Landscape Pattern Analysis
Related to Forest Wildlife Resources

by

Margaret Katherine Trani

Dissertation submitted to the Faculty of the
Virginia Polytechnic Institute and State University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY
IN
FISHERIES AND WILDLIFE SCIENCES

APPROVED:

Robert H. Giles, Jr. Chair

Paul L. Angermeier

Timothy G. Gregoire

James L. Smith

Dean F. Stauffer

June, 1996
Blacksburg, Virginia

Keywords: Forest Wildlife, Landscape Modification, Landscape Pattern, Spatial Scale, Habitat Assessment, Landscape Analysis, Fragmentation, Spatial Heterogeneity

c. 2

LD
5655
V856
1996
T736
c. 2

Landscape Pattern Analysis Related to Forest Wildlife Resources

by

Margaret Katherine Trani

Robert H. Giles, Jr. Chair

Fisheries and Wildlife Sciences

(ABSTRACT)

Wildlife management and natural resource policy decisions are increasingly being made at the landscape level. Understanding the relationship between modification and the pattern of land classes may minimize potential impacts and enhance the complement of wildlife species.

Twenty-four expressions were selected for landscape analysis that describe the spatial heterogeneity, fragmentation, edge characteristics, and connectivity of pattern. Metric relationships were characterized across a variety of landscapes. Cluster analysis organized the metrics into classes quite different from the classification categories used in the literature. Cluster membership reflected the number of land classes, the amount and distribution of forest cover, number of forest patches, patch position, patch shape, patch radius, and edge length.

Cartographic modeling was used to determine how modification influenced landscape pattern. The models depicted spatial relationships resulting from proposed landscape changes. Timber harvest schemes with a few large units and those in clustered arrangements led to less fragmentation than those schemes with several small units or those dispersed across the landscape. The placement of roads had either an invasive or partitioning effect on landscape pattern.

Discriminant analysis rated the effectiveness of pattern expressions for environmental assessment. Metric effectiveness differed among the timber harvest, road expansion, and deforestation modification schemes. The utility and limitations of each expression was discussed.

Sensitivity analyses examined the effects of changing spatial scale on pattern

description. Scale influence was dependent upon landscape complexity, distribution of land classes, and the size and shape of those classes. The loss of ability to detect localized variability, to differentiate among spatial patterns, and to represent boundary detail accompanied the use of large pixels ($420m^2$). There was evidence that spatial scale influences habitat evaluation. Semivariogram analysis assessed the constancy of expression behavior during changes in scale and presented the limits of tolerance for using large pixels in pattern analyses. The variability observed suggested that pattern misrepresentation occurred at coarse resolution levels.

The successful application of landscape analysis depends on the ability to quantify pattern. By analyzing and understanding selected aspects of landscape pattern, I have examined how wildlife management can be enhanced through a knowledge of the landscape.

ACKNOWLEDGMENTS

Several people assisted in bringing about the completion of this dissertation. I acknowledge my committee members for their advice and contributions during the course of my research program. Special thanks to Robert Giles for his role as committee chairman and his guidance while overseeing the research process. Acknowledgment goes to Drs. Dean Stauffer and Timothy Gregoire for their statistical and data presentation expertise, as well as their constant encouragement. Recognition is also offered to Drs. James Smith and Paul Angermeier for their expert counsel during the initial planning and completion stages of the research.

This study was funded in part by the Southern Regional Office of the U.S. Forest Service, without whose support the research would not have been possible. I appreciate the interest and support of Jesse Overcash, Dave Collins, and several people from the Blacksburg Ranger District and the George Washington-Jefferson National Forest.

I also thank the faculty of the Department of Fisheries and Wildlife Sciences, from whom I have learned the merits of different approaches towards wildlife research. My appreciation is also extended to Dr. James Campbell, Department of Geography, for introducing many of the spatial analysis techniques used in this dissertation. I have also valued the friendship of several graduate students, past and present.

Finally, my appreciation is offered to my grandparents, whose courage brought them to this country 90 years ago on a quest to improve their lives and those of their descendants. My parents, Garmino and Louise Trani, were supportive throughout my academic career, a luxury they could never afford themselves. Most importantly, I wish to thank my husband, Skip Griep, and our children Jennifer and Joshua, for their understanding and patience during the years of graduate research. We have lost invaluable time together.

TABLE OF CONTENTS

INTRODUCTION 1

Landscapes and Their Analyses 1
Objectives of the Dissertation 4

LITERATURE REVIEW 6

Spatial Heterogeneity 6
Fragmentation 9
    Plants 9
    Invertebrates 10
    Mammals 10
    Birds 11
Edge Characteristics 12
    Beneficial Edge Effects 13
    Detrimental Edge Effects 14
    Edge and Predation 15
    Edge and Brood Parasitism 16
Connectivity 16
    Travel Corridors 17
    Dispersal Corridors 17
    Connectivity to Critical Habitat 18
Describing Landscape Pattern 19
Conclusion 21

CHAPTER 1: Landscape Pattern Expression Selection 22

Spatial Heterogeneity Expressions 24
Fragmentation Expressions 29
Edge Characteristics Expressions 32
Connectivity Expressions 34
Data Requirements 37

CHAPTER 2: Classifying and Relating Landscape Pattern Expressions 39

Introduction 39
Methods 40
    Map Selection 40
    GIS Analysis 41

Statistical Analysis 41
Results and Discussion 42
Relationships Between Landscape Pattern Expressions..... 42
Classification of Landscape Pattern Expressions..... 57
Associations Between Landscape Pattern Expressions..... 61
Summary 63

CHAPTER 3: The Influence of Modification on the Pattern of Land Classes..... 64

Introduction 64
Methods 66
Map Selection 66
GIS Analysis 66
Statistical Analysis 68
Results and Discussion 71
The Influence of Timber Harvest on Landscape Pattern..... 71
Effectiveness of Expressions for Describing Harvest Alternatives..... 73
Effectiveness of Expressions for Describing Spatial Arrangement..... 76
Road Expansion Influence on Landscape Modification..... 82
Effectiveness of Expressions for Describing Roadless and Roaded Areas..... 88
Progressive Deforestation Influence on Landscape Modification..... 90
Effectiveness of Expressions for Describing Contiguous and Fragmented Landscapes..... 97
General Trends in Landscape Pattern Change 99
Forest Management within a Landscape Context..... 103
Discriminant Analysis and Landscape Pattern Analysis 105
Reducing the List of Landscape Pattern Expressions..... 110
Summary 112

CHAPTER 4: The Influence of Spatial Scale on Landscape Pattern Analysis 115

Introduction 115
Methods 117
Map Selection and Data 117
GIS Analysis Process 117
Statistical Analysis 119
Results and Discussion 123
The Influence of the Spatial Scaling Process 123
The Influence of Spatial Resolution on Metric Behavior 125
Underlying Landscape Pattern and Spatial Scale Analysis 140
General Trends in Landscape Pattern Change 143
Variability of Pattern Metrics and the Semivariogram 144

Spatial Scale and Wildlife Habitat Assessment 146
Management Implications 149
Summary 151

CHAPTER 5: Conclusion..... 152

LITERATURE CITED 159

Appendix 1. Batch Program for Landscape Expression Calculation..... 172

Appendix 2. Turbo Pascal program for Landscape Expression Calculation..... 178

VITA 183

LIST OF FIGURES

Figure 1. Relationships between selected spatial heterogeneity metrics..... 44

Figure 2. Relationships between selected fragmentation metrics..... 48

Figure 3. Relationships between patch size and selected fragmentation metrics..... 51

Figure 4. Relationships between selected edge metrics..... 54

Figure 5. Relationships between selected connectivity metrics..... 56

Figure 6. A classification of 24 landscape expressions by the oblique principal components clustering technique..... 59

Figure 7. Simplified representation of 1 of the harvest patterns developed under the timber harvest models..... 67

Figure 8. Graphical representation of 2 progressive fragmentation schemes evaluated on the model landscapes..... 69

Figure 9. The response of selected landscape expressions to the 5 standard harvest schemes..... 74

Figure 10. Changes in landscape pattern based on the dispersed and clustered timber harvest models..... 79

Figure 11. Relationship between edge habitat creation and timber harvest modification..... 81

Figure 12. Changes in selected spatial heterogeneity and edge metrics during the progressive addition of road lengths..... 84

Figure 13. The partitioning effect of road addition on landscape pattern..... 86

Figure 14. The invasive effect of road addition on landscape pattern..... 87

Figure 15. Changes in selected spatial heterogeneity and edge expressions during the progressive loss of forest cover..... 93

Figure 16. Changes in selected fragmentation and connectivity expressions

during progressive loss of forest cover..... 96

Figure 17. Projection of landscape maps onto the discriminant function axis..... 109

Figure 18. Location of the forested landscapes used in the pattern and spatial scale analyses..... 118

Figure 19. Schematic illustration showing the increase in pixel size as spatial resolution becomes coarse..... 120

Figure 20. General form of the semivariogram..... 122

Figure 21. The relationship between spatial resolution (pixel size) and sampling intensity (number of sampled pixels) within a map..... 126

Figure 22. The influence of spatial resolution on the Simpson and evenness indices..... 127

Figure 23. The influence of spatial resolution on interspersion and binary comparison matrix values and their corresponding semivariograms..... 130

Figure 24. The influence of spatial resolution on forest edge length, convexity, and Patton's index with their corresponding semivariograms..... 132

Figure 25. The influence of spatial resolution on fragmentation I, fragmentation II, and forest contiguity with their corresponding semivariograms..... 134

Figure 26. The influence of spatial resolution on percent forest cover and forest interior with their corresponding semivariograms..... 136

Figure 27. The influence of spatial resolution on the number of forest patches, mean patch size, and interpatch distance with their corresponding semivariograms..... 139

LIST OF TABLES

Table 1. Landscape pattern expressions selected for examination..... 23

Table 2. Input parameters for the pattern metrics described in this study..... 38

Table 3. Correlation coefficients between the expressions of spatial heterogeneity and other pattern groups..... 43

Table 4. Correlation coefficients between the expressions of fragmentation and other pattern groups..... 47

Table 5. Correlation coefficients between the expressions of edge and connectivity and other pattern groups..... 53

Table 6. Cluster membership of the pattern metrics based upon oblique principal components..... 58

Table 7. Pattern changes along a progression of increasing number of timber harvest units as derived from 38 maps..... 72

Table 8. Effectiveness of landscape metrics as discriminators of pattern differences resulting from changes in the number and size of harvest units as derived from 38 maps..... 75

Table 9. Effectiveness of landscape metrics as discriminators of pattern differences resulting from changes in spatial arrangement of harvest units as derived from 30 maps..... 77

Table 10. Pattern changes resulting from road addition and expansion as derived from 30 maps..... 83

Table 11. Effectiveness of landscape metrics as discriminators of pattern differences between roadless and heavily-roaded landscapes as derived from 49 maps. 89

Table 12. Pattern changes resulting from progressive deforestation as derived from 30 maps..... 91

Table 13. Effectiveness of landscape metrics as discriminators of pattern differences between fragmented and contiguous landscapes as derived from 30 maps.. 98

Table 14. The influence of spatial scale on landscape pattern description and potential wildlife habitat evaluation..... 148

INTRODUCTION

Landscapes and Their Analyses

Resource managers are facing some of their greatest challenges as the end of the twentieth century approaches. Landscape modification and loss of wildlife habitat continue to be the major factors impacting wildlife resources. The loss of forests world-wide may have contributed to global warming and escalating species extinctions (McNeely et al. 1990). These challenges are occurring at a time when there is a growing appreciation of the importance of dynamic processes at the landscape scale (Naveh and Lieberman 1984; Zonneveld and Forman 1990). Landscape-level phenomena are receiving increasing attention as questions of global change become more prominent (Turner 1990).

The emerging field of landscape ecology may provide an approach for evaluating possible solutions to these environmental problems. Landscape ecology encompasses many disciplines concentrating study on the spatial and temporal patterns of landscapes (Risser et al. 1984). Used in this context, pattern refers to the perception of spatial arrangement of communities and ecosystems. The development of spatial pattern, and the consequences of that pattern for the dynamics of populations and ecosystems (Levin 1992), is a fundamental theme within landscape ecology. It includes interest in the modifications occurring through landscape management and how landscapes in turn influence such management.

The role of landscape analysis in environmental planning and management has been established for several decades. Dutch researchers recognized the dynamic role of anthropogenic influence upon the creation of landscape heterogeneity and stability (Zonneveld and Forman 1990). A landscape perspective was also described in the early writings of Leopold (1949). By the middle of this century, the landscape was recognized as the highest integrative level of environmental processes (Dansereau 1957). The emphasis at a landscape level towards interrelating the patterns of people and their natural

ecosystems was made by Van der Maarel (1975), who spoke of landscape ecology as a relevant science.

A landscape often refers to the organizational level above that of community and ecosystem (Forman and Godron 1986). It may be viewed as an area distinguished by visible structural and functional qualities that are more similar within the landscape than with those of surrounding areas. A landscape may be hundreds of square kilometers, such as a large forest, or may comprise an area of much smaller extent. The definition of a landscape is dependent upon the context of its use. Its meaning is shaped by the scale at which ecological processes are being discussed, analyzed, and evaluated. A landscape then becomes the working topic for those protecting and managing areas and their resources.

Resource professionals are recognizing the influence that the landscape, and the pattern within that landscape, has on species and the dynamics of their populations. Species respond to spatial pattern in various ways, using certain areas for feeding and reproduction, and avoiding other areas entirely. Fragmentation, occurring when a landscape is divided into sections, results in the decline of some forest birds (Wilcove et al. 1986; Askins et al. 1990); in the decline of large, wide-ranging species (Picton 1979; Pelton 1985); and in the loss of other specialized species (Matthiae and Stearns 1981; Rosenberg and Raphael 1986). The existence of corridors within a landscape facilitates movement for some species (Harris and Gallagher 1989) while the size and shape of landscape patches influences both biotic and abiotic processes (Van Dorp and Opdam 1987). The distribution and availability of resources in space is an important component for the persistence of many species (Merriam 1990). Landscape connectivity (Noss and Harris 1986) and edge characteristics (Yahner 1988) may also have important implications, both positive and negative, for the persistence of an array of wildlife resources with very different requirements.

If a relationship between landscape pattern and species abundance exists, it is important to understand and describe such patterns. Landscape pattern must be identified

and quantified in meaningful ways before a clear relationship can be made between it and species responses to it. Several expressions describe landscape pattern; it is likely that no single expression will contain needed information for all analyses or behave consistently across a variety of landscape patterns. The utility and limitations of these expressions, which may vary by species, should be understood prior to their application for environmental assessment.

Several of the landscape metrics, of which pattern is composed, can be grouped into the categories of: (1) spatial heterogeneity, (2) fragmentation, (3) edge characteristics, and (4) connectivity. Relationships are expected to exist among these groups. For example, Franklin and Forman (1987) reported that forest edge per unit area increased while patch size decreased within their harvest model. This relation suggests that other relationships may exist among landscape expressions. Most importantly, it suggests questions about whether those relations continue to persist over a range of different landscapes. Knowledge about these relationships may be used to reduce the number of expressions required for landscape analysis, by identifying metrics that provide unique pattern information. It may prove valuable to classify pattern metrics (based on their behavior) into classes so that the nature of, and relationships between, can be easily understood.

Landscapes today support diverse, and often competing, uses of natural resources. These landscapes increasingly have anthropogenic influences in the form of intensive management. This force on the landscape has accompanied a shift in emphasis from monitoring specific management sites to considering cumulative impacts over the landscape (Noss 1983; Flather and King 1992). This emphasis has led to the United States Forest Service policy requiring the application of a landscape perspective within resource management issues (USDA 1990). This perspective has also arisen from federal laws requiring that resource agencies examine potential land use conflicts with wildlife resources.

An understanding of the effect of proposed management activities on the resulting

landscape pattern may promote strategies to minimize potential adverse impacts (Freemark 1988). For example, landscape characteristics such as patch size, edge length, and spatial arrangement can be significantly altered by forest management (Hunter 1990). By altering the distribution and availability of spatial resources, changes in landscape pattern may influence many of the components important for the persistence of species (Merriam 1990). Land management agencies monitor these components, along with other information, to predict species richness and occurrence (Stoms 1992). The development of spatial analyses for evaluating how landscape modification influences pattern may prove useful for resource managers.

The description and quantification of landscape pattern may depend on the scale of observation. Spatial scale issues are fundamental to landscape studies (Meetenmeyer and Box 1987; Cullinan and Thomas 1992; Holling 1992; Levin 1992). Landscapes described at one scale may not be the same at another, and information may be lost at both fine and coarse levels of resolution. If spatial scale influences the outcome of pattern analysis, then resource plans based on those analyses will also be affected. Integrating landscape measurements made at disparate scales (Musick and Grover 1991) and extrapolating those measurements from one spatial scale to another remains a problem. There is a fundamental need to consider how spatial scale may influence observable relationships (Allen and Starr 1982; Wiens et al. 1986) and the assessment of wildlife habitat.

Objectives of the Dissertation

The above introduction suggests that there is a connection between landscape pattern and the persistence of species and their habitats (Golley 1989). Species occupy landscapes and the pattern of the landscape is believed to either support or inhibit the survival of those species. Given this background, my goal in this study was to assess landscape pattern as it relates to forest wildlife resources.

This goal was pursued in the following manner. First, an extensive literature review is presented that documents the relevance of selected aspects of landscape pattern to

biological diversity. The remainder of the dissertation contains 4 sections. The first section (Chapter 1) presents and describes a large list of landscape expressions. Within the second section (Chapter 2), I compare and contrast the expressions for classifying them and identifying their relationship. Expressions are recognized that are highly correlated, that duplicate pattern information, and that appear to have no discernable relationship to the other expressions. In addition, a classification is presented based upon the behavioral properties of the expressions. The third section (Chapter 3) examines how landscape modification (in the form of timber harvest, road construction, and progressive deforestation) influences pattern and wildlife-habitat relationships. The effectiveness of the expressions for monitoring purposes is presented along with a discussion suggesting their usefulness and limitations. In the fourth section (Chapter 4), I explore how changes in spatial scale influence landscape pattern analysis, and describe the sensitivity, stability, and variability of pattern expression behavior. The analysis identifies how the underlying landscape pattern and the sampling process contribute to the results observed at different spatial scales. Finally, I discuss the management implications that selecting a spatial scale for the landscape analysis may have upon wildlife habitat assessment.

By analyzing and understanding selected aspects of landscape pattern, I have attempted to develop new approaches for examining how the management of wildlife resources may be enhanced through increasing knowledge of landscape pattern.

LITERATURE REVIEW

Landscapes are modified by both natural phenomena and human activities. Natural phenomena include climatic, geological, and ecological processes. Human activities include road construction, stream channelization, timber harvest, wetland drainage and conversion, and agricultural practices. The resulting landform modifies air and ground temperature, moisture, and nutrient availability (Swanson et al. 1988). These, in turn, influence the colonization, distribution, and persistence of plants and animals.

Resource professionals are recognizing the significant impact that these modifications have upon landscape pattern and related ecological processes dependent on those patterns. Fragmentation results in the decline of many bird populations; in the decline of large, wide-ranging species; and in the loss of other specialized species. Habitat corridors allow movement of some species between habitats. The size and shape of forest patches affect biotic and abiotic processes over various scales. Landscape connectivity may enhance habitat use by some species. Edge characteristics may have both positive and negative implications for biological diversity.

Understanding the connection between selected aspects of landscape pattern and the persistence of species is the first step towards developing strategies for minimizing impacts of proposed land use changes. Quantifying and describing landscape patterns is the logical next step. The purpose of this section is to present a general review of the evidence linking selected aspects of landscape pattern with wildlife resources.

Spatial Heterogeneity

Spatial heterogeneity is the variability in the number, type, size, and shape of land classes within a landscape. These factors, and the changes in them, are related to the number of species present in an area and the abundance of each species. For example, the spatial heterogeneity of a forest may be a function of the number of different plant communities, their shape and seral stages, and the elements of structural complexity within

those communities.

The role of spatial heterogeneity in regulating species diversity has been examined by several researchers. Heterogeneous stands often have greater species richness and abundance than that found in homogeneous stands (Roth 1976). Freemark and Merriam (1986) found that edge-adapted species were particularly abundant in heterogeneous stands. Management practices that enhance local habitat diversity however, may depress species richness at regional scales (Faaborg 1980). Species richness begins to decline when the average size of habitat units becomes small relative to required needs of species and species groups.

Wildlife species differ in their needs for spatial heterogeneity. Habitat generalists such as the bobcat (Felis rufus), coyote (Canis latrans), raccoon (Procyon lotor), and striped skunk (Mephitis mephitis) can exist in many different environments. Habitat specialists require certain conditions such as a fairly uniform area or a mixture of specific vegetative types. The northern spotted owl (Strix occidentalis caurina) is a specialist that requires relatively contiguous expanses of old-growth forest.

Many bird species require a diversity of habitats. The American woodcock (Scolopax minor) requires different habitat types in which to roost, forage, nest, and rear its broods. Canada geese (Branta canadensis) use both fields and shorelines. The royal eagle (Aquila chrysaetos) nests on steep cliffs and hunts in cultivated fields (Forman and Godron 1986). There is evidence that even within continuous forests, bird species may require a mosaic of patch types (Karr and Freemark 1983).

Many mammals use 2 or more habitat types. An animal may forage or rear young in one habitat and rest or hide in another. Moose (Alces americana) feed in marshes and sleep in thick woods. Elk (Cervus canadensis) and white-tailed deer (Odocoileus virginianus) move among the forest, shrublands, and fields for their needs (Roller 1978). Coyotes hunt in early successional stages for small mammals and may use downed logs within mature stands as denning sites (Thomas et al. 1979). Bighorn sheep (Ovis canadensis) make use of rugged area and desert streambeds; skunks make use of fields

and wooded fencerows.

Diverse habitats often have several successional stages that provide a variety of plant communities and resources for animals. Wild turkey (Meleagris gallopavo) forage for invertebrates in early seral stages and for mast in mature forest stands (Gustafson et al. 1994). Ruffed grouse (Bonasa umbellus) roost in regeneration stands, breed in pole stands, and forage in mature aspen stands.

Many animals forage within different habitats in each season. For example, black bear (Ursus americanus) will feed on mast from lowland oaks in winter and spring; then move to upland oaks for the remainder of the year (Rudis and Tansey 1995). Areas with diverse habitats may offer seasonal and temporal resources for a particular species. Ample variety is particularly important for waterfowl that migrate over long distances in order to meet both breeding and over-wintering requirements. Mallards (Anas playtrhynchos) will feed on agricultural crop residue during winter, while canvasback (Aythya valisineria) and redhead ducks (A. americana) will forage on aquatic vegetation and invertebrates (Risser et al. 1984).

The "interspersion" of different habitat types required by a species is often as important to the species as the number or types of habitats available. Interspersion suggests the arrangement and distribution of land classes (e.g., forest stands, ponds, corn fields). The proximity or "juxtaposition" of different habitats is also of importance. Energy expenditures may be greatly reduced when habitat types are juxtaposed appropriately (Giles 1978). Habitats that provide nesting cover adjacent to forage areas can support large numbers of wildlife. Yahner (1988) reported that the juxtaposition of farmstead shelterbelts with croplands results in a greater diversity of breeding birds than does the juxtaposition of shelterbelts with pasture. Pheasants (Phasianus colchicus) use agricultural fields for feeding, preferring fields juxtaposed with hay crops that provide nesting and roosting cover (Yahner 1988). Gray squirrels (Sciurus carolinesis) commonly occur in pine forests during summer and fall, but do not remain there throughout the year unless hardwoods are in close proximity (Dickson and Huntley 1987).

In addition, Harris (1984) suggested that the manner in which streams and riparian strips are juxtaposed with coniferous forest adds considerably to habitat value for grizzly bear (Ursus horribilis), river otter (Lutra canadensis), mink (Mustela vison), and the northern water shrew (Sorex palustris).

Fragmentation

Fragmentation occurs when a parcel of forest or other habitat is separated into "patches". Continual deforestation reduces further the size of forest patches, increasing the distance among remaining patches. Landscape fragmentation may result from geomorphic processes such as climatic disturbances (e.g., hurricanes and tornados) and from human activities (e.g., summer home construction).

Harris (1988a) cited fragmentation as the most serious threat to biological diversity and the primary cause of some present-day species extinctions. Not only is there loss of the original habitat, but those species that required that habitat are isolated. These isolated areas have been compared to oceanic islands, with declining rates of immigration and increasing rates of extinction. Area-sensitive species requiring large tracts of habitat decline or even become locally extinct.

The reduction of forest size by fragmentation may lead to the regional extirpation of some species. Often these species are distributed over a patchy environment, making the flow of animals between patches difficult. The persistence of a population may be linked to the number, size, and degree of isolation of forest patches. Indeed, Robbins et al.(1989) found that the degree of patch isolation is often a significant predictor of relative abundance for many bird species.

Plants. Ranney et al. (1981) documented that fragmentation influences major changes in the vegetational composition of forest patches. The microclimate at the perimeter of a forest patch differs markedly from that within the large, continuous forest. The perimeter will experience more wind and sun, creating a special climate in which a

different assemblage of plants can grow. Characteristic edge associations composed of xeric-adapted and shade-intolerant species become established. Species usually associated with early succession then grow under a forest canopy (Whitney and Runkle 1981) and may have a long-term effect on the composition of forest patches.

Fragmentation may also have a significant impact on vegetation dynamics in oak-pine forests. Gibson et al.(1988) found that density and diversity of saplings and trees were greater in forest patches than in the non-isolated stands. In apparent contrast, Hobbs (1988) reported that the entire array of vascular plants of upland forest patches was positively related to patch size.

Invertebrates. Invertebrates are also affected by fragmentation. One of the first studies to document the effects of forest fragmentation on insects in the tropics showed that dung and carrion beetle communities in forest patches differ less than those in contiguous forests (Klein 1989). Several types of invertebrates (e.g., ants, flies, beetles) become relatively abundant near forest edges created by fragmentation (Hunter 1990), promoting insect disturbance at those edges. It also gives further credence to the theory of a water imbalance likely at edges which results in tree mortality, a condition rapidly followed by insect infestations. Such edge phenomena relate well to insect species richness and their faunal associates.

Mammals. Although the effects of fragmentation on mammals have not been studied extensively, it appears that a species-area relationship exists. Matthiae and Stearns (1981) found that the relative abundance and number of mammalian species increased with forest size. Red squirrel (Tamiasciurus hudsonicus), grey squirrel, and fox squirrel (Sciurus niger), raccoon, and red fox (Vulpes vulpes) were more abundant in the large forests of Wisconsin. Fahrig and Merriam (1985) also reported that certain mammals were more common in large forest tracts than in smaller, isolated patches. Populations of white-footed mice (Peromyscus leucopus) and chipmunks (Tamias striatus) in small

forest patches decline seasonally and to such an extent that local extinctions occur (Merriam 1990).

Fragmentation stemming from landscape development impacts those mammalian species that do not adapt well to human presence. Rosenberg and Raphael (1986) reported that fisher (Martes pennati), gray fox (Urocyon cinereoargenteus), ringtail cat (Bassariscus astutus), and northern flying squirrel (Glaucomys sabrinus) were sensitive to fragmentation in the Pacific northwest. Picton (1979) found that the presence of several species of large mammals in Montana was correlated with the size of the mountain ranges where each species occurs. These species included mule deer (Odocoileus hemionus); elk; pronghorn antelope (Antilocapra americana); bighorn sheep; mountain goat (Oreamnos americanus); bison (Bison bison); and grizzly bear. The increase in mammal species with an increase in habitat size is attributed to the addition of new species as their requirements are met for minimum habitat size and specific habitat components. The insularity of these populations increased with continued landscape development while larger, undeveloped areas tended to protect these species from extinction.

Fragmentation often appears directly proportional to road density (Bennett 1991). Roads may act as barriers to the movement of species through the landscape, dividing and isolating populations to a varying extent. Extensive networks of roads have negative impacts upon moose, black bear, white-tailed deer, and mountain lion (Felis concolor). The distribution of the timber wolf is also influenced by road density (Mech et al. 1988). These negative impacts stem from both loss of habitat and increased hunter accessibility onto the landscape, increasing the harvest and disturbance of these mammals.

