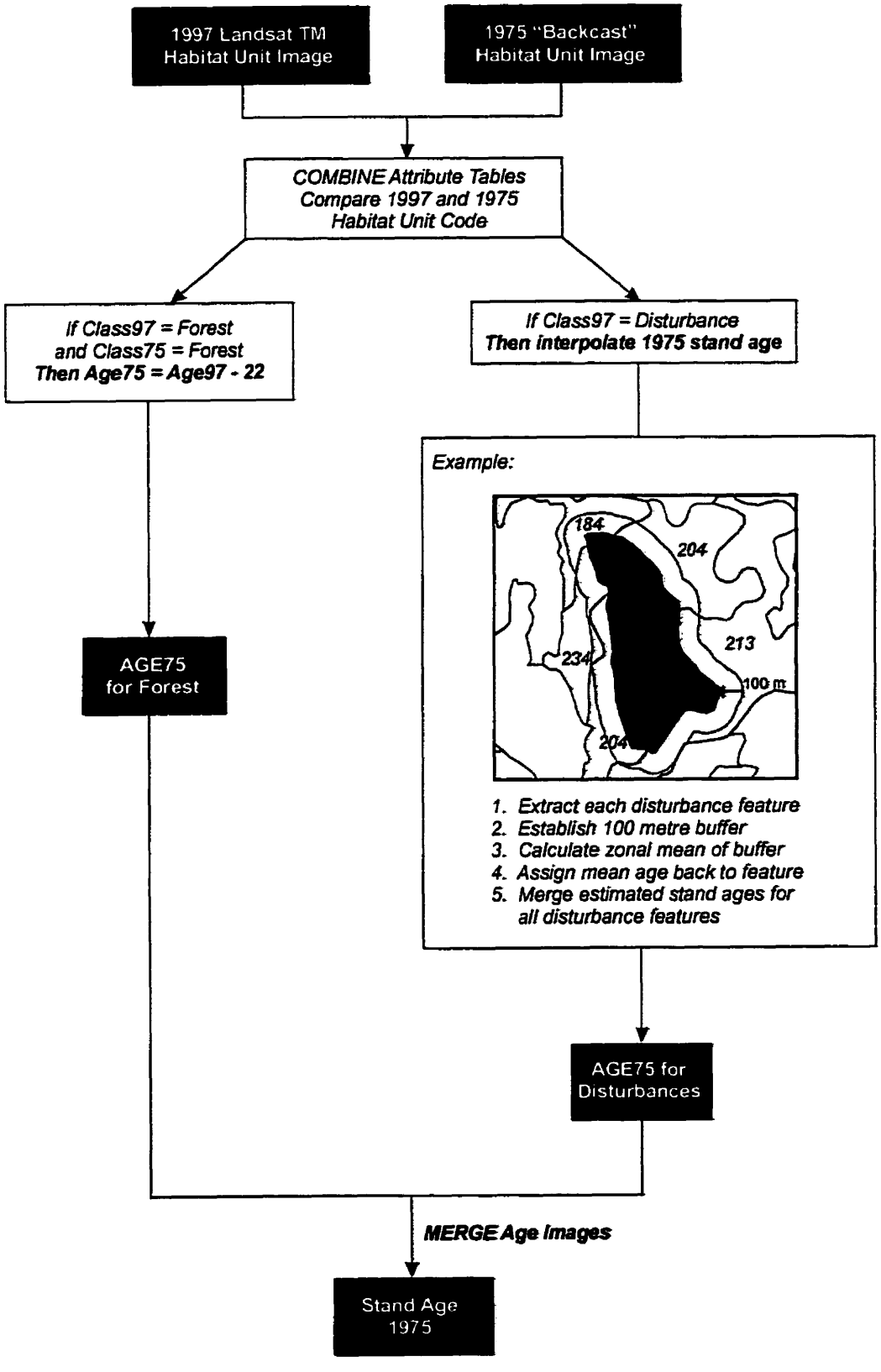


Figure 4.7 Estimating 1975 stand age for 1997 disturbances from surrounding stand age attribute data

4.4 Calculation and Comparison of Spatial Pattern Indices

Quantification and comparison of the spatial configuration of habitat patches between time periods was conducted to determine if fragmentation of early winter caribou habitat had occurred in the study area as a result of timber harvesting and wildfires. Rather than providing a comparative analysis of an extensive set of indices, a limited set of key landscape metrics were chosen based on the recommendations of several recent and similar forest fragmentation studies (as discussed in section 2.4.2). The equations for each spatial metric are listed below:

$$\text{Class Area}_i = \sum_{j=1}^{n} a_{ij} \left(\frac{1}{10{,}000}\right)$$

where a_{ij} = area (m^2) of patch ij — Eq. 3

Number of patches in the landscape of patch type (class) i $= n_i$ — Eq. 4

$$\text{Patch Density} = \frac{n_i}{A}(10{,}000)(100)$$

where A = Total landscape area (m^2) — Eq. 5

$$\text{Mean Patch Size} = \frac{\sum_{j=1}^{n} a_{ij}}{n_i}\left(\frac{1}{10{,}000}\right) \qquad \text{Eq. 6}$$

$$\text{Patch Size Coefficient of Variation} = \frac{\text{PSSD}}{\text{MPS}}(100) \qquad \text{Eq. 7}$$

where MPS = Mean Patch Size as calculated in Eq. 6
PSSD = Patch Size Standard Deviation

$$\text{PSSD} = \sqrt{\frac{\sum_{j=1}^{n}\left[a_{ij} - \left(\frac{\sum_{j=1}^{n} a_{ij}}{n_i}\right)\right]^2}{n_i}}\left(\frac{1}{10{,}000}\right) \qquad \text{Eq. 8}$$

$$\text{Edge Density} = \frac{\sum_{k=1}^{m'} e_{ik}}{A} \quad (10{,}000)$$

where e_{ik} = total length (m) of edge in landscape between patch type (class) i and k

Eq. 9

$$\text{Mean Shape Index} = \frac{\sum_{j=1}^{n} \left(\frac{0.25\, p_{ij}}{\sqrt{a_{ij}}} \right)}{n_i}$$

where p_{ij} = perimeter (m) of patch ij

Eq. 10

$$\text{Mean Proximity Index} = \frac{\sum_{j=1}^{n} \sum_{s=1}^{n} \frac{a_{ijs}}{h_{ijs}^2}}{n_i}$$

where s = 1,... , n patches, within specified neighbourhood
a_{ijs} = area (m^2) of patch ijs within specified neighbourhood (m) of patch ij
h_{ijs} = distance (m) between patch ijs [within neighbourhood]and patch ij, based on edge-to-edge distance

Eq. 11

$$\text{Mean Core Area} = \frac{\sum_{j=1}^{n} a_{ij}^{c}}{n_i} \left(\frac{1}{10{,}000} \right)$$

where a_{ij}^{c} = core area (m^2) of patch ij based on specified buffer width (m)

Eq. 12

$$\text{Interspersion/Juxtaposition} = \frac{-\sum_{k=1}^{m'} \left[\left(\frac{e_{ik}}{\sum_{k=1}^{m'} e_{ik}} \right) \ln \left(\frac{e_{ik}}{\sum_{k=1}^{m'} e_{ik}} \right) \right]}{\ln(m' - 1)} (100)$$

where m' = number of patch types (classes) present in landscape

Eq. 13

The metrics were generated using the Patch Analyst software extension for the ArcView GIS package. Patch Analyst was designed specifically for the quantification of landscape structure (Elkie *et al.* 1999). The program consists of several scripts written in the Avenue and C programming language, and calculates spatial indices for both vector and raster data sources. The raster version includes spatial statistics originally provided by the FRAGSTATS spatial pattern analysis program (McGarigal and Marks 1995). Patch Analyst calculates over twenty-four different landscape metrics or indices at both the landscape and class level. In their programming structure, a landscape includes all patches, polygons, contiguous cells or shapes, and a class includes all patches, polygons, contiguous cells or shapes in a landscape that have the same value for a given attribute (Elkie *et al.* 1999). Before each of the spatial metrics could be calculated, consideration had to be given to the appropriate definition of *landscape* and *patch* relative to the research objective.

The landscape boundary defining the region of analysis corresponds to the home range of the local subpopulation of mountain caribou and remained constant between time periods so that landscapes of similar extent were being compared. The past (1975) and most recent (1997) landscapes were constructed in such a way to represent a mosaic of habitat patches that differed based on the early winter habitat suitability rating. The HSI values for the habitat suitability models ranged from 0.0 to 1.0. The integer range was reclassified into 10 ordinal rank HSI classes to create discrete habitat suitability classes for fragmentation analysis (see Table 4.7) with the lowest habitat suitability represented by HSI class 1 and the highest or most suitable early winter habitat as HSI class 10. Spatial metrics were calculated for each of the 10 ordinal habitat suitability classes. This particular landscape configuration followed the concept of a habitat landscape provided by Dunning *et al.* (1992); a mosaic of habitat patches defined by nonrandom distribution of resource utilization (Wiens 1976) in which the "focal" or "target" habitat patch type is embedded. Of primary interest are the directional changes in spatial metrics for the high suitability classes (HSI classes 8, 9, 10) between 1975 and 1997. Therefore, the focus of the fragmentation analysis will be on changes in HSI class level indices rather than a

landscape level analysis. McGarigal and Marks (1995: p. 19) suggest that in landscape ecology applications, such as fragmentation, the analysis of class indices may serve to "...separately quantify the amount and distribution of each patch type in the landscape and thus can be considered indices of fragmentation for each patch type".

Table 4.7 Ordinal habitat suitability classes used for patch level analysis

Habitat Suitability Input Range	Ordinal HSI Class
0.00 to 0.10	HSI Class 1
0.10 to 0.20	HSI Class 2
0.20 to 0.30	HSI Class 3
0.30 to 0.40	HSI Class 4
0.40 to 0.50	HSI Class 5
0.50 to 0.60	HSI Class 6
0.60 to 0.70	HSI Class 7
0.70 to 0.80	HSI Class 8
0.80 to 0.90	HSI Class 9
0.90 to 1.00	HSI Class 10

4.5 Summary

Three different methodological components were incorporated into a single process of quantification and assessment of landscape changes that have affected early winter habitat suitability between 1975 and 1997. These components included 1) mapping or classification of habitat variables to develop a Habitat Suitability Index (HSI) model for the 1997 time period, 2) a "backcasting" procedure employed to reconstruct a 1975 HSI model that could be directly compared to the 1997 model in terms of pixel resolution, and 3) calculation of class level metrics for the 1975 and 1997 HSI model map products.

First, a HSI model was developed for the most recent time period (1997) based on the HSI equation and coefficients produced by McLellan *et al.* (1995). This particular model was chosen because it was specific to the study area and had been developed based on statistical analysis of telemetry observations. Required habitat variables for the HSI model included elevation, slope, forest stand age, and habitat unit. Habitat unit replaced

the primary tree species variable of the original model in order to provide consistent and historical coverage of the study area. Stand height was also omitted from the HSI model due to attribute accuracy issues. Elevation and slope data were extracted from the DEM created for the study area, while a forest stand age image was derived from the BCMOF forest inventory database. Habitat units had not been previously mapped and were classified by the integration of GIS and remote sensing analysis in a Hybrid Decision Tree (HDT) algorithm. This particular algorithm was chosen due to the flexibility to include different data types and multiple classification techniques within a single tree structure (Friedl and Brodley 1997).

The HDT algorithm was comprised of four general procedures:

1. disturbance mapping
2. classification of non-forested habitat units
3. classification of forested areas
4. post-classification sorting or aggregation

Embedded within the HDT algorithm were a series of decision rules used to assign image pixels to the appropriate habitat or disturbance category outlined in the habitat unit classification scheme. Binary decision rules constructed from the quantitative analysis of training sites collected from the BCMOF forest database and satellite imagery were used in the disturbance mapping procedure and the classification of non-forested habitat units (following the example of such research described in section 2.3.1 of this thesis). Similarly, a post-classification knowledge based approach (Kontoes *et al.* 1993; Wang and Civco 1994) was employed in step 4 of the HDT algorithm to assign output pixels from a Maximum Likelihood Classification to the appropriate habitat or disturbance category. The end product of the Hybrid Decision Tree algorithm, an image representing caribou habitat units in 1997, along with the elevation, slope and forest age images were then reclassified into the appropriate HSI coefficients and an early winter habitat suitability image was calculated.

An HSI model for the 1975 time period was then developed. The elevation and slope variables had already been classified for the 1997 HSI model; however, images for 1975 habitat units and forest stand age had to be constructed. A HDT algorithm, similar to that applied the 1997 TM imagery, was used to classify habitat units from the 1975 MSS imagery. Habitat unit boundaries and patch sizes were not consistent between the 1997 and 1975 classifications due to the differing pixel resolution of the TM (30 metre) and MSS (80 metre) imagery. A new 1975 habitat unit image was reconstructed or "backcast" from the 1997 habitat unit classification to overcome these difficulties. Pixels classified as a habitat category in both 1975 and 1997 were considered to have remained stable over time and were assigned the 1997 habitat unit category. Disturbance pixels from the 1997 classification were replaced by the previous 1975 cover category and written back to the backcast 1975 habitat image. The end result was a habitat unit image in which feature boundaries were directly comparable to the 1997 classification.

For those 1997 disturbance pixels that were determined to be forested habitat in 1975, the stand age for the previous time period was not included in the BCMOF forest inventory and had to be interpolated before calculating the 1975 HSI model. The 1975 stand age of each feature was determined by calculating the average age of all mature forested stands within a 100 metre proximity zone. The resulting 1975 stand age image, 1975 habitat units from the backcasting procedure, and the elevation and slope images were then reclassified into the appropriated HSI category and a 1975 habitat suitability image was created.

Finally, several landscape indices selected based on the recommendations of previous forest fragmentation research were calculated using the Patch Analyst ArcView extension. A comparative analysis of class level indices for the 1975 and 1997 HSI images was then initiated to determine the spatial effects of timber harvesting and wildfires on early winter caribou habitat composition and spatial configuration.

5.0 RESULTS AND ANALYSIS

5.1 Introduction

The classification accuracy assessment for the habitat unit classifications are summarized at the beginning of this chapter and relate directly to the first objective of this thesis research: the classification of required habitat variables and the development of 1975 and 1997 habitat suitability models. An analysis of overall classification accuracy, and individual class accuracy levels for both the 1997 and 1975 image products is provided to determine the success of the Hybrid Decision Tree classification algorithm. This discussion focuses primarily on the accuracy of the 1997 habitat unit image, as it forms the base data source from which the 1975 habitat suitability map was constructed. The classification accuracy assessment addresses the importance of understanding potential sources of error that may be propagated within a habitat suitability model, an issue that many previous studies of caribou habitat fragmentation have not addressed.

Results and analysis pertaining to the second research objective, the quantification and assessment of early winter caribou habitat fragmentation, are then presented in the following sequence. First, habitat fragmentation is summarized in terms of the amount of quality habitat lost between 1975 and 1997. The changes in habitat composition are quantified and assessed primarily by investigating the relative areas of habitat and disturbance cover types, forest stand age cohorts, and HSI classes in each time period. This type of analysis allows the quantification of the total amount of suitable early winter habitat that has been transformed to less desirable cover types by timber harvesting and wildfires. The second part of the fragmentation analysis focuses on changes in the spatial characteristics and configuration of suitable early winter habitat in the study area between 1975 and 1997. The results of the habitat fragmentation analysis are presented and the directional changes between time periods for each spatial metric are interpreted. This component of the investigation was designed to test the main hypothesis of this study, which is that fragmentation of critical early winter habitat is occurring in the study area as a result of timber harvesting and wildfires.

5.2 Habitat / Disturbance Mapping

5.2.1 1997 Habitat Unit Classification

The 1997 habitat unit classification derived from the Hybrid Decision Tree algorithm is presented in Figure 5.1. Visual assessment of the map product reveals a landscape pattern consistent with that outlined in the original description of the study area (section 3.2: page 30). Higher elevations within the study area consist of ice, snow, glaciers, rock, and small amounts of alpine tundra communities. The Subalpine Parkland zone (ESSFvcp), indicated in dark green in Figure 5.1, occurs at elevations between approximately 1900 metres and extends up to the rock/ice/glacier and Alpine Tundra (AT) zone. The parkland-to-ESSFvc transition zone marks the treeline boundary at about 1900 metres elevation, with the Interior Cedar-Hemlock (ICHwk) zone occurring at the lowest elevations in the valley bottoms. The distribution of the shrub/herb class, depicted in a light-violet colour, across the image illustrates the high number of avalanche paths in the study area. Due to the steep terrain and heavy snowfall in the Revelstoke region, avalanches are common in late winter and early spring naturally fragment the forested landscape. Riparian areas are limited to the Downie creek valley in the northwest portion of the study area.

Timber harvesting activities occurring since 1975 have resulted in a large number of clearcuts across the study area. Recent cuts, depicted in red, are located almost exclusively within the ICHwk biogeoclimatic zone, as illustrated by the army green colour surrounding most of the red recent cut features. Each of the major watersheds within the study area, with the exception of the La Forme valley directly north of Mount Revelstoke Park and the upper reaches of the Tangiers valley, have been the target of timber harvesting. Recent burns and recent disturbances with an unknown agent are less extensive in area compared to the recent cuts class and occur at higher elevations, mostly within the ESSFvc and ESSFvcp (parkland) zones. This pattern is a result of prevalent fire behaviour within the study area. Wildfires are most often the result of lightning strikes on prominent topographic features, such as mountain peaks, and tend to

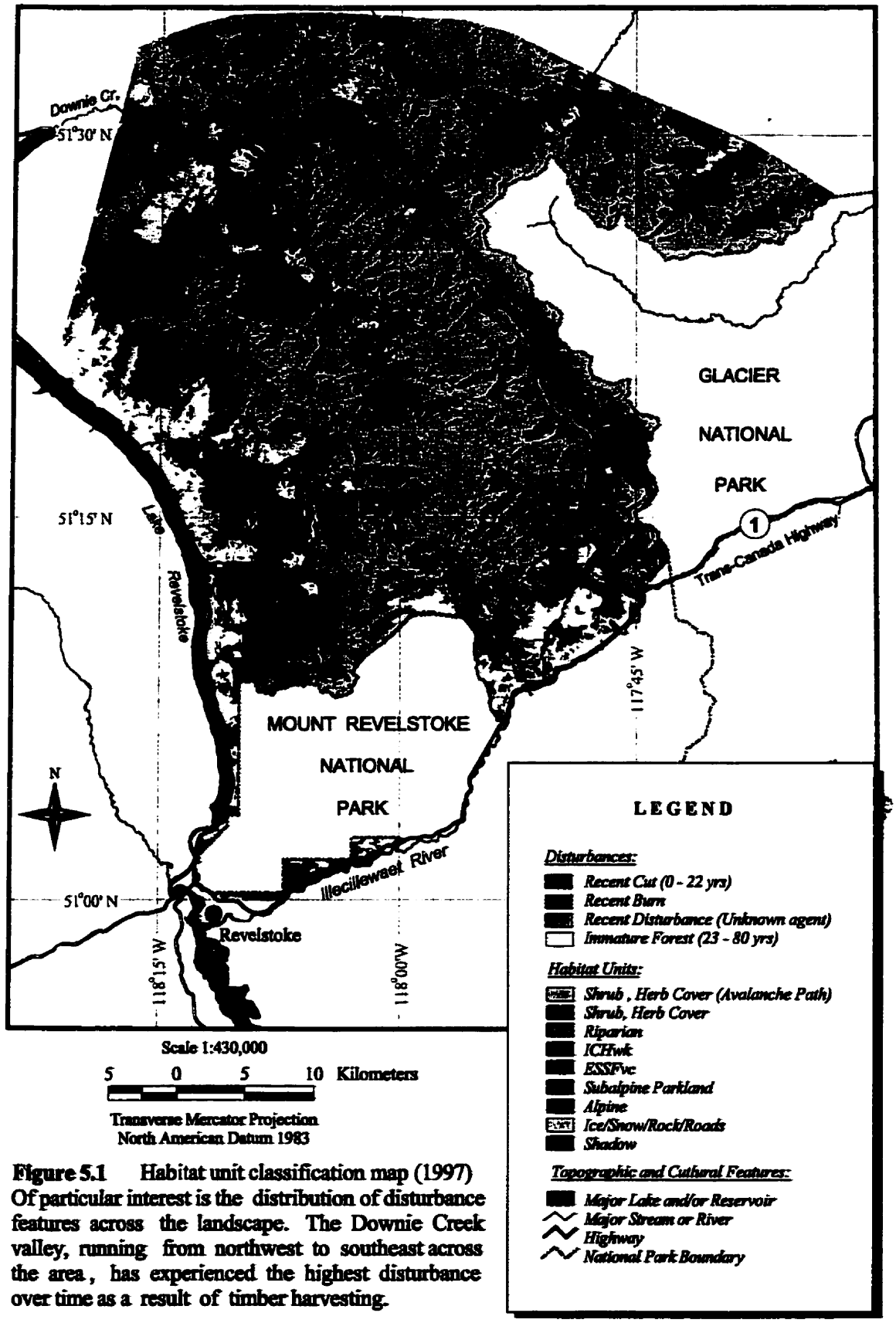


Figure 5.1 Habitat unit classification map (1997) Of particular interest is the distribution of disturbance features across the landscape. The Downie Creek valley, running from northwest to southeast across the area, has experienced the highest disturbance over time as a result of timber harvesting.

move downslope through the parkland and ESSFvc zones. The majority of the Immature Forest class, depicted in pale yellow, has resulted from wildfires occurring in the study area before 1975, with the exception of the Sale Mountain area directly northwest of Mount Revelstoke Park and the southern end of the Tangiers valley. Many of the immature forest features in these specific areas represent regenerating clearcuts that were logged before 1975.