Birds. The effects of fragmentation on bird species has been studied quite extensively. Long-term studies show major population declines of neotropical migrants, particularly in small forest patches. Breeding bird censuses for isolated forest patches indicate general reduction in abundance and diversity of species over the past 20 years (Lynch and Whitcomb 1977).

Species that are intolerant of fragmentation tend to be highly migratory, are forest-interior specialists, build open nests, and nest on the ground (Whitcomb et al. 1981). The worm-eating warbler (Helmintheros vermivorus), the hooded warbler (Wilsonia citrina), and the black-and-white warbler (Mniotilta varia) are intolerant of fragmentation (Hamel 1992). These and other similar species are generally absent in patches < 20 ha. Other species that appear sensitive to habitat size include the swallow-tailed kite (Elanoides forficatus), broad-winged hawk (Buteo platypterus), barred owl (Strix varia), pileated woodpecker, and the black-billed cuckoo (Coccyzus erythrophthalmus) (Hamel 1992). While many species avoid small patches, widespread permanent residents and short-distance migrants tend to predominate in small areas (Askins et al. 1990).

Forest size is also a primary determinant of the richness and size of bird assemblages. Galli et al.(1976) and Van Dorp and Opdam (1987) reported a direct relationship between number of bird species and the increase in forest area. Ambuel and Temple (1983) found that bird diversity increases with area. The increase was due primarily to increases in forest-dwelling, long-distance migrants. Freemark (1988) also reported that large forest size was important to long-distance migrants and species with large territory requirements.

Opdam et al. (1985) reported that the distance from a patch within agricultural landscapes to the nearest large forest was negatively correlated with species richness. Both sedentary and migratory species seemed to show the effects of isolation. Whitcomb et al. (1981) found that isolation negatively effects the number of forest interior birds in small patches. Lynch and Whigham (1984) found that the abundance of breeding birds is typically reduced by the isolation of forest patches resulting from fragmentation.

Edge Characteristics

Edge has been an important consideration in landscape analysis since the beginning of wildlife management. An edge is the place where 2 different plant communities, successional stages, or land uses come together. Edge is related to the broad concept of

boundary, which is common in the landscape ecology literature. Boundaries are recognized by spatial discontinuities in features of the soil, vegetation, and geology (Zonneveld and Forman 1990). Boundary dynamics emphasize how boundaries affect ecological processes, the movement of materials (including energy and species) over an area, and how imbalances in these transfers affect landscape configuration (Weins et al. 1985). This is a distinctly different focus from the concept of "edge effect" (Leopold 1933), which considers how transitions between community types influence species diversity (Weins 1992). In these transitional areas, there is often an increase in species composition and abundance. This may be due to an additive effect on the flora and fauna when 2 communities come together.

Five factors appear important in edge development: land class types; successional stages of those land classes; solar aspect; activity of herbivores; and the way an edge is maintained (Ranney et al. 1981). The relative importance of these factors changes through time and space.

Several factors work separately or in combination to produce "inherent" or natural-occurring edges. Such edges are characterized by abrupt changes in soil type, topographic or geomorphic differences, and changes in microclimate (Thomas et al. 1979). Inherent edges are influenced by the recurrence of environmental disturbances. In contrast, "induced edges" are those created by management practices or short-term natural phenomena. Induced edges can be created by manipulation (e.g., cultivation, fertilization, harvest) and by environmental factors such as fires, storms, and floods.

Beneficial Edge Effects. Many wildlife species occur within edge habitat, particularly in certain juxtapositions of communities offering food and cover. Rocky Mountain mule deer (Odocoileus hemionus hemionus), Rocky Mountain elk (Cervus elaphus nelsoni), black-tailed deer (Odocoileus hemionus), and white-tailed deer make heaviest use of their range near the edges of brush openings. Game birds such as the American woodcock (Scolopax minor) and northern bobwhite (Colinus virginianus), do

well in edge habitats. Many bird species of urban and agricultural landscapes are edge-adapted (Whitcomb et al. 1981) just as are many game species. Species requiring access to 2 or more habitat types are frequently associated with edges. These include the European sparrowhawk (Accipiter nisus) and the European badger (Meles meles) (Hunter 1990).

The distribution of songbirds near forest edges has also been investigated. Many bird species concentrate near edges and many forest birds favor edge habitat (Yahner and Scott 1988). Edge also provides enhanced forage and improved habitats for woodland passerines (Reese and Ratti 1988). In Toronto, Gotfryd and Hansell (1986) found that edge length was strongly associated with bird abundance and species richness, possibly due to the relatively abundant invertebrates that occur near edges. In addition, Gates and Gysel (1978) suggested that edges attract many passerine birds because they contain structural cues similar to those of the patchy habitats in which these species evolved.

Detrimental Edge Effects. Edge is not universally beneficial for all species and is likely to be detrimental to certain populations. Excessive edge may lead to reduced populations of species dependent on large blocks of forest interior (Robbins et al.1989) and a shift from characteristic forest species to generalist species at both local and regional scales (Wilcove et al. 1986).

There appears to be a group of species that avoids edges and does best in large continuous forest tracts distant from edges. Forest species such as the sugar maple (Acer saccharum) and American beech (Fagus grandifolia) do not thrive near edges (Ranney et al. 1981). Desiccation from the sun and occasional windthrow may make edges unsuitable habitat for these species. Southeastern breeding birds species that nest only in the interior of forests include the sharp-shinned hawk (Accipiter striatus), Cooper's hawk (Accipiter cooperii), hairy woodpecker (Picoides villosus), winter wren (Troglodytes troglodytes), and veery (Catharus fuscescens) (Hamel 1992).

Management for edge effect through habitat subdivision can negatively impact

interior species, particularly in forest patches with large perimeter-to-area ratios (Noss 1983). Robbins (1979) found increasing evidence that birds characteristic of forest interior habitats are unable to maintain their populations where edge is abundant.

An increase in density of forest-edge and farmland species along edges may exclude certain interior and long-distance migrant species. Competition for tree cavities by edge-adapted starlings (Sturnus vulgaris) and red-bellied woodpeckers (Melanerpes carolinus) exert a direct negative impact on less abundant forest species (Harris 1988a). This competition may influence bird community composition more than area-dependent changes in habitat (Ambuel and Temple 1983).

Edge and Predation. Many species that readily use edges are subject to high rates of mortality from predators attracted to edge habitats. Raccoon, short-tailed weasel (Mustela erminea), and striped skunks often travel along edges, preying on small animals. Ground nests can receive great predation pressure in edge habitats where mammals and reptiles are the dominant predators (Chasko and Gates 1982). This increase in predation reduces the fecundity of such birds as the Kentucky warbler (Oporornis formosus), scarlet tanager (Piranga olivacea), wood thrush (Hylocichla mustelina), yellow-throated vireo (Vireo flavifrons), and the ovenbird (Seiurus aurocapillus) (Temple and Cary 1988). For these species, increased edge alters both the quantity of habitat available and the quality of the remaining habitat.

Many bird species occurring in edge habitat exhibit reduced reproductive success there, often referred to as the "ecological trap" phenomenon (Gates and Gysel 1978). Nesting success of some songbirds is often low near forest edge due to high densities of nest predators such as the blue jay (Cyanocitta cristata), the American crow (Corvus brachyrhynchos), and the common grackle (Quiscalus quiscula) (Yahner and Scott 1988). Wilcove et al. (1986) reported that edge-related increases in predation on artificial nests can extend 300 meters from the forest edge. Finally, plant herbivory by deer and insects is often great along edges (Temple 1986). In Wisconsin, browsing by white-tailed deer

profoundly affects the abundance and structure of woody and herbaceous plant species. The effects of high deer densities along edges carries into the interior of old and mature forest stands, greatly modifying their composition. This leads to reduced regeneration of Canada yew (Taxus canadensis), eastern hemlock (Tsuga canadensis, and white cedar (Thuja occidentalis) (Alverson et al. 1988).

Edge and Brood Parasitism. Nest parasitism is also a factor concerning managers of edges. Brood parasites lay their eggs in the nests of other species that hatch and rear the parasite's young. Brood parasites substantially reduce the reproductive success of their hosts; few or none of the host's young fledge from a parasitized nest. Brood parasites such as the brown-headed cowbird (Molothrus ater) occur more frequently near edges and openings within large forest tracts. Brittingham and Temple (1983) suggested that cowbirds may contribute to the population declines of the endangered Kirtland's warbler (Dendroica kirtlandii) and in the reductions of the Acadian flycatcher (Empidonax virescens), veery (Catharus fuscescens), American redstart (Setophaga ruticilla), and Louisiana waterthrush (Seiurus motacilla).

Connectivity

Several researchers have presented evidence for the value of corridors (MacClintock et al. 1977; Whitcomb et al. 1981; Wilcove et al. 1986). Corridors may facilitate movement of animals, dispersal into new habitats, and improve habitat quality by connecting patches of critical habitat.

Forman and Godron (1981) described 4 types of corridors. Line corridors such as roads, fencerows, and powerlines generally support species characteristic of edge habitat. Strip corridors are wide bands that contain the interior environment required by many species. Riparian corridors are the most widespread. The fourth type of corridor is a network, which may provide access to resources that are patchily distributed. Each of the types of corridors are used by many species for travel within habitats and for dispersal.

Travel Corridors. The effectiveness of travel corridors depends on the habitat structure within the corridor, the width and position of the corridor relative to patches in the landscape, and the life histories of the animals involved (Noss and Harris 1986). Narrow strips of trees, fencerows, and networks can facilitate travel by some species and retard others. Forman and Godron (1986) described movement along corridors for a variety of bird and mammal species. Striped skunk travel long distances along hedgerows in search of prey. Bears and wolves travel along roads and frozen drainage ditches.

Stream and riparian corridors affect the movement of animals within watersheds, linking different patches of habitat for red fox, pine marten (Martes martes), and fisher. Hardwood streamside corridors allow wild turkey to move over areas that would otherwise constitute barriers to movement. Elk often use riparian corridors to travel between their winter and summer ranges (Thomas 1979). Major river corridors often serve as flyways for migratory waterfowl and raptors.

Travel corridors are often eliminated by agricultural and commercial development, restricting species movement. The home ranges of otter and mink become confined by human dwellings along the waterways. Harris (1988a) reported that intersections of roads and waterways with home range reduces the range and increases mortality due to collision with motorized vehicles. This was observed for the endangered panther (Felis concolor), Key deer (Odocoileus. v. clavium), and manatee (Trichechus manatus).

Dispersal Corridors. Simpson (1940) discussed the importance of land bridges as dispersal corridors for terrestrial species colonization. Heany and Patterson (1986) presented an extensive review of the regional patterns of mammal distribution as affected by dispersal corridors. Pelton (1986) described how the loss of movement corridors has come to restrict the distribution of black bears.

Jackson (1987) reported that red-cockaded woodpeckers (Dendrocopos borealis) were aided in colonizing existing habitat by corridors. Dickson and Huntley (1987) found that neither gray squirrel nor fox squirrel would disperse into isolated patches unless

wooded corridors of sufficient width were available. Similarly, Stauffer and Best (1980) found that among 40 bird species inhabiting riparian forests in Iowa, 9 species occurred only in strips wider than 90 meters.

Dispersal corridors may also facilitate recolonization. Corridors that link small patches may enrich those patches with species that otherwise could not survive there (Galli et al. 1976; Robbins 1979). Corridors were shown by modeling (Fahrig and Merriam 1985) to enhance recolonization of local extinctions and to reduce frequency of extinction by enhancing growth in the subpopulations.

Recolonization across agricultural landscapes is enhanced by fencerows serving as corridors for movement across croplands (Merriam 1990). The meadow vole (Microtus pennsylvanicus) extended its distribution range along a newly constructed interstate highway (Forman and Godron 1986). The grassy sides of the highway served as a long corridor that facilitated dispersal.

Another important aspect of dispersal corridors is promoting gene flow between habitat patches which allows species to persist in landscapes where they could not otherwise occur. Harris (1988b) reported that isolating a panther subpopulation resulted in restricting gene flow sufficient to cause inbreeding and the expression of distinct morphological traits.

Connectivity to Habitat. The connectivity of habitat in a landscape may enhance habitat quality. Connectivity refers to the degree of continuity of a landscape that facilitates movement both within and among habitats. Small tracts of forest can often be used by forest birds when connected to large tracts by means of wooded corridors (Robbins 1979; Freemark and Merriam 1986). Forest interior birds are able to breed and persist in forest fragments connected by woodland corridors (MacClintock et al. 1977). Merriam (1990) discussed how corridors between forest fragments provide ease of colonization by woodland mice.

The connectivity of landscapes is rapidly becoming an important consideration in

land-use planning. When the fragments of forest or other habitat are interconnected, the species occurring within those fragments operate demographically as a metapopulation. Such metapopulation dynamics allow the persistence of many species on the landscape. Corridors have linked large reserves allowing species to shift their geographic ranges in response to long-term climatic change (Hunter et al. 1988). Fragmentation and the isolation of habitats may be alleviated through a series of corridors, possibly permitting the survival of extinction-prone populations. In Florida, riparian and coastal corridors are being used to interconnect existing protected areas to facilitate the preservation of wide-ranging species (Noss and Harris 1986).

Describing Landscape Pattern

This review has shown that the pattern of a landscape is important in the way that it changes desirable life conditions for many species. This pattern may be altered by resource management by public agencies and private land owners. Environmental monitoring dictated by federal laws requires federal agencies to use landscape analyses and related techniques to project landscape conditions arising from proposed management activities while evaluating potential land-use conflicts. A remaining question is one of how these agencies evaluate landscape pattern; and after the values are determined, how are decisions altered and how is the land affected? One approach might be to compare visually the satellite imagery of an existing landscape with plots of the same landscape over which proposed activities have been superimposed. More often, however, a quantitative approach is followed using a series of pattern metrics that can describe the landscape before and after the proposed changes.

These pattern metrics serve as indirect measures of landscape pattern, analogous to the use of samples to describe population characteristics. Numerous metrics exist that describe the dimensions of pattern. How does an agency identify which pattern metrics are relevant for a species or for a specific problem? Knowing the landscape pattern needed by a species (e.g., for access to food) is the first of several steps in the selection process. The agency must also consider several criteria of adequacy: Does the metric

behave consistently over a variety of landscape patterns? Can it effectively discriminate the changes in pattern resulting from landscape modification? Upon what input parameters is the metric based, and what, precisely, is expressed? The availability of the data required for metric calculation is another criterion, as are the issues of the cost and the timeliness of data acquisition.

The correct selection of a pattern metric is very important. If the choice of a metric has been made poorly (i.e., when the metric describes a dimension of pattern different than the information desired), the interpretation of initial landscape conditions and projected landscape changes may be erroneous. The validity of the landscape evaluation (as well as the management program outcome) can be directly linked to the choice of the pattern metric.

As an example of how pattern evaluation may influence management outcome, suppose a land agency plans to develop a road system into a forested landscape to facilitate timber harvest. The landscape also supports a group of sensitive species that have a functional relationship to pattern. These species require relatively large expanses of unbroken forest and the spatial connectedness (La Gro 1991) of a forested landscape. The agency wants to determine how the proposed road system will change the existing landscape pattern and indirectly affect this species group. A series of road alternatives are designed, and 2 metrics are selected for the pattern analysis: mean patch size and connectivity. The analysis is completed, and a road system is chosen and implemented. However, the agency finds that these metrics may have failed to detect relevant changes in pattern. The new road system was invasive in nature and did not result in forest subdivision or patch isolation, criteria important in the calculation of patch size and connectivity. Because inappropriate metrics may have been used in the analysis, the evaluation of landscape change was in error, and species habitat may have been negatively affected. What the agency actually needed were measures describing the amount of forest cover away from edge (percent forest interior) and the connectivity of the forest land class irrespective of the connections between patches (forest contiguity). These alternative

metrics detect changes within the integrity of forest cover, regardless of whether forest subdivision or patch formation occurs. If the proper metrics had been implemented, perhaps a different alternative would have been chosen. It is also possible that the selection of the forest interior and contiguity metrics might have resulted in the identification of 2 almost-equal alternatives (i.e., in their degree of influence on pattern), one of which may be less expensive to implement than the other. Either way, the selection of the pattern metrics may directly influence the management outcome.

The above example illustrates how the knowledge of methods used to describe landscape pattern can play an important role in management programs and how such knowledge can influence the ability to manage landscapes. To direct the activities that produce the greatest benefit to the wildlife resource, land agencies must be able to evaluate precisely landscape conditions, to forecast successfully potential landscape modification, and to implement planned management activities. The successful application of landscape analyses will depend on the ability to quantify pattern. It is to this issue that the remaining chapters of this dissertation are directed.

CONCLUSION

This review has suggested that a connection exists between landscape pattern and the conservation of species and their habitats. Land management agencies make decisions affecting millions of hectares which affect the species occurring in these areas. The influence of resource management upon landscapes is profound; management activities directly alter landscape patterns and thus influence the abundance and diversity of species.

Successful resource management and the solutions to many environmental problems require an understanding of how spatial dimensions influence wildlife resources. However, landscape pattern must be identified and quantified in meaningful ways before connections between pattern and ecological process can be made. The expressions of pattern examined in this dissertation may serve as tools towards that goal.

CHAPTER 1

Landscape Pattern Expression Selection

Several words or phrases exist that are expressions of the pattern in the landscape. These expressions are of known or potential relevance to wildlife resources. The expressions that I selected and examined in this dissertation originate from the disciplines of wildlife management, forestry, landscape ecology, geography, and information theory. Their selection was based on their perceived utility in landscape analysis, and their potential for environmental monitoring of landscape modification. The metrics of landscape structure, of which pattern is composed, include the major categories of : (1) spatial heterogeneity, (2) fragmentation, (3) edge characteristics, and (4) connectivity.

Most expressions are applied directly to an entire map from which a single value for each expression is derived. Some expressions are based on the analysis of a spatially-defined neighborhood composed of a 3 x 3 pixel "roving-window". The 9 pixels of each window, including the focal central pixel, are examined for similarities and differences (depending on the specific pattern metric). The result is summarized into a single value characterizing a specific map location described within the context of its neighboring map locations. It is placed into the central pixel of the window. The roving-window is then moved across the map until every pixel has served as the central pixel for comparison with adjacent pixels. The expression of pattern for a landscape then, is the statistical mean of all the values produced by moving the window throughout the map. Values for the metrics of spatial diversity, number of different classes, interspersion, binary comparison matrix, and contiguity were derived using the roving-window procedure.

Table 1 summarizes an extensive selection of landscape pattern expressions arranged into the above-named 4 major categories. Several have been modified or applied differently by various authors. Multiple sources listed in the table indicate such modifications or differences.

Table 1. Landscape pattern expressions selected for examination.

Landscape Expression	Selected Reference
Spatial Heterogeneity	
Shannon Index	Shannon and Weaver 1963
Dominance Index	O'Neill et al. 1988
Number of Different Classes [a]	Murphy 1985
Simpson Index	Simpson 1949, Pielou 1977
Landscape Evenness	Romme 1982, Hoover and Parker 1991
Interspersion [a]	Eastman 1992
Binary Comparison Matrix [a]	Murphy 1985
Spatial Diversity [a]	Robinove et al. 1986
Fragmentation	
Fragmentation Index I	Monmonier 1982
Fragmentation Index II	Ripple et al. 1991
Percent Interior Forest	Levinson 1981, Dunn et al. 1991
Number of Forest Patches	
Mean Patch Density	Ripple et al. 1991
Mean Patch Size	Dunn et al. 1991
Interpatch Distance	Urban and Shugart 1986
Percent Forest Cover	Lauga and Joachim 1992
Edge Characteristics	
Total Forest Edge	Giles 1978, Ranney et al. 1981
Convexity Index	Berry 1991
Patton Index	Patton 1975
Compactness	Eastman 1992
Connectivity	
Connectivity	Forman and Godron 1986
Contiguity Index [a]	La Gro 1991
Spatial Integrity	Berry 1991
Patchiness	Roth 1976

[a] Expression computed using the roving window technique.

Spatial Heterogeneity Expressions

Shannon Index

The diversity index H' is based on information theory (Shannon and Weaver 1963):

$$H' = -\sum_{i=1}^{N} (P_i) \ln (P_i) \qquad (1)$$

where **H'** = Diversity metric;

N = Total number of different land classes;

ln = Natural logarithm; and

$\mathbf{P_i}$ = Area of land class (i) expressed as a proportion of the total landscape area. (Some observers use number of land class types expressed as proportions in subareas of a landscape. See equation No. 8).

Shannon values of zero describe a single-class landscape. Values change with the interplay between number of classes and their proportions. The index increases as the proportions in each land class become more equal and with the addition of new land classes.

Dominance

The dominance index measures the extent to which 1 or more land classes dominate the landscape. Dominance is calculated as the deviation from the maximum possible diversity (**ln N**). The index is an expression of the deviation of the observed diversity from the maximum possible diversity which occurs when the area in each land class is equal (O'Neill et al. 1988). The index is calculated by:

$$D = HMAX - H' \qquad (2)$$

where **D** = Dominance index;

H' = Shannon index (as described above in equation No. 1);

HMAX = ln (N); and

N = Total number of land classes.

Dominance values begin at zero, when the existing land classes are present in equal proportions (i.e., HMAX = H'). Large values occur when a few or less land classes dominate a landscape.

Number of Different Classes

The number of different classes (NDC) is calculated using the roving-window technique (Murphy 1985):

$$\mathbf{NDC} = \left[\sum_{i=1}^{N} \mathbf{NDC_i} \right] / \mathbf{NP} \qquad (3)$$

where **NDC** = Number of different classes;

$\mathbf{NDC_i}$ = Number of different classes for pixel (i); and

NP = Total number of map pixels.

Expression values for each window range from 1 (for the presence of 1 land class) to a maximum of 9 (a different class present within each pixel of the 3 x 3 window). The NDC expression for a landscape is the mean of all values produced by moving the roving-window throughout the map.

Simpson Index

The modified Simpson index of diversity (Simpson 1949; Pielou 1977) is a summation of the squared proportions of land classes present. The expression is determined by:

$$D' = 1 - \sum_{i=1}^{N} \left[a_i (a_i - 1) \right] / \left[A (A - 1) \right] \qquad (4)$$

where **D'** = Modified Simpson index;

A = Total area of land classes;

$\mathbf{a_i}$ = Area of land class (i); and

N = Total number of land classes.

The modified index is unbiased for finite samples. Subtracting the summation from 1.0 allows the expression to increase when the variety of spatial area in different land classes increases. The Simpson index increases with the addition of land classes and as the proportion of different land classes approach an even distribution on the landscape.

Landscape Evenness

The concept of evenness refers to how abundance is distributed among the land classes present. Landscape evenness describes the spatial complexity of a landscape by measuring the relative proportions of areal representation among land classes (Romme 1982; Hoover and Parker 1991):

$$E' = \left[-\ln \sum_{i=1}^{N} P_i^2 \right] / (\ln N) \qquad (5)$$

where **E'** = Evenness metric;

N = Number of different classes present; and

$\mathbf{P_i}$ = Proportion of the total landscape covered by land class (i).

This is a slightly revised form of equation (4) and parallels the relationship between

equation (1) and (2). The evenness index is expected to increase when the proportions of different classes are evenly distributed on a landscape, and to decrease with increasing dominance by a single class. The maximum possible value of this index (as other indices) varies depending on the number and proportions of different classes present.

Interspersion

Interspersion refers to the arrangement or mixture of different land classes. One type of interspersion index can be calculated using the roving-window technique (Eastman 1992):

$$I' = \left[\sum_{i=1}^{N} I_i \right] / NP \qquad (6)$$

where I' = Interspersion index;

I_i = Interspersion value for pixel (i); and

NP = Total number of map pixels.

Each pixel holds an interspersion value representing the number of neighboring pixels having land class designations that differ from the central pixel. Values range from 0 (no difference in any land classes from that of the central pixel) to 8 (the maximum mixture of land classes within the roving-window). The interspersion expression for a landscape is the mean of all values produced by moving the 9-pixel window throughout the map.

Binary Comparison Matrix

The binary comparison matrix characterizes variability within a 3 x 3 pixel neighborhood (Murphy 1985). The index uses both the number of classes and the frequency of occurrence of each class to assign a value to the central pixel of the neighborhood. The index is expressed as follows:

$$BCM = \frac{1}{2}\left[N^2 - \sum_{i=1}^{K} F_i^2 \right] \qquad (7)$$

where **BCM** = Binary comparison matrix;

N = Number of analytical elements (always 9, since 8 are neighbors);

K = Number of classes in the neighborhood (maximum of 8); and

$\mathbf{F_i}$ = Frequency of elements in class (i).

A predominance of neighborhood pixels in a single land class results in a BCM index near zero. Relatively low class frequencies across all classes produces a high index value (>20). The BCM expression for a landscape is the mean of all values produced by moving the roving-window throughout the map.

Spatial Diversity

The spatial diversity index provides a simple measure of complexity at a given location (Robinove et al. 1986). The index is calculated as follows:

$$SD = \left[\sum_{i=1}^{N} NDC_i - NP \right] / N - 1 \qquad (8)$$

where **SD** = Spatial diversity index;

$\mathbf{NDC_i}$ = Number of different classes for pixel (i);

NP = Total number of map pixels; and

N = Total number of land classes.

The spatial diversity expression for a landscape is the sum of all $\mathbf{NDC_i}$ values produced by moving the 9-pixel window throughout the map. Spatial diversity values begin at zero for a 1-class landscape (e.g., total forest) and increases with additional land classes.

Fragmentation Expressions

Fragmentation Index I

One fragmentation index (Monmonier 1982) is determined as follows:

$$\mathbf{FRAG_1 = (M - 1) / (NP - 1)} \qquad (9)$$

where **$FRAG_1$** = Fragmentation index I;

M = Number of distinct landscape regions; and

NP = Total number of map pixels.

A landscape region is defined as a group of contiguous map pixels assigned to the same land class (Monmonier 1982). This is analogous to a polygon in GIS, which is an area of space delineated by a boundary composed of line segments (Burrough 1986). Each region or polygon is a uniquely-defined distinct unit. There will always be at least 1 region. For example, a totally-forested landscape represents 1 region, whereas a forested landscape with 1 cornfield and 2 forest clearings represents 4 regions. Fragmentation values range between zero for a single-class landscape (e.g., a pattern in which all areal units are assigned to a single class) and 1.0 for a highly-complex map (e.g., a pattern in which no 2 contiguous areal units are allocated to the same land class).

Fragmentation Index II

Another fragmentation metric is based on the average distance of forest cover from non-forested areas (e.g., clearings, fields, harvested stands). Forest pixels are assigned values based on their distance to the nearest non-forest area; non-forest pixels are assigned to zero (Ripple et al. 1991). The index is calculated by:

$$\mathbf{FRAG_2} = \left[\sum_{i=1}^{NP} \mathbf{dist_i} \right] / \mathbf{NP} \qquad (10)$$

where **$FRAG_2$** = Fragmentation index II;

$dist_i$ = Distance to non-forest for pixel (i); and

NP = Total number of map pixels.

Large index values reflect a low degree of fragmentation (i.e., long distance to non-forest), while maximum fragmentation occurs when mean values approach zero (Ripple et al. 1991).

Percent Interior Forest

The interior forest expression is the percent of the total forest area remaining after removing a buffer from the edge of each forest tract. Edge buffers are often of considerable width and are established according to species-specific criteria (Ranney et al. 1981; Temple 1986; Wilcove et al. 1986). In this study, a buffer width of 100m (Levenson 1981; Dunn et al. 1991) is used to calculate the expression:

$$\textbf{Interior Forest} = \left[\frac{\textbf{Total area of interior forest}}{\textbf{Total landscape area}} \right] \qquad (11)$$

Number of Forest Patches

An expression for the number of forest patches is computed as follows:

$$\textbf{Patch Count} = \textbf{Total number of forest patches of any type on the landscape} \qquad (12)$$

For purposes of GIS analysis, a forest patch is defined as a group of contiguous map units (horizontal or vertical) of a forest land class. Only orthogonally contiguous pixels were considered to be contiguous.

Mean Patch Density

The density of forest patches is an expression used to reflect the "patchy nature" (or lack thereof) of forest cover on a landscape (Ripple et al. 1991). Patch density is calculated by:

$$\textbf{Patch Density} = \left[\textbf{Patch count / Total forested area}\right] \quad \textbf{(13)}$$

A forest patch is defined as described in Eq. 12. This expression differs from Eq. (12) in that the number of patches on the landscape are expressed on a per area basis. Mean patch density values begin at zero (e.g., the complete absence of forest) and then increase with additional forest patches on the landscape.

Mean Patch Size

The mean size of forest patches, the inverse of (Eq. 13), has been used to describe available resources for a variety of species (Dunn et al. 1991). The metric is computed as:

$$\textbf{Mean Patch Size} = \left[\textbf{Total forested area / Patch count}\right] \quad \textbf{(14)}$$

Values are typically expressed in hectares. Singular land use types other than forest may be similarly defined for both mean patch size and mean patch density.

Mean Interpatch Distance

Mean interpatch distance is an expression used to define patterns of spacing between forest patches. The average distance from the edge of a forest patch to the closest edge of every other forest patch is measured for each patch on the landscape (Urban and Shugart

1986). The expression is calculated as:

$$\mathbf{MID} = \sum_{i=1}^{N} \frac{\text{Distance between patch (i) and patch (i+1)}}{N(N-1)/2} \qquad \textbf{(15)}$$

where **MID** = Mean interpatch distance;

N = Total number of patches on the landscape; and

i = Patch number.

Mean interpatch distance values are typically expressed in meters or kilometers.

Percent Forest Cover

Percent forest cover describes the proportion of the landscape covered by forest (Lauga and Joachim 1992). The expression is calculated as:

$$\mathbf{FC} = \left[F / A \right] 100 \qquad \textbf{(16)}$$

where **FC** = Forest cover;

F = Total forested area in the landscape; and

A = Total area of the landscape.

Forest cover values range between zero (describing a landscape without forest) to 100 (a completely forested landscape).

Edge Characteristics Expressions

Total Forest Edge (17)

Total forest edge is the length of edge that exists at the interface between forest and

other land cover classes (Giles 1978; Ranney et al. 1981). Another land use type can be used as the base criterion.

Convexity Index

The convexity index (Berry 1991) is a perimeter-to-area ratio that describes the amount of edge per unit area of forest. It is computed as:

$$\textbf{Convexity} = \left[\frac{\textbf{Total forest edge length}}{\textbf{Total forest area}} \right] \qquad \textbf{(18)}$$

Convexity values increase with irregularity of forest boundaries and with complexity of forest arrangement.

Patton Index

The Patton index describes the deviation from circularity of a patch or forested area (Patton 1975). It expresses the perimeter-to-area ratio described for equation (18) and is computed as:

$$\mathbf{PI} = \mathbf{P} \;/\; \left[2\ (\mathbf{A}\ \Pi)^{0.5} \right] \qquad \textbf{(19)}$$

where **PI** = Patton index;

P = Perimeter of the forest patch;

A = Area of the forest patch; and

Π = 3.1416.

The index compares the perimeter of a forest patch to the circumference of a circle having an area equal of that of the patch. The Patton index approaches 1.0 with circular patch shapes. For irregular and complex patch shapes, the index increases without limit (Ripple

1991). However, a landscape patch must have a width (e.g., 30 m) to be recognizable. Thus, the index for any stated area does have a limit, once that minimum width is expressed.

Compactness

The compactness expression compares the area of a patch to the perimeter of that patch (Eastman 1992). It is computed as:

$$\mathbf{C} = \left[\mathbf{A_p} \ / \ \mathbf{A_c} \right]^{0.5} \qquad \textbf{(20)}$$

where $\mathbf{C}$ = Compactness ratio;

$\mathbf{A_p}$ = Area of a patch being calculated; and

$\mathbf{A_c}$ = Area of a circle having the same perimeter as that of the patch being calculated.

Compactness ratios range between zero and 1.0. Values near 1.0 describe forest patches approaching circularity. Values less than 1.0 indicate irregularly-shaped patches.

Connectivity Expressions

Connectivity

The connectivity index of Forman and Godron (1986), adapted from classical spatial analyses, expresses the degree to which forest patches on a landscape are connected to one another. The index compares the ratio of the number of existing connections between patches to the theoretical maximum number of patch connections for the same landscape:

$$CI = L / \left[3 (V - 2) \right] \quad (21)$$

where **CI** = Connectivity index;

L = Number of links between forest patches; and

V = Patch count.

For purposes of GIS analysis, a patch link was defined as 1 or more forest pixels linking 2 patches by connecting to them in a NW, NE, SW, or SE direction. Connectivity values range from zero, describing total patch isolation, to 1.0, where every patch is linked to every other patch.

Contiguity Index

The modified contiguity index assesses the spatial connectedness or unbroken adjacency of a forested landscape using the roving-window technique (La Gro 1991). A contiguous forest area consists of a group of forest pixels that touch horizontally, vertically, or diagonally. A new landscape image is created by assigning pixels within the class of interest (e.g., forest) a value of 1.0, and the remaining background pixels a value of zero. The window of 9 pixels is then moved successively over all pixels, 1 pixel at a time. A value of 2.0 is assigned to horizontal and vertical pixels matching the central forest pixel; a value of 1.0 is assigned to diagonal pixels matching the central forest pixel. Finally, the value of the center pixel is assigned the sum of the surrounding 8 pixel values. In essence, this final value is a function of the number and location of forest pixels within the window:

$$CI = \sum_{i=1}^{N} \left[C_i / N - 1 \right] / (M - 1) \quad (22)$$

where **CI** = Contiguity index;

C_j = Contiguity value for forest pixel (i);

N = Total number of forest pixels; and

M = Maximum possible sum of window values which occurs when all window pixels represent the forest land class.

Contiguity values range between 1.0 and 2.0; values approach 2.0 as forest connectedness increases.

Spatial Integrity

The spatial integrity index expresses the spatial "balance" of a landscape by contrasting the number of forest openings with the number of forest patches associated with a landscape. If a forested landscape is punctuated by a number of openings, or is broken into several patches surrounded by non-forest areas, its contiguous character is altered. This character is summarized by the Euler number (Berry 1991):

$$\mathbf{EN} = \mathbf{H} + \left[\mathbf{1} - \mathbf{V} \right] \qquad \textbf{(23)}$$

where **EN** = Euler number;

V = Total patch count; and

H = Number of openings in a forested landscape.

The index is positive when the number of openings exceeds the number of forest patches. The index becomes negative when the number of patches exceeds the number of openings on a landscape. Values near 1.0 indicate landscapes where the number of openings approximates the number of forest patches.

Patchiness

The "patchiness" expression (Roth 1976) describes the distance between habitat

features. The expression is based on the following:

$$\mathbf{P} = \left[100 \ \mathbf{SD} \right] / \ \mathbf{ID} \qquad \textbf{(24)}$$

where: **P** = Patchiness index;

SD = Standard deviation of interpatch distances; and

ID = Mean interpatch distance.

Interpatch distance is defined as described in Eq. 15. This is an application of the classical measure of general statistics, the coefficient of variation (Pielou 1977). Patchiness values begin at 0, for a completely forested landscape without patches, and increase as forest patches become further apart.

Data Requirements

A list of the inputs needed for all of the landscape expressions is presented in Table 2. There are 18 distinct units used for the 24 expressions selected and described above. People with different needs and perspectives have concentrated on the same measures and have used them in different ways or with different emphases. The relationships between these expressions will be explored further in Chapter 2, Classifying and Relating Landscape Expressions.

Table 2. Input parameters for the pattern metrics described in this study.

Area

Area of interior forest (11)[a]
Area of each forest patch (19, 20)
Total area of forest cover (14, 16, 18)
Total area of landscape (4, 11, 13, 16)
Area of each different land class (1, 2, 4, 5)

Distance

Pixel distance to non-forest locations (10)
Edge-to-edge distance between forest patches (15, 24)

Edge Length

Total forest edge length (17, 18)
Edge length of each forest patch (19, 20)

Number

Number of landscape regions (9)
Number of total map pixels (9, 10, 22)
Number of different land classes (1, 2, 5, 8)
Number of forest patches (12, 13, 14, 15, 21, 23)
Number of connections between forest patches (21)
Number of openings within a forested landscape (23)

Roving Window Operations

Number of different neighborhood classes (3, 6, 7, 8)
Number and position of contiguous forest cells (22)
Adjacency matrix (proportion of cells *i* adjacent to cells *j* (7)

[a]Expression equation number.

CHAPTER 2

Classifying and Relating Landscape Pattern Expressions

INTRODUCTION

A growing number of studies suggest that there is a connection between landscape pattern and the persistence of species and their habitats (Ranney et al. 1981; Van Dorp and Opdam 1987). If the relationship between pattern and the ability to manage habitat exists, then it will be very important to be able to understand and describe such patterns.

Knowledge of the expressions used to describe landscape pattern seems likely to become increasingly important. Species occupy landscapes and the pattern of the landscape either supports or inhibits the survival and well-being of species (Golley 1989). Before a connection between landscape pattern and ecological structures or process can be made, pattern must be identified and quantified in meaningful ways. Several groups of expressions exist that are currently used to describe selected aspects of landscape pattern. The major groups include: (1) spatial heterogeneity, (2) fragmentation, (3) edge characteristics, and (4) connectivity. Relationships are expected to exist among the expressions within these groups. However, there have been few studies that have explored the connection, if any, between the different groups of landscape expressions.

Characterizing the relations between pattern metrics into recognizable classes may prove useful in guiding metric selection for landscape analysis. It may also be worthwhile to determine if the structure of the pattern metrics, based upon their overall resemblance to one another, is similar or different from the classification categories used in the literature. Knowledge of such relations may permit the reduction of the number of metrics required for pattern analysis. For example, Franklin and Forman (1987) reported that forest edge increased while patch size decreased within their harvest model. This relation suggests that other relationships may exist among landscape expressions and, most importantly, questions whether those relations continue to persist over a range of different

landscapes.

There have been several published studies that have compared the classic diversity indices (Whittaker 1972; Peet 1974; Pielou 1977; Ludwig and Reynolds 1988). Most of these comparisons were made on theoretical grounds, and few empirical studies were done. Actual empirical trials on a variety of landscape maps may contribute additional understanding of the relations between the different types of landscape metrics.

In this chapter, I compare and contrast landscape expressions for purposes of classifying them and identifying relationships. First, a similarity matrix is presented along with graphical displays to illustrate which expressions are highly correlated, which duplicate landscape pattern information, and which appear to have no discernable relationship to the other expressions. Second, a classification is presented based on an oblique principal components clustering procedure. A dendrogram based upon the results of the cluster analysis is then displayed.

METHODS

Map Selection

The 24 expressions of landscape pattern (see Table 1, Chapter 1) were computed from 73 forested landscape maps (scale: 1:24,000) from the Blacksburg Ranger District of the Jefferson National Forest within the Craig Springs, Glenvar, Looney, McDonald's Mill, and Newport 7.5 minute quadrangles (U.S. Geological Survey, 1987). In order to obtain a variety of different landscape conditions, several subsets of maps were selected from each quadrangle which reflected a broad range of patterns. Additional maps were generated by modifying existing forest patterns, providing a range of "hypothetical" landscape patterns for examination (e.g., from fully-forested landscapes to those with a few, scattered forest patches with varied arrangements of forest cover). Each subset map consisted of a square 3 km x 3 km area of 900 hectares. All maps were restricted to the same size and shape, reducing the influence of these parameters during spatial analysis.

GIS Analysis

The maps were classified into forest and non-forest elements and digitized using Arc-Info (Environmental Systems Research Institute 1987). The resulting vector data were converted into an ERDAS-formatted grid image using the Arc-Info POLYGRID command. The maps were then rasterized using Idrisi (Eastman 1992; Version 4.1, Release 1993) using an image resolution of 30m (0.09 ha/pixel).

Each map was then analyzed for the suite of 24 landscape expressions presented in Table 1. The GIS analysis operations were completed using Idrisi functions (Appendix 1) and a program written in Turbo-Pascal (Appendix 2) developed by the author.

Statistical Analysis

Correlation Matrix. Prior to analysis, the data were standardized to account for differences in measurement units (e.g., km, ha, etc.) and to ensure that each metric was weighted equally. Metric values were converted into standardized form using a Z-transformation (Davis 1986). To investigate relations among the landscape expressions, Pearson product-moment correlation coefficients (SAS 1987) were calculated. The correlation coefficients between each landscape metric and each of the other metrics and the significance probabilities ($P < 0.001$) associated with those coefficients were calculated using PROC CORR.

Scatter plots were drawn for each pairwise comparison of metrics for examining the nature of their relationship (e.g., linear, curvilinear, or none). Regression analysis (based on the sum of least squares method) was used to generate trend lines illustrating the relationship between pairs of expressions. Due to the exploratory nature of this analysis, the trend lines were used solely to provide a useful summary of the data and have no statistical significance. The plotted lines having the best fit (linear, exponential, logarithmic, or power) were selected on a visual basis. Trend lines were omitted when no single line best summarized the data.

Classification. Cluster analysis was used to classify the 24 landscape expressions into distinct groups. The clustering procedure was based on oblique component analysis related to multiple group factor analysis (SAS 1987), a procedure that attempts to reveal a simple underlying structure that is presumed to exist within a set of multivariate observations. The structure is expressed in the pattern of variance and covariance between expressions, and in the similarities between observations (Davis 1986). The analysis is based on a measure of resemblance (e.g., similarity) computed between every pair of expressions; the correlation matrix described above was used for this purpose.

The basic sequence of steps for the VARCLUS cluster procedure were as follows: First, the entire group of variables was placed in a single cluster. A cluster of selected expressions was then chosen for splitting. The selected cluster had the largest eigenvalue associated with the second principal component. The chosen cluster was then split into 2 clusters by finding the first 2 principal components and assigning each expression to the component with which it had the highest squared correlation. The expressions were iteratively reassigned to clusters to maximize the variance accounted for by the cluster components. The procedure stopped when each cluster had only a single eigenvalue greater than 1.0. The levels of similarity were used to construct a dendrogram and represented the proportion of explained variance at which the expressions were assigned cluster membership .

RESULTS AND DISCUSSION

Relationships Between Landscape Pattern Expressions

Spatial Heterogeneity Expressions. Table 3 presents the matrix of correlation coefficients for the spatial heterogeneity metrics. Fig. 1 illustrates selected metric relations. Among the 4 landscape expression groups, the spatial heterogeneity group had the largest number of significant ($P<0.001$) correlations between its members.

Table 3. Correlation coefficients between the expressions of spatial heterogeneity and other pattern groups.

LANDSCAPE EXPRESSION	SPATIAL HETEROGENEITY							
	SHANN	DOMIN	NDC	SIMPSON	EVEN	INTERSP	BCM	SPATDI
SPATIAL HETEROGENEITY								
Shannon Index (SHANN)	1.000	-0.767*	0.897*	0.944*	0.803*	0.868*	0.893*	0.544*
Dominance Index (DOMIN)		1.000	-0.679*	-0.822*	-0.932*	-0.657*	-0.708*	-0.764*
Number of Different Classes (NDC)			1.000	0.890*	0.761*	0.976*	0.986*	0.727*
Simpson Index (SIMPSON)				1.000	0.925*	0.860*	0.898*	0.649*
Landscape Evenness (EVEN)					1.000	0.731*	0.788*	0.756*
Interspersion (INTERSP)						1.000	0.991*	0.726*
Binary Comparison M atrix (BCM)							1.000	0.752*
Spatial Diversity (SPATDIV)								1.000
FRAGMENTATION								
Fragmentation Index Monmonier (FRAG1)	0.504*	-0.333	0.745*	0.526*	0.415	0.707*	0.690*	0.580*
Fragmentation Index Ripple (FRAG2)	-0.326	0.055	-0.371	-0.295	-0.188	-0.301	-0.328	-0.083
Percent Interior Forest (INTERIO)	-0.288	0.003	-0.216	-0.211	-0.095	-0.128	-0.152	0.169
Number of Forest Patches (PATCHES)	0.625*	-0.403	0.742*	0.590*	0.459*	0.662*	0.673*	0.447*
Mean Patch Density (DENSITY)	0.620*	-0.387	0.740*	0.581*	0.441*	0.664*	0.672*	0.435*
Mean Patch Size (SIZE)	-0.142	-0.082	-0.140	-0.116	-0.025	-0.057	-0.084	0.147
Interpatch Distance (DIST)	0.156	-0.116	0.122	0.127	0.093	0.034	0.059	-0.001
Percent Forest Cover (COVER)	0.108	-0.285	0.229	0.182	0.236	0.332	0.303	0.514*
EDGE CHARACTERISTICS								
Total Forest Edge (EDGE)	0.663*	-0.644*	0.833*	0.724*	0.685*	0.864*	0.859*	0.845*
Convexity Index (CONVEX)	0.034	0.302	0.124	-0.050	-0.185	0.034	0.032	-0.221
Patton Index (PATTON)	0.266	-0.384	0.263	0.303	0.365	0.287	0.295	0.435*
Compactness (COMPACT)	-0.318	0.444*	-0.309	-0.362	-0.420	-0.329	-0.342	-0.476*
CONNECTIVITY								
Connectivity Index (CONNECT)	0.228	-0.233	0.414	0.298	0.296	0.413	0.409	0.500*
Contiguity Index (CONTIG)	-0.263	-0.109	-0.359	-0.173	-0.011	-0.287	-0.284	0.036
Spatial Integrity (INTEG)	-0.521*	0.294	-0.586*	-0.485*	-0.350	-0.516*	-0.530*	-0.269
Patchiness (PATCHI)	0.374	-0.251	0.574*	0.436*	0.362	0.587*	0.583*	0.466*

* Significant ($P < 0.001$), df = 71.

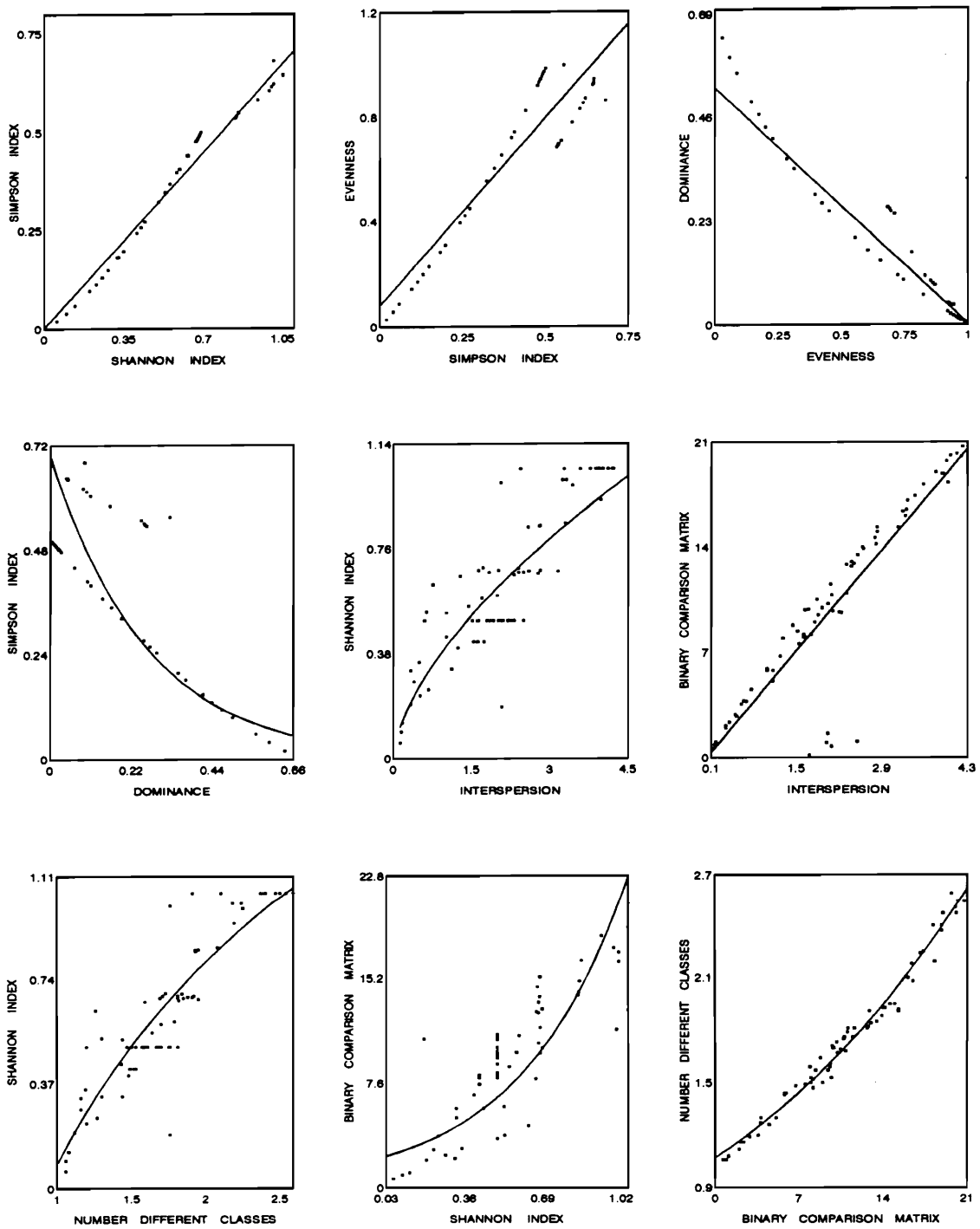


Fig. 1. Relationships between selected spatial heterogeneity metrics. Trend lines are derived from regression analysis.

The correlation analysis revealed that the relationship between the Shannon and Simpson indices was highly correlated (r = 0.944). In fact, these 2 expressions had a large number of relatively high correlations to other members of the heterogeneity group. Shannon was closely related to evenness (r = 0.803) and the roving-window measures of the number of different classes (r = 0.897); binary comparison matrix (r = 0.893); and interspersion (r =0.868). The Simpson index was highly correlated with evenness (r = 0.925) and the 3 roving-window measures binary comparison matrix (r = 0.898); number of different classes (r = 0.890); and interspersion (r = 0.860).

Dominance and evenness were negatively correlated (r = -0.932); the expressions are related inversely (i.e., high values of dominance logically occur with low values of evenness). Conversely, if several land classes are equally distributed across a landscape, evenness would be high and the dominance value would be low. Dominance was the only heterogeneity expression inversely related to each of the other members of this group; in particular with the Simpson index (r = -0.822) and with the Shannon index (r = -0.767). This relationship also seems reasonable: as the number of land classes increases on a landscape (e.g., from 10 to 30 classes), a decrease in the dominance of any 1 land class is expected.

Three of the roving-window expressions were highly correlated. Interspersion was highly related to the number of different classes (r = 0.976) and binary comparison matrix (r = 0.991); in turn, binary comparison matrix was highly correlated with the number of different classes (r = 0.986).

Spatial diversity was also significantly correlated with the other members of the heterogeneity group, but to a lesser degree than those discussed above. The highest negative correlation for spatial diversity was with dominance (r = - 0.764). This expression takes the number of different land classes in a roving-window and divides by the total number of land classes on a landscape minus 1. The procedure used results in a type of averaging, smoothing data, and obscuring localized differences.

Comparing the heterogeneity group with the fragmentation expressions, it appears

that fragmentation index I, number of forest patches, and patch density are somewhat related to spatial heterogeneity. For fragmentation index I, this was expected since as a landscape heterogeneity increases, the number of distinct map regions or polygons also increases (which is the numerator for this expression). As the number of forest patches and patch density increase, pattern complexity also increases. Within the heterogeneity group, number of different classes was the expression most highly correlated to fragmentation index I ($r = 0.745$); number of forest patches ($r = 0.742$); and patch density ($r = 0.740$).

Interspersion was correlated with the fragmentation index I ($r = 0.707$); number of forest patches ($r = 0.662$); and patch density ($r = 0.664$). Binary comparison matrix was also related to the 3 fragmentation expressions ($r = 0.707$, $r = 0.673$, and $r = 0.672$, respectively).

Many positive relationships were expected to exist between the spatial heterogeneity and edge expression groups (i.e., an increase in landscape heterogeneity would be accompanied by an increase in edge length). However, total forest edge was the only edge expression demonstrating such a relationship and was significantly correlated with each heterogeneity member. Edge had a stronger relationship to number of different classes, binary comparison matrix, and interspersion (each of which reflects arrangement of land classes), than with the remaining heterogeneity expressions. Forest edge length appears to be a potentially useful expression of pattern complexity. None of the remaining edge expressions demonstrated either a significant positive or negative linear relationship to landscape heterogeneity.

Fragmentation Expressions. Table 4 presents the matrix of correlation coefficients for the fragmentation metrics. Fig. 2 depicts selected metric relations. The relatively low positive and negative values between many of the fragmentation expressions suggest that few of these expressions are linearly related, (i.e., they are measuring different aspects of forest fragmentation, or there are strong non-linear relations).

Table 4. Correlation coefficients between the expressions of fragmentation and other pattern groups.

LANDSCAPE EXPRESSION	FRAGMENTATION							
	FRAG1	FRAG2	INTERIO	PATCHES	DENSITY	SIZE	DIST	COVER
SPATIAL HETEROGENEITY								
Shannon Index (SHANN)	0.504*	-0.326	-0.288	0.625*	0.620*	-0.142	0.156	0.108
Dominance Index (DOMIN)	-0.333	0.055	0.003	-0.404	-0.387	-0.082	-0.116	-0.285
Number of Different Classes (NDC)	0.745*	-0.371	-0.216	0.742*	0.740*	-0.140	0.122	0.229
Simpson Index (SIMPSON)	0.526*	-0.295	-0.211	0.590*	0.581*	-0.116	0.127	0.182
Landscape Evenness (EVEN)	0.415	-0.188	-0.095	0.459*	0.441*	-0.025	0.093	0.236
Interspersion (INTERSP)	0.707*	-0.301	-0.128	0.662*	0.664*	-0.057	0.034	0.332
Binary Comparison Matrix (BCM)	0.690*	-0.371	-0.152	0.673*	0.672*	-0.084	0.059	0.303
Spatial Diversity (SPATDIV)	0.580*	-0.083	0.169	0.447*	0.435*	0.147	-0.001	0.514*
FRAGMENTATION								
Fragmentation Index Monmonier (FRAG1)	1.000	-0.288	-0.095	0.769*	0.778*	-0.252	0.204	0.224
Fragmentation Index Ripple (FRAG2)		1.000	0.880*	-0.434*	-0.440*	0.662*	-0.420	0.642*
Percent Interior Forest (INTERIO)			1.000	-0.402	-0.406	0.713*	-0.530*	0.872*
Number of Forest Patches (PATCHES)				1.000	0.994*	-0.478*	0.511*	-0.124
Mean Patch Density (DENSITY)					1.000	-0.480*	0.515*	-0.126
Mean Patch Size (SIZE)						1.000	-0.601*	0.674*
Interpatch Distance (DIST)							1.000	-0.490*
Percent Forest Cover (COVER)								1.000
EDGE CHARACTERISTICS								
Total Forest Edge (EDGE)	0.728*	0.034	0.289	0.524*	0.519*	0.198	-0.107	0.668*
Convexity Index (CONVEX)	0.210	-0.732*	-0.771*	0.429	0.432[d]	-0.634*	0.444*	-0.710*
Patton Index (PATTON)	0.052	0.037	0.137	-0.127	-0.139	0.479*	-0.280	0.317
Compactness (COMPACT)	-0.042	-0.094	-0.213	0.145	0.160	-0.564	0.340	-0.401
CONNECTIVITY								
Connectivity Index (CONNECT)	0.665*	-0.126	0.097	0.356	0.364	-0.202	0.009	0.299
Contiguity Index (CONTIG)	-0.318	0.804*	0.814*	-0.561*	-0.569*	0.627*	-0.427	0.637*
Spatial Integrity (INTEG)	-0.602*	0.381	0.391	-0.919*	-0.923*	0.528*	-0.524*	0.205
Patchiness (PATCHI)	0.671*	-0.324	-0.110	0.579*	0.591*	-0.418	0.299	0.161

* Significant ($P < 0.001$), df = 71.