5.2.2 Accuracy Assessment of 1997 Habitat Unit Classification

The contingency table summarizing the classification accuracy of the 1997 habitat unit map is reported in Table 5.1. The overall map accuracy for all habitat and disturbance classes is 91.8% with a confidence interval of +/- 3.5% at a 99% confidence level. This value represents the estimated error for the overall classification accuracy and takes into consideration the total number of pixels sampled and the expected distribution of the standard error of the estimates. Therefore, there is a 99% probability that the 1997 habitat unit classification has a minimum accuracy of 88.3%, but is no more than 95.3% accurate.

A total of 575 pixel locations were verified in order to determine the accuracy in each class. All class accuracy levels are consistently high (greater than 80% accuracy). By analyzing the nature of the classification errors for each class, the success of the Hybrid Decision Tree (HDT) algorithm for mapping the range of disturbance and habitat classes is evaluated.

Assessment of 1997 Disturbance Features:

The producer's and user's accuracy levels for the disturbance classes (Cuts, Burns, Recent Unknown, and Immature Forest) reflect the success of the first component of the HDT algorithm, in which disturbance features were identified through a combination of GIS attribute selection and image brightness differencing. As such, the disturbance class accuracy levels provide an indication of the attribute and positional accuracy of the BCMOF forest inventory database, as most of these features were extracted from this

Table 5.1 Contingency table resulting from 1997 habitat unit classification accuracy assessment

		Reference Data (Field and Aerial Photography Samples)											
		Cut	Burn	Unk.	Imm.	Shrub	AT	I/R/R/L	ICHwk	ESSFvc	ESSFvcp	Row Total	% User's
Classification Data	Recent Cut	**40**	-	-	-	-	-	-	1	-	-	41	**98**
	Recent Burn	1	**26**	-	1	-	-	1	-	-	1	30	**87**
	Recent Unknown	-	-	**25**	3	-	-	-	-	-	2	30	**83**
	Immature Forest	-	1	-	**39**	-	-	-	1	-	-	41	**95**
	Shrub	-	-	-	-	**97**	-	-	8	1	-	106	**92**
	AT	-	-	-	-	-	**45**	3	-	-	-	48	**94**
	I/R/R/L	-	-	-	-	1	4	**60**	-	-	-	65	**92**
	ICHwk	-	-	-	-	3	-	-	**94**	3	-	100	**94**
	ESSFvc	2	-	-	-	4	-	-	2	**53**	-	61	**87**
	ESSFvcp	-	-	-	-	-	-	4	-	-	**50**	54	**93**
	Column Total	43	27	25	43	105	49	68	106	57	53	**529**	
	% Producer's	**93**	**96**	**100**	**91**	**92**	**92**	**88**	**89**	**93**	**94**		

Map Accuracy = 91.8 % 99 % Confidence Interval = +/- 3.5 % (88.3 % - 95.3 %)

data source. Many of the recent cuts and burns (occurring since 1995) were mapped using the Brightness differencing technique and their accuracy provides insight on the utility of this particular technique for updating the existing disturbance database.

The disturbance class with the lowest producer's accuracy (highest error of commission) is Recent Cuts with 3 pixels committed to other classes (1 pixel as Recent Burn, 2 as ESSFvc). These errors appear to be a result of the residual effects of misregistration and filtering in the two images rather than an incorrect attribute code in the BCMOF database, as all 3 locations occur in close proximity to a class boundary. No attempt was made to restrict the accuracy assessment pixels to interior areas of polygons, which is a common technique used in accuracy assessment to limit the effects of locational error (Stehman and Czaplewski 1998). Therefore, a logical result is that some of these pixels located near class edges or boundaries are likely to be misclassified in the assessment. The disturbance class with the lowest user's accuracy and highest omission error rate is Recent Disturbance (Unknown Agent – most likely burns for which no code was provided in the BCMOF inventory database). The error pattern for this class arises from a few locations that were omitted from this class and subsequently grouped with the Immature Forest or Parkland class. The 3 pixels classified as a Recent Disturbance (instead of Immature Forest) appear to be a result of inaccurate BCMOF forest inventory attribute information. Specifically, there were a few incorrect dates in the BCMOF database; i.e. areas in which the database indicated stand age < 23 years but the stand actually existed in 1975 as Immature Forest. Based on each individual class accuracy levels, it is apparent that the HDT algorithm was successful at mapping disturbance features occurring between 1975 and 1997, despite the examples cited above.

Assessment of 1997 Non-Forested Habitat Features:

The accuracy levels for the non-forested and forested habitat units indicate the success of the other three components of the HDT algorithm: the K-Means unsupervised classification of non-forested cover types, the MLC supervised classification of forested habitat classes, and the post-classification sorting procedure. The non-forested habitat

class with the lowest accuracy level (see Table 5.1 - 88% producer's accuracy and 92% user's accuracy) is the combined Ice/Rock/Road/Lake class and is most often confused with the Alpine Tundra cover class, probably due to the similar spectral character of alpine vegetation and rock. For example, alpine tundra field sites were characterized by open alpine meadows of heather and sedge and terrestrial lichen species sparsely distributed on a rock substrate. Based on the observed errors of commission and omission of this class, the TM imagery appears to have resolved some of the sparse vegetation cover (heather, lichen, moss, bryoids) in the rocky terrain; but is perhaps not sensitive to smaller vegetation patches which were often classified as rock.

Assessment of 1997 Forested Habitat Features:

The forested habitat class with the lowest producer's accuracy is the Mature Cedar/Hemlock (ICHwk); 89% accuracy was obtained based on a sample of 106 locations. The highest commission error occurs between the ICHwk and the Shrub Cover class. These locations represent pixels that were committed to or classified as Shrub cover instead of as the correct ICHwk forest class. A cursory review of some of these errors suggests that they have originated from the misclassification of mixed pixels located on the edge of a shrub/ICHwk boundary. In this situation, the spectral value of a pixel representing a transition from shrub to ICHwk is a mixture of the spectral signatures of the shrub understory class and the ICHwk class. As a result, the classification algorithm often committed ICHwk pixels to the shrub class (8 locations). Wang and Civco (1994) cited the misclassification of mixed pixels as a common problem of statistical-based image classification. A minor commission error (2 locations) also exists between the ESSFvc class. The user's accuracy for the ICHwk class is 94%. The forested habitat class with the highest rate of omission error (user's accuracy) is the Mature Engelmann Spruce/Subalpine Fir (ESSFvc) which is most often misclassified as Shrub or ICHwk.

The minor ESSFvc and ICHwk confusion may be a consequence of the transition in forest species composition across an elevation gradient. For example, western hemlock

dominant/subalpine fir codominant stands at the higher elevations of the ICHwk zone were often confused with subalpine fir dominant/western hemlock codominant stands in the lower elevation range of the ESSFvc zone. As a result of the transition area between biogeoclimatic zones, it was expected that the misclassification between ESSFvc and ICHwk pixels would be higher, resulting in lower accuracy levels for each of these classes. One explanation for the higher than expected forest class accuracies may be found within the field sampling protocol. As outlined in section 3.4.1 of this thesis, to overcome potential geometric inaccuracies in the imagery when matching field data to a pixel UTM locations, it was decided that plot locations should be approximately three pixel lengths from any stand boundary (75 metres). However, by sampling away from stand boundaries, it is possible that the majority of the field points used in the accuracy assessment were located outside of the ICHwk/ESSFvc transition zone.

5.2.3 1975 Habitat Unit Classification

The 1975 habitat unit image derived from the Hybrid Decision Tree classification of the Landsat MSS satellite imagery and BCMOF forest inventory database is presented in Figure 5.2. Visual assessment of the map product relative to the 1997 habitat unit map depicted in Figure 5.1 reveals that the 1975 HDT classification results are generally consistent with those of the 1997 HDT algorithm. The spatial pattern of the habitat classes (Shrub, Riparian, ICHwk, ESSF, Parkland, Alpine etc.) is similar to that of the 1997 habitat unit product, as discussed in section 5.2.1. Comparison of the disturbance classes between the 1975 and 1997 map products illustrates the intensive timber harvesting that has occurred in the study area over the 22 year time period. In the 1975 classification the Recent Cuts are limited to the Sale Mountain area, Woolsey Creek, and Tangiers River valleys. The Recent Burns and Recent Unknown disturbance classes in the 1975 classification are much more extensive in area compared to the 1997 classification, while the occurrence of Immature forests in 1975 is less frequent than in the 1997 classification.

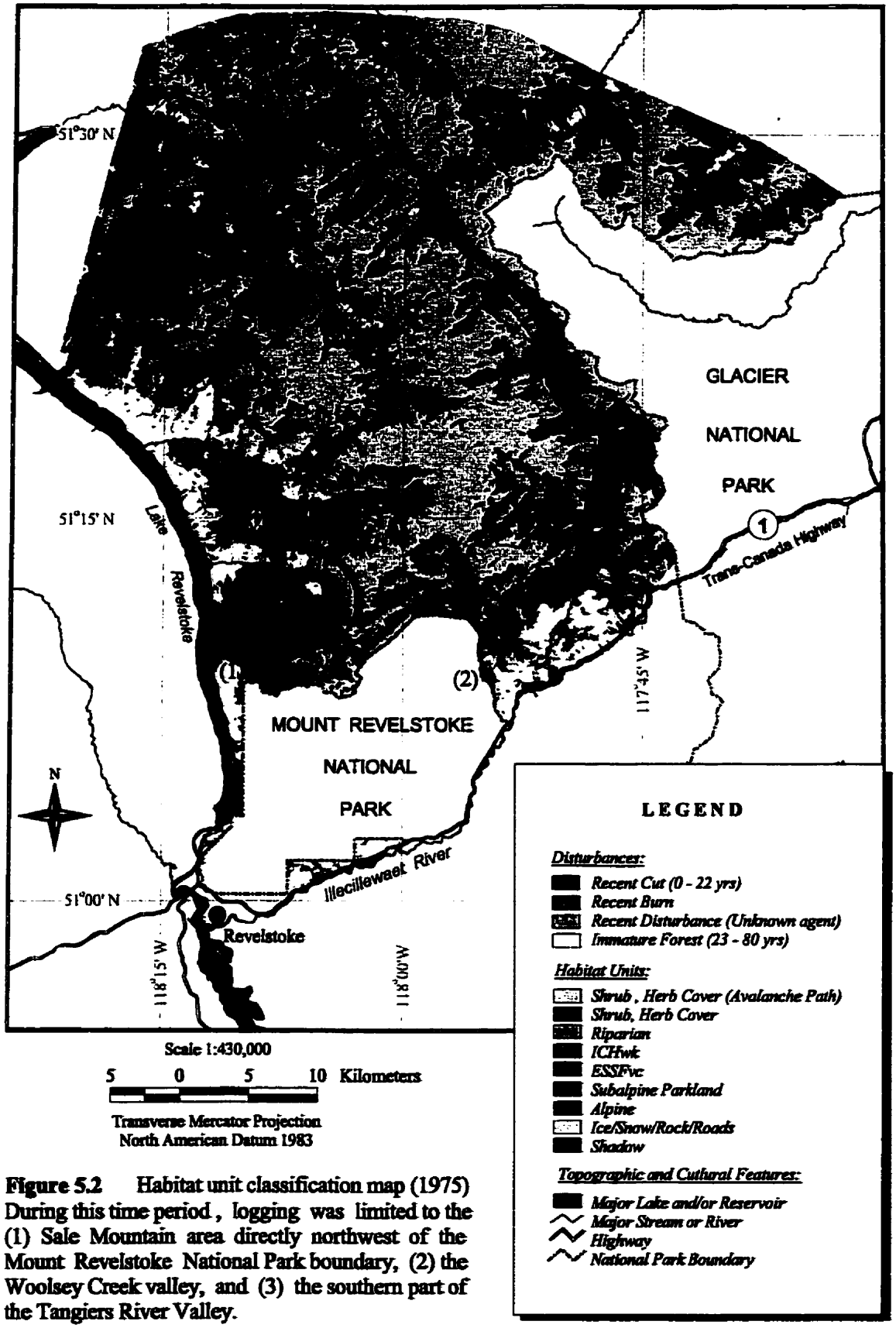


Figure 5.2 Habitat unit classification map (1975) During this time period, logging was limited to the (1) Sale Mountain area directly northwest of the Mount Revelstoke National Park boundary, (2) the Woolsey Creek valley, and (3) the southern part of the Tangiers River Valley.

5.2.4 Accuracy Assessment of 1975 Habitat Unit Classification

In the 1975 MSS image habitat classification (Table 5.2) an overall accuracy of 89.5% was obtained with a confidence interval of +/- 4.8% at a 99% confidence level. A total of 451 pixel locations were checked in this accuracy assessment. The individual class accuracy levels are generally consistent with those reported for the 1997 map product, with a few notable exceptions. First, both the user's and producer's accuracy for the Alpine Tundra class is substantially lower than that of the 1997 Alpine Tundra class accuracy. At 52%, the user's accuracy shows a large error of omission with the mixed class of Ice/Rock/Road/Lake. This error makes sense with such a diverse class and low MSS radiometric resolution (i.e. little sensitivity to differences in reflectance). For example, the 30 metre pixel resolution of the TM imagery appears to more effectively resolve the sparse alpine vegetation cover; but the 80 metre spatial resolution of the MSS image is less sensitive to these small vegetation patches and classified those areas as rock. The high commission error of the combined Ice/Rock/Road/Lake class also supports this explanation, as almost half of the Rock field locations (11) are committed to the Alpine Tundra class.

Similarily, the ESSFvcp (Parkland) class is often confused with the combined Ice/Rock/Road/Lake class, as five pixels were misclassified as ESSFvcp instead of rock. These errors appear to be a result of uncertainty within shadowed areas of the MSS imagery. For example, areas having a shaded relief value less than 50 were not included in the MLC supervised classification and were assigned a habitat unit based on the aspect and elevation values. A pixel reporting an elevation and aspect range consistent with the ESSFvcp zone is included in this class when the pixel may have actually represented a rock location.

Table 5.2 Contingency table resulting from 1975 habitat unit classification accuracy assessment

		Reference Data (Field and Aerial Photography Samples)											
		Cut	Burn	Unk.	Imm.	Shrub	AT	I/R/R/L	ICHwk	ESSFvc	ESSFvcp	Row Total	% User's
Classification Data	Recent Cut	**31**	-	-	1	-	-	-	2	-	-	34	**91**
	Recent Burn	-	**41**	-	-	-	-	-	1	2	-	44	**93**
	Recent Unknown	-	-	**40**	-	-	-	-	2	-	-	42	**95**
	Immature Forest	-	-	-	**24**	-	-	-	3	-	-	27	**89**
	Shrub	-	-	1	1	**66**	-	-	3	1	-	72	**92**
	AT	-	-	-	-	-	**12**	11	-	-	-	23	**52**
	I/R/R/L	-	-	-	-	-	3	**23**	-	-	2	28	**82**
	ICHwk	-	-	-	-	4	-	-	**78**	1	-	83	**94**
	ESSFvc	-	-	-	-	2	-	-	2	**49**	-	53	**92**
	ESSFvcp	-	-	-	-	-	-	5	-	-	**40**	45	**89**
	Column Total	31	41	41	26	72	15	39	91	53	42	**404**	
	% Producer's	**100**	**100**	**98**	**92**	**92**	**80**	**59**	**86**	**92**	**95**		

Map Accuracy = 89.6 % 99 % Confidence Interval = +/- 4.8 % (84.8 % - 94.4 %)

Shadowing of the imagery is most apparent at higher elevations due to the rugged terrain. Therefore, the confusion between rock and parkland pixels is more common than the misclassification of the lower elevation forest classes (ICHwk and ESSFvcp). Perhaps the Ice/Rock/Road/Lake pixels falling within the ESSFvcp zone should have been merged into a single class, based on the observation that each habitat unit has the same habitat suitability index (0.20) in terms of early winter habitat use by the local caribou population.

Finally in Table 5.2, the ICHwk class shows a higher error of commission than that of the 1997 classification (86% producer's accuracy), this time with most of the other disturbance and forest classes. These errors may be a result of the residual misregistration and filtering errors in the two images.

The overall map accuracy of the 1997 and 1975 habitat images and class by class accuracy statistics are reasonable and consistent with reports in the literature for similar mapping studies of forested, mountainous environments. For example, in one study by Franklin *et al.* (1994) in the Kananaskis Valley of Alberta, a straightforward maximum likelihood classification of ten general land cover types using 1992 Landsat TM imagery yielded an overall mean classification accuracy of 66%. This was improved to 78.7% with access to variables derived from a high resolution DEM. In 28 detailed cover classes, an overall accuracy of 72.7% was obtained. One main difference between Franklin *et al.* (1994) and this thesis was the availability of extensive GIS data in which the collection of training samples was optimized. And, as expected, the Hybrid Decision Tree classifier in the present study appears to have provided a significant improvement in accuracy over that which would have been obtained using a straightforward statistical decision rule (such as the MLC). In the New Jersey Pine Barrens study reported by Luque *et al.* (1994), overall classification accuracies for Landsat MSS data of 87.9% and for TM data of 88.9% were obtained in five land cover class maps using a supervised maximum likelihood classification procedure. Sachs *et al.* (1998) applied a combination of unsupervised and supervised classification techniques to produce a 1992 land cover map

from Tasseled Cap TM images in the Cariboo and Prince George forest districts of British Columbia. The authors reported an overall accuracy of 79% for 12 land cover classes. A possible reason for the higher accuracy level of the present mapping project (91.8% compared to 79% for Sachs *et al.* 1998) is that immature forest stands were extracted from the forest inventory database, while Sachs *et al.* relied on the Tasseled Cap imagery alone for discrimination between conifer age classes. As a result, mature and old conifer pixels were often confused and 56 percent of the young conifer pixels within the Sachs *et al.* (1998) study were omitted from that class and grouped with one of the other closed canopy classes.