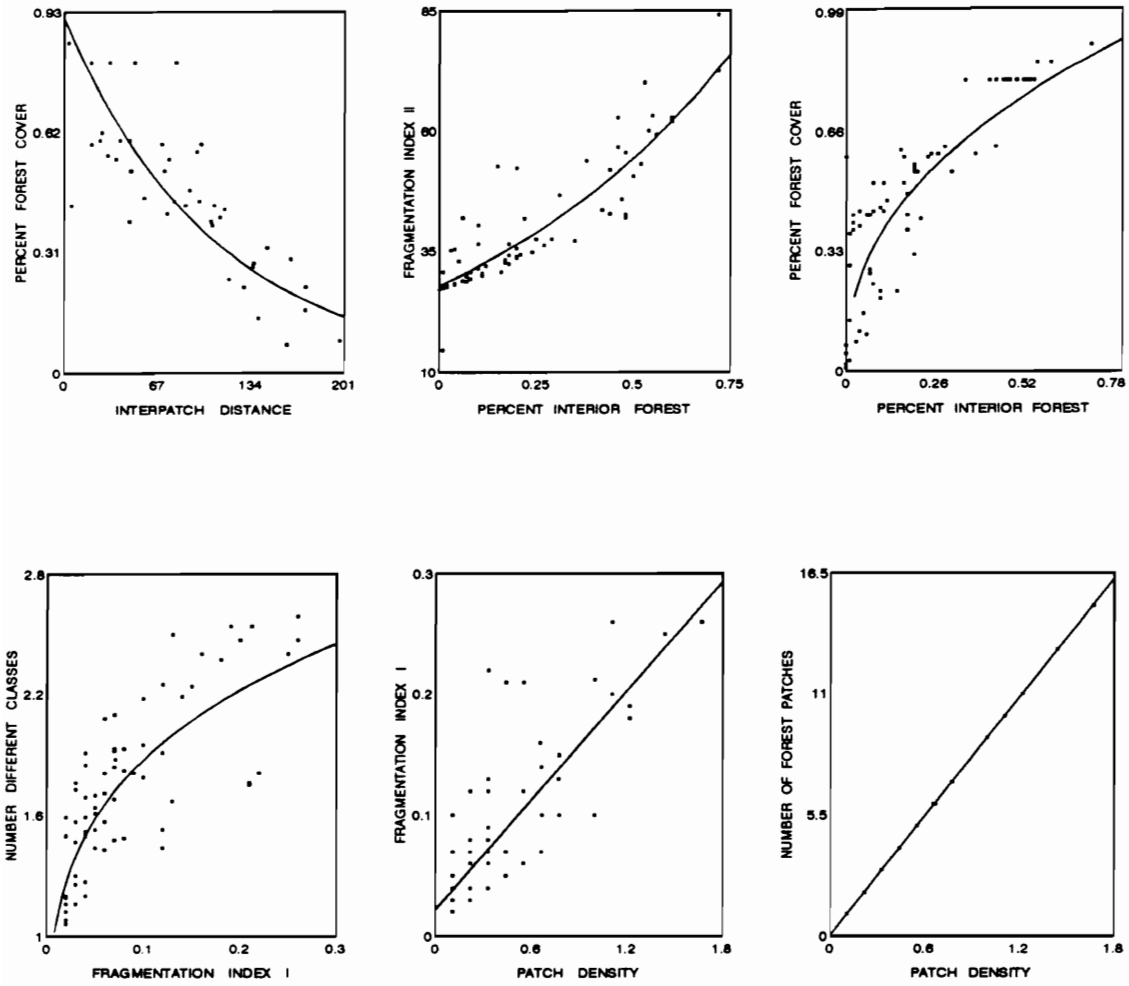


Fig. 2. Relationships between selected fragmentation metrics. Trend lines are derived from regression analysis.

The number of forest patches and patch density were highly correlated. Patch density is a simple adaptation of the patch count; it is computed by dividing the number of patches by the total forest area. This fact, along with the correlation coefficient between these 2 expressions ($r = 0.994$), suggests these expressions are redundant. Fragmentation index I was positively related to both of these expressions ($r = 0.769$, $r = 0.778$, respectively).

Fragmentation index II was strongly related to interior forest ($r = 0.880$). One is a linear measure, the other is an area measure. The fragmentation index represents the average distance on the landscape to non-forest pixels, while interior forest values represent the area remaining after an edge buffer (or zone of potential non-forest influence) is removed from total forest cover. The edge buffer is determined by the distance from forest to non-forest pixels. Fragmentation index II was linearly related to percent forest cover ($r = 0.642$) to a lesser degree than that observed for forest interior. For example, several landscapes may each have 80% forest cover but with differing arrangements of that cover; the fragmentation index value varies in response to both the amount of cover and its arrangement.

There was a strong linear relationship between percent interior forest and forest cover ($r = 0.872$). The variability at the lower end of the trend line in Fig. 2 reflects the influence of different arrangements of forest cover. The absence of a negative relationship between interior forest and number of patches was initially surprising; an increase in patch number did not always reflect a decline in percent interior forest. There was a significant interplay with patch size. In several landscapes, the influence of a large number of patches was offset by a few, larger patches which contained a majority of interior forest.

A strong relationship between interpatch distance and forest cover was not evident ($r = -0.490$). Many landscapes with dwindling forest cover may exist as 1 forest patch, however irregular in shape. Interpatch distance is zero in these cases. The singular patch is a general limitation to patch metrics now reported. These metrics equal zero unless 2 or more patches are present, regardless of the existing landscape pattern. To counter this

problem, a correlation coefficient was recomputed using only those maps with 2 or more patches. The resulting correlation coefficient rose dramatically (r = -0.815). It seems likely that as forest cover recedes on most patchy landscapes, interpatch distance increases.

Mean patch size illustrates several interesting relationships with the other expressions (Fig. 3). Patch size was slightly correlated to fragmentation index II (r = 0.662). This was somewhat expected; as a patch becomes larger, the mean distance between forest and non-forest pixels becomes greater. However, both the shape and size of the forest patch will influence fragmentation index II values (e.g., the more compact a patch, the greater the distance to non-forest). The role of patch shape may explain the variability displayed in the plot of these 2 expressions as well as those in the plot of mean patch size and percent interior forest (r = 0.713).

There was no strong negative relationship between number of patches and mean patch size (r = -0.478). A small average patch size did not necessarily reflect a landscape with numerous patches, but was influenced by the amount of forest cover present.

A moderate relationship was observed between patch size and interpatch distance (r = - 0.601). When a correlation coefficient was recomputed between patch size and interpatch distance using only those maps with 2 or more patches, the correlation rose slightly (r = -0.709). As patch size continues to increase, the interpatch distance becomes reduced.

Another notable relationship was the correlation between patch size and forest cover (r = 0.674). One would expect that the more forest cover a landscape has, the larger the mean patch size becomes. However, both the distribution and amount of forest cover influenced patch size. For example, a 70% forested landscape might comprise numerous, small patches or a few, large patches. As a landscape becomes more forested, it seems logical that patch size would approach the size of the total area.

Edge Expressions. The matrix of correlation coefficients for the edge metrics

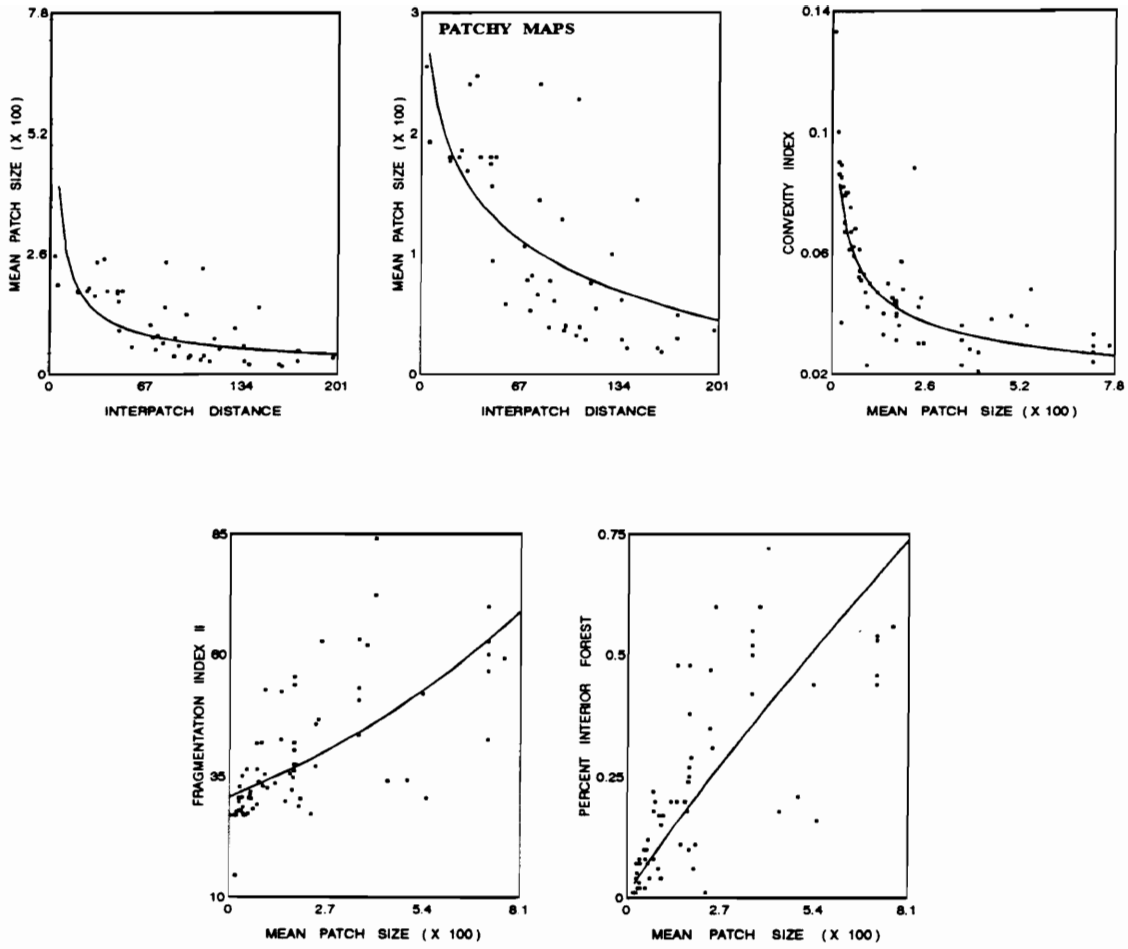


Fig. 3. Relationships between patch size and selected fragmentation metrics. Trend lines are derived from regression analysis.

is presented in Table 5. The relatively low positive and negative values between the majority of the edge expressions, (with the exception of Patton's index and compactness), suggest that these expressions are measuring different aspects of edge characteristics or exhibit non-linear relations (Fig. 4).

Patton's index and compactness were highly correlated. The correlation coefficient between these 2 edge expressions (r = -0.943) indicates these expressions have a close, inverse relationship. Both are primarily expressions of forest patch perimeter relative to the area of a circle having the same perimeter as that of the forest patch being measured. A perfectly circular patch has a Patton index and compactness value of 1.0. As patch shape becomes increasingly irregular, Patton values surpass 1.0 while compactness approaches zero (Fig. 4). Also noteworthy is that neither Patton's index nor compactness were significantly related to total forest edge itself.

Positive relationships were expected to exist between the edge and fragmentation expression groups (i.e., an increase in landscape fragmentation would be accompanied by an increase in edge). Total forest edge demonstrated 3 such relationships with the fragmentation expression group (Fig. 4). Edge was correlated with the fragmentation index I (r = 0.728); as the number of map regions (fragmentation index I) on a landscape increase, the edge length also increases. Edge was also positively correlated with number of forest patches (r = 0.524) and patch density (r = 0.519). There were no other positive correlations between the edge and fragmentation groups. The lack of relationship between edge and the fragmentation expressions of forest interior (r = 0.289), fragmentation index II (r = 0.034), and convexity (r = -0.263) were unexpected.

In contrast, the edge expression convexity was inversely related to 2 fragmentation expressions: forest interior (r = -0.771) and fragmentation index II (r = -0.732). It is noteworthy that each of these 3 expressions involve both edge length and forest area in their formulations (I shall return to that topic later). The inverse relationship between convexity and forest cover (r = -0.709) suggests that as forest cover increases, this edge-to-area expression continues to decrease. Notice the pattern of variability for the

Table 5. Correlation coefficients between the expressions of edge and connectivity and other pattern groups.

LANDSCAPE EXPRESSION	EDGE				CONNECTIVITY			
	EDGE	CONVEX	PATTON	COMPACT	CONNECT	CONTIG	INTEG	PATCHI
SPATIAL HETEROGENEITY								
Shannon Index (SHANN)	0.663*	0.034	0.266	-0.318	0.228	-0.263	-0.521*	0.374
Dominance Index (DOMIN)	-0.644*	0.302	-0.384	0.445*	-0.233	-0.109	0.294	-0.251
Number of Different Classes (NDC)	0.833*	0.124	0.263	-0.309	0.414	-0.359	-0.586*	0.574*
Simpson Index (SIMPSON)	0.724*	-0.050	0.303	-0.362	0.298	-0.173	-0.485*	0.436*
Landscape Evenness (EVEN)	0.685*	-0.185	0.365	-0.420	0.296	-0.011	-0.350	0.362
Interspersion (INTERSP)	0.864*	0.034	0.287	-0.329	0.413	-0.287	-0.516*	0.587*
Binary Comparison Matrix (BCM)	0.859*	0.032	0.295	-0.342	0.409	-0.284	-0.530*	0.583*
Spatial Diversity (SPATDIV)	0.845*	-0.221	0.435	-0.476*	0.500*	0.036	-0.269	0.467*
FRAGMENTATION								
Fragmentation Index Monmonier (FRAG1)	0.728*	0.210	0.052	-0.042	0.665*	-0.317	-0.602*	0.671*
Fragmentation Index Ripple (FRAG2)	0.034	-0.732*	0.037	-0.094	-0.126	0.804*	0.381	-0.324
Percent Interior Forest (INTERIO)	0.289	-0.771*	0.137	-0.213	0.097	0.814*	0.391	-0.110
Number of Forest Patches (PATCHES)	0.524*	0.429	-0.127	0.145	0.356	-0.561*	-0.919*	0.579*
Mean Patch Density (DENSITY)	0.519*	0.432*	-0.139	0.160	0.364	-0.569*	-0.923*	0.591*
Mean Patch Size (SIZE)	0.198	-0.634*	0.479*	-0.564*	-0.202	0.627*	0.528*	-0.418
Interpatch Distance (DIST)	-0.107	0.444*	-0.280	0.340	0.009	-0.427	-0.524*	0.299
Percent Forest Cover (COVER)	0.668*	-0.709*	0.317	-0.401	0.299	0.637*	0.205	0.161
EDGE CHARACTERISTICS								
Total Forest Edge (EDGE)	1.000	-0.263	0.401	-0.467*	0.523*	0.074	-0.352	0.577*
Convexity Index (CONVEX)		1.000	-0.319	0.409	-0.032	-0.906*	-0.418	0.140
Patton Index (PATTON)			1.000	-0.943*	0.067	0.183	0.325	-0.110
Compactness (COMPACT)				1.000	-0.053	-0.276	-0.335	0.109
CONNECTIVITY								
Connectivity Index (CONNECT)	0.523*	-0.032	0.067	-0.053	1.000	-0.037	-0.222	0.584*
Contiguity Index (CONTIG)	0.074	-0.906*	0.183	-0.276		1.000	0.536*	-0.262
Spatial Integrity (INTEG)	-0.352	-0.418	0.325	-0.335			1.000	-0.523*
Patchiness (PATCHI)	0.577*	0.140	-0.110	0.109				1.000

* Significant ($P < 0.001$), df = 71.

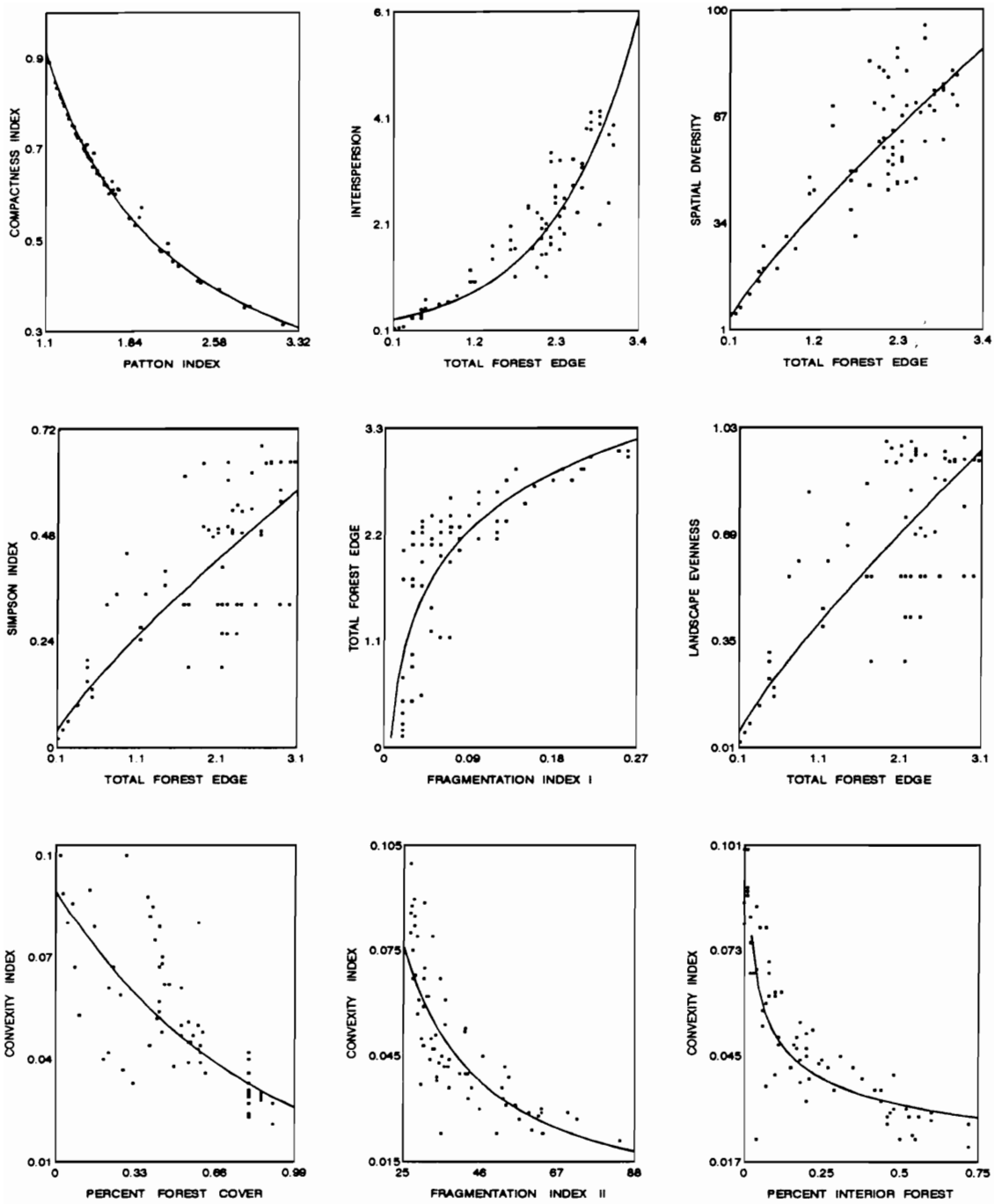


Fig. 4. Relationships between selected edge metrics. Trend lines are derived from regression analysis.

convexity values in Fig. 4. High variability is observed up to 60% of forest cover and then lessens as 100% cover is approached. This can be readily explained by imagining the landscape as a checkerboard whose "squares" are progressively filled in by forest. If a filled-in square is isolated from the other squares, or touches only diagonally to another square, the amount of edge increases along with the increase in cover. However, if a square is filled-in directly adjacent to others, the amount of edge may increase slightly, remain the same, or actually decrease. As the checkerboard becomes increasingly filled-in, a point is reached where the amount of existing edge consistently decreases along with the increase in forest cover.

A few inverse relationships were expected between the edge and connectivity expression groups (i.e., as edge increases, connectivity decreases). However, only one such relationship existed; this occurred between convexity and contiguity ($r = -0.906$).

Connectivity Expressions. The matrix of correlation coefficients for the connectivity expressions is also presented in Table 5. The absence of relatively high correlation values between any of the members of this group suggests that none of these expressions are linearly related. In fact, patchiness showed no discernable linear relationship to any other expression, regardless of the group.

The connectivity and fragmentation expression groups express contrasting aspects of landscape pattern; it seems logical to expect inverse relationships to exist between these 2 groups. Only 1 connectivity metric, spatial integrity, exhibited such a relation with 2 fragmentation metrics (Fig. 5). Spatial integrity displayed a strong negative relationship with number of forest patches ($r = -0.919$) and patch density ($r = -0.923$). As the number of forest patches on a landscape increases, the first term in the spatial integrity formula increases in a negative direction (e.g., 10 patches on a landscape is reflected by -10 in the formula). Patch density, as a transform of the number of forest patches, is related to spatial integrity in the same way.

In contrast, Fig. 5 presents a number of positive relations between forest contiguity

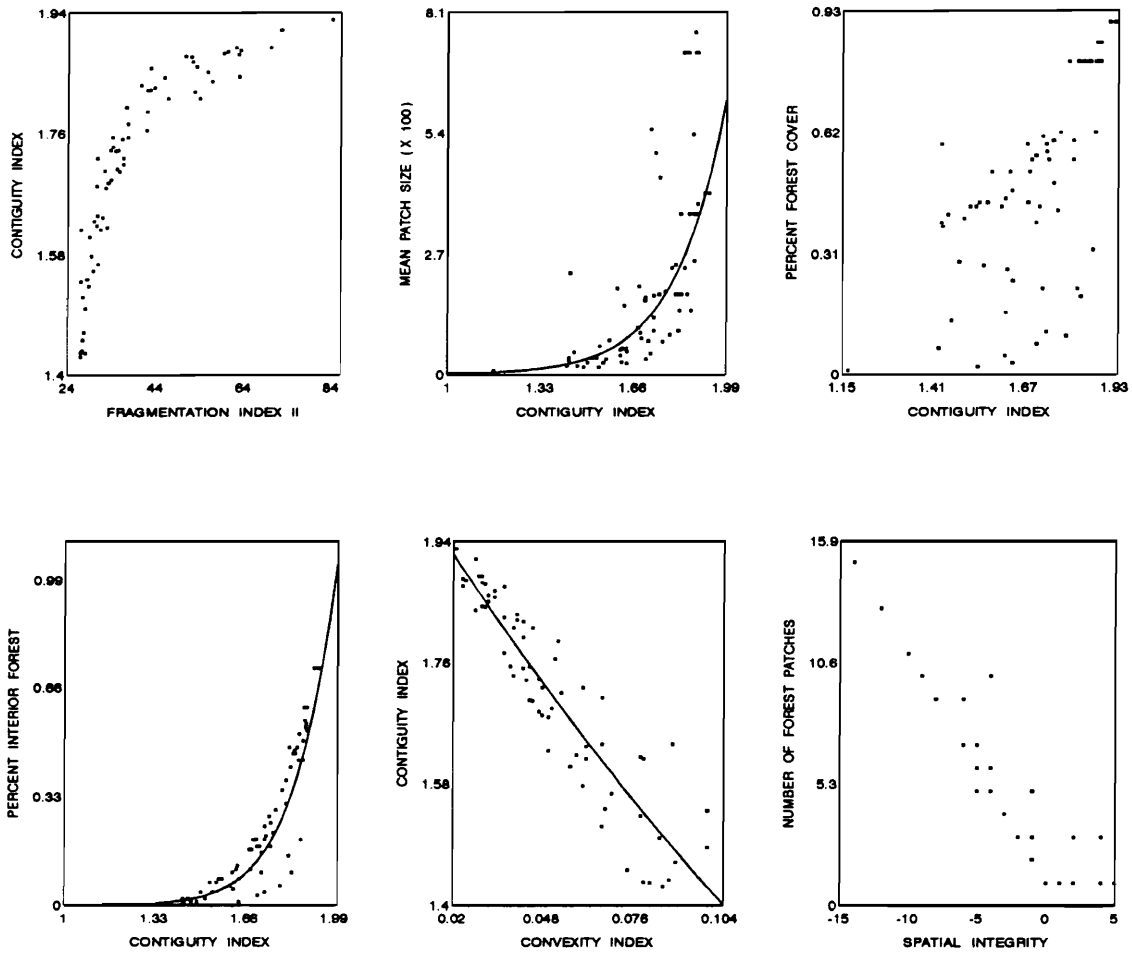


Fig. 5. Relationships between selected connectivity metrics. Trend lines are derived from regression analysis.

and selected fragmentation metrics. Contiguity was positively related to fragmentation index II (r = 0.804); as contiguity within a forested landscape increased, the mean distance to non-forest pixels (fragmentation index II) also increased. A similar relationship existed between contiguity and interior forest (r = 0.814). Contiguity was slightly correlated with mean patch size (r = 0.627) and percent forest cover (r = 0.637), relations influenced by the distribution of forest cover.

Classification of Landscape Pattern Expressions

The classification of pattern metrics produced 6 clusters of related metrics (Table 6). The dendrogram (Fig. 6) illustrates the level of clustering and the resulting expression groups. The expressions were divided into a set of hierarchical clusters, organized so that 1 cluster was entirely contained within another. The pattern metrics were assigned cluster membership based on levels of similarity, representing the proportion of explained variance. The edge, fragmentation, and connectivity groups were not combined into like clusters; however, all of the spatial heterogeneity expressions were grouped together. Total forest edge was classified with the heterogeneity group; as discussed previously, this expression had many significant correlations to members of that group. Another edge expression, convexity, was grouped with fragmentation expressions. Of the remaining edge expressions, only Patton's index and compactness were clustered together. Two of the connectivity expressions, contiguity and spatial integrity, were classified with fragmentation expressions. Connectivity and patchiness were the only connectivity expressions clustered together.

Perhaps the natural structure and behavioral properties of these landscape expressions are quite different from the classification categories used in the literature. Exploring this idea, the following characteristics were noted within the 6 metric clusters:

Cluster 1. The members of this cluster, interpatch distance and mean patch size,

Table 6. Cluster membership of the pattern metrics based upon oblique principal components. Metrics were assigned to clusters based on levels of similarity.

Variable	R^2 Own Cluster[a]	R^2 Next Closest[b]	1 - R^2 Ratio[c]
Cluster 1			
Interpatch Distance	0.801	0.277	0.276
Mean Patch Size	0.801	0.533	0.426
Cluster 2			
Percent Forest Cover	0.720	0.423	0.486
Convexity	0.827	0.363	0.272
Fragmentation Index II	0.804	0.366	0.308
Percent Interior Forest	0.915	0.483	0.164
Contiguity	0.847	0.347	0.234
Cluster 3			
Patton Index	0.971	0.180	0.035
Compactness	0.971	0.255	0.038
Cluster 4			
Patchiness	0.741	0:331	0.387
Connectivity	0.736	0.163	0.315
Fragmentation Index I	0.804	0.534	0.421
Cluster 5			
Patch Density	0.982	0.443	0.033
Number of Forest Patches	0.979	0.428	0.037
Spatial Integrity	0.930	0.346	0.106
Cluster 6			
Shannon Index	0.824	0.360	0.275
Dominance	0.722	0.177	0.338
Simpson Index	0.896	0.317	0.151
Evenness	0.813	0.181	0.228
Binary Comparison Matrix	0.936	0.416	0.109
Interspersion	0.890	0.429	0.192
Number of Different Classes	0.908	0.494	0.182
Spatial Diversity	0.676	0.351	0.500
Forest Edge	0.757	0.491	0.478

[a] The squared correlation of the metric with its own cluster.
[b] The next highest squared correlation between the metric and another cluster.
[c] The ratio of (1.0 - R^2 Own Cluster) / (1.0 - R^2 Next Closest). The smaller the ratio, the better the cluster separation.

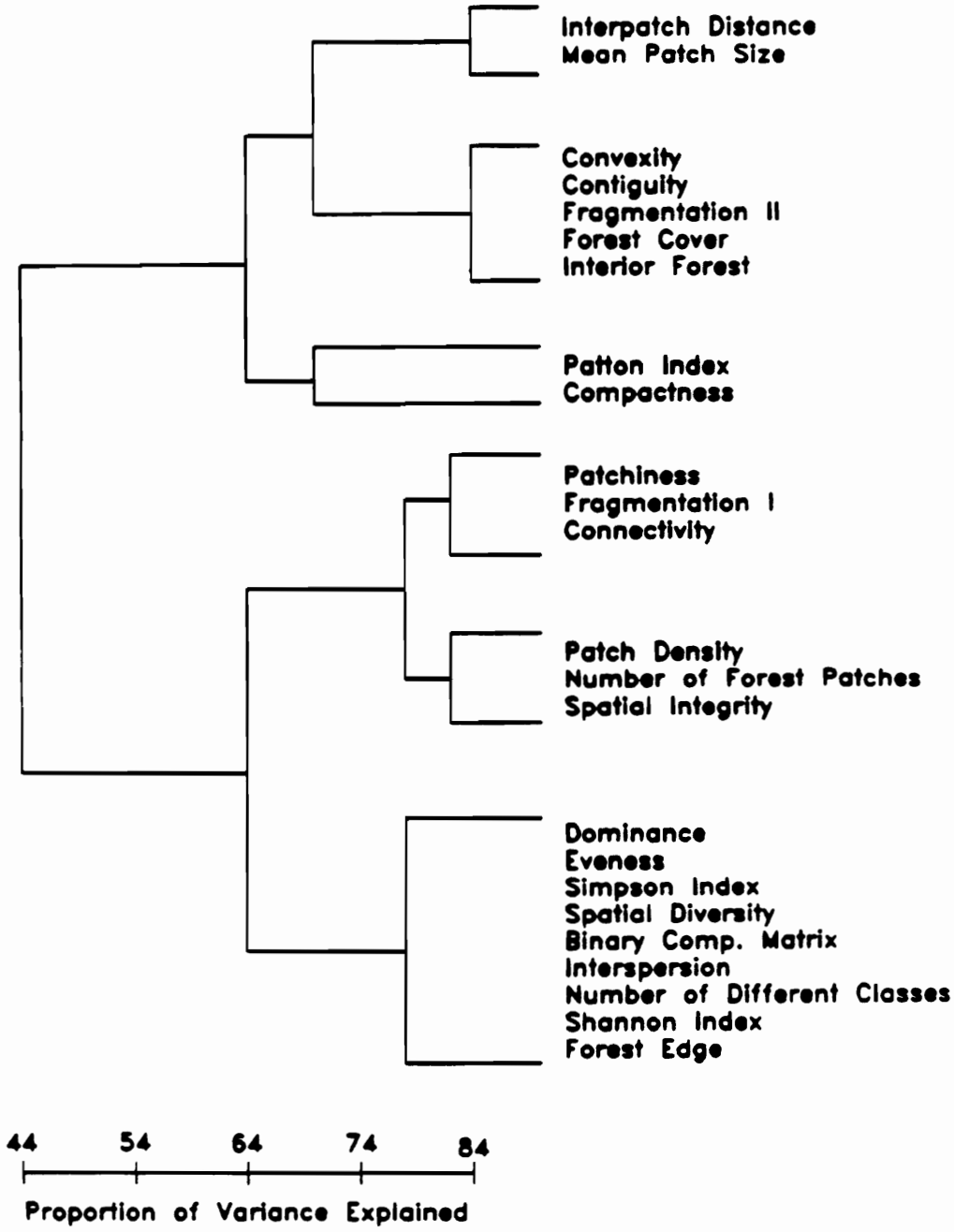


Fig. 6. A classification of 24 landscape expressions by the oblique principal components clustering technique.

are influenced by the amount of forest cover. The less forest cover there is in an area, the more likely that patches will be farther apart and will consist of a relatively small patch size. However, as discussed earlier, patch distribution and shape will influence these metricss. Patch size and interpatch distance use patch radius in their respective formulae (e.g., an increase in patch radius translates into a decrease in mean interpatch distance). This relationship will also be affected by irregular patch shapes and patch distribution.

Cluster 2. The members of this cluster are from the fragmentation, edge, and connectivity groups. They are moderately correlated with one another. Each member reflects the amount of forest cover on a landscape. In addition, the expressions for convexity, fragmentation index II, percent interior forest, and contiguity are influenced by the arrangement or distribution of that forest cover.

Cluster 3. The 2 members of this cluster, the Patton and compactness indices, both relate the amount of forest edge and area in a landscape to that of a circle. In that manner, both actually could be classified as shape measures. This cluster grouping indicates the Patton index is more related to shape than to either heterogeneity or edge characteristics.

Cluster 4. The members of this group, patchiness, connectivity, and fragmentation index I, were only slightly correlated. This group is indirectly influenced by the number of forest patches. What ties them together is the influence of patch position relative to connectedness. Patchiness expresses lack of connectedness while connectivity reflects the connectedness of forest patches. Fragmentation index I is related to patch position in the following way: if patches are positioned apart, the index increases. If patches are positioned side by side (i.e., they touch in any way), the index decreases because these patches are considered as 1 map region (the numerator term in fragmentation index I).

Cluster 5. The 3 members of this cluster were also highly correlated. The 2

fragmentation expressions, number of forest patches and patch density, along with the connectivity expression spatial integrity, all are influenced by the number of forest patches.

Cluster 6. The members of this cluster are all highly correlated with one another. However, although relationships exist, some of the expressions provide different types of information. Each member directly or indirectly expresses the number of different land classes on a landscape. Interspersion and edge reflect this indirectly; as the number of classes increases, the edge between these classes increases and the more interspersed the landscape becomes. The Shannon, Simpson, binary comparison matrix, dominance, and evenness expressions are also based on the proportions of land classes. The arrangement or distribution of land classes across a landscape, in addition to the number of land classes, are reflected in the spatial diversity, number of different classes, interspersion, and edge expressions.

Associations Between Landscape Pattern Expressions

The findings of this chapter can be highlighted by 2 recurring themes. The first is that one cannot assume the relationships between specific metrics and among the pattern metric groups. The second theme is that if relations do exist, landscape factors can influence them.

A manager selecting a metric for analysis may benefit by examining metric relationships. For example, suppose a manager needs a metric that reflects strongly the aspects of edge that is relevant to a particular species. If the manager assumes all edge metrics are comparable, the evaluation may be misleading. With few exceptions, the lack of strong relationships within the edge, fragmentation, or connectivity groups suggests that these metrics express different dimensions of pattern. The associations that did exist within these groups were metric-specific. The strong relationships observed among all of the spatial heterogeneity metrics proved to be the exception, rather than the rule. The

selection of an expression should be based on whether the metric provides the desired information, rather than on assumed relationships.

When a strong relation between 2 metrics exists, one must examine whether applying both metrics results in duplication of information. The Patton and compactness indices provide equivalent pattern detail; the application of both metrics may lead to double-counting in weighted analyses. One can apply knowledge about metric relations to reduce the number of expressions required, to limit the risk of introducing intercorrelated variables, and to restrict the analysis to metrics providing unique information.

The assumption of relations among groups of pattern metrics may be misleading. For example, the connectivity and fragmentation metric groups appear to represent contrasting dimensions of landscape pattern. A manager may omit the connectivity expressions from analysis on the assumption that this group simply expresses opposite information to that provided by the fragmentation group. In this scenario, important pattern detail may be lost as there is an absence of strong inverse relations between the metrics of these 2 groups. Similarly, there was an absence of relations between the edge and fragmentation metric groups. There was a positive association, however, between the spatial heterogeneity group and selected fragmentation metrics during certain conditions (e.g., when the landscape consisted of a series of numerous forest patches, pattern complexity increased).

The results of the cluster classification clarifies the associations between the individual metrics and the pattern groups. The coupling of presumably different pattern metrics suggests there is a bridging of pattern information between the groups from which they originate. This relation appears based upon shared input parameters and the pattern detail that is expressed by the metrics. Cluster membership reflected 7 dimensions of pattern. During metric selection, a manager may profit by concentrating on these dimensions of pattern, rather than the nominal categories used in the literature.

Finally, the findings of this chapter suggest that landscape analysis may benefit by examining the factors that influence metric relationships. For example, the relation

between forest interior and forest cover was influenced by the arrangement of forest cover. The shape of a forest patch influenced the relationship between mean patch size and the measured distance to non-forest as well as the relationship between patch size and interior forest. The correlation between the number of patches and patch size was influenced by the distribution and amount of forest cover present. Caution is advised when making assumptions concerning the constancy of expected relationships.

SUMMARY

Several expressions of landscape pattern were compared. Some of the expressions within a particular group had direct correlates; others were not correlated. The expressions within spatial heterogeneity group proved to be the most highly correlated. The relatively low correlation coefficients within the fragmentation, edge, and connectivity groups suggested a lack of linear relations within those groups.

Relationships between expression groups were not simple or straightforward. Positive linear relationships were expected between the spatial heterogeneity and edge expression groups. However, total forest edge was the only edge metric demonstrating such a relationship. Positive relationships were also expected between the edge and fragmentation metric groups; however, this occurred infrequently. Inverse relationships were expected to occur between the connectivity and fragmentation expression groups; only 1 connectivity expression, spatial integrity, demonstrated such a relationship.

Cluster analysis displayed complex relationships among the landscape expressions. The classification based on metric behavior appeared different from the categories used in the literature. Cluster membership reflected the number, proportion, and arrangement of land classes; the number of forest patches; patch and radius; and edge length.

The behavior of the pattern metrics will be explored further in Chapter 3, The Influence of Landscape Modification on the Pattern of Land Classes.

CHAPTER 3

The Influence of Modification on the Pattern of Land Classes

INTRODUCTION

One of the most significant challenges to present-day wildlife management is landscape modification. This is not a new phenomenon; since the sixteenth century, native inhabitants of the United States have traditionally practiced land modification, including burning and clearing, creating anthropogenic landscapes (Hammett 1992). Some of these landscapes are being intensively managed. Anthropogenic disturbances have become so extensive that their influence is evaluated on a landscape scale for capturing the cumulative effects of site-specific activities on wildlife (Flather and King 1992).

There are many forms of landscape modification, such as those caused by fire, flood, mineral extraction, channelization, and structural development. Many are less noticeable, such as timber harvest, road construction, and deforestation, but may have far-reaching effects. Ecological consequences can differ depending on the pattern imposed on a landscape by land-use activities (Franklin and Forman 1987). These activities alter spatial pattern and the habitat needed by many species for life needs. These changes may be beneficial for some species, harmful to others. Pattern change is often accompanied by changes in species composition and the persistence of those species dependent on that habitat.

Wildlife management and forest policy decisions are increasingly being made at the landscape level. Much of the motivation for the landscape context of resource planning comes from federal laws requiring resource agencies to examine the potential impacts from land management on wildlife resources. Concern for wildlife was a major impetus of the National Forest Management Act (1976), requiring maintenance of a diversity of plant and animal communities while meeting multiple-use objectives. The National

Environmental Policy Act (1969), the first legislation to address the analysis of cumulative effects, dictated the analysis of environmental impacts associated with federal projects. The Endangered Species Act (1973) prescribed land management analyses on critical habitat for endangered, threatened, and sensitive species and protection for the ecosystems encompassing those habitats. The environmental monitoring dictated by these laws could be enhanced by the inclusion of landscape pattern analysis. If there is a connection between landscape modification and the persistence of species, then the influence of land management activities may be profound. It appears to many that knowledge of the relationship among landscape modification, pattern, and wildlife should become an integral part of effective forest management for minimizing potential impacts and enhancing the entire complement of wildlife species, their habitats, and resource use.

The landscape context of management planning makes examining potential methods for evaluating modification increasingly important. With the advent of GIS technology, it is possible to simulate placement of a proposed management modification on a landscape map and examine the effect on selected aspects of landscape pattern. Implicit within my work is that resource benefits are a function of such pattern. Cartographic models have been used for a variety of purposes, from exploring natural processes of landscape modification to evaluating proposed management activities (Schultz and Joyce 1992; Stoms 1992). Cartographic models generalize spatial relationships using the location and configuration of landscape elements. Proposed changes are projected and change in landscape pattern determined. Quantitative measures of effects can be seen in lieu of conducting landscape-level experiments. These models allow the processes taking place to be understood and may aid in predicting future landscape conditions and their effects on the wildlife resource.

The objective of this chapter is to examine how landscape modification may influence landscape pattern and wildlife habitat. First, changes in landscape pattern resulting from timber harvest, road construction, and progressive deforestation are presented with tables, graphs and schematic diagrams. Second, the effectiveness of the

expressions for describing and detecting these modifications is presented along with a discussion suggesting their usefulness and limitations. Next, the management implications for wildlife resources are discussed. Finally, I use results of these analyses to reduce the landscape pattern metric list for subsequent analyses.

METHODS

Map Selection

The 24 expressions of landscape pattern (see Table 1) were computed from 98 forested maps (scale: 1:24,000) from the Blacksburg Ranger District of the Jefferson National Forest. Maps were selected that reflected a broad range of land class patterns, providing a variety of different landscape conditions. To further extend the range of potential landscapes, additional maps were generated by modifying existing land-use patterns. A minimum of 30 unique maps were used for examining each landscape modification (i.e., the maps used for the analysis of one type of landscape modification (e.g., timber harvest, road addition, or progressive deforestation) were not used for any other modification.

GIS Analysis

Cartographic modeling within Idrisi was used to simulate the landscape modification scenarios described below. These models consisted of GIS functions combined in a particular sequence for each specific modification.

A timber harvest of 20% total forest cover was modeled for each landscape. Each landscape was harvested in 5 different ways: with 1 initial unit comprising 20%; 2 units of 10%; 4 units of 5%; 8 units of 2.5%; and 16 units of approximately 1.25% (Fig. 7). This scheme resulted in changes in harvest unit number, size, and shape. To investigate the influence of harvest unit arrangement on pattern, half of the maps contained units dispersed evenly across the landscape; the remaining maps had units placed in clustered

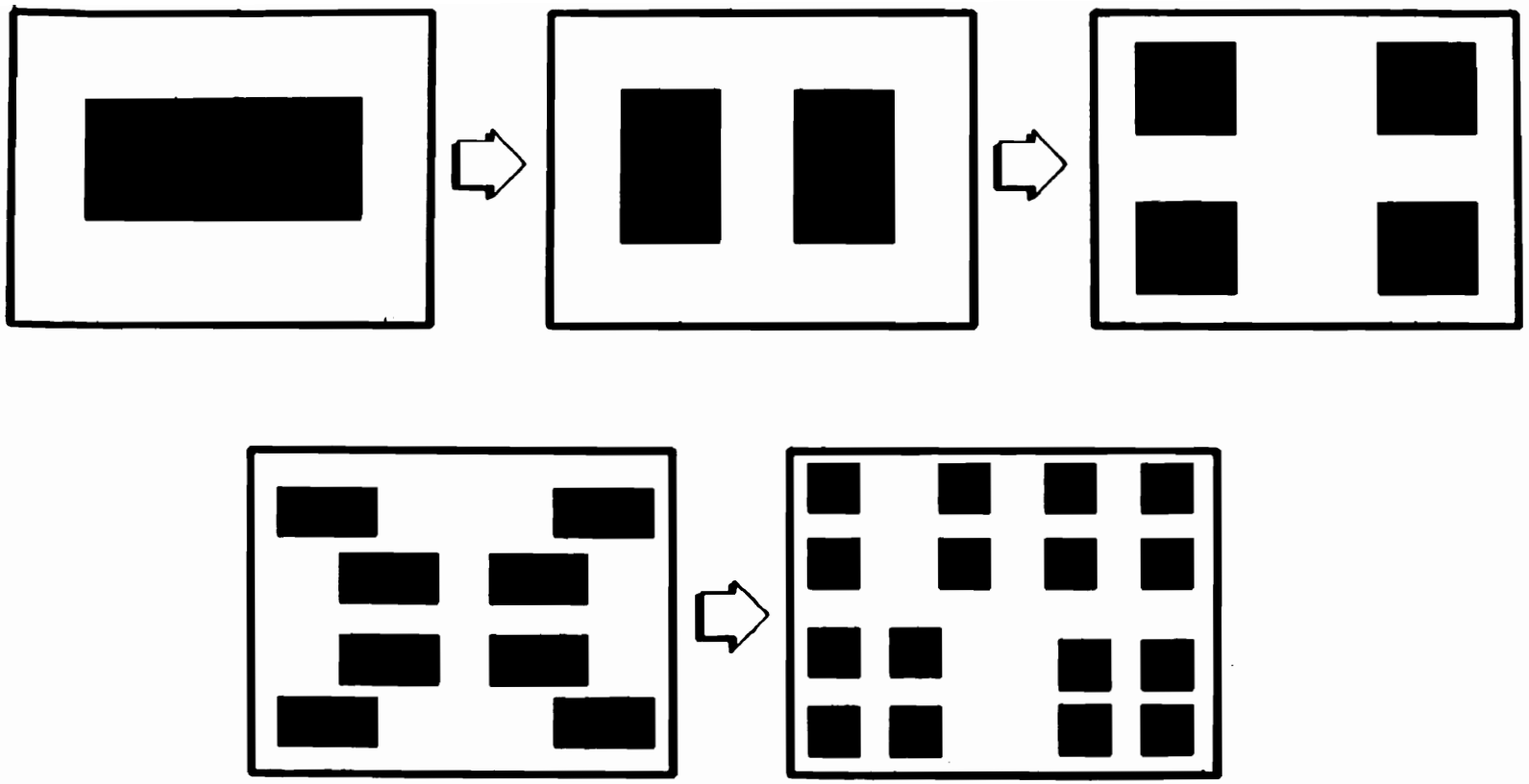


Fig. 7. Simplified representation of 1 of the harvest patterns developed under the timber harvest models. Black squares symbolize the potential placement of harvest units.

arrangements. It was sufficient for this analysis to specify that all units represented cuts using the same technique.

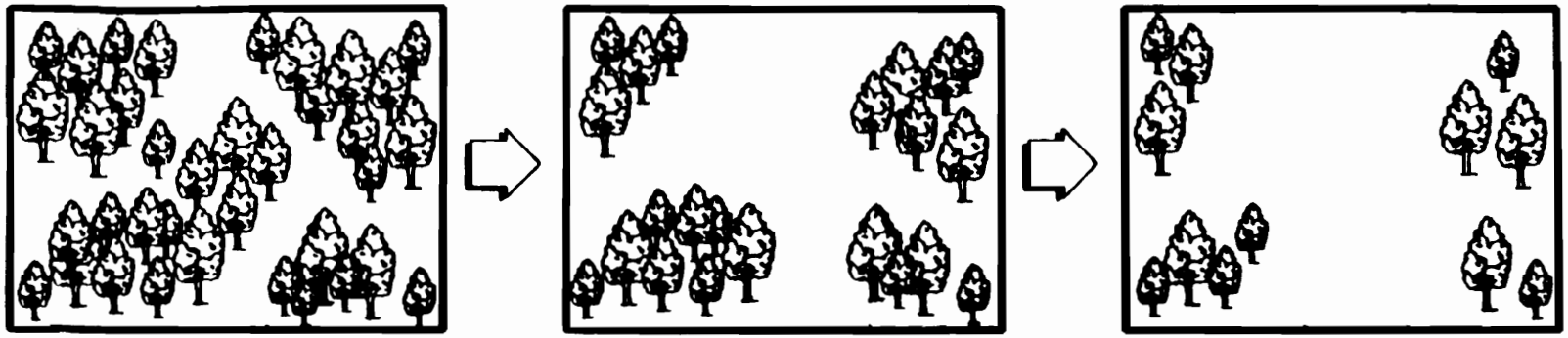
During the modeling of road construction and expansion, 4 successive road lengths of 1.5 km were overlaid onto each forested landscape. Dependent on placement, 1 road length was sufficient to reach the center of a landscape map from any starting point along the map edge; 2 lengths were sufficient to completely span a map. These hypothetical road additions were placed based on the following criteria: existing road placement, if applicable; and guidance from U. S. Forest Service personnel regarding placement of system roads.

Progressive deforestation was modeled by duplicating a variety of documented forest cover loss patterns (Levinson 1981; Dunn et al. 1991, and others). This modification was applied to continuous, unbroken forest blocks; to contiguous forest blocks perforated by openings; and to landscapes comprised of several forest patches (Fig. 8). First, a buffer zone 300m wide was generated along the inside edge of each forested area. Next, this zone was reclassified as non-forest, followed by measurement of the 24 landscape expressions. A new buffer zone of 300m was then generated along the newly-created forest edges, and the process repeated for each successive stage. This continued until the entire forested landscape was replaced by non-forest land class.

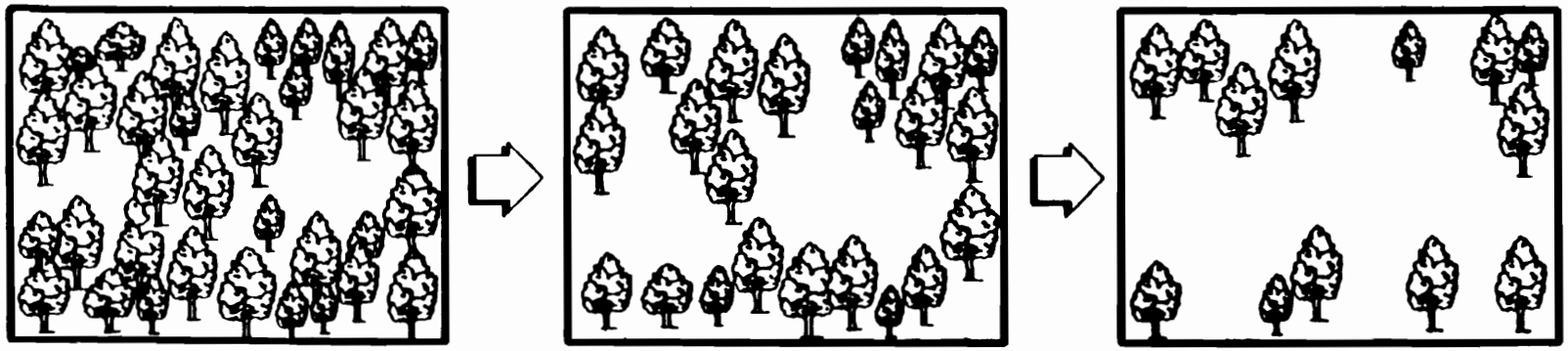
The maps for each landscape modification scheme were digitized and processed as described in Chapter 2. Maps were analyzed at each stage of modification for the entire suite of landscape expressions. The cartographic modeling and GIS analysis operations were completed using Idrisi functions (Appendix I) and a Turbo Pascal program that I developed (Appendix II).

Statistical Analysis

Descriptive Statistics. Values for each expression of pattern were computed from the landscape modification maps. These values were analyzed using SAS (1987)



(A) Shrinking forest patches.



(B) Expanding forest clearings.

Fig. 8. Graphical representation of 2 progressive fragmentation schemes evaluated on the model landscapes

to produce descriptive statistics for each stage of the modification process to detect progressive changes in landscape pattern. Graphical displays of selected expressions were then created.

Discriminant Analysis. Discriminant analysis was used to examine the ability of an expression to discriminate between changes in landscape modification. A subset of the maps was regrouped for 4 separate discriminant analyses. To assess the effectiveness of an expression for detecting pattern differences arising from the spatial arrangements of harvest units, subsets of the maps were classified as having a dispersed (i.e., scattered relatively evenly across the landscape) or clustered (i.e., bunched in close proximity to one another) pattern of harvest units. A subset of the harvest maps was also classified into either a few-unit (1, 2, or 4) or into a many-unit (8 or 16) group for assessing the effectiveness of expressions for detecting pattern changes arising from variation in the number and size of harvest units on a landscape. A subset of the road modification maps was selected and classified into 2 groups, 1 representing roadless landscapes, the other representing heavily-roaded (6.0 km / 900 ha) landscapes. Finally, subsets of the progressive deforestation modification maps were placed into 2 categories: contiguous landscapes (large, continuous blocks of forest); or fragmented landscapes (numerous small forest patches or large forest blocks punctuated with many openings). The suite of landscape expressions was then tabulated for each of the 4 paired groupings.

For each landscape modification, the discriminant analysis proceeded as follows: First, the multivariate means for each expression was calculated for each group and then subtracted from one another to produce the vector of mean differences. A sum-of-squares matrix containing all expressions was computed and used to construct the pooled variance and covariance matrix. Mahalanobis distance, based upon the pooled covariance matrix, was computed to measure the separation of the 2 groups . The significance of Mahalanobis distance was tested using the Hotelling T^2 test and transformed to an F test (Davis 1986). F-values and standardized differences were computed to assess the

discrimination effectiveness of the expressions. Standardization was accomplished by taking the difference between each multivariate mean for an expression and dividing by the corresponding pooled standard deviation.

Two assumptions were made about the nature of the data to test the significance of the separation between the 2 groups: (1) The metrics are normally distributed within each group, and (2) The variance-covariance matrices of the groups are equal in size. However, the discriminant function is not seriously affected by limited departures from normality or by limited inequality of variances (Davis 1986). Three other assumptions required for discriminant analysis were met: (1) The probability of an unknown map belonging to either group was equal, (2) None of the metrics used to calculate the discriminant function were misclassified, and (3) There was no missing data (i.e., all metrics are measured for each map).

RESULTS AND DISCUSSION

The Influence of Timber Harvest on Landscape Pattern

Landscape pattern was greatly affected by the variation in the number and size of harvest units (Table 7). The progression from a few, large harvest units to several small units produced a steady increase in the spatial heterogeneity of a landscape. This increase was reflected by the number of different classes, binary comparison matrix, spatial diversity, and interspersion expressions. These expressions reflected the localized differences occurring as changes were made in the number and size of harvest units. These changes were not reflected in the Shannon, Simpson, dominance, and evenness metrics. Each of these metrics rely on the proportion of land classes, which in this analysis was held constant. The interspersion metric was highest when there were several, small units, reflecting the increase mix of forest and non-forest classes.

The different harvest models also resulted in differences in the expressions of

Table 7. Pattern changes along a progression of increasing number of timber harvest units as derived from 38 maps.

Landscape Expression	Number of Harvest Units: 1		2		4		8		16	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Spatial Heterogeneity										
Shannon Index	0.500	0.00	0.500	0.00	0.500	0.00	0.500	0.00	0.500	0.00
Dominance Index	0.193	0.00	0.193	0.00	0.193	0.00	0.193	0.00	0.193	0.00
Number of Different Classes	1.418	0.10	1.495	0.03	1.592	0.02	1.694	0.06	1.743	0.05
Simpson Index	0.323	0.00	0.323	0.00	0.323	0.00	0.323	0.00	0.323	0.00
Landscape Evenness	0.556	0.00	0.556	0.00	0.556	0.00	0.556	0.00	0.556	0.00
Interspersion	1.098	0.09	1.544	0.16	2.072	0.11	2.234	0.11	2.456	0.09
Binary Comparison Matrix	6.310	0.44	7.936	0.52	9.586	0.32	10.238	0.54	10.632	0.37
Spatial Diversity	41.800	3.06	49.500	2.83	59.200	2.46	69.400	5.87	74.300	5.22
Fragmentation										
Fragmentation Index I	0.020	0.00	0.037	0.01	0.104	0.03	0.131	0.03	0.198	0.01
Fragmentation Index II (km)	0.744	0.04	0.644	0.05	0.544	0.07	0.541	0.04	0.454	0.05
Percent Interior Forest	62.500	1.56	55.600	2.44	49.000	2.32	43.700	2.57	40.000	3.62
Number of Forest Patches	1.000	0.00	1.700	0.37	3.200	0.85	3.500	1.41	4.100	1.21
Mean Patch Density (no./ha)	0.111	0.00	0.189	0.04	0.356	0.09	0.478	0.19	0.455	0.13
Mean Patch Size (ha)	720.000	0.00	570.000	77.00	432.800	96.00	487.200	96.00	416.600	101.00
Interpatch Distance (km)	0.000	0.00	0.056	0.04	0.464	0.22	0.184	0.09	0.358	0.16
Percent Forest Cover	0.800	0.00	0.800	0.00	0.800	0.00	0.800	0.00	0.800	0.00
Edge Characteristics										
Total Forest Edge (km)	17.880	0.89	19.560	0.81	23.460	1.19	26.280	1.46	27.060	0.49
Convexity Index	0.025	0.01	0.027	0.01	0.033	0.01	0.036	0.01	0.038	0.01
Patton Index	1.880	0.09	1.811	0.11	2.109	0.17	2.350	0.24	2.405	0.19
Compactness	0.546	0.03	0.577	0.04	0.515	0.05	0.481	0.06	0.447	0.04
Connectivity										
Connectivity Index	0.000	0.00	0.050	0.05	0.166	0.07	0.162	0.07	0.209	0.07
Contiguity Index	1.920	0.01	1.889	0.01	1.845	0.01	1.828	0.01	1.809	0.01
Spatial Integrity	0.600	0.16	0.300	0.56	0.500	0.95	4.100	1.89	4.300	1.84
Patchiness	0.000	0.00	40.500	21.26	53.900	18.21	46.500	19.31	66.500	22.32

fragmentation of a landscape. These expressions increased when there were several, small harvest locations. The increase in fragmentation resulted from dividing the forest into isolated patches or from creating dispersed clearings throughout the forest. There was an associated decline in both the average distance to non-forest and the amount of interior forest as harvest units were added. The decline in forest interior continued as additional harvest units were added. When 16 harvest locations were placed, an average of 40% of interior forest remained.