5.2.5 Propagation of Error in 1975 Habitat Suitability Model

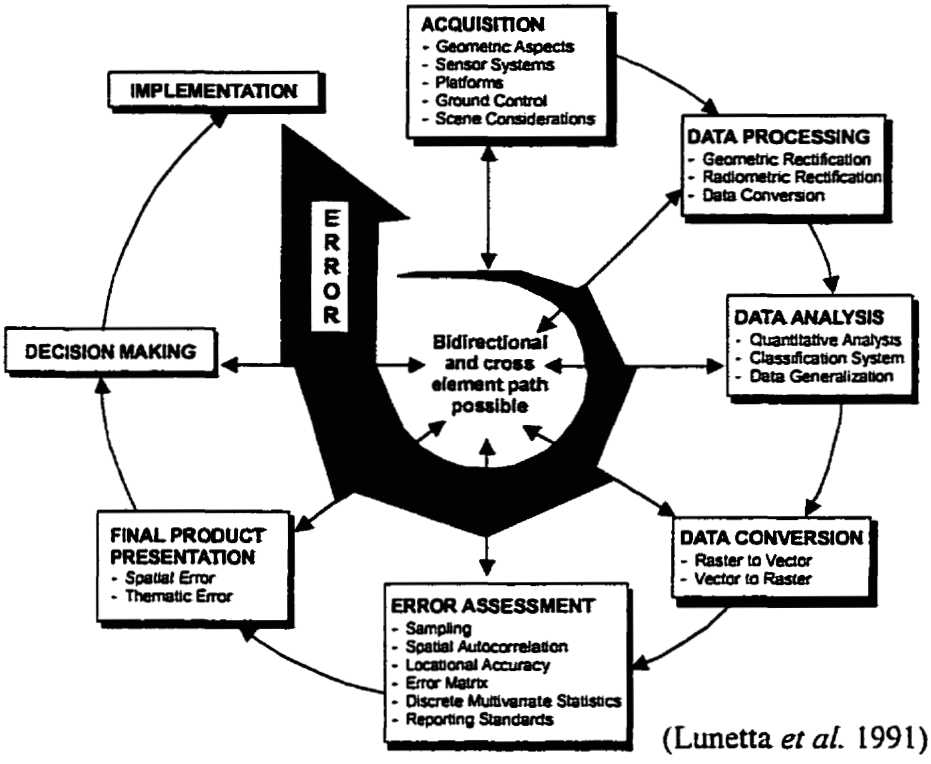
The accuracy of the 1997 and 1975 habitat maps discussed above allowed the evaluation of the success of the Hybrid Decision Tree algorithm for mapping habitat and disturbance features from multiple data sources. The effects of combining these two classification products using GIS overlay operations should also be assessed to determine how error from each image source may have been propagated to the final overlay output image. Sections 4.3.3 summarized a procedure used to "backcast" or reconstruct a 1975 habitat unit map from the 1997 database. Specifically, the Arc/Info COMBINE function was used to overlay the 1997 and 1975 habitat unit images. One potential problem of a post-classification comparison approach of digital image sources is that the combined product or overlay image is likely to exhibit a combined accuracy level that is a product of each individual classification (Stow *et al.* 1980). Therefore, the overall map accuracy of the backcast 1975 habitat unit image used in the HSI model equation would be a joint classification rate of the image products produced by the HDT algorithm. To simply calculate the joint classification accuracy rate of the 1975 overlay image, the two accuracy levels are multiplied and converted to a percentage (Singh 1989):

$$\text{Joint Accuracy} = 0.918 \times 0.869 \times 100 = 82.25\% \qquad \text{Eq. 14}$$

This value indicates that the average classification accuracy of the backcast 1975 habitat unit image is 82.25%. However, it should be noted that the 1997 habitat unit image formed the base map for all areas that remained unchanged in terms of cover type between 1975 and 1997. These areas form a large proportion (96.25%) of the study area, and the 1997 classification accuracy level (91.8%) best represents the attribute certainty of these locations. The joint image accuracy level applies only to those areas that had been harvested, burned or regenerated from the Immature forest class (< 80 years stand age) to one of the Mature forest classes (> 80 years of age). For these locations (3.75% of study area), the joint accuracy level of 82.25% applies.

5.2.6 Other potential sources of error

Figure 5.3 summarizes error introduced through GIS overlay operations and other potential sources of error that may be introduced through the data processing flow of an integrated remote sensing/GIS classification approach. Satellite data acquisition errors, geometric and radiometric rectification, data generalization, vector to raster conversion, telemetry location uncertainty, field sampling design, error assessment and reporting standards, and thematic error in the final image products are recognized as some of the most common sources of uncertainty encountered in a GIS application. The end result of error propagation places limits on the confidence with which resource managers may use remote sensing and GIS derived images and output maps into the decision making process (Lunetta *et al.* 1991).



(Lunetta *et al.* 1991)

Figure 5.3 Sources of error that may accumulate in an image classification project

To assess and describe each potential source of error and their accumulation through the information flow is beyond the scope of this research. For a comprehensive discussion of error propagation in remote sensing and GIS analysis, the reader is referred to Lunetta *et al.* (1991). On a related issue, Stoms *et al.* (1992) investigated the sensitivity of wildlife habitat models to uncertainties in model assumptions and input data. This thesis research has attempted to quantify and minimize the amount of error introduced into the GIS habitat suitability models and resulting maps wherever possible. For example, ground truth information was collected and root mean square error values were calculated to determine geometric accuracy of the TM and MSS satellite imagery and the attribute accuracy of the BCMOF forest inventory database. The analysis of the classification error matrices for the 1997 and 1975 habitat unit images serve to quantify some of the uncertainty introduced by the Hybrid Decision Tree algorithm. However, it seems probable that additional error has accumulated in the final habitat suitability maps

through the combination of elevation and slope data, forest stand age information, and the habitat unit images.

Of particular concern is the estimation of 1975 stand age for disturbance features identified in the 1997 classified scene (as described in section 4.3.4). Consequently, the pre-disturbance age may have been underestimated resulting in lower habitat suitability ratings than expected for many areas. This may be reflected in the total amount of quality habitat removed. For example, if a large number of pixels were placed in HSI class 7 instead of HSI class 8 because the 1975 interpolated stand age was lower than expected, these locations may not be considered to represent a loss of critical early winter habitat. Based on this interpretation, it is possible that the amount of habitat lost to timber harvesting and wildfires will also be underestimated.

Other potential sources of error may have been introduced in the fragmentation analysis itself. The landscape metrics used to quantify the spatial characteristics and configuration of habitat were chosen based on a review of those applied in related research projects. However, no sensitivity analysis was carried out in this thesis to determine if other indices may have been more suitable, or if alternative definitions of the landscape boundary and patch attributes may have produced different results. Instead, most of the variables identified as having an effect on metric sensitivity, as identified by Haines-Young and Chopping (1996) and Franklin *et al.* (1999), were consistent between the 1975 and 1997 landscapes to limit the effect on the fragmentation analysis results. For example, the landscape boundary, raster data format, and input metric parameters such as edge distance required by the Patch Analyst software were identical for both the 1975 and 1997 habitat suitability landscapes. By addressing this potential source of uncertainty along with the examples provided in Figure 5.3, the results of the habitat fragmentation analysis may now be reported and interpreted with a general idea of the overall confidence in the accuracy of the habitat suitability maps and the spatial pattern metrics calculated for each.

5.3 Quantification and Assessment of changes in Habitat Composition

The first obvious spatial effect of timber harvesting and wildfires on critical early winter habitat areas would appear to be the loss of forested areas or the reduction of the total amount of suitable habitat available for utilization by local mountain caribou. When harvest or fire activity removes a forest stand, there are two subsequent changes in terms of habitat suitability. First, the cover type designation changes temporarily from a forested to a non-forested habitat unit and the stand age cohort or structural characteristics reverse from a climax community back to an early seral stage of regeneration. Both types of changes are captured in the habitat suitability model equation, while the other two model variables, elevation and slope are considered to remain constant between time periods. The following sections report the change in relative areas by forest cover type, stand age, and habitat suitability classes as a result of forest disturbances occurring between 1975 and 1997 in the Revelstoke region in order to address research objective 2a: the quantification and assessment of habitat fragmentation in terms of changes in overall habitat composition.

5.3.1 Changes in Forest Cover Type

A comparison of the area in each of the habitat unit classes of interest mapped in the study area is contained in Table 5.3. The most obvious change expressed in the two image products (Figures 5.1 and 5.2) is evident within the Recent Cuts class. The area of this class increased 150.75% from 1263 ha in 1975 to approximately 3168 ha in 1997. The area in Recent Burns decreased substantially from 1975 to 1997, perhaps as a direct result of fire suppression activities, but perhaps also a result natural variability in fire regimes for this region.

Other significant changes in forest cover type in Table 5.3 are apparent in each of the ICHwk and ESSFvc biogeoclimatic zones. In the ICHwk class the area in 1975 is approximately 23177 ha, and in 1997 the area is approximately 22488 ha, a decrease of almost 3%. It is important to note that the 3% decrease in area refers to all mature and

old growth stands (greater than 80 years of age) within the ICHwk biogeoclimatic zone over the whole study area. The actual decrease due to conversion of ICHwk to recent cuts and burns may be greater, but this decrease is mitigated by the conversion of Immature Forest to ICHwk by the definition of age classes. For example, there is a net increase of Immature Forest of more than 25% in the time period of this study, while 2142 ha of Immature Forest moves into the ICHwk class. This pattern is isolated in Table 5.4, which contains the summary of land cover class transitions for each of the classes presented in Table 5.3.

An important point to bear in mind while interpreting the results of Table 5.3 and 5.4 is that although it appears that natural regeneration of immature forests has offset the loss of mature and old growth habitat within the ICHwk zone, a more accurate interpretation is that only mature (80 to 180 years stand age) ICHwk class areas have increased due to regeneration. As will become apparent in section 5.3.2, almost all of the 3% ICHwk habitat disturbance or loss has occurred within old growth ICHwk stands.

A net loss of habitat is observed in Table 5.4 for the ICHwk and ESSFvc biogeoclimatic zones, while a slight increase or net gain in forested habitat is recorded for the Subalpine Parkland zone. The total net forested habitat loss is approximately 888.83 ha. Approximately 3737 ha converted from forest habitat to other classes, offset by approximately 2848 ha which converted from Immature Forest to the forested classes. Figure 5.4 shows the proportion of change in cover type within the forest habitat loss that can be traced to harvesting and to natural disturbances (burns). More than 66% of the change can be attributed to the forest harvesting in the ICHwk class compared to less than 10% change induced by natural disturbance; in the ESSFvc class this pattern is 15% and 6%, respectively. Clearly, the major differences in the forest habitat classes are directly attributable to the increased area of cutblocks in the ICHwk forest stands. The recent cut disturbance features appear to have been accurately depicted in the integrated remote sensing/ GIS classification procedure based on the accuracy levels reported for the 1997 and 1975 habitat unit images.

Table 5.3 Comparison of disturbance and forested habitat class areas

Cover Type	1975 Area (Ha)	1997 Area (Ha)	Change (Ha)	% Change
Recent cuts	1263.62	3168.56	1904.94	150.75
Recent burns	2411.38	316.06	-2095.32	-86.89
Recent unknown	2684.31	752.94	-1931.37	-71.95
Immature forest	13270.88	16617.81	3346.93	25.22
ICHwk	23177.06	22488.69	-688.37	-2.97
ESSFvc	26572.19	26242.19	-330.00	-1.24
ESSFvcp	35867.31	35997.12	129.81	0.36

Table 5.4 Summary of forest cover class transitions between 1975 and 1997

1975 Cover Type	1997 Disturbance Class	Total Area (Hectares)	Gain / Loss
ICH	Harvest	2489.75	-
ESSF	Harvest	577.38	-
ICH	Natural	341.69	-
ESSF	Natural	229.25	-
Parkland	Natural	99.13	-
Immature Forest	ICH	2142.38	+
Immature Forest	ESSF	477.25	+
Immature Forest	Parkland	228.94	+

Total ICH Habitat Loss	= 2830.76 Ha	Total ICH Habitat Gain	= 2142.38 Ha
Total ESSF Habitat Loss	= 806.63 Ha	Total ESSF Habitat Gain	= 477.25 Ha
Total Parkland Habitat Loss	= 99.13 Ha	Total Parkland Habitat Gain	= 228.94 Ha
Total Forested Habitat Loss	**= 3737.40 Ha**	**Total Forested Habitat Gain**	**= 2848.57 Ha**

Net ICH Habitat Loss	= 688.38 Ha
Net ESSF Habitat Loss	= 329.38 Ha
Net Parkland Habitat Gain	= 129.81 Ha
Net Forested Habitat Loss	**= 888.83 Ha**

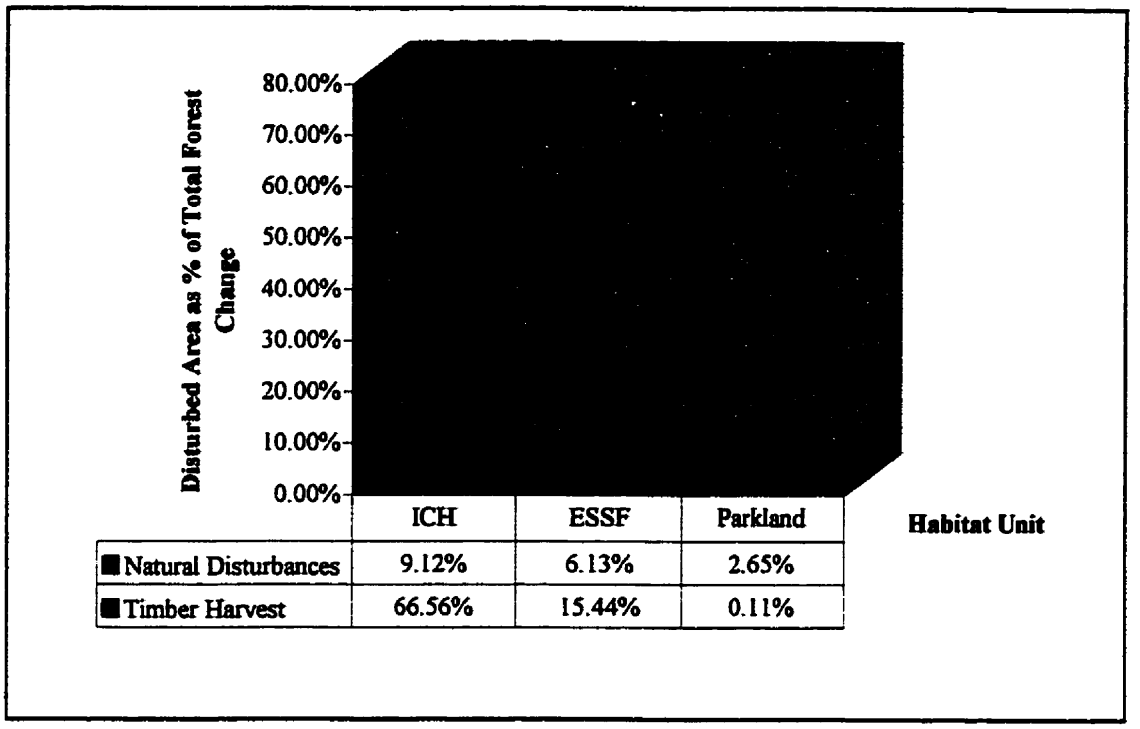


Figure 5.4 Relative disturbance areas per habitat unit zone

5.3.2 Changes in Forest Stand Age

Table 5.5 contains a summary of the age class transitions observed between the 1975 and 1997 forest habitat classifications across the landscape. Note that this age class division represents an amalgamation of the 10 original age classes within the B.C. Forest Inventory age classification scheme into 5 broad age categories. The table diagonal represents the area in hectares per age class that remained unchanged between 1975 and 1997 (retained the same age class). Values to the lower left of the diagonal report the total area per age class that moved into a higher age class (regeneration) while the values to the upper right of the diagonal summarize forested landscape area that moved to a lower age class (disturbance). By subtracting the observed decrease in class area from the increase, a total change value per class indicates the amount and nature of the age class change (i.e. whether the observed change indicated regeneration or disturbance).

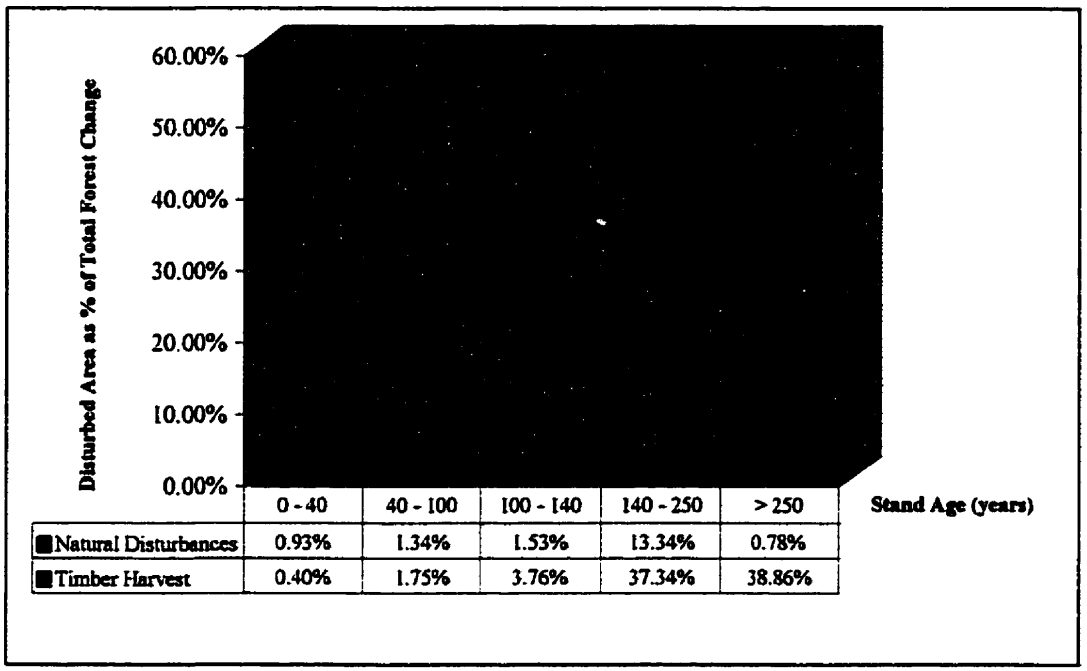
Table 5.5 Stand age transition matrix within ICHwk, ESSFvc, and ESSFvcp biogeoclimatic zones

		Stand Age Class Distribution 1975					
		Class 1 0 - 40	Class 2 40 -100	Class 3 100 - 140	Class 4 140 - 250	Class 5 > 250	1997 Area
Age Class 1997	Class 1: 0 - 40 years	**5100**	133	201	1923	1493	8850
	Class 2: 40 - 100 years	3569	**9320**	0	0	0	12888
	Class 3: 100 - 140 years	0	2095	**1390**	0	0	3485
	Class 4: 140 - 250 years	0	0	2734	**27542**	0	30276
	Class 5: > 250 years	0	0	0	1439	**12048**	13487
	1975 Class Area (Ha)	8669	11548	4325	30904	13541	
	Total Increase (Ha)	3569	2095	2734	1439	0	
	Total Decrease (Ha)	0	133	201	1923	1493	
	Total Change (Ha)	3569	1962	2533	-484	-1493	
	% Change	**41.17%**	**16.99%**	**58.57%**	**-1.57%**	**-11.03%**	

Net Decrease in Age Class = 3751 Ha (5.44 % of Forest Matrix)
Net Increase in Age Class = 9837 Ha (14.26 % of Forest Matrix)
No Change in Age Class = 55399 Ha (80.30 % of Forest Matrix)

The % Change statistic presents the Total Change as a percentage of the 1975 class area. Approximately 80% of the forest matrix did not experience any change in age class over the 22 year time period of this study. However, more than 14% of the area experienced an increase in age class (regeneration), offset by approximately 5% of the area which experienced a decrease in age class (disturbance). The distribution of these changes reveals that the negative values were located in the older age class stands. These stands were converted from older ages to younger, primarily through forest harvesting and fire. The younger age classes all experienced a positive increase in area. Class 3 (100-140 yrs) experienced the largest single change in age class, with a +58% change. It is possible that this significant increase in Class 3 area is due to the transition of extensive areas burned in the 1880's (MacDonald 1996) from Class 2 (40 – 100 yrs) in 1975 into Class 3 in 1997.

Figure 5.5 shows the breakdown of the age class changes by disturbance type in the total forested area. Clearly, from 1975 to 1997, the two older classes experienced the largest negative change in area and that change can be directly attributed to the forest harvesting activity in the ICHwk forest habitat class. Burns accounted for approximately 14% of the total forest habitat lost, primarily in the 140-250 yr. age class, while harvesting accounted for more than 75% of the total area removed in these two age classes of the forested habitat.



	0 - 40	40 - 100	100 - 140	140 - 250	> 250
Natural Disturbances	0.93%	1.34%	1.53%	13.34%	0.78%
Timber Harvest	0.40%	1.75%	3.76%	37.34%	38.86%

Figure 5.5 Relative disturbance areas per stand age class

5.3.3 Changes in Habitat Suitability

The map products produced from the Habitat Suitability Index (HSI) models for the 1975 and 1997 time periods are provided in Figure 5.6(a) and Figure 5.6(b). By comparing each of the HSI maps to the habitat unit images for the 1997 (Figure 5.1) and 1975 (Figure 5.2) time periods, the link between the variables entered into the HSI model (elevation, slope, habitat unit, and stand age) can be visualized and interpreted. For example, the lowest HSI classes (HSI classes 1-3) correspond to the

Ice/Snow/Rock/Roads, Alpine Tundra, and Subalpine Parkland habitat units depicted in Figures 5.1 and 5.2. Each of these habitat unit classes was assigned low HSI values as a result of the telemetry analysis of preferential caribou habitat selection in the study area (McLellan *et al.* 1995) (refer to Table 4.4 for HSI value assignment per habitat variable). In addition, each of these classes is limited to higher elevations, also resulting in a low HSI value for the elevation variable. The highest or most suitable HSI classes in Figures 5.6(a) and 5.6(b) (HSI classes 8-10) occur generally at lower elevations (between elevations of 850 and 1300 metres), across a range of slopes between 20 and 50 per cent, and in cedar-hemlock stands greater than 250 years of age.

The most extensive areas of suitable early winter habitat in 1975 (Figure 5.6(a)) occur in the Carnes Creek, Downie Creek, and La Forme Creek watersheds. A visual comparison of each of these areas between 1975 and 1997 reveals that large portions of suitable early winter habitat (HSI classes 8-10) have transformed from a high HSI class in 1975 to a lower HSI class in 1997. In particular, in the southern slopes of the Carnes Creek valley and the central portion of the Downie Creek valley, the spatial extent of HSI classes 8, 9, and 10 and the spatial characteristics and configuration of these important habitat areas has been affected by disturbance. Large, contiguous areas of suitable habitat patches in 1975 have been reduced in size by the conversion to low quality habitat. The grey patches occurring in these areas in 1997 correspond primarily to recent timber harvest cuts depicted in Figure 5.1. Based on these observations, the HSI maps provided in Figures 5.6(a) and 5.6(b) suggest that timber harvesting and wildfires have affected the composition and configuration of quality early winter habitat in the study area between time periods.

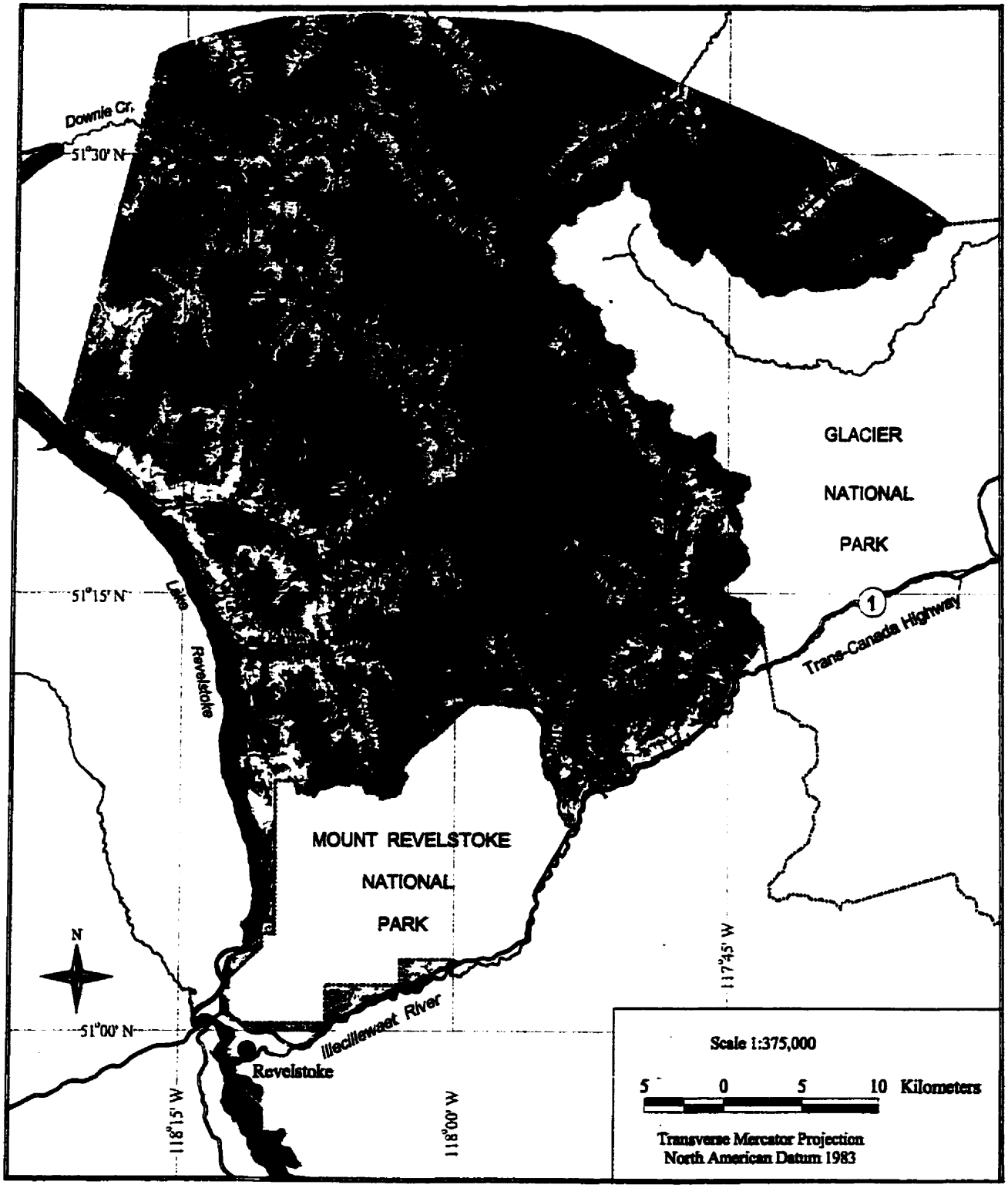


Figure 5.6 (a) Spatial distribution of habitat suitability classes for the early winter season in 1975.

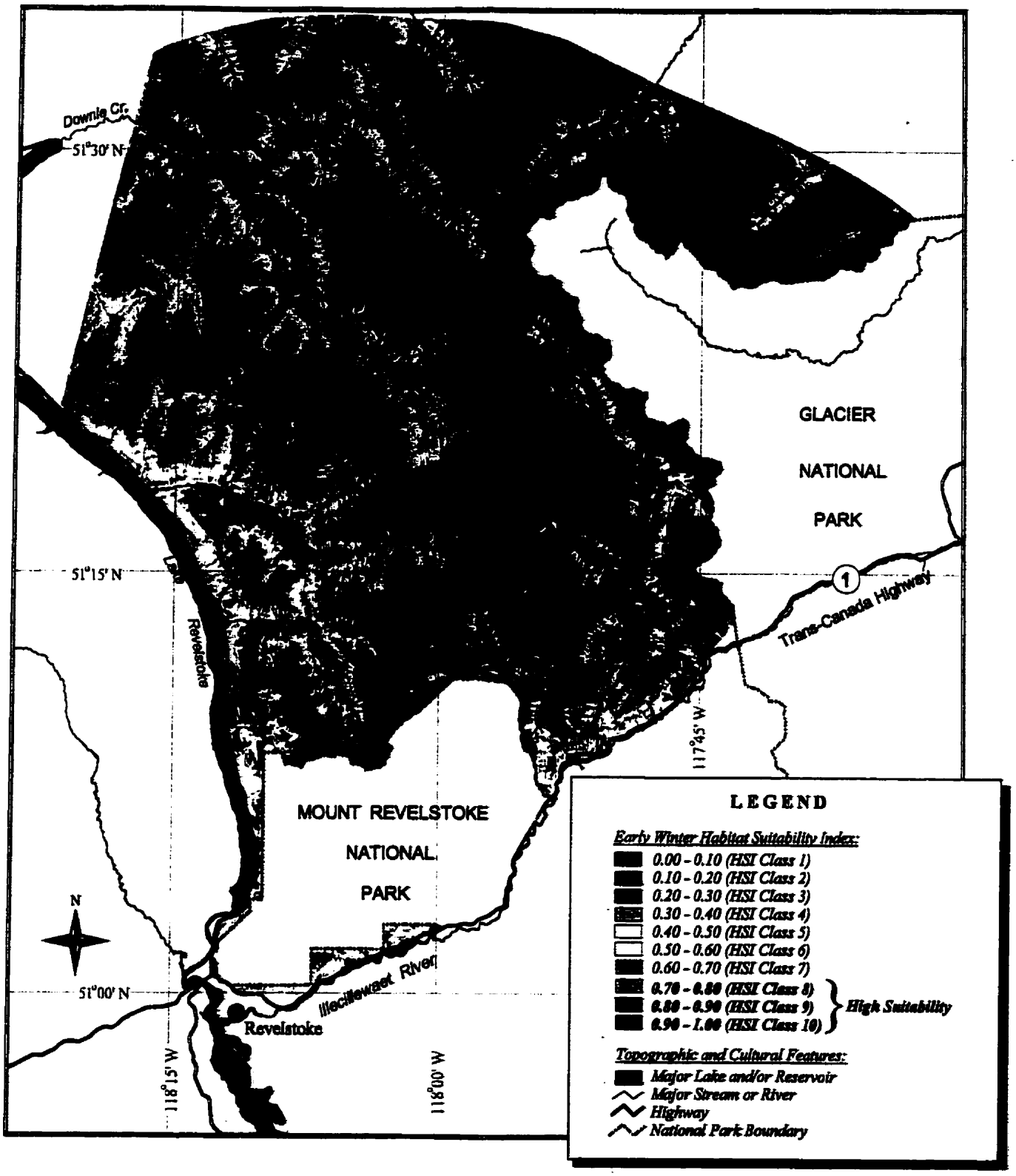


Figure 5.6 (b) Spatial distribution of habitat suitability classes for the early winter season in 1997. Those areas experiencing a negative change in HSI value between 1975 and 1997 represent a forest disturbance feature in Figure 5.1, while those areas experiencing an increase in HSI value are indicative of stand regeneration.

However, a visual assessment of these changes is limited in that the total amount of habitat loss and changes in the spatial characteristics and configuration of habitat has not been quantified or objectively compared. The following sections address this research objective by quantifying and assessing the fragmentation of early winter caribou habitat in the study area.

Table 5.6 summarizes changes in HSI class area for the study area between 1975 and 1997. The table diagonal represents the amount of area where the habitat suitability rating did not change between 1975 and 1997. Values to the lower left indicate the area per HSI class that has experienced an increase in habitat suitability as a result of regeneration or increase in stand age over time. The areas of HSI class that declined as a result of timber harvesting or fire are reported to the upper right of the table diagonal. As compared to the stand age transition matrix (Table 5.5) which only reports changes in stand age for the forested matrix, the HSI transition matrix was computed from the complete study area including both forested and non-forested sites. Therefore, the total amount of area associated with a decrease in HSI for the study area is quite small (only 2.55%), while a large proportion has increased in habitat suitability as a result of stand regeneration (4.39%) or retained the same habitat suitability rating (93.06% of the study area). This would suggest that the impact of harvesting and fire has not been that substantial when considering the entire study area. However, by interpreting these results at the HSI class level rather than the landscape level of analysis, it becomes apparent that the most suitable early winter habitat areas (e.g. HSI classes 8, 9, 10) have experienced the highest disturbance rate relative to the 1975 class area. For example, the highest negative % change statistics are reported for HSI class 8, 9, and 10. The effects of stand regeneration and disturbance reported in the HSI transition matrix are balanced by simply comparing the overall change in area of each habitat suitability class between 1975 and 1997 (Table 5.7) and the change as a percentage of the 1975 class area (Figure 5.7). HSI classes 8, 9, and 10 report the highest decrease in overall area between 1975 and 1997 indicating that forested areas providing high quality early winter habitat are being disturbed at a faster rate than replacement by stand regeneration.

Table 5.6 HSI class transition matrix summarizing changes in habitat suitability rating between 1975 and 1997 for complete study area (forested and non-forested habitat)

		1975 HSI Class Area (Ha)										
		Class 1 0.0 - 0.1	Class 2 0.1 - 0.2	Class 3 0.2 - 0.3	Class 4 0.3 - 0.4	Class 5 0.4 - 0.5	Class 6 0.5 - 0.6	Class 7 0.6 - 0.7	Class 8 0.7 - 0.8	Class 9 0.8 - 0.9	Class 10 0.9 - 1.0	1997 Area
1997 HSI Class Area (Ha)	Class 1	**8963**	0	0	0	0	0	0	0	0	0	8963
	Class 2	7	**87487**	338	190	247	81	68	0	0	0	88418
	Class 3	0	716	**23066**	407	196	515	737	791	434	52	26914
	Class 4	0	113	2553	**13410**	207	71	25	55	0	0	16434
	Class 5	0	3	651	721	**8944**	28	8	0	0	0	10355
	Class 6	0	1	180	802	207	**7506**	15	0	0	0	8711
	Class 7	0	0	23	707	1	231	**5674**	5	0	0	6642
	Class 8	0	0	2	205	0	2	251	**4942**	0	0	5403
	Class 9	0	0	0	0	0	0	44	241	**2800**	0	3085
	Class 10	0	0	0	0	0	0	0	39	0	**408**	447
1975 Class Area (Ha)		8970	88321	26814	16442	9801	8435	6821	6074	3234	460	
Total Increase (Ha)		7	834	3410	2435	208	233	295	280	0	0	
Total Decrease (Ha)		0	0	338	597	650	696	852	851	434	52	
Total Change (Ha)		7	834	3072	1839	-442	-463	-557	-571	-434	-52	
% Change		**0.08%**	**0.94%**	**11.46%**	**11.18%**	**-4.51%**	**-5.48%**	**-8.17%**	**-9.40%**	**-13.43%**	**-11.39%**	

Net Decrease in HSI = 4470 Ha (2.55 % of Landscape area)
Net Increase in HSI = 7703 Ha (4.39 % of Landscape area)
No Change in HSI = 163198 Ha (93.06 % of Landscape area)

If high quality habitat is considered to be that above 0.70 HSI, then 8.53% of the quality early winter habitat has been lost since 1975. Figure 5.8 shows the breakdown of the habitat loss by disturbance type and compares the relative effects of fire and timber harvesting on the observed habitat loss. Stand disturbance due to timber harvesting appears to be the primary cause of the net decline in area of high quality habitat. These results support the conclusion that quality early winter habitat has experienced a decrease in area and that timber harvesting and wildfires have altered the composition of the landscape over the 22 year period of this investigation. The effects of the observed disturbance regime on the spatial characteristics and configuration of the remaining habitat patches are discussed in section 5.4.

Table 5.7 Comparison of habitat suitability class areas between time periods

HSI Class	1975 Area (Ha)	1997 Area (Ha)	Change (Ha)	% Change
1 : 0.00 – 0.10	8970.19	8970.10	0.00	0.00 (+)
2 : 0.10 – 0.20	88320.62	88417.76	97.14	0.11 (+)
3 : 0.20 – 0.30	26813.88	26913.95	100.07	0.37 (+)
4 : 0.30 – 0.40	16442.00	16434.44	-7.56	0.05 (-)
5 : 0.40 – 0.50	9801.32	10354.69	553.37	5.65 (+)
6 : 0.50 – 0.60	8434.65	8711.06	276.42	3.28 (+)
7 : 0.60 – 0.70	6821.20	6641.76	-179.44	2.63 (-)
8 : 0.70 – 0.80	**6073.76**	**5402.64**	**-671.13**	**11.05 (-)**
9 : 0.80 – 0.90	**3233.82**	**3084.94**	**-148.88**	**4.60 (-)**
10 : 0.90 – 1.00	**460.01**	**447.14**	**-12.87**	**2.80 (-)**

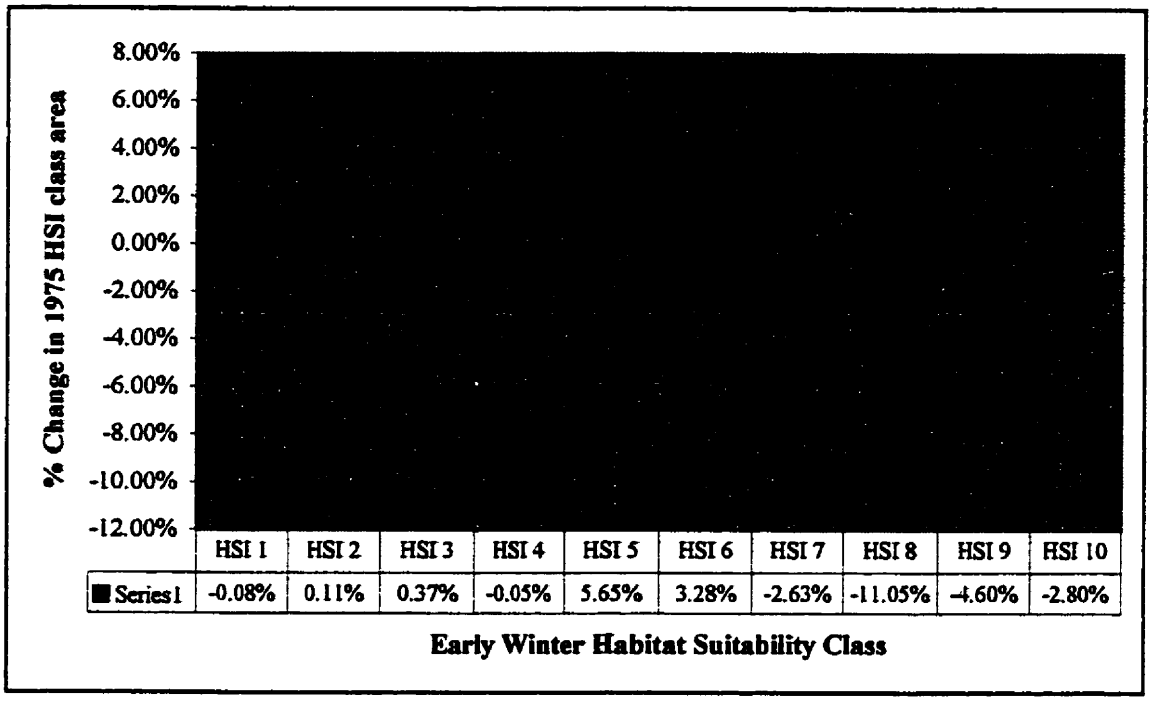


Figure 5.7 Summary of habitat loss per HSI class between 1975 and 1997

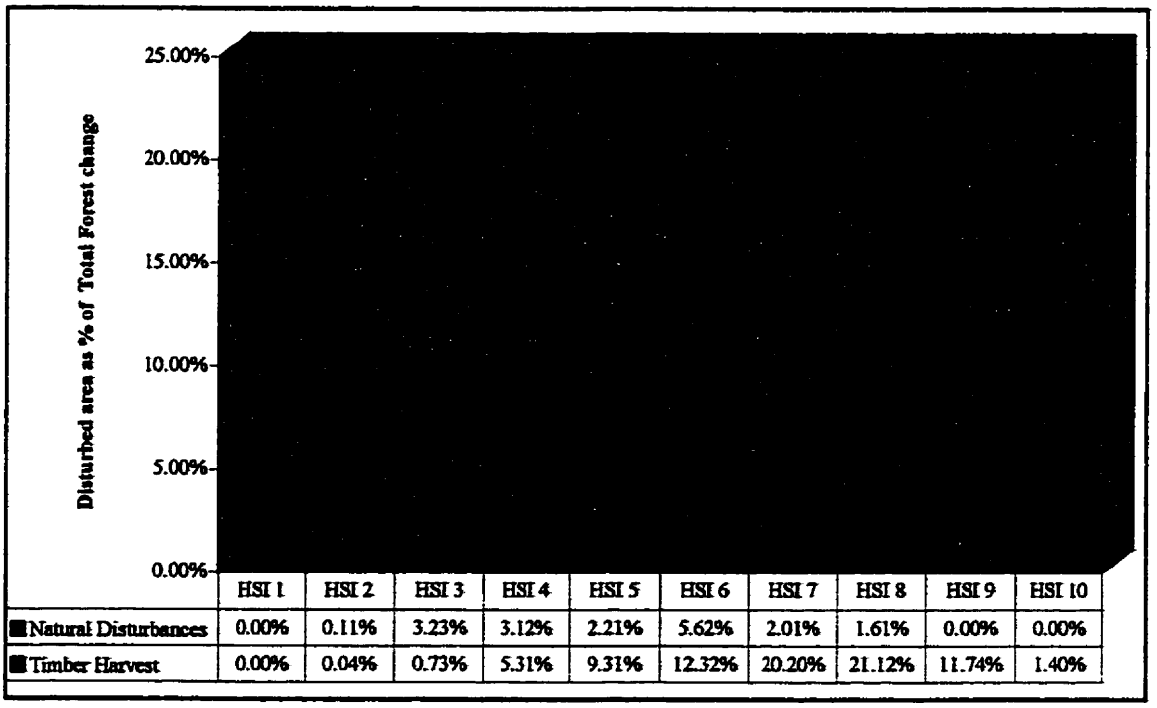


Figure 5.8 Relative disturbance areas per habitat suitability class

5.4 Quantification and Assessment of changes in Habitat Configuration

Table 5.8 contains the computed values for the following spatial metrics in the 1975 and 1997 habitat suitability maps:

- Class Areas (also presented in Table 5.7)
- Number of Patches
- Patch Density
- Mean Patch Size
- Patch Size Coefficient of Variation
- Edge Density
- Mean Shape Index
- Mean Proximity Index
- Mean Core Area
- Interspersion / Juxtaposition

The areas with the highest habitat suitability rating (classes 8, 9, 10) represent the focal patch types of the fragmentation analysis and are indicated in bold type for each spatial metric in Table 5.8. The first metric presented in Table 5.8 is Class Area and relates to the previous discussion provided in section 5.3.3 concerning changes in the total amount of suitable habitat in 1975 and 1997. Based on Tables 5.7 and 5.8, the change in Class Area between time periods indicates that the total area or amount of quality early winter habitat decreased by 8.53% between 1975 and 1997. Those areas having habitat suitability ratings between 0.70 and 0.80 (HSI class 8) experienced the largest reduction in area (11.05%). The patch abundance (Number of Patches) and Patch Density for each focal HSI class also increased as a result of landscape disturbances, as would be expected with more habitat patches in the same total area.