While forest interior declined rapidly with the number of harvest units, the total edge length between forest and non-forest areas rose linearly to a high when 16 units were harvested. Continued addition of harvest units resulted in high edge-to-area ratios, reflecting the influence of increasing edge to forest area. (The area remained constant for each harvest model).

There was a steady decline in forest connectivity during the addition of harvest units. The contiguity of the forested landscapes was higher when a few, large units were harvested in contrast to the harvest of several small units. The response of contiguity and several other expressions during the different harvest alternatives is depicted in Fig. 9.

Effectiveness of Expressions for Describing Harvest Alternatives

The results of the discriminant analysis for the differences in resulting landscape pattern is given in Table 8. The expressions are listed in order of their effectiveness for detecting pattern changes arising from timber harvest modeling.

The measure of the difference between the multivariate means, Mahalanobis distance, was significant (6.45, $P < 0.001$). The group centroids (or means) on the discriminant axis were 88.73 (**Few Units**) and 62.85 (**Many Units**). I reject the null hypothesis that the 2 multivariate means are equal, or that the distance between them is zero (Davis 1986).

The magnitude of the standardized differences provides the first means for assessing the effectiveness of each landscape expression in discriminating among the patterns

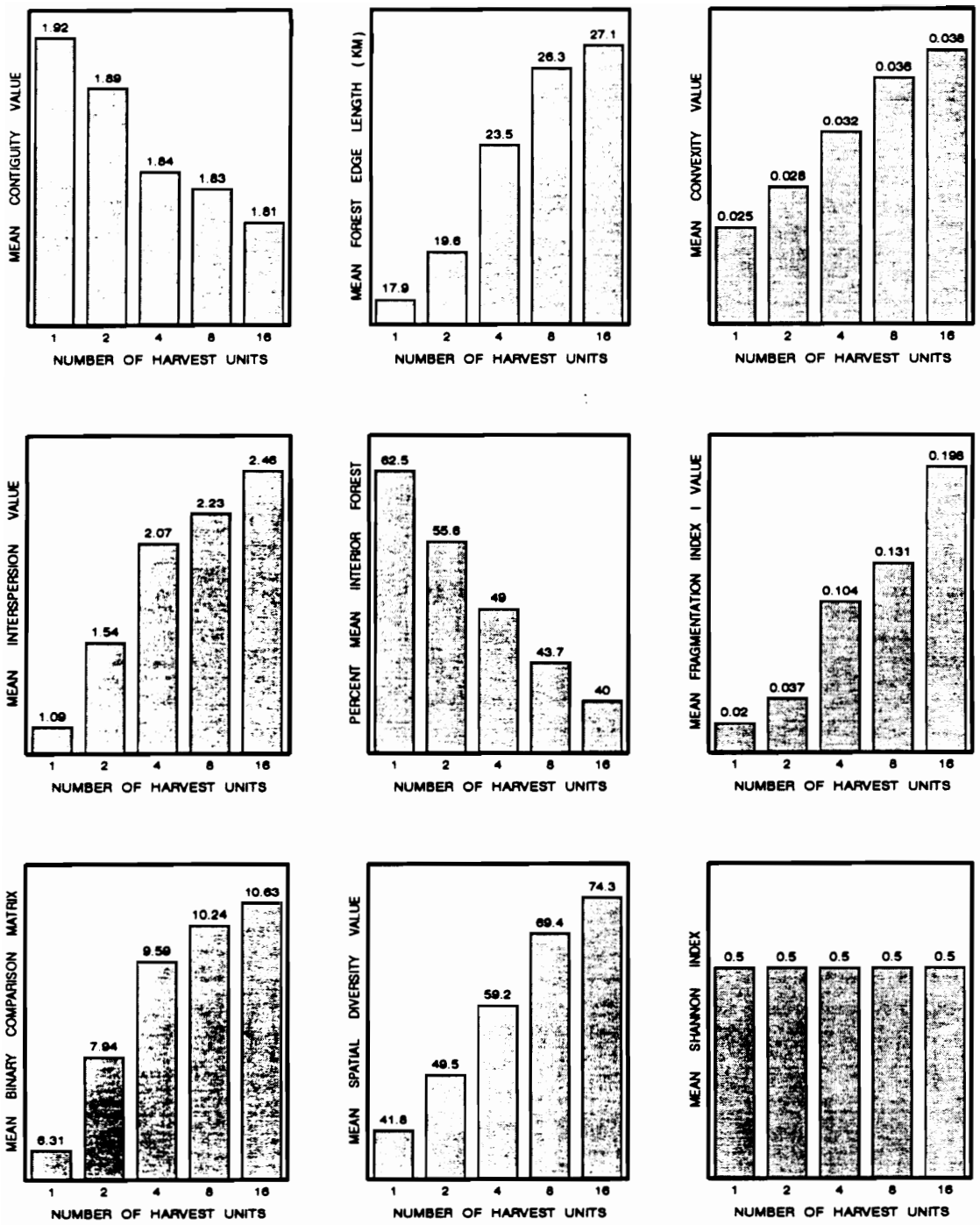


Fig. 9. The response of selected landscape expressions to the 5 standard harvest schemes.

Table 8. Effectiveness of landscape metrics as discriminators of pattern differences resulting from changes in the number and size of harvest units as derived from 38 maps.

Landscape Expression	Mean I Few Units[a]	Mean II Many Units[b]	Mean Differences[c]	Standardized Differences[d]	F Value[e]
Contiguity Index	1.903	1.817	0.086	3.076	89.7***
Interspersion	1.329	2.345	-1.016	-2.646	66.3***
Convexity Index	0.026	0.037	-0.011	-2.395	54.3***
Total Forest Edge (km)	18.930	26.670	-7.740	-2.386	53.9***
Fragmentation Index I	0.035	0.164	-0.129	-2.365	53.0***
Binary Comparison Matrix	7.285	10.435	-3.149	-1.905	34.4***
Percent Interior Forest	0.584	0.418	0.165	1.677	26.6***
Spatial Diversity	47.222	71.850	-24.628	-1.636	25.4***
Number of Different Classes	1.472	1.718	-0.246	-1.636	25.4***
Fragmentation Index II (km)	0.676	0.499	0.177	0.995	9.4**
Spatial Integrity	0.333	4.200	-3.867	-0.899	7.7**
Patton Index	1.907	2.377	-0.470	-0.865	7.1**
Connectivity Index	0.047	0.185	-0.139	-0.737	5.1*
Number of Forest Patches	1.555	3.800	-2.244	-0.731	5.1*
Mean Patch Density (no./ha)	0.173	0.422	-0.249	-0.730	5.1*
Mean Patch Size (ha)	628.000	451.900	176.100	0.662	4.2*
Patchiness Index	20.444	56.500	-36.056	-0.619	3.6
Compactness	0.545	0.464	0.081	0.583	3.2
Interpatch Distance (km)	0.120	0.271	-0.151	-0.371	1.3
Percent Forest Cover	0.800	0.800	0.000	0.000	0.0
Landscape Evenness	0.556	0.556	0.000	0.000	0.0
Dominance Index	0.193	0.193	0.000	0.000	0.0
Shannon Index	0.500	0.500	0.000	0.000	0.0
Simpson Index	0.323	0.323	0.000	0.000	0.0

[a] The mean for each metric representing 1, 2, or 4 harvest unit locations.
[b] The mean for each metric representing 8 or 16 harvest unit locations.
[c] The difference between the means for each metric.
[d] Represents the mean difference for a metric divided by the corresponding pooled standard deviation. Absolute values greater than or equal to 1.0 indicate effective discrimination.
[e] Indicates the significance of a metric in the discriminant equation.
* Significant ($P < 0.05$).
** Significant ($P < 0.01$).
*** Significant ($P < 0.001$).

created. The 9 landscape expressions listed at the top of Table 8 were considered effective based on this criterion. The role of each metric in the discriminant equation is indicated within the F-value column; 16 landscape expressions proved to be significant discriminators using this criterion.

These results indicate that contiguity and interspersion were the expressions most successful in detecting landscape pattern differences. Convexity, total forest edge, fragmentation index I, binary comparison matrix, percent interior forest, spatial diversity, and number of different classes were also sensitive to changes resulting from the different harvest patterns.

Those expressions highly significant ($P<0.001$) in the discriminant analysis were consistent in their behavior. They exhibited either a progressive increase (number of different classes, interspersion, binary comparison matrix, spatial diversity, fragmentation index I, forest edge, and convexity) or a progressive decrease (fragmentation index II, percent interior forest, and contiguity) in values with little variability at each stage of the harvest alternatives (Table 7). The less significant expressions ($P < 0.01$, $P < 0.05$) were somewhat unpredictable, with value shifts in opposite directions.

Not all of the expressions were useful in distinguishing pattern changes that may arise from timber harvest. The expressions that were not significant demonstrated either high variability about their values (patchiness, compactness, and interpatch distance) or did not change in value (Shannon index, Simpson index, dominance, evenness, and forest cover).

Effectiveness of Expressions for Describing Spatial Arrangement

Table 9 presents the results of the discriminant analysis for the dispersed and clustered harvest models. The Mahalanobis distance (26.62) and its associated F statistic (3.75, $P < 0.05$) indicated the multivariate distance between the dispersed and clustered harvest landscapes was significant. The group centroids on the discriminant axis were -34.01 (**Dispersed**) and -60.61 (**Clustered**). Based upon the values for the

Table 9. Effectiveness of landscape metrics as discriminators of pattern differences resulting from changes in spatial arrangement of harvest units as derived from 30 maps.

Landscape Expression	Mean I Dispersed[a]	Mean II Clustered[b]	Mean Differences[c]	Standardized Differences[d]	F Value[e]
Fragmentation Index II (km)	0.385	0.641	-0.256	-2.151	34.7***
Number of Different Classes	1.790	1.563	0.227	2.057	31.7***
Spatial Diversity	79.000	56.267	22.733	2.057	31.7***
Percent Interior Forest	0.368	0.517	-0.149	-1.989	29.7***
Connectivity Index	0.025	0.333	-0.307	-1.981	29.4***
Binary Comparison Matrix	11.068	9.236	1.832	1.845	25.5***
Patchiness Index	15.467	95.800	-80.333	-1.709	21.9***
Mean Patch Size (ha)	638.900	252.100	386.800	1.673	21.0***
Number of Forest Patches	1.800	5.400	-3.600	-1.134	9.7**
Mean Patch Density (no./ha)	0.200	0.600	-0.400	-1.133	9.6**
Patton Index	2.576	2.000	0.576	1.013	7.7**
Compactness	0.413	0.550	-0.137	-0.959	6.9*
Interspersion	2.411	2.097	0.313	0.958	6.9*
Spatial Integrity	5.200	0.733	4.467	0.927	6.5*
Contiguity Index	1.817	1.834	-0.017	-0.620	2.9
Interpatch Distance (km)	0.226	0.445	-0.219	-0.434	1.4
Fragmentation Index I	0.133	0.156	-0.023	-0.282	0.6
Total Forest Edge (km)	25.440	25.760	-0.320	-0.083	0.1
Convexity Index	0.035	0.036	-0.001	0.000	0.0
Dominance Index	0.193	0.193	0.000	0.000	0.0
Percent Forest Cover	0.800	0.800	0.000	0.000	0.0
Landscape Evenness	0.556	0.556	0.000	0.000	0.0
Shannon Index	0.500	0.500	0.000	0.000	0.0
Simpson Index	0.323	0.323	0.000	0.000	0.0

[a] The mean for each metric representing regularly distributed units.
[b] The mean for each metric representing grouped units in close proximity.
[c] The difference between the means for each metric.
[d] Represents the mean difference for a metric divided by the corresponding pooled standard deviation. Absolute values greater than or equal to 1.0 indicate effective discrimination.
[e] Indicates the significance of a metric in the discriminant equation.
* Significant ($P < 0.05$).
** Significant ($P < 0.01$).
*** Significant ($P < 0.001$).

standardized differences, 11 landscape expressions were effective discriminators of spatial arrangement. The F-values support this and further suggest the potential importance of number of forest patches, mean patch density, and Patton index for discriminating among harvest arrangements.

The landscape expressions contributing the most classification information were fragmentation index II, number of different classes, spatial diversity, percent interior forest, connectivity, binary comparison matrix, patchiness, and mean patch size. Fragmentation index II topped the list of expressions that were sensitive to pattern changes arising from the spatial arrangement of harvest units. The mean value of this metric was twice as high on those landscapes with clustered compared to dispersed harvest units. It is interesting that the roving-window heterogeneity expressions that were important discriminators in the earlier analysis were again valuable for detecting differing spatial arrangements. The useful ones included binary comparison matrix, spatial diversity, interspersion, and number of different classes.

The differing arrangements of harvest units were also reflected in the values of percent forest interior and mean patch size. Mean forest patch size decreased under the clustered arrangement. This decline reflected the isolation of forest patches that may occur when harvest areas are in a clustered formation. This isolation is also reflected in value changes for connectivity, number of forest patches, patchiness, and patch density values.

Fig. 10 depicts the response of each pattern metric to changing spatial configuration. Fragmentation index II values show a trend of steady decline in distance to non-forest (e.g., distance to edge) when harvest units are evenly dispersed across the landscape. The interior forest area disappeared rapidly on those landscapes with dispersed harvest units. Binary comparison matrix values dramatically increased in heterogeneity with the dispersed harvest arrangement. There was a variable upward trend for the clustered arrangement. Number of different classes and spatial diversity (not pictured) exhibited a similar response. Mean patch size and patchiness illustrate that isolated forest patches had

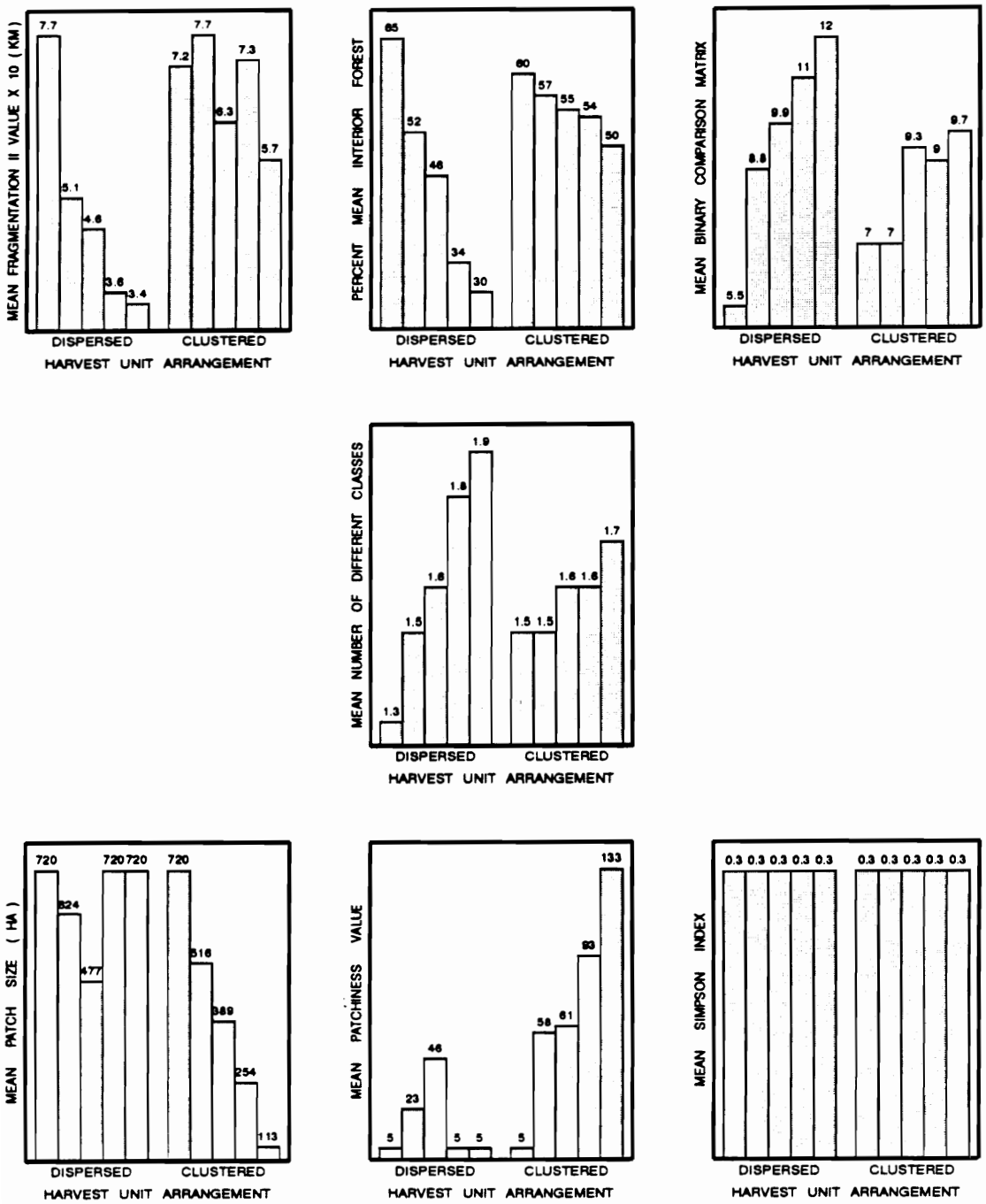
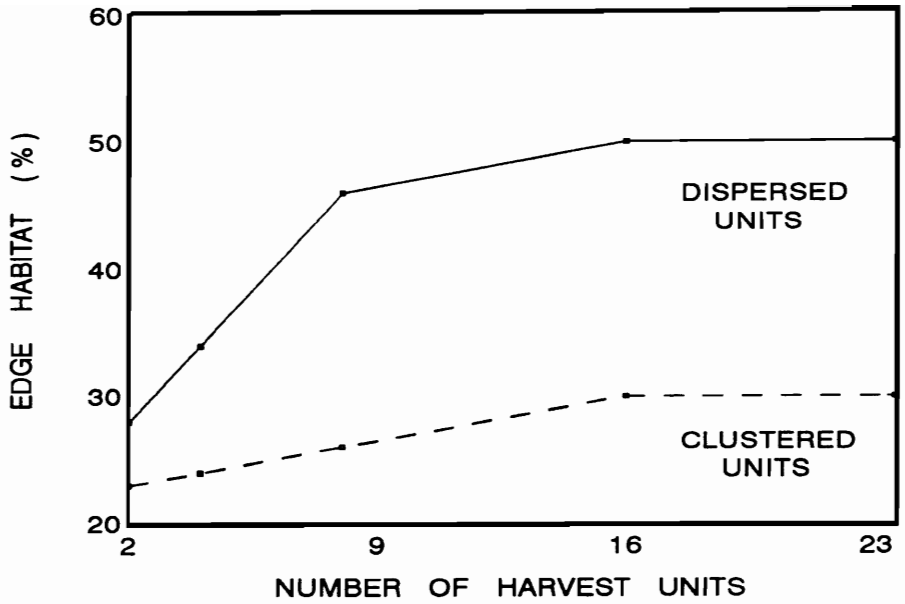


Fig. 10. Changes in landscape pattern based on the dispersed and clustered timber harvest models. The 5 bars within each arrangement represent 1, 2, 4, 8, and 16 harvest units respectively.

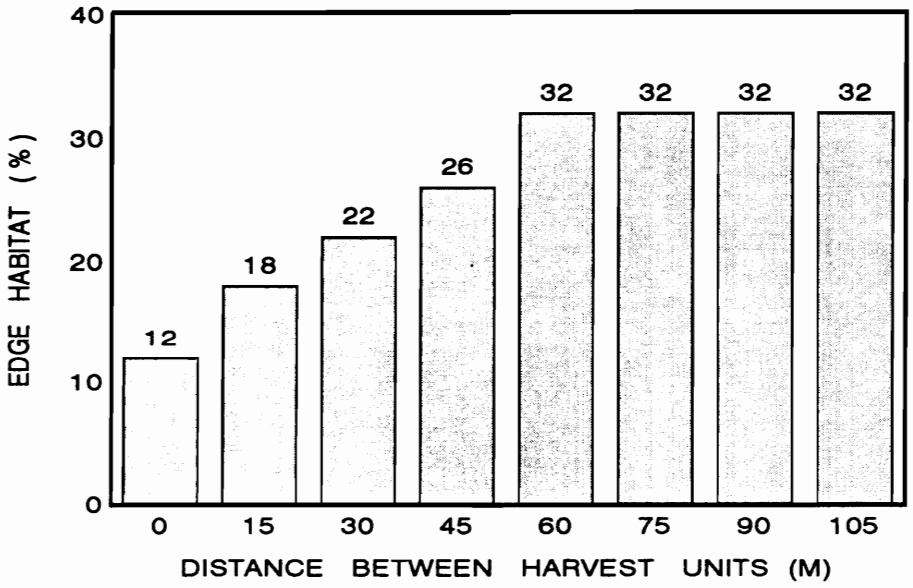
been created from clustered harvest arrangements. Finally, forest edge length and the Simpson index were ineffective discriminators of different spatial arrangements.

Several expressions included in the discriminant analysis were not useful in distinguishing between the dispersed and clustered harvest arrangements. The Shannon, Simpson, evenness, and dominance expressions were not sensitive to changes in spatial configuration. As discussed in the previous section, sole reliability on the number and proportion of land classes may prove a limitation for evaluating harvest alternatives. Values for contiguity, forest cover, convexity, and total forest edge changed very little during the different cutting patterns. The dispersed harvest arrangement was expected to lower forest contiguity values. This metric represents the mean contiguity value for a landscape; it is likely that localized differences are masked by the averaging procedure.

Although edge length was not significant for detecting changes in the spatial arrangement of harvest locations, potential edge habitat would differ between the 2 harvest models. The area of edge habitat in a landscape is related to both total edge length and a width which together comprise a "border." The border width is species-specific. Edge habitat differs in microclimate and vegetational composition from the forest interior, and can be a concentrated area for edge-related species (see Literature Review). The top of Fig. 11 illustrates the difference in edge habitat creation using a 30m buffer width along the edge of each harvest unit. It is apparent that of the 2 spatial arrangements, the dispersed harvest scheme creates more edge habitat than the clustered scheme. The bottom of Fig. 11 compares the proportion of edge habitat created along a gradient of increasing distance between harvest locations, again based on the arbitrary constraint of 30m zone width around each unit. Once the distance between harvest units reaches and exceeds twice the set buffer length, the proportion of edge habitat remains constant. (The curve for edge habitat levels off because more than 16 units could not retain their individuality by having at least some distance between them). As harvest units are placed proximal to one another, the areas of edge buffer overlap and become progressively



(A)



(B)

Fig. 11. Relationship between edge habitat creation and timber harvest modification. (A) The number and arrangement of harvest units. (B) The spacing of harvest units using a 30m buffer on a 9 ha sample tract.

smaller amounts of edge habitat are created.

Road Expansion Influence on Landscape Modification

Table 10 presents the progressive changes in landscape pattern resulting from road construction. Road construction dramatically altered the spatial heterogeneity of a landscape. A new road essentially introduced a new land class onto a landscape. There was a reduction in dominance values, as expected, as roads were placed onto the map, since the proportion in each land class declined. This change in proportions also produced expected increases in evenness values. The progression from roadless to 6.0 km of roads produced a consistent increase in value for Shannon and Simpson indices, also related to the change in proportions. Values for the number of different classes, interspersion, and binary comparison matrix also increased. After the first stage of road expansion, spatial diversity also rose. Values of this index initially dropped, reflecting the addition of "road" as a new land class in the denominator of this expression. The denominator did not change after each successive road addition, so the spatial diversity values subsequently climbed linearly with road length addition.

With each additional road segment, land classes became interspersed with forested areas. This augmentation of interspersion resulted in changes in edge length. Patton's and compactness indices fluctuated in different directions during the process of road expansion. Areas became more circular; the road increase added a "non-compact" component. Additional roads resulted in high edge-to-area ratios. Forest edge length escalated with each successive road addition. The position of road placement altered this rate of increase, however. If a road was placed along the boundary of a landscape, or within non-forested areas, then additional edge development was slight or unchanged.

Fig. 12 depicts the influence of road placement on the heterogeneity and edge metrics. The binary comparison matrix and number of different classes (not shown), followed the same pattern as interspersion. The heterogeneity values were dependent on how many land classes were replaced by roads. On landscapes where roads removed

Table 10. Pattern changes resulting from road addition and expansion as derived from 30 maps.

Landscape Expression	Road Length: None		1.5 km		3.0 km		4.5 km		6.0 km	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Spatial Heterogeneity										
Shannon Index	0.334	0.06	0.519	0.06	0.632	0.06	0.714	0.05	0.772	0.05
Dominance Index	0.011	0.01	0.376	0.02	0.264	0.19	0.212	0.03	0.154	0.03
Number of Different Classes	1.390	0.07	1.570	0.07	1.710	0.07	1.860	0.07	1.990	0.08
Simpson Index	0.242	0.04	0.316	0.04	0.382	0.04	0.439	0.03	0.483	0.03
Landscape Evenness	0.470	0.08	0.417	0.05	0.533	0.05	0.642	0.04	0.739	0.03
Interspersion	1.160	0.22	1.630	0.21	2.070	0.21	2.490	0.19	2.880	0.18
Binary Comparison Matrix	6.230	1.20	8.490	1.10	10.500	1.00	12.400	0.94	14.100	0.88
Spatial Diversity	36.400	7.10	33.100	2.90	44.300	2.40	55.300	1.90	64.900	1.80
Fragmentation										
Fragmentation Index I	0.037	0.01	0.063	0.01	0.086	0.01	0.108	0.01	0.131	0.01
Fragmentation Index II (km)	0.202	0.04	0.765	0.08	0.609	0.06	0.471	0.03	0.460	0.02
Percent Interior Forest	63.900	6.80	52.800	6.20	43.900	5.50	35.000	4.70	27.100	4.10
Number of Forest Patches	1.870	0.23	2.170	0.29	3.030	0.37	4.230	0.58	5.000	0.73
Mean Patch Density (no./ha)	0.207	0.02	0.241	0.03	0.337	0.04	0.470	0.06	0.611	0.08
Mean Patch Size (ha)	583.000	63.00	516.000	60.00	393.000	56.00	343.000	53.00	237.000	41.00
Interpatch Distance (km)	0.165	0.04	0.208	0.06	0.278	0.06	0.362	0.07	0.468	0.08
Percent Forest Cover	80.000	3.70	75.400	3.60	70.600	3.60	66.000	3.50	61.200	3.50
Edge Characteristics										
Total Forest Edge (km)	17.140	1.04	18.880	0.89	21.080	0.87	23.280	0.78	24.980	0.76
Convexity Index	0.027	0.01	0.031	0.01	0.037	0.01	0.044	0.01	0.051	0.01
Patton Index	1.500	0.10	1.700	0.09	1.690	0.07	1.750	0.07	1.660	0.06
Compactness	0.727	0.03	0.632	0.02	0.624	0.02	0.603	0.02	0.633	0.02
Connectivity										
Connectivity Index	0.082	0.03	0.092	0.03	0.155	0.04	0.163	0.04	0.186	0.04
Contiguity Index	0.885	0.02	0.840	0.02	0.794	0.03	0.746	0.03	0.690	0.03
Spatial Integrity	-0.570	0.31	-0.730	0.34	-1.700	0.40	-2.900	0.57	-4.200	0.71
Patchiness	30.000	8.70	30.600	8.50	43.600	9.00	53.500	9.00	66.400	11.00

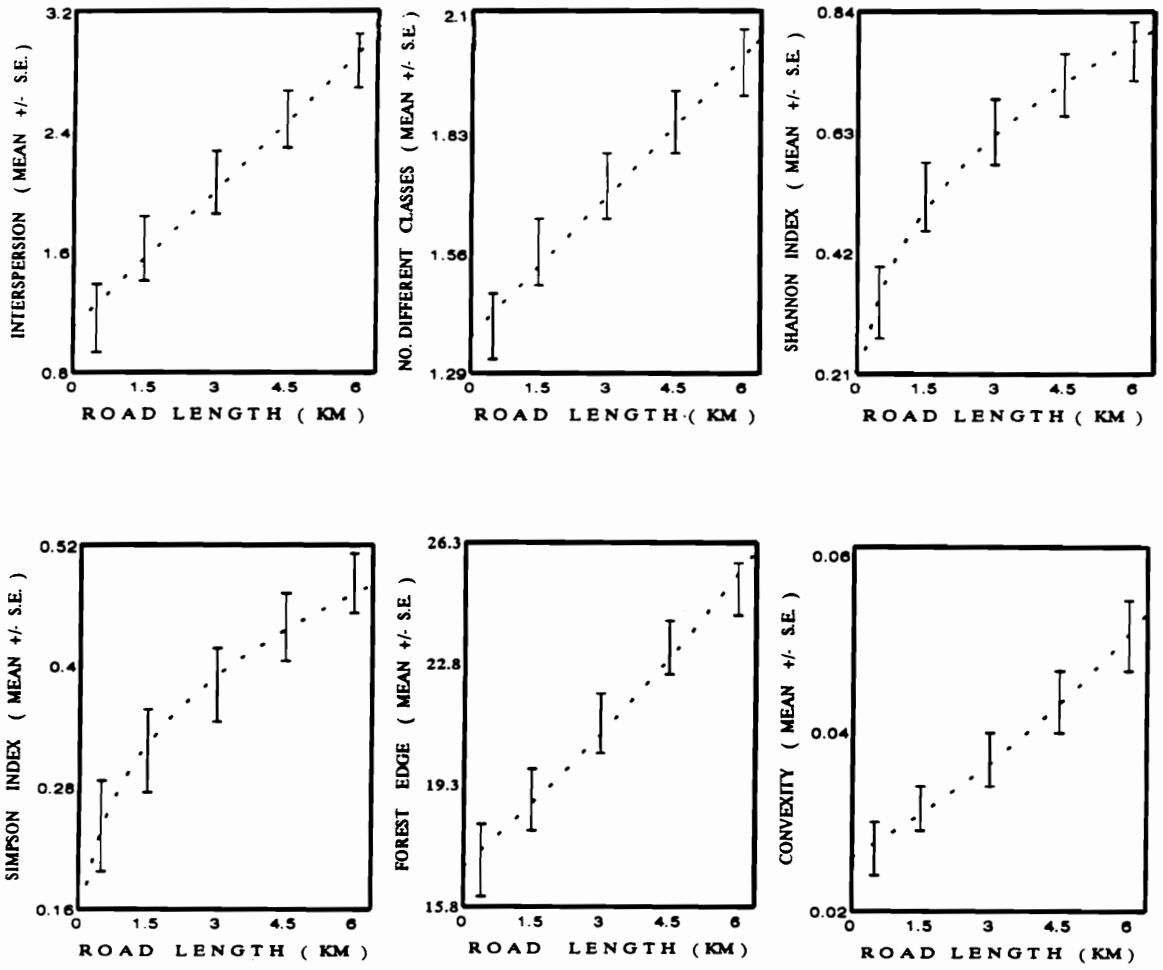
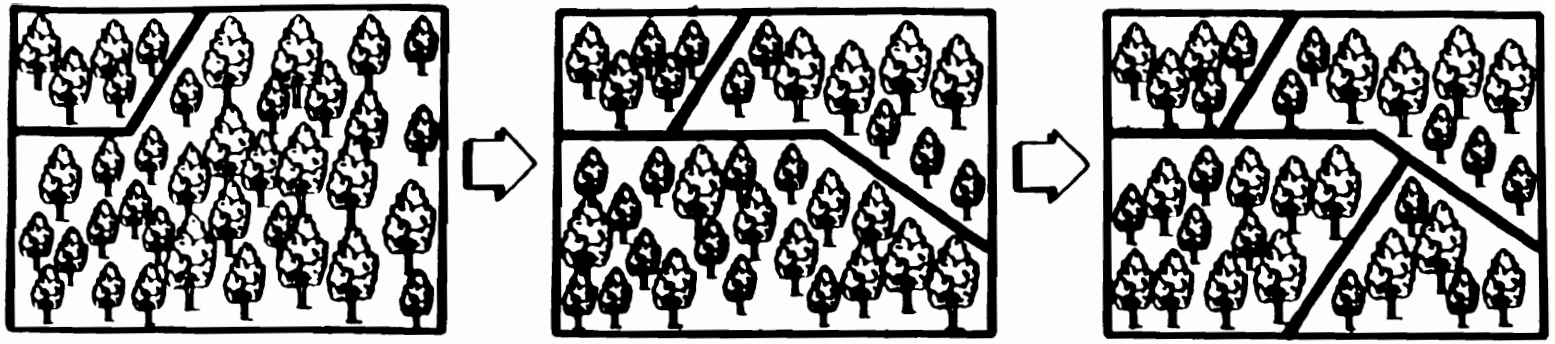


Fig. 12. Changes in selected spatial heterogeneity and edge metrics during the progressive addition of road lengths. Trend lines are derived from regression analysis.

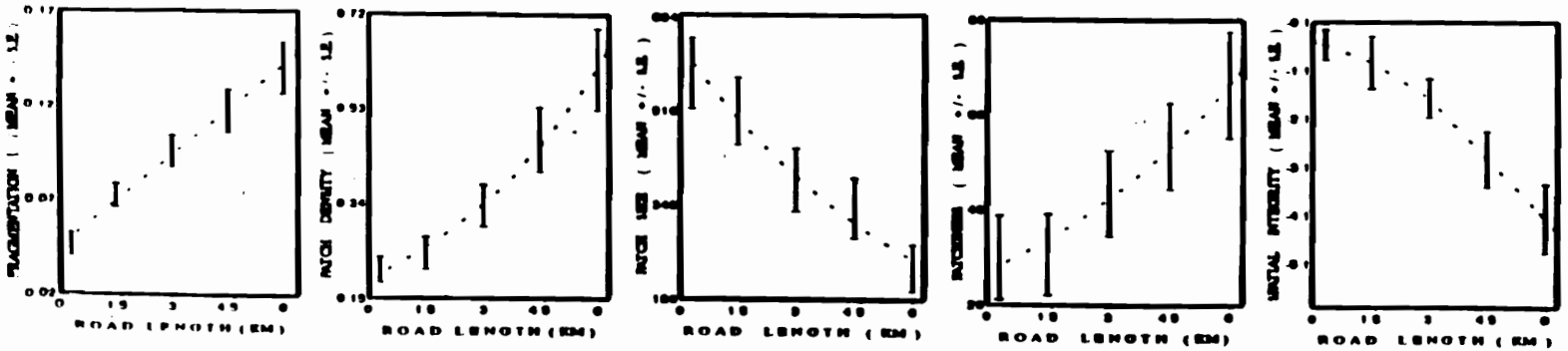
portions of the predominant land class, spatial heterogeneity expanded. However, when roads were placed through less common land classes, heterogeneity values did not perceptibly increase. It becomes increasingly clear that the condition resulting after a land use change is a function not only of the change (e.g., road expansion in this case) but also of the pre-existing landscape condition. The variability in convexity values during the last 2 road additions reflected the influence of increasing edge to forest area. At this stage, there was a likelihood of dividing the landscape into isolated forest segments. The tendency of road placement to partition landscapes is discussed below.

Road expansion had either a partitioning or an invasive effect on resulting landscape pattern. This effect was most notable in those metrics used to monitor fragmentation and connectivity. Both of these attributes were greatly altered during road establishment. When the placement of roads partitioned a landscape into forest patches, there was an increase in patch density, number of patches, interpatch distance, patchiness, and fragmentation index I (Fig. 13). This was also accompanied by consistent declines in mean patch size and spatial integrity. These 2 expressions declined steadily with each consecutive road addition, as positioned roads created isolated forest patches. Each expression incorporates some aspect of patch measurement (number, size, position, or distance apart), which reflects their performance during this landscape modification. There was an increase in standard error, as evident from the length of each bar, during the final 2 stages of road expansion for the fragmentation index I, patch density, and spatial integrity metrics. (Interpatch distance and number of patches, not shown, emulated patch density). With additional road segments, the possibility rose for partitioning the landscape into forest segments with each consecutive road extension.

There were instances when road expansion had an invasive rather than a partitioning effect on landscape pattern. When a road was invasive, pattern changes were best characterized by the 4 expressions displayed in Fig. 14. Forest interior dropped rapidly with each road extension, with an average of 30% remaining after the final road addition. Values of interior forest were variable during the initial stages of road placement.



(A)



(B)

Fig. 13. The partitioning effect of road addition on landscape pattern. (A) Road placement depicting landscape partitioning. (B) Selected expression response to this type of road placement.

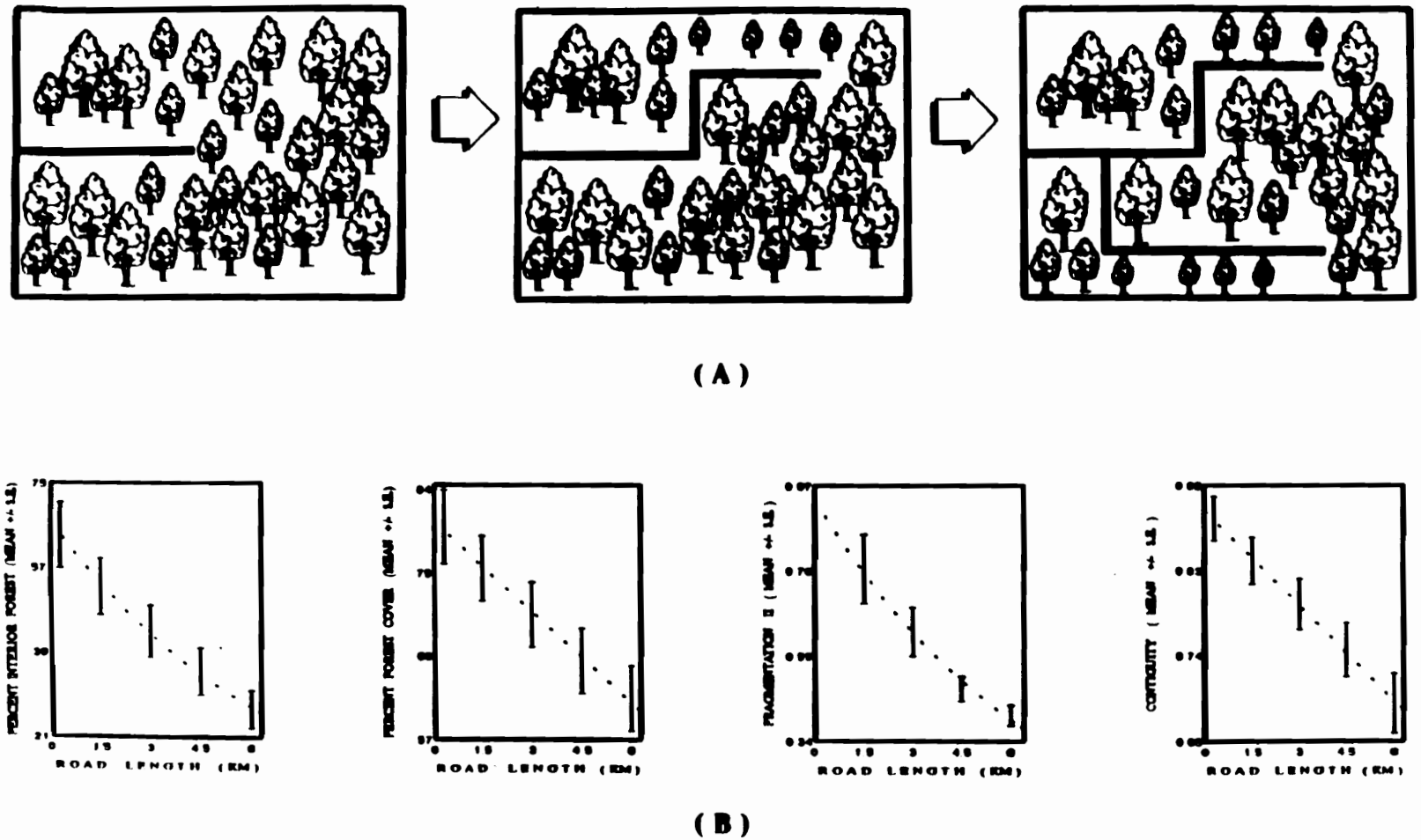


Fig. 14. The invasive effect of road addition on landscape pattern. (A) Invasive pattern of road placement. (B) Selected expression response to this type of road placement.

If roads were placed along landscape boundaries (not shown), losses of forest interior were less (11 - 20%) than if roads were established directly into the center of the forested landscape (15 - 29%).

The loss of forest cover to road length was linear. High variability at each stage reflected which land class type was replaced by a road. There was also a decline in fragmentation index II with each consecutive road extension. With each extension, distance to non-forest areas receded. Finally, the linear reduction in forest contiguity at each stage further demonstrates the invasive property of road addition.

There was 1 expression from the fragmentation and connectivity groups whose behavior during the road modification was inconsistent with the changes observed in the other expressions. The connectivity index, which is dependent on the number of patches and the connections between them, actually increased rather than declined with each successive road extension. This can be explained by the following example: A road extension severs portions of a forest patch while leaving a strip of connective forest between these patch sections. The connectivity index increases as this new landscape pattern translates into 2 connected patches with a link between them, in contrast to the original large single patch without connections.

Effectiveness of Expressions for Describing Roadless and Roaded Areas

Table 11 presents the results of the discriminant analysis for the differences in pattern resulting from road addition. The difference between the multivariate means, Mahalanobis distance, was significant (97.38, $P < 0.001$) and indicates the uniqueness of the roadless and heavily-roaded landscapes. The group centroids were 529.26 (**Roadless**) and 431.88 (**Roaded**). The F-statistic (35.52, $P < 0.001$) indicates that the separation between the 2 groups is significant. Using the standardized differences for assessing effectiveness, 16 of the landscape expressions were highly effective ($P < 0.001$). The "**F-value**" column indicates there were 19 significant expressions in the discriminant

Table 11. Effectiveness of landscape metrics as discriminators of pattern differences between roadless and heavily-roaded landscapes as derived from 49 maps.

Landscape Expression	Mean I Roadless[a]	Mean II Roaded[b]	Mean Differences[c]	Standardized Differences[d]	F Value[e]
Fragmentation Index I	0.056	0.148	-0.092	-1.745	37.3***
Total Forest Edge (km)	20.090	26.220	-6.130	-1.689	34.9***
Interspersion	1.700	3.149	-1.449	-1.585	30.7***
Number of Different Classes	1.573	2.109	-0.535	-1.540	29.0***
Convexity Index	0.033	0.057	-0.025	-1.469	26.4***
Mean Patch Density (no./ha)	0.234	0.707	-0.473	-1.442	25.5***
Number of Forest Patches	2.120	6.375	-4.255	-1.440	25.4***
Contiguity Index	1.842	1.646	0.196	1.424	24.8***
Shannon Index	0.497	0.844	-0.346	-1.400	24.0***
Spatial Integrity	-0.720	-4.958	4.238	1.392	23.7***
Binary Comparison Matrix	8.908	15.456	-6.548	-1.387	23.6***
Mean Patch Size (ha)	463.200	164.600	298.600	1.234	18.6***
Fragmentation Index II (km)	0.658	0.369	0.289	1.125	15.5***
Percent Interior Forest	0.489	0.210	0.279	1.123	15.5***
Simpson Index	0.339	0.524	-0.185	-1.080	14.3***
Interpatch Distance (km)	0.198	0.563	-0.365	-1.036	13.1***
Percent Forest Cover	0.727	0.573	0.154	0.846	8.8**
Patchiness Index	36.002	72.020	-36.018	0.713	6.2*
Spatial Diversity	53.680	67.479	-13.799	-0.660	5.3*
Dominance Index	0.195	0.101	0.095	0.571	4.0
Connectivity Index	0.109	0.219	-0.110	0.535	3.5
Landscape Evenness	0.639	0.788	-0.149	-0.499	3.0
Patton Index	1.752	1.667	0.084	-0.216	0.6
Compactness	0.604	0.628	-0.024	-0.213	0.6

[a] The mean for each metric representing roadless landscapes.
[b] The mean for each metric representing landscapes with 6.0 km of roads.
[c] The difference between the means for each metric.
[d] Represents the mean difference for a metric divided by the corresponding pooled standard deviation. Absolute values greater than or equal to 1.0 indicate effective discrimination.
[e] Indicates the significance of a metric in the discriminant equation.
* Significant ($P < 0.05$).
** Significant ($P < 0.01$).
*** Significant ($P < 0.001$).

equation, the highest number for any of the different landscape modifications examined in this chapter. The large number of significant expressions suggests that road establishment and expansion has a profound influence on changes of landscape pattern within the 4 categories of landscape structure examined here.

Those expressions that were consistent in behavior (exhibiting either progressive increase or decrease in value) are listed at the top of the Table 11. Each member of the fragmentation group was highly significant ($P< 0.001$ or $P< 0.01$). Fragmentation index I topped the list of fragmentation expressions sensitive to pattern changes resulting from road modification. Mean patch density, the number of patches, and mean patch size were also rated highly; these expressions (along with fragmentation index I) reflect the partitioning influence of roaded landscapes. Total forest edge and interspersion were the second and third most successful expressions for detecting landscape pattern differences. Number of different classes, convexity, contiguity, and several of the spatial heterogeneity expressions were also sensitive to changes resulting from establishing roads.

Those expressions not useful for characterizing roadless and heavily-roaded landscapes are presented at the bottom of Table 11. Dominance and evenness were the only heterogeneity members not significant in the analysis, although both metrics did change as roads were added. Perhaps road expansion does not alter landscape pattern to a great extent or has counter-balance effects preventing detection by these 2 expressions. Connectivity, the standard error of which was large relative to computed mean values, was also not significant. Finally, the values of the Patton and compactness indices rose and fell throughout the road modification modeling, and were ineffective as discriminators.

Progressive Deforestation Influence on Landscape Modification

Table 12 presents the changes in landscape pattern resulting from the progressive deforestation. Each stage represents a forest zone shrinkage of 300m from the inside edge of all remaining forest land classes. This simulates deforestation and the progressive reduction in forested area. By Stage 5 (not presented), the entire forest land class had

Table 12. Pattern changes resulting from progressive deforestation as derived from 30 maps.

Landscape Expression	None		Progressive Forest Shrinkage: Stage 1[a]		Stage 2		Stage 3		Stage 4	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Spatial Heterogeneity										
Shannon Index	0.000	0.00	0.490	0.01	0.663	0.01	0.541	0.03	0.294	0.04
Dominance Index	0.000	0.00	0.207	0.01	0.050	0.02	0.177	0.03	0.299	0.04
Number of Different Classes	1.000	0.00	1.581	0.03	1.626	0.03	1.427	0.02	1.166	0.05
Simpson Index	0.000	0.00	0.331	0.01	0.461	0.01	0.371	0.02	0.184	0.03
Landscape Evenness	0.000	0.00	0.572	0.01	0.909	0.02	0.661	0.05	0.325	0.06
Interspersion	0.000	0.00	1.720	0.12	1.613	0.10	1.039	0.25	0.439	0.06
Binary Comparison Matrix	0.000	0.00	8.671	0.50	9.313	0.54	6.043	0.27	2.959	0.48
Spatial Diversity	0.000	0.00	58.167	3.49	62.567	3.51	42.700	2.04	17.567	2.60
Fragmentation										
Fragmentation Index I	0.010	0.00	0.082	0.01	0.052	0.01	0.043	0.01	0.022	0.01
Fragmentation Index II (km)	0.000	0.00	0.579	0.04	0.495	0.03	0.429	0.03	0.310	0.03
Percent Interior Forest	100.000	0.00	51.233	2.23	30.367	3.08	17.267	2.92	6.533	1.80
Number of Forest Patches	1.000	0.00	2.233	0.31	3.900	0.48	2.967	0.32	1.033	0.15
Mean Patch Density (no./ha)	0.111	0.00	0.244	0.03	0.422	0.05	0.344	0.04	0.118	0.02
Mean Patch Size (ha)	900.000	0.00	539.300	47.60	254.800	43.60	174.200	35.40	105.400	23.40
Interpatch Distance (km)	0.000	0.00	0.176	0.05	0.639	0.10	1.036	0.13	0.380	0.16
Percent Forest Cover	100.000	0.00	79.967	0.03	51.233	2.23	30.367	3.08	12.600	2.54
Edge Characteristics										
Total Forest Edge (km)	12.000	0.00	21.740	1.18	18.240	0.82	12.180	0.61	5.040	0.74
Convexity Index	0.013	0.00	0.030	0.01	0.043	0.01	0.058	0.01	0.065	0.01
Patton Index	1.130	0.00	1.795	0.11	1.568	0.06	1.465	0.04	1.208	0.11
Compactness	0.886	0.00	0.606	0.03	0.664	0.02	0.703	0.02	0.592	0.05
Connectivity										
Connectivity Index	0.000	0.00	0.139	0.04	0.082	0.03	0.032	0.02	0.000	0.00
Contiguity Index	2.000	0.00	1.869	0.01	1.787	0.02	1.717	0.03	1.395	0.12
Spatial Integrity	0.000	0.00	0.200	0.60	-2.700	0.48	-2.133	0.33	-0.067	0.12
Patchiness	0.000	0.00	45.170	11.32	35.490	6.12	25.157	4.61	0.850	0.85

[a] Each stage represents a forest zone shrinkage of 300m from the inside edge of all existing forest land classes.

been eliminated from the landscape. Each expression at that point was equal to zero with 3 exceptions: number of different classes (1.0), contiguity (1.0), and fragmentation index I (0.01).

The spatial heterogeneity expressions demonstrated pronounced changes during gradual loss of forest cover on the maps. The transition from a completely-forested landscape to one totally deforested produced curvilinear trends for 7 of the expressions and 1 bimodal distribution (dominance). All of the heterogeneity expressions returned to their initial values once forest cover was eliminated. However, the distributions of their values differed in both symmetry and shape. These are characterized in the first 5 graphs of Fig. 15. (The Simpson index, not shown, duplicated the Shannon index distribution; number of different classes and spatial diversity emulated the binary comparison matrix distribution).

The Shannon, Simpson, and evenness indices were characterized by rather symmetrical curves. These expressions increased steadily until 50% of the landscape remained forested, at which point their values steadily declined until the forest disappeared. These reflect the proportionate changes in land classes and the shift in "half-forested" and "half-unforested" proportions. Because the predominant land class (forest) was being replaced by non-forest, heterogeneity escalated; however, once non-forest comprised over half of the landscape, spatial heterogeneity declined.

Each of the roving-window expressions (binary comparison matrix, spatial diversity, number of different classes, and interspersion) displayed positively-skewed distributions. Initial values for the first 3 expressions escalated rapidly until 50% forest cover was reached and then tapered off gradually. Interspersion of land classes, however, reached a threshold during the first stage of progressive fragmentation (approximately 80% forest cover). As discussed during the timber harvest modification, these roving-window expressions reflect both spatial arrangement and land class proportion. It is possible that the influence of the arrangement component (absent in the traditional heterogeneity expressions discussed above) strongly influences the asymmetry observed.

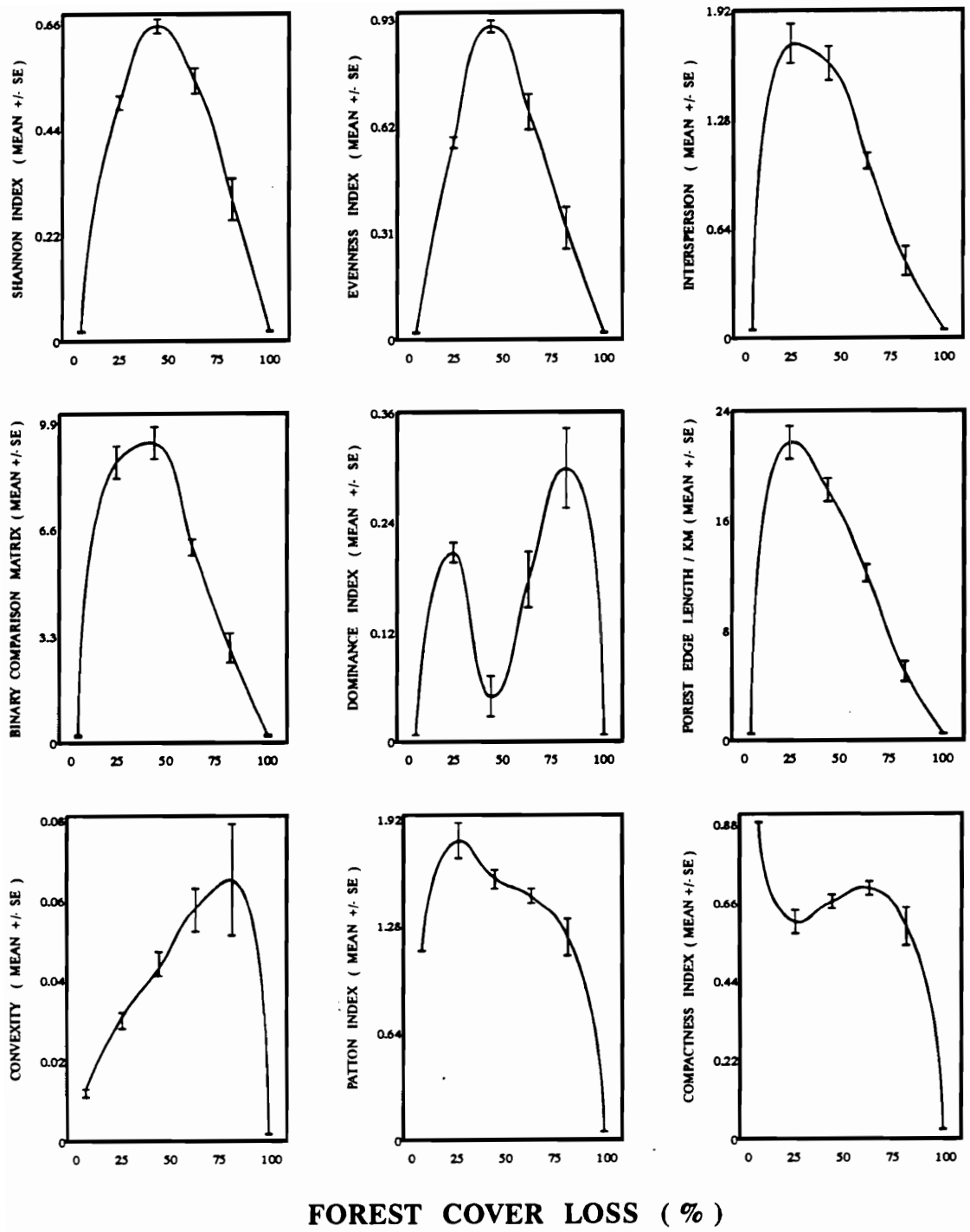


Fig. 15. Changes in selected spatial heterogeneity and edge metrics during the progressive loss of forest cover. Trend lines connect the mean expression values at each stage of forest loss.