Table 5.8 Comparison of metrics summarizing distribution and spatial configuration of habitat suitability patches (1975 vs. 1997)

Class Areas (Ha)

HSI Class	1975	1997	Change	%
1	8970.19	8970.19	0.00	0.00
2	88320.69	88418.75	98.06	0.11
3	26814.88	26913.94	99.06	0.37
4	16442.00	16434.44	-7.56	-0.05
5	9801.31	10354.69	553.38	5.65
6	8434.38	8712.06	277.68	3.29
7	6821.19	6641.75	-179.44	-2.63
8	**6073.75**	**5402.62**	**-671.13**	**-11.05**
9	**3233.81**	**3084.69**	**-149.12**	**-4.61**
10	**460.00**	**447.12**	**-12.88**	**-2.80**

Number of Patches

HSI Class	1975	1997	Change	%
1	5126	5126	0.00	0.00
2	5609	5534	-75.00	-1.34
3	7896	7918	22.00	0.28
4	6895	6663	-232.00	-3.36
5	3799	3779	-20.00	-0.53
6	3484	3590	295.00	8.47
7	2976	3093	117.00	3.93
8	**1566**	**1752**	**186.00**	**11.88**
9	**554**	**617**	**63.00**	**11.37**
10	**248**	**264**	**16.00**	**6.45**

Patch Density

HSI Class	1975	1997	Change	%
1	2.92%	2.92%	0.00	0.00
2	3.20%	3.16%	0.00	-1.34
3	4.50%	4.51%	0.00	0.28
4	3.93%	3.80%	0.00	-3.36
5	2.17%	2.15%	0.00	-0.53
6	1.99%	2.05%	0.00	8.47
7	1.70%	1.76%	0.00	3.93
8	**0.89%**	**1.00%**	**0.00**	**11.88**
9	**0.32%**	**0.35%**	**0.00**	**11.37**
10	**0.14%**	**0.15%**	**0.00**	**6.45**

Mean Patch Size (Ha)

HSI Class	1975	1997	Change	%
1	1.75	1.75	0.000	0.00
2	15.75	15.98	0.230	1.46
3	3.40	3.40	0.000	0.00
4	2.38	2.47	0.090	3.78
5	2.58	2.74	0.160	6.20
6	2.42	2.43	0.010	0.41
7	2.29	2.15	-0.140	-6.11
8	**3.88**	**3.08**	**-0.800**	**-20.62**
9	**5.84**	**5.00**	**-0.840**	**-14.38**
10	**1.85**	**1.69**	**-0.160**	**-8.65**

Patch Size Coefficient of Variation

HSI Class	1975	1997	Change	%
1	491.44	491.44	0.00	0.00
2	6387.92	6324.87	-63.05	-0.99
3	600.44	543.37	-57.07	-9.50
4	550.04	606.43	56.39	10.25
5	396.75	400.51	3.76	0.95
6	333.94	334.25	0.31	0.09
7	336.88	354.55	17.67	5.25
8	**418.32**	**308.01**	**-110.31**	**-26.37**
9	**545.13**	**452.34**	**-92.79**	**-17.02**
10	**200.59**	**194.95**	**-5.64**	**-2.81**

Edge Density

HSI Class	1975	1997	Change	%
1	17.14	17.14	0.000	0.00
2	56.11	56.09	-0.020	-0.04
3	43.68	43.94	0.260	0.60
4	29.11	29.19	0.080	0.27
5	17.44	18.03	0.590	3.38
6	15.76	16.16	2.270	14.40
7	13.37	13.30	-0.070	-0.52
8	**9.29**	**9.04**	**-0.250**	**-2.69**
9	**4.22**	**4.43**	**0.210**	**4.98**
10	**1.05**	**1.06**	**0.010**	**0.95**

Table 5.8 (cont'd) Comparison of metrics summarizing distribution and spatial configuration of habitat suitability patches

Mean Shape Index

HSI Class	1975	1997	Change	%
1	1.35	1.35	0.00	0.00
2	1.38	1.38	0.00	0.00
3	1.49	1.49	0.00	0.00
4	1.43	1.45	0.02	1.40
5	1.45	1.47	0.02	1.38
6	1.44	1.45	0.01	0.69
7	1.45	1.44	-0.01	-0.69
8	1.48	1.46	-0.02	-1.35
9	1.54	1.54	0.00	0.00
10	1.48	1.47	-0.01	-0.68

Mean Proximity Index

HSI Class	1975	1997	Change	%
1	69.08	69.08	0.00	0.00
2	196665	194565.09	-2100.16	-1.07
3	325.41	280.56	-44.85	-13.78
4	107.10	128.10	21.00	19.61
5	91.92	102.15	10.23	11.13
6	88.05	81.87	14.10	16.01
7	88.86	73.38	-15.48	-17.42
8	153.50	82.37	-71.13	-46.34
9	247.77	155.92	-91.85	-37.07
10	17.96	14.63	-3.33	-18.54

Mean Core Area (Ha)

HSI Class	1975	1997	Change	%
1	3.41	3.41	0.00	0.00
2	38.36	37.97	-0.39	-1.02
3	4.01	3.82	-0.19	-4.74
4	3.65	3.84	0.19	5.21
5	2.65	2.89	0.24	9.06
6	2.16	2.45	0.29	13.43
7	1.65	1.70	0.05	3.03
8	2.14	1.84	-0.30	-14.02
9	3.20	2.20	-1.00	-31.25
10	0.61	0.60	-0.01	-1.64

Interspersion / Juxtaposition

HSI Class	1975	1997	Change	%
1	3.07	2.91	-0.16	-5.21
2	64.49	64.74	0.25	0.39
3	55.84	57.72	1.88	3.37
4	74.47	74.18	-0.29	-0.39
5	73.19	73.40	0.21	0.29
6	77.02	76.37	-3.62	-4.70
7	76.51	77.12	0.61	0.80
8	80.53	81.68	1.15	1.43
9	83.84	86.20	2.36	2.81
10	48.38	51.77	3.39	7.01

Mean Patch Size decreased for each of the classes of interest between 1975 and 1997, and this change was significant ($p < 0.0891$, two-way ANOVA) for HSI class 8, but not statistically significant for the other focal patch types (HSI class 9 or 10). The significant decrease in patch size for HSI class 8 probably results from the disproportional removal of this habitat class by timber harvesting (see Figure 5.7) compared to the other habitat suitability categories. In contrast, areas falling within HSI classes 9 and 10 were less frequently targeted by harvest operations. The decrease in patch size for these 2 classes was statistically insignificant ($p > 0.10$, two-way ANOVA). However, the ecological significance of the observed decrease in mean patch size (as indicated by a change of -14.38% for HSI class 9 and –8.65% for HSI class 10) should not be underestimated. The Patch Size Coefficient of Variation decreased from 1975 to 1997 (refer to Table 5.8). One reasonable interpretation of this result is that the largest, contiguous areas of quality habitat are being decreased in area by the introduction of disturbance patches into the forest matrix. As a result, the remaining habitat patches become not only smaller, but also more similar in size over time.

Edge Density (the amount of edge relative to the area of all classes) increased the most for HSI class 9 (4.98% change) with only a slight increase for class 10 (0.95%). In contrast, this metric actually decreased for HSI class 8, a result that at first seemed to contradict the expected results of fragmentation. However, upon close visual inspection of the changes in the spatial distribution of HSI class 8 in 1975 compared to 1997, it appears that the main effect of harvesting and fires on this particular habitat class is patch attrition or disappearance. For example, Figure 5.9 illustrates a portion of the Downie Creek valley in which many of the HSI class 8 patches in 1975 are almost completely replaced by clearcuts in 1997. As a result, the amount of edge between forest and non-forest cover types actually decreases as low quality habitat patches coalesce over time. A comparison of the Class Area and Patch Density metrics per HSI class also supports this interpretation. For example, the change in class area for HSI class 8 is –11.05% compared to only –4.61 % for HSI class 9; however, the patch density of each of these classes is almost identical (11.88% change for class 8 and 11.37% for class 9).

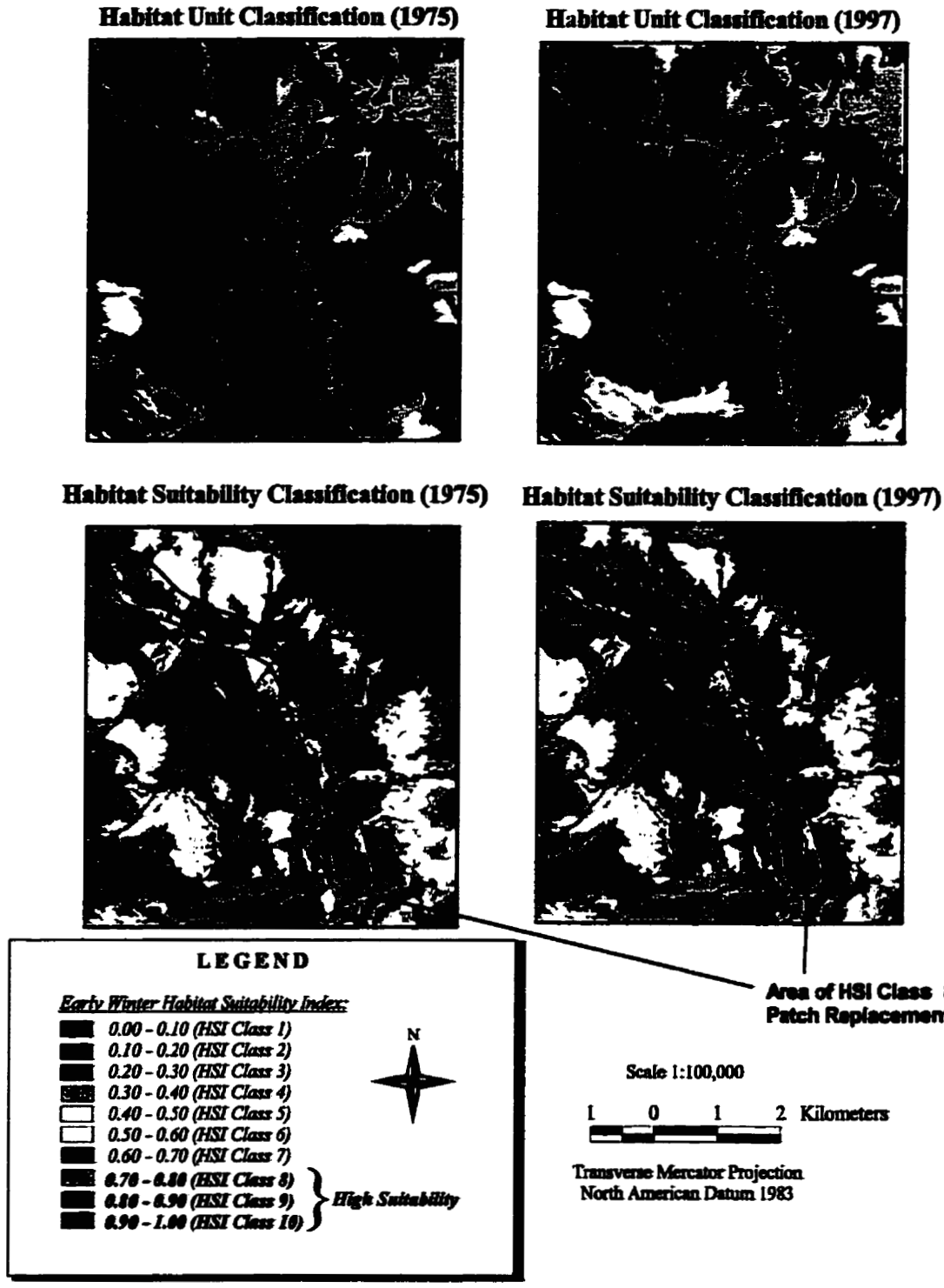


Figure 5.9 An example of HSI class 8 patch disappearance in the Downie Creek area. The area within the black outline, in the habitat suitability classifications for 1975 and 1997, represents a situation where a substantial portion (if not all) of the HSI class 8 patches have been transformed to a lower HSI Class in 1997. The habitat unit classifications for each time period indicate that the observed changes are due to the harvest of mature Cedar-Hemlock stands.

This indicates that the abundance of class 8 patches has not increased in proportion to the total amount of area removed and suggests that many class 8 patches that existed in 1975 have disappeared rather than having been split into smaller habitat fragments.

The Mean Shape index in Table 5.8 shows a decrease between time periods for HSI classes 8 and 9, indicating that suitable habitat patches are becoming less geometrically complex over time, although the amount of change in this index is not large. The mean shape index may also serve to quantify the location or position of disturbance features within natural vegetation patches. For example, the placement of a clear cut in the central portion of a habitat patch may increase the perimeter in relation to the patch area. Consequently, the geometric complexity and shape index of the patch would actually increase. In contrast, by positioning a disturbance so that it extends from an existing landscape edge, the patch shape index may decrease. Figure 5.10 provides a graphic example of this situation. Given the general decrease in shape complexity reported in Table 5.8 for the focal habitat classes, the latter situation may be the prevalent pattern of disturbance position in the study area. By focusing on the Downie Creek valley in the habitat unit classification map (Figure 5.1: page 84), it can be seen that many of the cutblocks introduced between 1975 and 1997 have indeed been placed along an existing edge between forested and non-forested cover types (such as a riparian/ICHwk edge boundary). As a result, it appears that the class level shape index has quantified the overall reduction in shape complexity per habitat patch and the positioning of disturbance features relative to existing landscape edges.

Habitat Patch Configuration before disturbance:

Area (m^2)	1268750
Perimeter (m)	5950
Shape Index	1.45

Disturbed Habitat Patch in which cut is positioned in patch centre:

Area (m^2)	1076250
Perimeter (m)	8500
Shape Index	2.05
% Change	+41.37

Disturbed Habitat Patch in which cut is positioned along existing edge:

Area (m^2)	1080625
Perimeter (m)	5700
Shape Index	1.37
% Change	- 5.52

Figure 5.10 Changes in patch shape relative to disturbance position.
By placing a cutblock (indicated in red) in the central portion of the habitat patch, the shape index has increased from 1.45 before disturbance (see top diagram) to 2.05 (41.37%) after disturbance (bottom left diagram). A cutblock located along the patch edge (bottom right) has actually decreased the shape index from 1.45 before disturbance to 1.37 after disturbance (5.52%).

The Mean Proximity Index (MPI) decreased substantially for each of the focal HSI classes, suggesting that the isolation and degree of fragmentation for suitable early winter habitat patches has increased as a result of timber harvesting and fire activity between 1975 and 1997. The proximity index was first introduced by Gustafson and Parker (1992) and modified in the FRAGSTATS software (McGarigal and Marks 1995) to distinguish between sparse distributions of small habitat patches (indicating a fragmented landscape) from a situation in which habitat patches form a complex cluster of larger patches (less fragmented). The metric equation (Equation 11: page 77) for MPI is a product of both the average patch area of a class and the mean edge-to-edge distance between patches. Based on this formula, it follows that a larger MPI value indicates a class in which patches are distributed in larger, more contiguous areas that are located in closer proximity to patches of the same type (McGarigal and Marks 1995). Therefore, the decrease in MPI reported in Table 5.8 appears to confirm that areas of suitable habitat are being fragmented into smaller patches that are more dispersed or isolated from each other over time.

Mean Core Area is the average patch size remaining after removing an edge buffer of 100 metres. This particular edge zone width has been selected by several researchers investigating fragmentation issues in boreal forests (Ripple *et al.* 1991; Mladenoff *et al.* 1994; Sachs *et al.* 1998; Kushla and Ripple 1998) to represent the area influenced by the "edge effect". Edge effects result from biotic and abiotic factors such as increased wind speed, solar insulation, and an altered soil moisture regime that combine to alter the environmental conditions along patch boundaries compared to the interior or core conditions. A conservative rule-of-thumb for estimating the width of the edge environment is a measure equivalent to two tree heights (Franklin and Forman 1987; Chen *et al.* 1992). A width of 100 metres was therefore chosen to represent the equivalent of two tree heights in the study area. The Mean Core Area index computed for the remaining interior habitat patches decreased substantially for HSI classes 8 and 9 by 14.02 and 31.25% respectively. Only a slight decrease was reported for the highest suitability class (HSI class 10) with a 1.64% change from the 1975 Mean Core Area

value. A noteworthy point for HSI class 10 is the relatively small Mean Core Area values measured for both the 1975 (0.61 hectares) and 1997 (0.60 hectares) landscapes. This indicates that the average interior core area for the highest quality habitat is less than 1 hectare in size and raises concerns about the viability of such a small core area for providing adequate protective cover and forage for local mountain caribou during the critical early winter season.

Finally, the interspersion and juxtaposition index (IJI) has increased for each of the focal habitat classes, although the amount of change is small indicating that the adjacency and interspersion of each class has not been significantly altered by the disturbance regime.

The results of the habitat patch analysis reported above indicate that both the landscape composition and configuration have been altered over the 22 year period of the study as a result of timber harvesting and wildfires. The observed directional changes in the spatial metrics calculated for each time period are consistent with results for similar studies and appear to confirm the observed spatial effects of forest fragmentation recognized in the literature and discussed in section 2.4.2. These include a loss of total habitat area, increased patch abundance and density, decreased patch size, more edges, less core area, simplified patch shape, and a wider, more dispersed patch configuration over time.

In a review paper discussing the general principles of landscape and regional ecology, Forman (1995) brought forth the idea of different phases or spatial processes of landscape change which increase habitat loss and isolation over time. These processes include:

- *Perforation:* a process in which holes are created in a habitat or land type by disturbance features.

- *Dissection:* the subdivision of an area by equal-width lines such as roads or utility corridors

- *Fragmentation:* the splitting or breaking of a land cover type into pieces that are often widely and unevenly distributed across space

- *Shrinkage*: the decrease in size of landscape patches

- *Attrition*: the disappearance of landscape patches

Forman argues that habitat fragmentation is one phase within this broader sequence of landscape processes that transform the natural land cover type of a region as a result of human or natural disturbance events. The five processes listed above overlap in both sequential order and importance, with perforation and dissection peaking in importance in the early phases of landscape change. Fragmentation and shrinkage are typical of the middle phases and attrition predominates at the end of the landscape transition cycle. Each of the spatial processes serves to increase the amount of habitat loss and isolation in a region but each is also distinctive in terms of the resulting spatial pattern on the landscape. Forman cites the following examples. In the first four phases, average patch size decreases but would typically increase upon attrition. Connectivity across an area decreases with dissection and fragmentation, and total boundary length or Edge Density increase in the first three phases but decreases with shrinkage and attrition.

Based on the results provided in Table 5.8, Mean Patch Size, Patch Size Coefficient of Variation, and the Mean Proximity indices showed the most change between 1975 and 1997 for the HSI classes of interest in the study area. This suggests a stage of landscape evolution that fits with both the *fragmentation* and *shrinkage* phases described by Forman (1995). If the proximity between suitable habitat patches can be considered similar to Forman's concept of connectivity, then a decrease in Mean Proximity would be consistent with the fragmentation phase. The decrease in the average size and variability of suitable early winter habitat patches seems consistent with the shrinkage phase of landscape change.

5.5 Ecological implications of early winter habitat fragmentation

The long-term effects of habitat fragmentation on the mountain caribou of the Revelstoke region are a topic of paramount importance to local residents and resource managers. The Revelstoke herd is the most viable southerly population, numbering approximately 400 animals. Although the population trend has remained relatively stable since 1975, (MacDonald 1996; Parks Canada 1999) the direct and indirect ecological consequences of early winter habitat fragmentation identified in this research have to be interpreted, understood, and managed in order to ensure viable populations into the future.

Some evidence suggests that timber harvesting in winter areas may provide some short-term advantages to mountain caribou in terms of lichen availability and foraging strategy. For example, Simpson (1985) found that caribou were attracted to cutblocks during early winter timber operations as fallen trees and skid trails provided access to abundant sources of lichen which would have been otherwise inaccessible on standing trees. Mountain caribou in the Selkirk mountains were observed to use hemlock habitat types that had been 60% burned or cut (Servheen and Lyon 1989) and may have been taking advantage of the increased availability of lichen litterfall caused by blowdown and deadfalls that resulted from increased wind turbulence along clear cut and burn edges (Golding and Swanson 1978).

Unfortunately, several long-term effects that may have potentially negative impacts on future species persistence accompany these apparent short-term advantages provided to caribou by habitat disturbance. First, the removal of ICHwk old growth forest cover removes sources of lichen, and understory species such as falsebox and wintergreen, that caribou rely on during early winter months. Areas removed by disturbance may not reach full habitat potential for over 250 years, based on the preferential use of stands greater than 250 years of age (McLellan *et al.* 1994). Furthermore, once recently disturbed areas begin to regenerate, the densely stocked nature of young stands are avoided by caribou, perhaps due to the reduced visibility of possible predators (Illius and Fitzgibbon 1994;

Rominger *et al.* 1996). Immature forests may therefore act as a barrier to movement, forcing animals to travel further distances around such impediments. An increase in early successional habitat provided by timber harvesting and burns would also appear to promote the proliferation of other ungulate populations such as white-tailed deer and moose. These species may adversely affect the survival of caribou by the introduction of parasites and disease (Cumming 1992) and the attraction of carnivore predators such as wolves and coyotes (MacDonald 1996; Seip 1992) into the study area. Another potential negative impact of timber harvesting is the construction of a road network. Cumming and Beange (1993) suggest that logging roads may provide access for vehicles, hunters and natural predators. Thus, the spatial effects of timber harvesting and burns quantified by the integrated mapping and analysis methodology of this project and the identified fragmentation of critical early winter habitat may have associated cumulative effects. Most of these effects are recognized by wildlife ecologists as potential threats to long-term survival of the species.

From a management perspective, a Minister's Advisory Committee has been established within the study area and is currently investigating the potential impacts of past and proposed timber harvesting practices on important mountain caribou habitat areas in the study area. A draft report released by the Revelstoke and Area Land Use Planning Minister's Advisory Committee in October of 1997 (Government of British Columbia 1997) suggested that the proposed land use changes resulting from the implementation of the Kootenay-Boundary Land Use Plan (Government of British Columbia 1996) represent only a moderate habitat risk to the Revelstoke caribou population. This analysis was based on a timber supply model in which future timber resource amounts were simulated and projected 45 years into the future based on alternative land use plans. However, these preliminary results appear to be based on the amount of habitat that would be available to local caribou populations and did not address the potential effects of land use policies on the spatial characteristics or configuration of habitat areas. The Committee draft report did recognize key resource management initiatives including the need to develop a monitoring strategy to assess the effects of the land use plan on the local caribou

population, important habitat areas, and biodiversity within the Revelstoke region. An integrated remote sensing, GIS, and landscape ecology approach, as developed and illustrated within this thesis, could provide the baseline data set, and a potential set of tools that would enable the Revelstoke advisory committee and other resource managers to establish a landscape-level monitoring program within the study area. An ecological monitoring program of this nature could be used to identify areas of landscape change and ultimately may facilitate comprehensive land use practices that minimize the adverse effects of landscape disturbance and fragmentation on local wildlife populations.

5.6 Summary

An accuracy assessment of the 1997 and 1975 habitat unit images indicated overall classification accuracy levels of 91.8% and 89.5% respectively. Individual class accuracy levels were consistently high (greater than 80% accuracy) for both disturbance and habitat categories. An analysis of omission and commission errors indicated that the residual effects of image misregistration and post-classification filtering and the misclassification of mixed pixels were the source of most of the reported inaccuracies. The reduced spatial resolution of the Landsat MSS satellite sensor (80 metre spatial resolution compared to 30 metre TM pixel) was also cited as a possible source of classification error. Based on the reported map and class accuracy levels relative to similar mapping projects using Landsat satellite imagery, an integrated GIS/remote sensing classification procedure appears to improve upon traditional statistical based algorithms such as a Maximum Likelihood Classification approach.

A discussion of error propagation within the "backcast" 1975 habitat unit image and other potential sources of error accumulated through the information processing flow was also provided. This enabled the results of the habitat fragmentation analysis to be reported and interpreted with respect to the possible inaccuracies embedded in the digital landscapes and the spatial metrics calculated from these data sources.

A comparative analysis of total class areas and the nature of the disturbances responsible for landscape change revealed that over 3737.40 hectares of forested habitat had been lost between 1975 and 1997 as a result of timber harvesting and wildfires. Timber harvesting of old growth cedar hemlock stands was responsible for the majority of this landscape change. However, the disturbance rate was mitigated by natural regeneration of stands from immature to mature successional stages. Therefore, the overall net forest habitat loss was approximately 888.83 hectares. As a result of the disproportionate removal of old growth cedar hemlock habitat areas, 8.53 % of the quality early winter habitat (HSI > 0.70) had been lost between 1975 and 1997.

Changes in the spatial characteristics and configuration of habitat patches were explained through a comparative analysis of several spatial metrics calculated for each time period. Directional changes in patch abundance (Number of Patches) and Patch Density, Mean Patch Size, Patch Size Coefficient of Variation, Edge Density, Patch Shape, Mean Proximity, Mean Core Area, and Interspersion and Juxtaposition were each analyzed and appeared to confirm the *fragmentation* and *shrinkage* (Forman 1995) of critical early winter habitat areas over time as a result of timber harvesting and wildfires.

Currently, the observed fragmentation pattern seems to have had little effect on the survival of the mountain caribou of the study area, as population levels have remained stable since 1975. Caribou research in other study areas indicate that animals may actually benefit from the short term effects of timber harvesting and burns as increased amounts of lichen become available along disturbance boundaries. However, these changes have negative implications for the long-term survival of the species should a continued trajectory of landscape fragmentation persist.

6.0 CONCLUSIONS AND RECOMMENDATIONS

6.1 Summary

This thesis has applied an integrated remote sensing, Geographic Information System, and landscape ecology approach to determine the spatial effects of timber harvesting and wildfires on a forested landscape near Revelstoke, British Columbia. It was hypothesized that fragmentation of early winter habitat critical to a local subpopulation of mountain caribou is occurring in the study area as a result of these specific disturbance factors, and that these landscapes changes could be quantified and assessed using the proposed methodology. The data collection process, methodology, and results of work toward two interconnected project components have been presented. These include:

1. Mapping and classification of habitat variables required to produce a habitat suitability model for the past (1975) and more recent (1997) time periods;

2. Quantitative assessment of fragmentation of early winter caribou habitat in the study area. This involved:
   a) Analysis of habitat loss and changes in overall habitat composition, and
   b) Comparative analysis of spatial metrics for quantifying changes in the spatial characteristics and configuration of suitable early winter habitat occurring over the 22 year period of the study.

Elevation, slope, and forest stand age information for the 1997 Habitat Suitability Index (HSI) model was extracted from the GIS database, the development of which was described in Chapter 3. Habitat units (as described by McLellan *et al.* 1995) were classified for the 1975 and 1997 time periods using Landsat MSS and TM imagery, respectively, and a Hybrid Decision Tree Classifier, which combined maximum likelihood decision rules and a spatial/contextual rule base. A brightness differencing procedure was used to isolate recent cuts and burns occurring between 1975 and 1997. A

"backcasting" technique was applied to permit classes in the earlier image (1975) to be accurately mapping based on their 1997 condition.

The accuracy of the resulting habitat unit images was 91.8% overall in the 1997 TM classification, and 89.5% overall in the 1975 MSS classification. The error assessment of important habitat variables, including habitat land cover unit and stand age as extracted from the BCMOF forest inventory database, raised the issue of the propagation of potential errors throughout the information flow of the thesis methodology and analysis. Many of the potential sources of error identified in the literature were not quantified within this research project. Therefore, despite the attempts to minimize classification error wherever possible, there is always the potential that some other source of attribute uncertainty has not been quantified or analyzed. Such is the nature of landscape modeling exercises that involve the generalization or simplification of real-world features into a digital form (Heywood *et al.* 1998). That is to say, by capturing and simulating the real-world landscape in a form that is compatible and manageable by the hardware and software currently available, the simulated model will certainly differ from the original.

Based on a class-by-class analysis of the habitat unit classifications for 1975 and 1997, the recent cuts class was observed to increase in area by approximately 150%. Much of this change (approximately 67%) occurred in the wet-cool Interior-Cedar-Hemlock (ICHwk) biogeoclimatic zone. Recent burns decreased between 1975 and 1997, perhaps as a result of fire suppression activities. The total net forest habitat (ICHwk, ESSFvc, Subalpine Parkland) loss was 888.83 ha. Observed changes in age class included a 14% increase, while approximately 5% of the forest habitat experienced a decrease in age class. Most of the decrease was recorded in older stands of ICHwk as a result of the disturbances caused by fire and harvesting. This latter disturbance accounted for three-quarters of the change in age class observed in the study area. Due to these changes in forest cover type and successional characteristics, an overall net decline in area of 8.53% was reported in the most suitable early winter habitat locations (HSI value > 0.70)

supporting the position that changes in habitat composition had occurred in the study area due to the observed disturbance regime.

Class level metrics, such as Patch Density, Mean Patch Size, Edge Density, Mean Proximity, and Mean Core Area were selected based on the literature and an understanding of the type of landscape change that had occurred in the Revelstoke area. In general, directional changes measured in the metrics were consistent with the observed spatial effects of forest fragmentation. For example, patch abundance and density, edge density, and interspersion increased while mean patch size, patch size variation, patch shape, proximity, and core area all decreased between time periods. This suggests that, as a result of timber harvesting and wildfires, suitable early winter habitat areas are becoming smaller, with more edges, less core area, a simplified patch shape, and a wider, more dispersed patch configuration over time. Mean Patch Size, Patch Size Coefficient of Variation, and the Mean Proximity indices showed the most change between 1975 and 1997, suggesting a stage of landscape evolution consistent with both the *fragmentation* and *shrinkage* phases described by Forman (1995).

6.2 Conclusions

Based on the research objectives of this thesis and the results and discussion provided in the previous section, the following conclusions are drawn:

1. A combined remote sensing, GIS and landscape ecology approach has been shown to be effective in the production of 1975 and 1997 disturbance and forest habitat unit maps, documentation of change in land cover, stand age, and caribou habitat suitability between 1975 and 1997, and analysis of habitat fragmentation and spatial structure in 1975 and 1997.

2. The most significant change observed in the forested habitat of the study area is the increased number of recent cutblocks. The majority (two-thirds) of these harvest areas are in the ICHwk biogeoclimatic zone, and three-quarters of that change can be found in the oldest ICHwk stands. This disproportionate disturbance of old growth cedar hemlock stands by harvesting was suggested to be the leading cause of a net decline in the amount of high quality early winter habitat in the study area between 1975 and 1997.

3. Suitable early winter habitat patches have decreased in Mean Patch Size, Mean Core Area, geometric complexity (Patch Shape), and Interspersion and Juxtaposition, and increased in the Number of Patches and Patch Density, Edge Density, and in Mean Proximity. In essence, the matrix of early winter habitat has been reduced in size and coherence, has many more edges and less interior, and has become more widely dispersed over time. This pattern of change is consistent with Forman's (1995) *fragmentation* and *shrinkage* phases of landscape evolution and supports the main research hypothesis, that fragmentation of critical early winter habitat is occurring in the study area as a result of timber harvesting and wildfires.

6.3 Recommendations for Further Research and Applications

This research has adopted a historical perspective by comparing the present landscape structure to the baseline conditions that existed 22 years ago. The methodology developed for this particular application could easily be implemented into an annual monitoring program for forest fragmentation and ecological integrity. The following recommendations are offered to resource managers and researchers wishing to establish an integrated spatial and temporal approach to landscape-level analysis.

- Annual change detection (with same sensor/platform imagery) should be considered in order to determine the changes more accurately in the future and with the view to create a 'time-series' that would be a more powerful input to the landscape metric analysis. In effect, this study has produced a 'baseline' 1997 landscape structure that could be considered the standard against which annual updates can be conducted for the area and in specific subareas of interest;

- Following this protocol, future cut plans could also be superimposed on the 1997 baseline landscape in order to simulate the spatial effects of proposed development on the forest matrix. The modeling of alternative 'what if' land use planning situations could provide a powerful predictive tool in a spatial landscape level analysis of potential forest fragmentation. In this manner, a continued trend of habitat fragmentation may be mitigated or prevented by *a priori* comprehensive analysis of the spatial effects of planning decisions;

- The focus of this research was on habitat mapping and changes in spatial distribution and structure over time and did not directly address the behavioural response of caribou in the study area to the observed changes in early winter habitat. Perhaps some of the spatial landscape metrics shown to be sensitive to changes in habitat patch distribution and configuration in this study could be incorporated into habitat selection analysis. This may provide a more comprehensive understanding of

preferential habitat use and changes in behaviour as a result of environmental disturbances. For example, by investigating the spatial character of habitat patches selected by caribou (as indicated by telemetry observations) and calculating mean patch size, interior core area, mean proximity and other landscape indices, researchers could determine the minimum size of a viable habitat patch, the amount of core area required, and the average distance that animals are traveling for resource utilization. These findings could then be applied to the location and design of proposed cutblocks that are more accommodating to the habitat requirements of mountain caribou in the study area.

- In the present analysis, a selection of metrics was used that appeared to provide the best or optimal interpretation of the changing landscape structure. A complete and thorough test of the selected or potential landscape metrics (and others as they are developed or become available) should be conducted to reveal those metrics most sensitive to the type of change encountered in the landscapes under study.

- Finally, the question of scale has not been addressed explicitly but needs to be considered in long-term ecological monitoring. This study addressed landscape change from a species-driven perspective and defined the landscape boundary in correspondence with the home range of a local caribou subpopulation. The landscape extent and spatial resolution was held constant in each time interval to provide a direct comparison of structural changes. However, if the particular environmental phenomenon under investigation changes, there are several scale-related issues that should be considered when designing the monitoring protocol. For example, researchers will have to decide over what area a landscape metric will be calculated and against what standard a change in metric values will be compared. Discussion will have to be initiated on acceptable tolerances in change over time to clarify what constitutes a significant change in landscape structure as captured in a selection of metrics.

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APPENDIX A: Disturbance Classification Algorithm

Objective: To classify disturbance features (recent cuts, recent burns, recent unknown, immature forests). Disturbances occurring before 1995 were extracted from the GIS forest inventory, while the most recent cuts and burns were mapped from satellite imagery.

Part I: Calculating disturbance ages from GIS / Coding Disturbance Types

1) Add necessary database items to Forest inventory coverage.

```
ADDITEM REV_FOR.PAT REV_FOR.PAT DDATE 2 2 N 0 PROJ_AGE
ADDITEM REV_FOR.PAT REV_FOR.PAT PDATE 2 2 N 0 DDATE
ADDITEM REV_FOR.PAT REV_FOR.PAT DCODE 2 2 I 0 PDATE
ADDITEM REV_FOR.PAT REV_FOR.PAT AGE60 3 3 N 0 DCODE
ADDITEM REV_FOR.PAT REV_FOR.PAT CODE60 2 2 I # AGE60
ADDITEM REV_FOR.PAT REV_FOR.PAT AGE75 3 3 N 0 CODE60
ADDITEM REV_FOR.PAT REV_FOR.PAT CODE75 2 2 I # AGE75
ADDITEM REV_FOR.PAT REV_FOR.PAT AGE97 3 3 N 0 CODE75
ADDITEM REV_FOR.PAT REV_FOR.PAT CODE97 2 2 I # AGE97
ADDITEM REV_FOR.PAT REV_FOR.PAT CUT_ND 2 2 I # CODE97
ADDITEM REV_FOR.PAT REV_FOR.PAT BURN_ND 2 2 I # CUT_ND
```

2) Code item DCODE as disturbance (Burn or logging) or not disturbance

```
TABLES
SEL REV_FOR.PAT
RESEL ACTIVITY = 'L' OR ACTIVITY = 'B'
CALC DCODE = 1
```

3) Calculate ages for cuts or burns

```
RESEL ACT_YEAR1 > 0
CALC AGE97 = 97 - ACT_YEAR1
CALC AGE75 = 75 - ACT_YEAR1
```

4) Calculate ages for burns or cuts with no activity year but established year and code as recent, or immature

```
ASEL
RESEL DCODE = 1 AND ACT_YEAR1 = 0 AND ESTAB_YEAR > 0
CALC AGE97 = 1997 - ESTAB_YEAR
CALC AGE75 = 1975 - ESTAB_YEAR
ASEL
RESEL ACTIVITY = 'L' AND AGE60 > 0 AND AGE60 <= 22
CALC CODE60 = 1
ASEL
RESEL ACTIVITY = 'L' AND AGE60 >= 23 AND AGE60 <= 80
CALC CODE60 = 3
```

```
ASEL
RESEL ACTIVITY = 'B' AND AGE60 > 0 AND AGE60 <= 22
CALC CODE60 = 2
ASEL
RESEL ACTIVITY = 'B' AND AGE60 >= 23 AND AGE60 <= 80
CALC CODE60 = 3
ASEL
```

5) Code 1975 cuts and burns as recent, or immature

```
RESEL ACTIVITY = 'L' AND AGE75 > 0 AND AGE75 <= 22
CALC CODE75 = 1
ASEL
RESEL ACTIVITY = 'L' AND AGE75 >= 23 AND AGE75 <= 80
CALC CODE75 = 3
ASEL
ASEL
RESEL ACTIVITY = 'B' AND AGE75 > 0 AND AGE75 <= 22
CALC CODE75 = 2
ASEL
RESEL ACTIVITY = 'B' AND AGE75 >= 23 AND AGE75 <= 80
CALC CODE75 = 3
```

6) Code 1997 cuts and burns as recent, or immature

```
ASEL
RESEL ACTIVITY = 'L' AND AGE97 > 0 AND AGE97 <= 22
CALC CODE97 = 1
ASEL
RESEL ACTIVITY = 'L' AND AGE97 >= 23 AND AGE97 <= 80
CALC CODE97 = 3
ASEL
ASEL
RESEL ACTIVITY = 'B' AND AGE97 > 0 AND AGE97 <= 22
CALC CODE97 = 2
ASEL
RESEL ACTIVITY = 'B' AND AGE97 >= 23 AND AGE97 <= 80
CALC CODE97 = 3
```

7) Tag cuts and burns with no activity year or no established year as "unknown"

```
ASEL
RESEL ACTIVITY = 'L' AND ACT_YEAR1 = 0 AND ESTAB_YEAR = 0
CALC CUT_ND = 1
ASEL
RESEL ACTIVITY = 'B' AND ACT_YEAR1 = 0 AND ESTAB_YEAR = 0
CALC BURN_ND = 1
```

8) Calculate ages for non-disturbance polygons

```
ASEL
RESEL DCODE = 0
RESEL PROJ_AGE > 0 AND STAND_AGE > 0
CALC DDATE = PROJ_AGE - STAND_AGE
CALC PDATE = REF_YEAR + DDATE
RESEL PDATE = 98
CALC AGE97 = PROJ_AGE - 1
CALC AGE75 = PROJ_AGE - 23
ASEL
RESEL DCODE = 0 AND PDATE = 97
CALC AGE97 = PROJ_AGE
CALC AGE75 = PROJ_AGE - 22
ASEL
RESEL DCODE = 0 AND PDATE = 96
CALC AGE97 = PROJ_AGE + 1
CALC AGE75 = PROJ_AGE - 21
ASEL
```

9) Non-Disturbance polygons: Assign disturbance codes for 1975 CUTS AND BURNS

```
RESEL DCODE = 0 AND AGE75 > 0 AND AGE75 <= 22 /* 115 POLYGONS
CALC CODE75 = 4 /* UNKNOWN RECENT DISTURBANCES
ASEL
RESEL DCODE = 0 AND AGE75 >= 23 AND AGE75 <= 80 /* 1017 POLYGONS
CALC CODE75 = 3
ASEL
```

10) Non-Disturbance polygons: Assign disturbance codes for 1997 cuts and burns

```
RESEL DCODE = 0 AND AGE97 > 0 AND AGE97 <= 22 /* 67 POLYGONS
CALC CODE97 = 4 /* UNKNOWN RECENT DISTURBANCES
ASEL
RESEL DCODE = 0 AND AGE97 >= 23 AND AGE97 <= 80
CALC CODE97 = 3
ASEL
QUIT
```

11) Separate disturbances and burns and cuts into coverages

```
&STAT 9999
ARCEDIT
EC REV_FOR
EF POLYGON
DE POLY
DRAW
```

```
SEL CODE75 > 0  /* 1260 POLYGONS
PUT DIS75
SEL CODE97 > 0  /* 1212 POLYGONS
PUT DIS97
CODE75 = 1 /* 45 POLYGONS
PUT CUT75
SEL CODE97 = 1 /* 182 POLYGONS
PUT CUT97
SEL CODE75 = 2 /* 79 POLYGONS
PUT BURN75
SEL CODE97 = 2 /* 8 POLYGONS
PUT BURN97
SEL CUT_ND = 1 /* 2 POLYGONS
PUT CUT_ND
SEL BURN_ND = 1 /* 61 POLYGONS
PUT BURN_ND
QUIT
BUILD DIS75 POLY
BUILD DIS97 POLY
BUILD CUT75 POLY
BUILD CUT97 POLY
BUILD BURN75 POLY
BUILD BURN97 POLY
BUILD CUT_ND POLY
BUILD BURN_ND POLY
```

12) Compare unknown date cuts and burns visually on MSS and TM imagery to determine time period of disturbance (Pre-1975 or Post 1975)
Use UNKNOWN.AML to code unknown cuts and burns with CODE75 and CODE97

```
&RUN UNKNOWN.AML
```

13) Put CUT_ND and BURN_ND into disturbance coverages

```
ARCEDIT
EC CUT_ND
EF POLYGON
SEL CODE75 = 3
PUT DIS75
SEL CODE97 = 3
PUT DIS97
EC BURN_ND
EF POLYGON
SEL CODE75 > 0
PUT BURN75
```

```
PUT DIS75
PUT DIS97

SEL CODE97 = 2
PUT BURN97
PUT DIS97
QUIT
BUILD DIS75 POLY
BUILD CUT97 POLY
BUILD DIS97 POLY
BUILD BURN75 POLY
BUILD BURN97 POLY
```

14) Dissolve cuts, burns and disturbance polygons based on code for mapping and training

```
DISSOLVE DIS75 DIS75_DS CODE75 POLY
DISSOLVE DIS97 DIS97_DS CODE97 POLY
DISSOLVE CUT75 CUT75_DS ACT_YEAR1 POLY
DISSOLVE CUT97 CUT97_DS ACT_YEAR1 POLY
DISSOLVE BURN75 BURN75_DS CODE75 POLY
DISSOLVE BURN97 BURN97_DS CODE97 POLY
```

Part II: Extracting training areas for cuts, burns, avalanches, riparian areas etc. to construct spatial/contextual rule base for separating cuts and burns from similar cover types

Use known cuts, burns, avalanches, riparian areas from GIS as training areas to extract brightness difference, minimum elevation, maximum elevation, elevation ranges, maximum slope, distance from roads, shape index, ratio index. Use to determine rules for classifying cuts and burns occurring between 1995 and 1997 from the satellite imagery.

15) Extract training areas

```
/* CALCULATE CODE = 0 POLYGONS WITH NO COVER ID FOR GRID
CONVERSION
TABLES
SEL CUT75_DS.PAT
RESEL ACT_YEAR1 = 0
CALC CUT75_DS-ID = 0
ASEL
RESEL ACT_YEAR1 > 0
CALC CUT75_DS-ID = CUT75_DS#
SEL CUT97_DS.PAT
```

```
RESEL ACT_YEAR1 = 0
CALC CUT97_DS-ID = 0
ASEL
RESEL ACT_YEAR1 > 0
CALC CUT97_DS-ID = CUT97_DS#
SEL BURN75_DS.PAT
RESEL CODE75 = 0
CALC BURN75_DS-ID = 0
ASEL
RESEL CODE75 > 0
CALC BURN75_DS-ID = BURN75_DS#
SEL BURN97_DS.PAT
RESEL CODE97 = 0
CALC BURN97_DS-ID = 0
ASEL
RESEL CODE97 = 2
CALC BURN97_DS-ID = BURN97_DS#
QUIT

/* CONVERT COVERAGES TO GRIDS
POLGRID CUT75_DS CUT75_GR CUT75_DS-ID
25
Y
POLYGRID CUT97_DS CUT97_GR CUT97_DS-ID
25
Y
POLGRID BURN75_DS BURN75_GR BURN75_DS-ID
25
Y
POLYGRID BURN97_DS BURN97_GR BURN97_DS-ID
25
Y
POLYGRID AVALANCHE AVAL_GR AVALANCHE-ID
25
Y
POLYGRID RIPARIAN RIPARIAN_GR RIPARIAN-ID
25
Y
POLYGRID DIS97_DS DIS97_GR CODE97
25
Y
```

```
/* In GRID, calculate shape and ratio values for cuts, burns, avalanche, and riparian
&STAT 9999
GRID
MAPEXT DEM_GR
SETWINDOW DEM_GR
SETCELL 25
CUT75A = ZONALAREA (CUT75_GR)
CUT97A = ZONALAREA (CUT97_GR)
BURN75A = ZONALAREA (BURN75_GR)
BURN97A = ZONALAREA (BURN97_GR)
AVALA = ZONALAREA (AVAL_GR)
RIPA = ZONALAREA (RIPARIAN_GR)

CUT75P = ZONALPERIMETER (CUT75_GR)
CUT97P = ZONALPERIMETER (CUT97_GR)
BURN75P = ZONALPERIMETER (BURN75_GR)
BURN97P = ZONALPERIMETER (BURN97_GR)
AVALP = ZONALPERIMETER (AVAL_GR)
RIPP = ZONALPERIMETER (RIPARIAN_GR)

/* CONVERT AREA AND PERIMETER GRIDS TO FLOATING POINT TO
CALCULATE REAL SHAPE VALUES


/* Calculate shape and ratio index for each feature

CUT75_SH = (0.25 * CUT75PF) / (SQRT(CUT75AF))
CUT97_SH = (0.25 * CUT97PF) / (SQRT(CUT97AF))
BURN75_SH = (0.25 * BURN75PF) / (SQRT(BURN75AF))
BURN97_SH = (0.25 * BURN97PF) / (SQRT(BURN97AF))
AVAL_SH = (0.25 * AVALPF) / (SQRT(AVALAF))
RIP_SH = (0.25 * RIPPF) / (SQRT(RIPAF))
CUT75_R = CUT75AF / CUT75PF
CUT97_R = CUT97AF / CUT97PF
BURN75_R = BURN75AF / BURN75PF
BURN97_R = BURN97AF / BURN97PF
AVAL_R = AVALAF / AVALPF
RIP_R = RIPAF / RIPPF

/* Calculate Brightness Difference (TM - MSS)
BDIFF = TBRIGHT - MBRIGHT

/* Extract Zonal Stats from each disturbance grid
Example:
```

```
CUT75A.DAT = ZONALSTATS (CUT75_GR, TBRIGHT, ALL)
CUT75B.DAT = ZONALSTATS (CUT75_GR, TGREEN, ALL)
CUT75C.DAT = ZONALSTATS (CUT75_GR, TWET, ALL)
CUT75D.DAT = ZONALSTATS (CUT75_GR, TNDVI, ALL)
CUT75E.DAT = ZONALSTATS (CUT75_GR, MBRIGHT, ALL)
CUT75F.DAT = ZONALSTATS (CUT75_GR, MGREEN, ALL)
CUT75G.DAT = ZONALSTATS (CUT75_GR, MYELLOW, ALL)
CUT75H.DAT = ZONALSTATS (CUT75_GR, MNDVI, ALL)
CUT75I.DAT = ZONALSTATS (CUT75_GR, DEM_GR, ALL)
CUT75J.DAT = ZONALSTATS (CUT75_GR, SLOPE_GR, ALL)
CUT75J.DAT = ZONALSTATS (CUT75_GR, RDS_DS, ALL)
CUT75K.DAT = ZONALSTATS (CUT75_GR, CUT75_SH, ALL)
CUT75L.DAT = ZONALSTATS (CUT75_GR, CUT75_R, ALL)
CUT75M.DAT = ZONALSTATS (CUT75_GR, BDIFF, ALL)

/* Convert each DAT file to DBASE to import into Excel
INFODBASE command
```

Part III: Based on spatial/contextual rules, separate cuts and burns occurring between 1975 and 1997 from avalanche, riparian areas, and other areas having high brightness difference due to illumination and misregistration.

16) Create mask of 1997 disturbances and the parks to remove from change detection

```
&STAT 9999
GRID
MAPEXT DIS97_GR

SETWINDOW DIS97_GR
SETCELL 25
TEMP1 = DIS97_GR OVER GNPBND
TEMP2 = MRNPBND OVER TEMP1
MASKTEMP = ^ TEMP2

/* Convert zero values to NODATA

MASK = SETNULL (MASKTEMP == 0, MASKTEMP)
```

17) Select Brightness difference values outside of known disturbance areas

```
BDIFF = TBRIGHT - MBRIGHT
BDIFF2 = SELECTMASK (BDIFF, MASK)
```

18) Determine brightness threshold by sampling known clearcuts and burns occurring between 1995 and 1997 (outside parks)

```
/* Select Brightness threshold 70 - 375

IF (BDIFF2 > 70 & BDIFF2 < 376) BTHRESH = 1
ENDIF
```

19) Select elevation between minimum cut (517 m) and max. burn (2099 m)

```
IF (DEM_GR >= 517 & DEM_GR <= 2099) DEM2 = 1
ENDIF

/* Multiply elevation mask with brightness threshold

DEM3 = DEM2 * BTHRESH
```

20) Select areas < 55 degrees slope (cut maximum slope)

```
IF (SLOPE_GR <= 55) SLOPE = 1
ENDIF

SLOPE1 = SLOPE * SHADOW2
```

21) Separate potential cuts/burns from avalanche paths based on observed TM Wetness of training areas. Avalanche paths will not be considered as disturbances.

```
IF (TWET <= -20) CUTBURN = 1
ENDIF

IF (TWET > -20 & TWET < 25) AVALDIS = 1
ENDIF

CUTBURN2 = SLOPE1 * CUTBURN
AVALDIS2 = SLOPE1 * AVALDIS

/* Convert zero values to NODATA before REGIONGROUP

CUTBURN3 = SETNULL (CUTBURN2 == 0, CUTBURN2)
AVALDIS3 = SETNULL (AVALDIS2 == 0, AVALDIS2)

REGIONCB = REGIONGROUP (CUTBURN3, #, EIGHT)
REGIONAV = REGIONGROUP (AVALDIS3, #, EIGHT)

CUTBAREA = ZONALAREA (REGIONCB)
AVALAREA = ZONALAREA (REGIONAV)
```

22) Select areas having feature area greater than known minimum area for cuts and burns - select all areas >= 1.06 Ha. But keep small areas if adjacent to previous disturbance

```
/* separate small disturbances from large

IF ((CUTBAREA / 10000) >= 1.06) LARGE = 1
ENDIF

IF ((CUTBAREA / 10000) < 1.06) SMALL = 1
ENDIF

/* Small areas - merge with adjacent previous disturbances (DIS97_GR)

PREVDIS = SETNULL (DIS97_GR == 0, DIS97_GR)
CLOSEZONE = EUCALLOCATION (PREVDIS, DISTCLOSE)

REGIONSM = REGIONGROUP (SMALL, #, EIGHT)
SM_MIN = ZONALMIN (REGIONSM, DISTCLOSE)
SMALLCLOSE = COMBINE (SM_MIN, CLOSEZONE)

IF (SMALLCLOSE.SM_MIN <= 25) SMALLCB = SMALLCLOSE.CLOSEZONE
ENDIF

/* Add small disturbances to Previous disturbances

SMALLCB2 = CON (ISNULL (SMALLCB), 0, SMALLCB)
PREVDIS2 = SMALLCB2 OVER DIS97_GR
```

23) For potential Cuts and burns larger than 1.06 HA calculate elevation ranges, shape, perimeter/area ratio, distance to closest existing disturbance, distance from roads

```
REGIONLRG = REGIONGROUP (LARGE, #, EIGHT)
AREA1 = ZONALAREA (REGIONLRG)
PERIM1 = ZONALPERIMETER (REGIONLRG)
SHAPE = (0.25 * PERIM1) / (SQRT(AREA1))
RATIO = AREA1 / PERIM1
EMIN = ZONALMIN (REGIONLRG, DEM_GR)
EMAX = ZONALMAX (REGIONLRG, DEM_GR)
ERNG = ZONALRANGE (REGIONLRG, DEM_GR)
MINDIS = ZONALMIN (REGIONLRG, DISTCLOSE)
RDMIN = ZONALMIN (REGIONLRG, RDSNEW_DS)
RDMAX = ZONALMAX (REGIONLRG, RDSNEW_DS)
BDIF2 = ZONALMEAN (REGIONLRG, BDIFF)
```

SHAPES = COMBINE (SHAPE, RATIO, MINDIS, CLOSEZONE, EMIN, EMAX, ERNG, RDMIN, RDMAX, BDIF2)

24) Remove areas having perimeter/area ratio less than 25. But keep these areas if adjacent to previous disturbance

A ratio of 25 was the lowest recorded for training areas. Features with ratio less than 25 tend to be long thin sliver areas of high brightness difference resulting from misregistration between TM and MSS images.

Remove these areas if beyond 25 metres of a pre-existing disturbance, keep these areas if adjacent to existing disturbance.

```
IF (SHAPES.RATIO <= 25 & SHAPES.MINDIS <= 25 & SHAPES.RDMIN >= 0 &
SHAPES.RDMAX > 80 & SHAPES.CLOSEZONE == 1) SLIVER =
SHAPES.CLOSEZONE
ELSE IF (SHAPES.RATIO <= 25 & SHAPES.MINDIS <= 25 & SHAPES.RDMIN >= 0
& SHAPES.RDMAX > 80 & SHAPES.CLOSEZONE == 2) SLIVER =
SHAPES.CLOSEZONE
ELSE SLIVER = 0
ENDIF

/* Merge with previous disturbances PREVDIS2

SLIVER2 = CON (ISNULL (SLIVER), 0, SLIVER)
PREVDIS3 = SLIVER2 OVER PREVDIS2
```

25) Remove areas not conforming to maximum elevation, elevation range, distance from roads rules determined for cuts and burns. Features passing rules are kept in the output image.

```
ELSE IF (SHAPES.EMAX > 1990) SHAPECB = 0
/* Removes features in which elevation higher than maximum elevation for cuts and
burns

ELSE IF (SHAPES.ERNG < 9 OR SHAPES.ERNG > 580) SHAPECB = 0
/* Removes areas in which elevation range is lower than minimum for cuts and burns or
the elevation range /* is higher than observed

ELSE IF (SHAPES.EMAX > 1840 & SHAPES.ERNG < 120) SHAPECB = 0
/* Removes areas that are not cuts because maximum elevation is too high but elevation
range is too low
/* for a burn.

ELSE IF (SHAPES.MINDIS > 25 & SHAPES.RATIO <= 25) SHAPECB = 0
/* Remove areas that have a lower perimeter/area ratio than cuts or burns and are not
adjacent to previous /* disturbances
```

```
ELSE IF (SHAPES.ERNG < 120 & SHAPES.RDMIN > 500) SHAPECB = 0
/* Remove features that are not burns because elevation range is too low
/* Features are not cuts because distance from roads is too high (greater than 500 metres)

ELSE IF (SHAPES.SHAPE > 5) SHAPECB = 0
/* Remove features that have a higher shape index than cuts and burns

ELSE SHAPECB = SHAPES
/* All features passing the above rules remain in the image file

ENDIF
QUIT

/* Join original rule values from SHAPES to output grid SHAPECB

JOINITEM SHAPECB.VAT SHAPES.VAT SHAPECB.VAT VALUE COUNT
```

26) Code detected features as either a recent cut (1) or recent burn (2) based on distance from roads

```
IF (SHAPECB.VALUE > 3 & SHAPECB.RDMAX <= 875) DETCB97 = 1
ELSE IF (SHAPECB.VALUE > 3 & SHAPECB.RDMAX > 875) DETCB97 = 2
ELSE IF (SHAPECB.VALUE == 3) DETCB97 = 1
ELSE DETCB97 = 0
ENDIF
```

27) Merge detected cuts and burns with previous disturbances map

```
DETCB97B = CON (ISNULL (DETCB97), 0, DETCB97)
PREVDIS4 = DETCB97B OVER PREVDIS3
```

APPENDIX B: Non-Forested Habitat Classification Algorithm

Objective: To classify non-forested pixels as identified by an unsupervised classification procedure into the appropriate habitat units.

1) In PCI execute K-Means unsupervised classification (iterative method after Cohen et. al. 1995). Three output images resulting must be assigned appropriate non-forested habitat codes using decision tree algorithm.

Input images:
KCLASS1
KCLASS2
KCLASS3

2) In Arc GRID, create mask to remove disturbance areas from classification

```
TEMP1 = ^ PREVDIS4
MASK1 = SETNULL (TEMP1 == 0, TEMP1)

KCLASS1B = SELECTMASK (KCLASS1, MASK1)
KCLASS2B = SELECTMASK (KCLASS2, MASK1)
KCLASS3B = SELECTMASK (KCLASS3, MASK1)
```

KCLASS1 RULES

```
KCLASS1C = RECLASS (KCLASS1B, CLASS1.TXT, DATA)
```

3) *Separate lake (12) from shadow (13)*
   Based on average shaded relief and average slope value of features. Shadow has low relief and may have high slope. Threshold values based on sampling of shadow and lake areas in the imagery.

```
IF (KCLASS1C == 13) KRULE1 = 13
ENDIF

REGIONK1 = REGIONGROUP (KRULE1, #, EIGHT)
RMN = ZONALMEAN (REGIONK1, REL_TM)
SMN = ZONALMEAN (REGIONK1, SLOPE_GR)
COMB1 = COMBINE (RMN, SMN)
IF (COMB1.VALUE > 0 & COMB1.RMN > 100 & COMB1.SMN < 20) KRULE2 = 12
ELSE KRULE2 = 13
ENDIF
```

4) *Separate alpine meadow vs. low elevation shrub vs. avalanche*

```
IF (KCLASS1C == 8) KRULE3 = 8
ENDIF

REGIONK2 = REGIONGROUP (KRULE3, #, EIGHT)
EMIN1 = ZONALMIN (REGIONK2, DEM_GR)
EMAX1 = ZONALMAX (REGIONK2, DEM_GR)
ERNG1 = ZONALRANGE (REGIONK2, DEM_GR)
SMN1 = ZONALMEAN (REGIONK2, SLOPE_GR)
COMB2 = COMBINE (EMIN1, EMAX1, ERNG1, SMN1)

/* Separate Alpine Meadow from Avalanche
/* Features meeting the following requirements based on minimum and maximum
/* elevation, elevation range, and mean slope were coded as alpine meadow.
/* Features not meeting requirements are considered to be avalanche paths

IF (COMB2.VALUE > 0 & COMB2.EMIN1 > 1900) KRULE4 = 5
ELSE IF (COMB2.VALUE > 0 & COMB2.EMAX1 > 1900 & COMB2.ERNG1 < 500)
KRULE4 = 5
ELSE IF (COMB2.VALUE > 0 & COMB2.EMAX1 > 1900 & COMB2.EMIN1 <= 1800
& COMB2.SMN1 < 20) KRULE4 = 5
ELSE IF (COMB2.VALUE > 0 & COMB2.EMAX1 > 1900 & COMB2.EMIN1 <= 1800
& COMB2.ERNG1 < 152) KRULE4 = 5
ELSE IF (COMB2.VALUE > 0 & COMB2.EMAX1 > 1900 & COMB2.EMIN1 <= 1800
& COMB2.ERNG1 >= 152 & COMB2.SMN1 < 20) KRULE4 = 5

/* Separate Low elevation shrub from Avalanche and alpine meadow
ELSE IF (COMB2.VALUE > 0 & COMB2.EMAX1 <= 1800 & COMB2.ERNG1 < 152)
KRULE4 = 6
ELSE KRULE4 = 8
ENDIF

/* Merge classes

KRULE2B = CON (ISNULL(KRULE2), 0, KRULE2)
KRULE4B = CON (ISNULL(KRULE4), 0, KRULE4)
KCLASS1D = RECLASS (KCLASS1C, CLASS2.TXT, DATA)
TEMP2 = KRULE2B OVER KRULE4B
KCLASS1_OUT = KCLASS1D OVER TEMP2

OUTPUT CLASSES
5 = alpine meadow
6 = low elev shrub
8 = avalanche
```

10 = rock
11 = snow/ice
12 = lakes
13 = shadow

KCLASS2 RULES

KCLASS2C = RECLASS (KCLASS2B, CLASS3.TXT, DATA)

OUTPUT CLASSES
10 = rock
11 = snow/ice
12 = lakes
13 = shadow
14 = forest

KCLASS3 RULES

KCLASS3C = RECLASS (KCLASS3B, CLASS5.TXT, DATA)

5) CLASS 1 - Separate sparse vegetation on rock substrate (7) from alpine forest (4)

```
IF (KCLASS3C == 4) KRULE7 = 4
ENDIF

/* Elevation zones (above parkland transition 1800m or below)
IF (DEM_GR <= 1800 & DEM_GR > 0) EZONES1 = 1
ELSE IF (DEM_GR > 1800 & DEM_GR NE 0) EZONES1 = 2
ENDIF

COMB3 = COMBINE (KRULE7, EZONES1)

IF (COMB3.KRULE7 == 4 & COMB3.EZONES1 == 1) KRULE8 = 7
ELSE IF (COMB3.KRULE7 == 4 & COMB3.EZONES1 == 2) KRULE8 = 4
ENDIF
```

6) CLASS 5 - Separate forest on bright slopes (14) from riparian & avalanche on shaded slopes (8)

```
IF (KCLASS3C == 8) KRULE9 = 8
ENDIF
```

```
RIPDIS = EUCDISTANCE (RIVSRIP_GR)
REGIONK4 = REGIONGROUP (KRULE9, #, EIGHT)
SHMN = ZONALMEAN (REGIONK4, REL_TM)
SLPMN = ZONALMEAN (REGIONK4, SLOPE_GR)
RIPMIN = ZONALMIN (REGIONK4, RIPDIS)
RIPMAX = ZONALMAX (REGIONK4, RIPDIS)
COMB4 = COMBINE (SHMN, SLPMN, RIPMIN, RIPMAX)

/* Riparian rules
/* Riparian areas in study area are adjacent to main streams and rivers in the Downie
/* Creek and Tangier River valleys.  Areas classified as riparian if lower slope than
/* avalanche paths and in close proximity to streams.

IF (COMB4.SLPMN < 20 & COMB4.RIPMAX < 651) KRULE10A = 9

/* Forest rules
/* Forested areas separated from avalanche and riparian based on shaded relief, slope and
/* distance from streams.

ELSE IF (COMB4.SHMN >= 200 & COMB4.RIPMAX >= 651 & COMB4.SLPMN >=
0) KRULE10 = 14
ELSE IF (COMB4.SHMN > 180 & COMB4.SHMN <= 200 & COMB4.RIPMAX >=
651 & COMB4.SLPMN < 20) KRULE10A = 14

/* Avalanche
/* Features with lower shaded relief than forested areas in this K-Means class
/* Not in close proximity to streams
/* Mean slope value of feature is greater than riparian areas

ELSE IF (COMB4.SHMN <= 200 & COMB4.SHMN > 180 & COMB4.RIPMAX >=
651 & COMB4.SLPMN >= 20) KRULE10A = 8

ELSE IF (COMB4.SHMN <= 180 & COMB4.RIPMAX < 651 & COMB4.SLPMN >=
20) KRULE10A = 8
ELSE IF (COMB4.SHMN <= 180 & COMB4.RIPMAX >= 651 & COMB4.SLPMNA <
20) KRULE10A = 8
ELSE KRULE10A = 8
ENDIF

KCLASS3D = RECLASS (KCLASS3C, CLASS6.TXT, DATA)
KCLASS3_OUT = KCLASS3D OVER KRULE10A

OUTPUT CLASSES
4 = alpine forest
```

8 = avalanche
9 = riparian
10 = rock
14 = forest

7) Merge all three KMEANS maps to create nonveg map

```
TEMP6 = KCLASS1_OUT OVER KCLASS2_OUT
KMEANS_OUT = TEMP6 OVER KCLASS3_OUT
```

8) Merge avalanche, alpine meadow, low elev shrub, & riparian classes and refine classification. Groups of contiguous pixels in these classes will be re-defined as avalanche if range and slope are appropriate.

```
AVRCL = RECLASS (KMEANS_OUT, CLASS7.TXT, DATA)
AVRCL2 = SETNULL (AVRCL == 0, 1)
REGIONK5 = REGIONGROUP (AVRCL2, #, EIGHT)
EMIN2 = ZONALMIN (REGIONK5, DEM_GR)
EMAX2 = ZONALMAX (REGIONK5, DEM_GR)
ERNG2 = ZONALRANGE (REGIONK5, DEM_GR)
SLPMN2 = ZONALMEAN (REGIONK5, SLOPE_GR)

IF (RIPDIS < 651) RIP = 1
ELSE RIP = 0
ENDIF

IF (SLOPE_GR < 16) SLOPE = 1
ELSE SLOPE = 0
ENDIF

RIPSLOPE = RIP * SLOPE

COMB7 = COMBINE (EMIN2, EMAX2, ERNG2, SLPMN2, RIPSLOPE)

/* Separate Riparian
IF (COMB7.RIPSLOPE == 1) AVRCL3 = 9

/* Separate alpine meadow from avalanche
/* Features are classified as alpine meadow if meeting following requirements of
/* minimum and maximum elevation, average slope, and elevation range.

ELSE IF (COMB7.EMIN2 > 1800) AVRCL3 = 5
ELSE IF (COMB7.EMAX2 > 1800 & COMB7.EMIN2 <= 1800 & COMB7.SLPMN2
<= 20) AVRCL3 = 5
```

```
ELSE IF (COMB7.EMAX2 > 1800 & COMB7.EMIN2 <= 1800 & COMB7.ERNG2 <
450) AVRCL3 = 5

/* Separate shrub from avalanche
/* Features are classified as shrub and not avalanche if elevation range is lower than
/* expected for an avalanche path.

ELSE IF (COMB7.EMAX2 <= 1800 & COMB7.ERNG2 < 450 & COMB7.RIPSLOPE
== 0) AVRCL3 = 6

/* All else is Avalanche
/* All features not meeting above requirements are coded as avalanche paths.

ELSE AVRCL3 = 8
ENDIF
```

10) Merge intermediate maps to create final map of non-forested habitat units.

```
AVRCL4 = CON (ISNULL(AVRCL3), 0, AVRCL3)
AVRCL5 = RECLASS (KMEANS_OUT, CLASS8.TXT, DATA)
NONFOR_OUT = AVRCL4 OVER AVRCL5
```

APPENDIX C: Post-classification Class Aggregation Algorithm

Objective: To aggregate disturbance, non-forested and tree species classes from the Maximum Likelihood Classification into the appropriate habitat units to create a final habitat map.

Objective: To aggregate disturbance, non-forested, tree species composition classes into final habitat units.

Part I: Prepare aspect and elevation grids for selection of biogeoclimatic subzones

1) Create grid designating south and north aspects as defined by field guide
South aspect = 90 – 300 degrees
North aspect = 300 – 90 degrees
ASPCODE 1 = south ASPCODE 2 = north

```
IF (ASPECT > 90 AND ASPECT < 300) ASPCODE = 1
ELSE IF (ASPECT <= 90 AND ASPECT >= 0) ASPCODE = 2
ELSE IF (ASPECT >= 300 AND ASPECT <= 360) ASPCODE = 2
ELSE ASPCODE = 0
ENDIF
```

2) Combine elevation and aspect code grids

```
DEM_ASP = COMBINE (DEM, ASPCODE)
```

3) Select aspect codes and suitable elevation ranges for subzones
ICHwk = 1
ESSFvc = 2
ESSFvcp (Parkland) = 3
Alpine Tundra = 4

```
IF (DEM_ASP.ASPCODE == 1 AND DEM_ASP.DEM > 550 AND DEM_ASP.DEM
<= 1350) ZONES = 1
ELSE IF (DEM_ASP.ASPCODE == 2 AND DEM_ASP.DEM > 550 AND
DEM_ASP.DEM <= 1350) ZONES = 1
ELSE IF (DEM_ASP.ASPCODE == 1 AND DEM_ASP.DEM > 1350 AND
DEM_ASP.DEM <= 1800) ZONES = 2
ELSE IF (DEM_ASP.ASPCODE == 2 AND DEM_ASP.DEM > 1250 AND
DEM_ASP.DEM <= 1800) ZONES = 2
ELSE IF (DEM_ASP.DEM > 1800 AND DEM_ASP.DEM <= 2300) ZONES = 3
ELSE IF (DEM_ASP.DEM > 2300) ZONES = 4
ELSE ZONES = 0
ENDIF
```

4) *Select suitable elevation ranges for ICHwk, ESSFvc, ESSFvcp, and AT when aspect could not be calculated*

```
IF (DEM_ASP.DEM > 550 AND DEM_ASP.DEM <= 1350) ELEV = 1
ELSE IF (DEM_ASP.DEM > 1350 AND DEM_ASP.DEM <= 1800) ELEV = 2
ELSE IF (DEM_ASP.DEM > 1800 AND DEM_ASP.DEM <= 2300) ELEV = 3
ELSE IF (DEM_ASP.DEM > 2300) ELEV = 4
ELSE ELEV = 0
ENDIF
```

5) *Reclassify second most probable class from MLC into 1 = ICH or 2 = ESSF*

```
CLASS2 = RECLASS (IMAGECLASS2, CLASS.TXT)
```

6) *Combine PCI image class with highest probability (IMAGECLASS) with PCI image class with second highest probability (IMAGECLASS2) and with Subzones and non-forested areas classification (nonfor2) and elev zones*

```
IMAGE_COM = COMBINE (IMAGECLASS, CLASS2, ZONES, NONFOR2, ELEV)
```

Part II: Assign Final Habitat Units

7) *Assign disturbance and non-forested classes final habitat code*

```
IF (IMAGE_COM.NONFOR == 1) MAP = 1 /* RECENT CUTS
ELSE IF (IMAGE_COM.NONFOR == 2) MAP = 2 /* RECENT BURNS
ELSE IF (IMAGE_COM.NONFOR == 3) MAP = 3 /* RECENT (UNKNOWN)
ELSE IF (IMAGE_COM.NONFOR == 4) MAP = 4 /* IMMATURE FORESTS
ELSE IF (IMAGE_COM.NONFOR == 8) MAP = 7 /* OPEN COVER (ROCK)
ELSE IF (IMAGE_COM.NONFOR == 9 & IMAGE_COM.ZONES < 3) MAP = 5 /*
AVALANCHE
ELSE IF (IMAGE_COM.NONFOR == 10) MAP = 8 /* RIPARIAN
ELSE IF (IMAGE_COM.NONFOR == 11) MAP = 13 /* ROCK
ELSE IF (IMAGE_COM.NONFOR == 12) MAP = 13 /* SNOW, ICE
ELSE IF (IMAGE_COM.NONFOR == 13) MAP = 13 /* LAKES
ELSE IF (IMAGE_COM.NONFOR == 7 AND IMAGE_COM.ZONES < 3) MAP = 6 /*
OPEN COVER (SHRUB)
ELSE IF (IMAGE_COM.NONFOR == 6 AND IMAGE_COM.ZONES < 3) MAP = 6 /*
ALPINE MEADOW TO OPEN COVER (SHRUB)
```

8) *Assign appropriate non-forested classes as Alpine Tundra zone*

```
(KEEP SHADOW AS SHADOW IN ALPINE)
ELSE IF (IMAGE_COM.NONFOR == 6 AND IMAGE_COM.ZONES == 4) MAP = 12
```

```
/* ALPINE MEADOW
ELSE IF (IMAGE_COM.NONFOR == 14 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.ELEV == 4) MAP = 14 /* KEEP SHADOW AS SHADOW IN ALPINE
ELSE IF (IMAGE_COM.IMAGECLASS == 0 AND IMAGE_COM.NONFOR == 0
AND IMAGE_COM.ZONES == 4) MAP = 14 /* SHADOW
ELSE IF (IMAGE_COM.NONFOR == 14 AND IMAGE_COM.ZONES == 4) MAP =
14 /* SHADOW
ELSE IF (IMAGE_COM.NONFOR == 5 AND IMAGE_COM.ZONES == 4) MAP = 12
/* ALPINE FOREST
ELSE IF (IMAGE_COM.NONFOR == 8 AND IMAGE_COM.ZONES == 4) MAP = 12
/* SPARSE VEG
ELSE IF (IMAGE_COM.NONFOR == 9 AND IMAGE_COM.ZONES == 4) MAP = 12
/* AVALANCHE IN ALPINE
ELSE IF (IMAGE_COM.IMAGECLASS > 0 AND IMAGE_COM.ZONES == 4 &
IMAGE_COM.NONFOR == 0) MAP = 12 /* FOREST CLASSES IN ALPINE
```

9) *Assign appropriate non-forested and tree species composition classes to ESSFvcp (Parkland) subzone*

```
ELSE IF (IMAGE_COM.NONFOR == 7 AND IMAGE_COM.ZONES == 3) MAP = 11
/* OPEN COVER (SHRUB)
ELSE IF (IMAGE_COM.NONFOR == 9 AND IMAGE_COM.ZONES == 3) MAP = 11
/* AVALANCHE IN PARKLAND = PARKLAND
ELSE IF (IMAGE_COM.NONFOR == 6 AND IMAGE_COM.ZONES == 3) MAP = 11
/* ALPINE MEADOW
ELSE IF (IMAGE_COM.NONFOR == 14 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.ELEV == 3) MAP = 11 /* SHADOW
ELSE IF (IMAGE_COM.NONFOR == 14 AND IMAGE_COM.ZONES == 3) MAP =
11 /* SHADOW
ELSE IF (IMAGE_COM.IMAGECLASS == 0 AND IMAGE_COM.NONFOR == 0
AND IMAGE_COM.ZONES == 3) MAP = 11 /* SHADOW
ELSE IF (IMAGE_COM.NONFOR == 5 AND IMAGE_COM.ZONES == 3) MAP = 11
/* ALPINE FOREST
ELSE IF (IMAGE_COM.NONFOR == 8 AND IMAGE_COM.ZONES == 3) MAP = 11
/* SPARSE VEG
ELSE IF (IMAGE_COM.IMAGECLASS == 1 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 2 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 3 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 4 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
```

```
ELSE IF (IMAGE_COM.IMAGECLASS == 5 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 6 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 7 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 8 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 9 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 10 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 11 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 12 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 13 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 14 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 15 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 16 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 17 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 18 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 19 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 21 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 22 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 23 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 24 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 25 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 26 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
```

```
ELSE IF (IMAGE_COM.IMAGECLASS == 27 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
ELSE IF (IMAGE_COM.IMAGECLASS == 28 AND IMAGE_COM.ZONES == 3 &
IMAGE_COM.NONFOR == 0) MAP = 11
```

10) Assign appropriate tree species composition classes to ICHwk (Cedar-Hemlock) subzone

```
ELSE IF (IMAGE_COM.NONFOR == 14 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.ELEV == 1) MAP = 9 /* SHADOW
ELSE IF (IMAGE_COM.NONFOR == 14 AND IMAGE_COM.ZONES == 1) MAP = 9
/* SHADOW becomes ICHwk if in elevation range of ICH zone
ELSE IF (IMAGE_COM.IMAGECLASS == 0 AND IMAGE_COM.NONFOR == 0
AND IMAGE_COM.ZONES == 1) MAP = 9 /* SHADOW
ELSE IF (IMAGE_COM.IMAGECLASS == 9 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 10 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 11 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 12 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 13 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 14 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 15 AND IMAGE_COM.ZONES == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* HB
ELSE IF (IMAGE_COM.IMAGECLASS == 16 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 17 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 18 AND IMAGE_COM.ZONES == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* HS
ELSE IF (IMAGE_COM.IMAGECLASS == 19 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 24 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 25 AND IMAGE_COM.ZONES == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* HBsh
ELSE IF (IMAGE_COM.IMAGECLASS == 26 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 27 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONFOR == 0) MAP = 9
```

```
ELSE IF (IMAGE_COM.IMAGECLASS == 28 AND IMAGE_COM.ZONES == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* HSsh
ELSE IF (IMAGE_COM.IMAGECLASS == 15 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* HB
ELSE IF (IMAGE_COM.IMAGECLASS == 18 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* HS
ELSE IF (IMAGE_COM.IMAGECLASS == 25 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* HBsh
ELSE IF (IMAGE_COM.IMAGECLASS == 28 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* HSsh
ELSE IF (IMAGE_COM.IMAGECLASS == 15 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 NE 2 & IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 18 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 NE 2 & IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 25 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 NE 2 & IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 28 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 NE 2 & IMAGE_COM.NONFOR == 0) MAP = 9
ELSE IF (IMAGE_COM.IMAGECLASS == 23 AND IMAGE_COM.ZONES == 1 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* SHsh
ELSE IF (IMAGE_COM.IMAGECLASS == 8 AND IMAGE_COM.ZONES == 1 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* SH
ELSE IF (IMAGE_COM.IMAGECLASS == 21 AND IMAGE_COM.ZONES == 1 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* BSsh
ELSE IF (IMAGE_COM.IMAGECLASS == 7 AND IMAGE_COM.ZONES == 1 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* SB
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* SCW
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 1 & IMAGE_COM.NONFOR == 0) MAP = 9 /* SCW
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 NE 2 & IMAGE_COM.NONFOR == 0) MAP = 9 /* SCW
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 NE 2 AND IMAGE_COM.ELEV == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* SCW
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 == 2 AND IMAGE_COM.ELEV == 1 &
IMAGE_COM.NONFOR == 0) MAP = 9 /* SCW
ELSE MAP = 0
ENDIF
```

11) Assign appropriate tree species composition classes to ESSFvc (Engelmann Spruce – Subalpine Fir) subzone

```
IF (IMAGE_COM.NONVEG2 == 14 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.ELEV == 2) MAP2 = 10 /* SHADOW
ELSE IF (IMAGE_COM.NONVEG2 == 14 AND IMAGE_COM.ZONES == 2) MAP2 =
10 /* SHADOW
ELSE IF (IMAGE_COM.IMAGECLASS == 0 AND IMAGE_COM.NONVEG2 == 0
AND IMAGE_COM.ZONES == 2) MAP2 = 10 /* SHADOW
ELSE IF (IMAGE_COM.NONVEG2 == 5 AND IMAGE_COM.ZONES == 2) MAP2 =
10 /* ALPINE FOREST IN ESSF
ELSE IF (IMAGE_COM.IMAGECLASS == 1 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* B100
ELSE IF (IMAGE_COM.IMAGECLASS == 2 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* BH
ELSE IF (IMAGE_COM.IMAGECLASS == 3 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* BS
ELSE IF (IMAGE_COM.IMAGECLASS == 4 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HM
ELSE IF (IMAGE_COM.IMAGECLASS == 5 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HMB
ELSE IF (IMAGE_COM.IMAGECLASS == 6 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HMS
ELSE IF (IMAGE_COM.IMAGECLASS == 7 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* SB
ELSE IF (IMAGE_COM.IMAGECLASS == 8 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* SH
ELSE IF (IMAGE_COM.IMAGECLASS == 21 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* BSsh
ELSE IF (IMAGE_COM.IMAGECLASS == 22 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* SBsh
ELSE IF (IMAGE_COM.IMAGECLASS == 23 AND IMAGE_COM.ZONES < 3 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* SHsh

/* HS AND HB IN ESSF ZONE WITH 2ND CLASS AS ESSF = ESSF
ELSE IF (IMAGE_COM.IMAGECLASS == 15 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 2 & IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HB
ELSE IF (IMAGE_COM.IMAGECLASS == 18 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 2 & IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HS
ELSE IF (IMAGE_COM.IMAGECLASS == 25 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 2 & IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HBsh
ELSE IF (IMAGE_COM.IMAGECLASS == 28 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 2 & IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HSsh
```

```
ELSE IF (IMAGE_COM.IMAGECLASS == 15 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 == 2 AND IMAGE_COM.ELEV == 2 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HB
ELSE IF (IMAGE_COM.IMAGECLASS == 18 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 == 2 AND IMAGE_COM.ELEV == 2 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HS
ELSE IF (IMAGE_COM.IMAGECLASS == 25 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 == 2 AND IMAGE_COM.ELEV == 2 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HBsh
ELSE IF (IMAGE_COM.IMAGECLASS == 28 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 == 2 AND IMAGE_COM.ELEV == 2 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* HSsh

/* CS IN ESSF ZONE WITH 2ND CLASS AS ESSF = ESSF
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 2 AND
IMAGE_COM.CLASS2 == 2 & IMAGE_COM.NONVEG2 == 0) MAP2 = 10
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 == 2 AND IMAGE_COM.ELEV == 2 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* SCW
ELSE IF (IMAGE_COM.IMAGECLASS == 20 AND IMAGE_COM.ZONES == 0 AND
IMAGE_COM.CLASS2 NE 2 AND IMAGE_COM.ELEV == 2 &
IMAGE_COM.NONVEG2 == 0) MAP2 = 10 /* SCW
ELSE MAP2 = 0
ENDIF
MAP3 = MAP2 OVER MAP
MAP1F = MAJORITYFILTER (MAP3, EIGHT, MAJORITY)

/* Output classes
/* 1 = Recent cuts
/* 2 = Recent burns
/* 3 = Recent disturbances (agent unknown)
/* 4 = Immature forests
/* 5 = Shrub, Grass, Forbe cover (Avalanche path)
/* 6 = Shrub, Grass, Forbe cover
/* 7 = Sparse Vegetation (Open cover with rock soil understory)
/* 8 = Riparian
/* 9 = ICHwk
/* 10 = ESSFvc
/* 11 = ESSFvcp parkland
/* 12 = Alpine Tundra
/* 13 = Ice/Snow/Rock/Road/Lakes
/* 14 = Shadow (in Alpine Tundra zone)
```