The bimodal pattern of the dominance index during deforestation was quite different from the other heterogeneity expressions. Dominance values began at zero, then increased until the landscape was approximately 75 - 80% forested. Dominance was reduced as forest loss continued until the proportion of forest equalled non-forest. Dominance values began to climb once more as non-forest predominated the landscape; values returned to zero when deforestation was complete. (Note: manual calculations indicate that dominance values would continue to increase past the point depicted in Fig. 15, as long as 2 land classes remain. The nature of the modeling scheme dictated the timing of metric measurement at the start of each successive stage. The point at which dominance values peaked in Fig. 15 was approximately 15% forest cover).

The progressive loss of forest cover also brought about changes in the edge characteristics of a landscape (Fig. 15). The changes in forest edge length followed those depicted by interspersion. Edge length increased steadily until a peak was reached at approximately 80% forest cover; then displayed a linear decline with continued deforestation. There was an incremental increase in edge-to-area ratios (convexity) as forest cover receded; more variability was displayed with each successive stage. Once forest loss became complete, the values were zero.

Compactness values fluctuated as forest patch shape was altered during the continued loss of forest. Patton's index also reflected changes in forest shape, but did not fluctuate to the degree portrayed by compactness. Both the Patton and compactness values were quite variable during the beginning and latter stages of deforestation. As deforestation began, the initial pattern of a landscape was established (i.e., as a series of patches, forest clearings, or a landscape with irregular forest boundaries). This landscape pattern was then retained somewhat as forest cover continued to be lost. Variability in values escalated once the majority of forest was removed, reflecting the splitting of 1 patch into 2 remnants and the disappearance of other patches. (Note: the variability in convexity values was also greatest during this last stage of deforestation).

The behavior of the fragmentation metrics during progressive deforestation were

varied (Fig. 16). Percent interior forest and mean patch size decreased exponentially as forest cover was lost. There was a rapid drop in both expressions up until the point when half of the forest cover remained; at that point, values fell slowly. The reduction in patch size reflected not only that existing patches shrunk but also that new patches were created.

Patch density rose steadily until 50% of the forest cover was removed; patches were then lost with each successive stage. Number of forest patches, not pictured, exhibited a similar response. Interpatch distance increased past the 50% deforestation stage, followed by a fairly rapid fall. The variability displayed during the later stages of deforestation reflect the variability in patch disappearance.

Fragmentation index II values reflected the steady decline in distance to non-forest at each interval once the deforestation process began. Fragmentation index I, not pictured, exhibited a similar response. (Note: the values beginning at zero reflect the modeling process. If a buffer area of non-forest had been included around the forest maps, initial values for fragmentation index II would have exceeded those from any successive interval of forest loss. Overall expression response would have then reflected consistent declines during each interval).

The progressive loss of forest cover also brought about changes in connectivity (Fig. 16). Contiguity declined with each stage of deforestation, dropping rapidly after forest loss surpassed the 75% level. At this point, variability in values was 3 times that observed during the earlier intervals. This variability reflects the distribution of the remaining forest area at this stage. If the remaining forest area occurred within the corners of a landscape map, contiguity values were lower than if the forest area occurred within a single map quadrant.

Patchiness displayed a curvilinear trend during deforestation. (Connectivity, not pictured, showed a similar response). Both expressions started at zero with a totally forested landscape and then rose as patches were formed. Values sharply declined between 50-75% forest loss, as patches disappeared from the landscape. When total deforestation was reached, values returned to zero. Patchiness was expected to

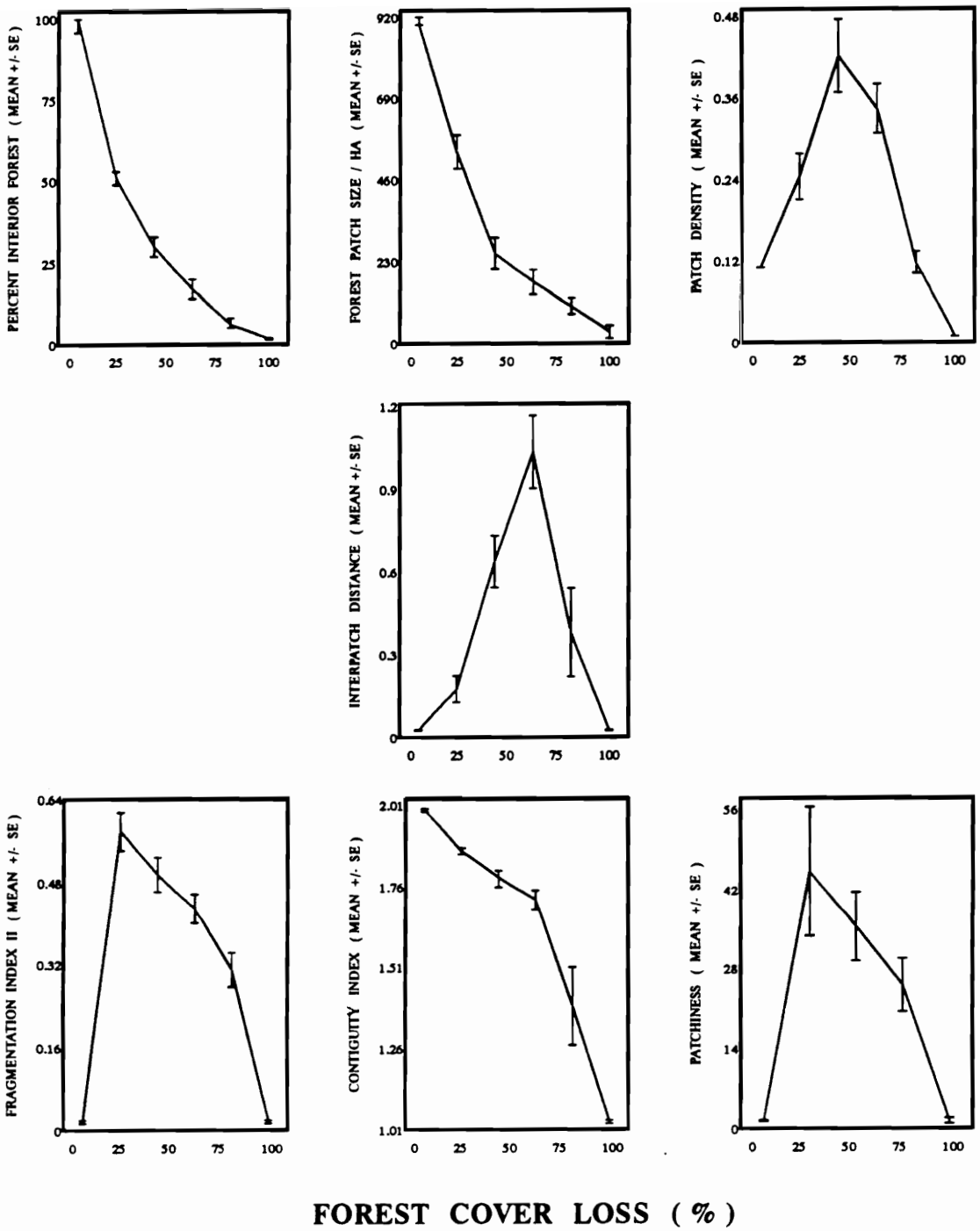


Fig. 16. Changes in selected fragmentation and connectivity metrics during progressive loss of forest cover. Trend lines connect the mean metric values at each stage of forest loss.

consistently escalate until forest loss was complete, reflecting the increasing distance between patches. However, as discussed for interpatch distance, patches are continually disappearing during the deforestation process and forest patches farthest away from others are eliminated as quickly as those patches situated closely together.

There was little change in the behavior of spatial integrity during the initial stages of deforestation, when patches were shrinking and forest openings expanding (the 2 parameters for this expression). With 50% of the forest lost, values plummeted as the number of patches exceeded the number of forest openings. The expression then reversed direction and approached zero as openings coalesced and patches disappeared on the landscape.

Effectiveness of Expressions for Describing Contiguous and Fragmented Landscapes

The results of the discriminant analysis for the differences in landscape pattern are displayed in Table 13. The Mahalanobis distance (157.58) and its associated F statistic (29.22, $P < 0.001$) indicate a significant difference between the contiguous and fragmented groups. The group centroids on the discriminant axis were 11.29 (**Contiguous**) and -146.28 (**Fragmented**). When the group means are well-separated (as in this case), discrimination between the 2 pattern groups will be relatively easy. The magnitude of the standardized differences indicate 15 expressions were effective as discriminators of these landscape patterns. Based on the F-values, 18 expressions were significant in the classification equation. This landscape modification was second (after road addition) in the number of significant expressions resulting from discriminant analysis.

The results imply that percent forest interior was the most effective expression for discriminating between fragmented and contiguously-forested landscapes. There was approximately 3 times more interior forest area measured in the contiguous forest maps than measured on the fragmented landscapes. Contiguity and convexity were the next most efficient expressions for discrimination. Moving from contiguous forest to a

Table 13. Effectiveness of landscape metrics as discriminators of pattern differences between fragmented and contiguous landscapes as derived from 30 maps.

Landscape Expression	Mean I Contiguous[a]	Mean II Fragmented[b]	Mean Differences[c]	Standardized Differences[d]	F Value[e]
Percent Interior Forest [a]	0.650	0.203	0.446	3.874	131.2***
Contiguity Index	1.929	1.715	0.214	2.786	67.9***
Convexity Index	0.020	0.056	-0.036	-2.442	52.1***
Number of Different Classes	1.367	1.706	-0.339	-2.197	42.2***
Mean Patch Size (ha)	652.300	195.800	456.500	2.054	36.9***
Percent Forest Cover	0.798	0.470	0.328	1.997	34.9***
Number of Forest Patches	1.389	4.059	-2.670	-1.929	32.5***
Mean Patch Density (no./ha)	0.154	0.451	-0.297	-1.928	32.5***
Binary Comparison Matrix	5.443	10.121	-4.678	-1.808	28.6***
Interspersion	0.948	1.885	-0.937	-1.785	27.9***
Interpatch Distance (km)	0.103	0.887	-0.784	-1.652	23.9***
Fragmentation Index I	0.032	0.074	-0.042	-1.420	17.6***
Spatial Diversity	36.722	64.647	-27.925	-1.418	17.6***
Total Forest Edge (km)	14.570	20.890	-6.330	-1.314	15.1***
Patchiness Index	15.389	60.518	-45.129	-1.022	9.1**
Landscape Evenness	0.556	0.738	-0.181	-0.916	7.3**
Simpson Index	0.319	0.405	-0.086	-0.906	7.2**
Shannon Index	0.487	0.586	-0.098	-0.816	5.8*
Dominance Index	0.167	0.107	0.060	0.678	4.0
Spatial Integrity	-0.278	-2.176	1.899	0.661	3.8
Fragmentation Index II (km)	1.144	0.385	0.759	0.584	3.0
Connectivity Index	0.028	0.103	-0.075	-0.526	2.4
Patton Index	1.490	1.678	-0.188	-0.446	1.7
Compactness	0.693	0.641	0.052	0.404	1.4

[a] The metric mean representing landscapes with large, continuous forest blocks.
[b] The metric mean representing landscapes with patchy forest cover.
[c] The difference between the means for each metric.
[d] Represents the mean difference for a metric divided by the corresponding pooled standard deviation. Absolute values greater than or equal to 1.0 indicate effective discrimination.
[e] Indicates the significance of a metric in the discriminant equation.
* Significant ($P < 0.05$).
** Significant ($P < 0.01$).
*** Significant ($P < 0.001$).

fragmented landscape pattern, edge length increases while forest cover recedes. Convexity expresses both forest edge and forest cover. The division and loss of forest area that is characteristic of many fragmented landscapes was reflected in the significance of the expressions of mean patch size, percent forest cover, number of patches, mean patch density, interpatch distance, and fragmentation index I. The roving-window heterogeneity expressions, which were important as discriminators in the harvest analyses, were quite valuable for analyzing deforestation. The expressions included number of different classes, binary comparison matrix, interspersion, and spatial diversity.

Not all of the expressions were useful in distinguishing between the contiguous and fragmented landscapes. Compactness, Patton's index, and spatial integrity were virtually the same for both groups. Both connectivity and fragmentation index II were not sensitive to changes in these patterns. Mean connectivity values actually increased in the fragmented group. The low position of fragmentation index II in this list was unexpected (again probably reflecting the modeling process discussed above with respect to this expression). Distance to non-forest was one-third less in the fragmented landscapes than it was in the contiguous setting. Fragmented patterns also made little difference in dominance values, the only spatial heterogeneity expression that was a non-significant discriminator.

General Trends in Landscape Pattern Change

It is possible to make some general observations about how landscape modification changes the different attributes of landscape pattern. Landscape modification from timber harvest and road expansion, along with progressive deforestation, result in an increase in spatial heterogeneity, fragmentation, and edge characteristics of landscape pattern. These are logical, confirmed by the above analyses, but most importantly, quantified as to their extent and relative value. In contrast, changes within the connectivity attributes of a landscape were quite variable and are dependent upon harvest unit arrangement, road placement, and deforestation stage.

Each of the 3 modification schemes were characterized by increases in forest edge length, and by reductions of forest area, forest patch size, percent interior forest, and percent forest cover. Such changes in attributes are often cited as the consequences of forest fragmentation (Li et al. 1993), a concept that has not been universally defined by ecologists. Herein, fragmentation is defined as any process reducing forest coverage and interior forest, and increasing edge length. The creation of forest remnants by forest subdivision may also occur. Numerous openings (e.g., fields, roads, clearcuts) interrupting a continuous forest canopy can also be described as a fragmented landscape.

If the changes within the above expressions are used as a guide, both timber harvest and road expansion contribute to forest fragmentation. The changes in pattern resulting from timber harvest and roads may be temporary, however. Once a stand is harvested, the area will revert to mature forest as succession proceeds. The direction and rate of succession will depend upon the structure and composition of the residual stand, browsing by herbivores, and subsequent management activities (Wigley and Roberts 1994). The influence of roads on landscape pattern may be temporary (e.g., forest roads closed to vehicular travel or reseeded), or permanent if maintained as a travel way (e.g., dirt, gravel, or pavement). Some species may eventually repopulate these areas as they return to previous condition.

Timber Harvest. The extent of fragmentation resulting from timber harvest was dependent on the number, size, and arrangement of harvest units. Higher levels of fragmentation occurred when the landscape was dispersed with small, numerous harvest units than when the landscape contained clustered units. The dispersed harvest scheme, similar to deforestation, influenced landscape pattern through the increase in spatial heterogeneity, patchiness, patch density, number of landscape regions, and forest edge length. There were also declines in the interior forest expression and in distance to non-forest. Forest edge length was greater and patch size smaller on those landscapes with several small harvest units (8-16) than on those with fewer, larger units (1-4). When the

large harvest units were used, there was retention of forest interior and landscape patchiness was minimal, producing a moderately-fragmented landscape. The large unit harvest scheme minimized the amount of forest edge created and provided large areas of contiguous forest.

Harvest schemes with large units may provide other benefits (in addition to minimizing forest fragmentation). Large harvest areas create (temporarily) areas of open habitat for species that are characteristic of early successional habitat and forest edges. Welsh and Healy (1993) found that early successional species requiring seedling-sapling habitat make use of large regenerating stands. These findings along with documented positive relationships between bird species richness and clearcut size (Galli et al. 1976; Robbins et al. 1989; Rudnicky and Hunter 1993), support the species-area relationship predicted by island biogeography theory (MacArthur and Wilson 1967). Maintenance of such habitat for shrubland specialists is a current management goal in the Northeast where many of these species have shown severe population declines (Askins 1994). However, use of large units for timber harvest may be constrained by the delay of forest recovery and sediment production associated with timber harvest (Li et al. 1993).

Although forest edge length was the same for both the dispersed and clustered harvests, the percentage of edge habitat was less in the clustered than in the dispersed arrangement (Fig. 11). Clustered harvest units also minimized the index of landscape patchiness, localizing disturbance and heterogeneity changes within affected quadrants of the landscape.

The dispersed pattern of timber harvest has been used on public forests over the past several decades; its use has been encouraged for creating edge and enhancing early-seral habitat. This harvest scheme was also used for promoting forest regeneration, developing road networks for fire suppression, and for dispersing hydrologic and sediment production effects (Wallin et al. 1994). However, its contribution to fragmentation suggests the need for careful consideration of this harvest scheme prior to implementing it.

Road Expansion. Modifying landscapes by expanding roads also resulted in several of the changes in pattern that were observed for progressive deforestation. The majority of fragmentation expressions were important descriptors for both modifications. The degree of similarity between road expansion and deforestation depended on whether the road network partitioned the landscape or was more invasive in nature (Compare Figs. 13 and 14). A partitioning road network mimicked the early stages of deforestation, exhibiting a reduction in forest patch size and spatial integrity, expanding the number of forest patches, increasing the distance between patches, and increasing the number of landscape regions. An invasive road network was similar to deforestation with respect to reductions in percent interior forest, forest cover, distance to non-forest, and contiguity. However, roads located along the periphery of a landscape, or those placed close to one another, had the least influence on resulting pattern of the different placement schemes modeled. With additional road expansion, each landscape (regardless of road placement) showed increases in spatial heterogeneity and forest edge length as compared to that depicted during initial deforestation.

The changes in pattern described above, considered with the number of significant expressions from the discriminant analysis, suggest that for a given area, road expansion can have a tremendous influence on landscape pattern. When roads are established within a forested landscape, there is a physical disruption to the continuous vegetative community and the structure of wildlife habitat. The creation of forest edge along the road alters microclimate conditions which influences seedling establishment, competitive interactions between individual plants, and changes in forest structure and composition (Chen et al. 1992); these effects persist hundreds of meters into the forest interior. The newly-created edge habitat becomes interspersed with forest and any other land class adjacent to the road. The new edge habitat may serve as a corridor connecting the forest to open habitat, attracting cowbirds into the interior of the forested landscape (Askins 1994). Deleterious effects of edges on fauna from nest predation and brood parasitism are well documented (Chasko and Gates 1982; Wilcove et al. 1986). In areas where

forest fragmentation occurs primarily from roads and powerlines, these corridors encourage high nest predation rates from red fox, striped skunk, and raccoon (Small and Hunter 1988). Nevertheless, many early-seral species benefit from the creation of edge habitat. These species include mule deer, elk, bobwhite quail, bobcat, and other species requiring access to more than 1 habitat type.

Forest Management within a Landscape Context

There are well-known effects on wildlife from changes in pattern arising from landscape modification. Some of these impacts were alluded to in the previous section and others will be briefly mentioned here. For additional detail, the reader may see the Literature Review section which discusses the influence of landscape pattern on wildlife resources.

Changes in habitat patch size have been recognized as a major component of fragmentation (Harris 1984; Lovejoy et al. 1986). Population declines for a number of species have been attributed to reduced "species-area" effects (Ambuel and Temple 1983; Lynch and Whigham 1984; Askins et al. 1990), the phenomenon of species richness increasing (to a limit) as "island" area increases. Such species are absent from all but the largest forest patches. Remnant patches of forest surrounded by open habitat constitute unfavorable habitat for many wildlife species; these remnants also have increased susceptibility to windthrow disturbance and other processes (Franklin and Forman 1987). Although most of this evidence comes from woodlots surrounded by agricultural land, patches isolated from other tracts of forest also show this pattern (Askins 1994). It has been suggested that for some species, fragmented landscapes become population sinks that are sustained by immigration from nearby forest tracts (Robinson and Wilcove 1994).

The influence of creating small forest patches should be considered within a landscape context when evaluating forest management activities. The results of this analysis indicate that management alternatives differ in the extent to which they alter landscape pattern. Resource managers and land planners may contemplate the number,

size, shape, and arrangement of harvest units and the placement as well as length of road expansion, since these factors influence subsequent pattern in different ways. Each can be used to meet different management goals.

The spatial arrangement of timber harvest units can be tailored to meet managerial objectives. Using a few, large harvest units or locating units in clustered formations creates the least amount of edge and maximizes interior forest retention. These harvest schemes represent a viable strategy for maintaining habitat for fisher, ringtail cat, barred owl, pileated woodpecker, worm-eating warbler, and other species requiring large forest expanse. This approach diminishes the risk of disturbance associated with edge creation, concentrates disturbance into localized areas, and reduces the need for extensive road networks for accomplishing harvest objectives. If maintaining large forest patches is critical for management of interior species, then these 2 harvest schemes represent a preferred harvest approach.

In comparison, management objectives may specify providing wildlife habitat for species requiring early seral stages, shrubland habitat, or a diversity of vegetative communities within an extensively-forested landscape. Harvest schemes using several, small units or those schemes which disperse units across a landscape improve spatial heterogeneity, interspersion of land classes, and edge development. This benefits bobcat, coyote, striped skunk, gray fox, and American woodcock; it also favors species that thrive in disturbed environments (e.g., blue jay, raccoon, and opossum). The edge zone surrounding each dispersed unit increases the amount of available edge habitat, providing foraging areas for species such as wild turkey and white-tailed deer. Newly-harvested areas relatively far from neighboring agricultural areas may present few opportunities for brood parasitism, however high management costs may be associated with this harvest scheme if establishing additional road networks is required for access.

The placement of road networks should also be considered within landscape management. Many species avoid roads: for example, mule deer, elk, mountain lion, and grizzly bear (Beringer et al. 1990). Road expansion has been a leading factor affecting

black bear behavior and habitat quality; the potential for human-bear encounters increase with road density. Brody and Pelton (1989) hypothesized that black bear response to thresholds in road density may result in geographical shifts in home ranges. The impact on bear and other wildlife populations should be considered prior to road construction; roads allow hunter access and illegal hunting in particular regions of the United States.

Resource managers may use cartographic modeling to limit the isolating effects that roads may have on a forested landscape and to reduce the disturbance to the surrounding environment (Bennett 1991). There are ways to locate optimally roads and corridors, limiting the influence on both landscape pattern and wildlife habitat. They may be placed along the periphery of forested landscapes, avoiding centers of forested tracts when possible. Consolidating roads into a single landscape quadrant would retain large areas of interior forest, reduce the number of openings interrupting large expanses of forest, and provide roadless areas in the remaining quadrants of the landscape that are required by many large mammals (Brody and Pelton 1989). Logging roads placed near critical habitat, if closed and reseeded after harvest, permit regeneration and reduce potential cowbird foraging habitat. Careful consideration should be given before partitioning forested landscapes with road networks; as discussed above, road length may be held in check using large or clustered units.

Discriminant Analysis and Landscape Pattern Analysis

Discriminant analysis has been used to interpret a wide range of community patterns including streams, deserts, and forests (Ludwig and Reynolds 1988). This procedure was used for characterizing woodpecker nesting habitat in Virginia (Conner and Adkisson 1976) and for identifying habitat selection attributes for desert avian communities in Arizona (Rice et al. 1983). The analysis is this study has proven to be useful for identifying a subset of quantitative variables that best reveal the differences among landscape patterns. Using standardized differences and the F-statistic as guidelines, it was possible to rank the effectiveness of expressions for classifying landscapes and for

quantifying the patterns that result from modifying landscapes. Whether a manager is monitoring the impacts from progressive deforestation or evaluating potential locations for timber harvest and roads, knowledge about which landscape expressions are sensitive enough to detect those landscape changes is important. The relevance of the expression to 1 or more resources is also important.

Identifying sensitive expressions also fosters an understanding of the underlying attributes that characterize landscapes. For example, Table 13 listed metric effectiveness for discriminating between fragmented and contiguous forest landscapes. The difference between these 2 groupings is much more than simply isolation of patches. These patterns differ in the amount of forest interior present, the degree of forest contiguity, the characteristics of forest edge, the mean size of forest patches, and in several spatial heterogeneity attributes. Such understanding can be useful for resource managers by reducing the number of expressions required for landscape analysis. Considerable time and money may be saved by limiting the analysis to effective metrics that may be of ecological relevance. Using numerous metrics introduces the risk of including highly-correlated variables in addition to the high costs of data collection, storage, and retrieval.

The dominance, evenness, and compactness indices were not significant in the discriminant analyses. These expressions were not sensitive enough to characterize the pattern changes examined in the test maps; however, these metrics have proved useful for descriptive purposes in other studies (O'Neill et al. 1988; Hoover and Parker 1991; Eastman 1992). Nevertheless, careful consideration of dominance, evenness, and compactness should be made prior to their inclusion for landscape analysis.

Three landscape metrics were consistently significant ($P<0.001$) during the different modification schemes. Binary comparison matrix, number of different classes, and percent interior forest were sensitive descriptors of pattern; their inclusion for any landscape analysis is highly recommended. Other metrics that were regularly found to be significant in the discriminant analyses included spatial diversity, mean patch size, interspersion, fragmentation index I, total edge length, convexity, and contiguity. I recommend the

inclusion of these expressions for general use and for preliminary examination of landscape pattern involving modification categories different from those tested here.

Certain metrics were effectively used with a specific landscape modification in addition to those discussed above. Those metrics recommended for detecting pattern changes resulting from harvest arrangement include fragmentation index II, connectivity, and patchiness. Metrics recommended for describing road modification include patch density, number of forest patches, Shannon and Simpson indices, spatial integrity, fragmentation index II, and interpatch distance. Those recommended for describing deforestation are forest cover, number of patches, patch density, and interpatch distance.

Selecting a metric should not be based solely on the level of significance presented in Tables 8, 9, 11, and 13. An understanding of how the metric behaves (i.e., why it is useful for detecting certain pattern changes), is of equal importance. For example, fragmentation index I, patch density, and mean patch size were effective metrics for road modification because each detects the partitioning effect that often occurs in roaded landscapes. The roving-window metrics were effective in expressing the pattern changes relating to harvest arrangement and road placement because they can detect local area variation. The Shannon, Simpson, evenness, and dominance indices were not effective for discerning differences among harvest alternatives because the proportion of forest to non-forest area remained the same within the different harvest alternatives. Selecting a metric should also consider the initial underlying pattern of the landscape. If a landscape contains forest patches, or if the potential for patch formation exists from modification, then using patch metrics is warranted (patchiness, number of patches, patch density, interpatch distance). However, if the landscape comprises expansive contiguous forest cover, then using other metrics (e.g., percent forest interior, forest cover, fragmentation index II, and contiguity) is preferable.

It is apparent that selecting landscape pattern descriptors should include expressions from more than 1 category. For example, from the discriminant analyses it is seen when forest fragmentation occurs, it may be analyzed with the specific fragmentation

expressions as well as half of the spatial heterogeneity and edge expressions.

Detecting pattern changes resulting from harvest suggests it may be done well by several spatial heterogeneity expressions as well as selected expressions from the edge, fragmentation, and connectivity categories. (Harvest unit arrangement however, was an exception. There was no indication for including edge metrics for describing pattern arising from different arrangements). For detecting the effects from adding roads to a landscape, the most efficient descriptors of change in pattern included a predominance of heterogeneity and fragmentation expressions, along with members of the edge and connectivity groups. Limiting landscape analysis to 1 or 2 categories of expressions loses the opportunity for obtaining useful information offered by other relevant expressions.

In addition to identifying metric usefulness as discriminators of landscape pattern, discriminant analysis allows the simultaneous effects of multiple factors to be studied. During analysis, a discriminant function transforms the suite of metrics for a landscape into a single discriminant score. Such a score is the projection of data points onto the discriminant function axis, and can be thought of as a way of collapsing a multivariate problem into a single-variable problem (Davis 1986). Following an initial discriminant analysis (such as that described earlier in this Chapter), unclassified maps representing proposed management alternatives can be brought into the analysis. A discriminant score can be calculated for each alternative and plotted onto the discriminant axis with the original classified groups. Based on the position of the score for the alternative along this axis, each may then be visually ranked in reference to the established 2 groups.

For example, Fig. 17 depicts the positions along the discriminant axis for the roadless and heavily-roaded landscape groups used in Table 11. The scores for each landscape map within those groups are also plotted. (The discriminant index, **R0**, is the point along the axis that is exactly halfway between the group means). The position of the 3 alternatives on the axis relative to the group means can be used for evaluating the differences among the 3 alternatives and their potential for landscape modification. The resulting pattern of Alternative **A3** has a discriminant score close to the periphery of the

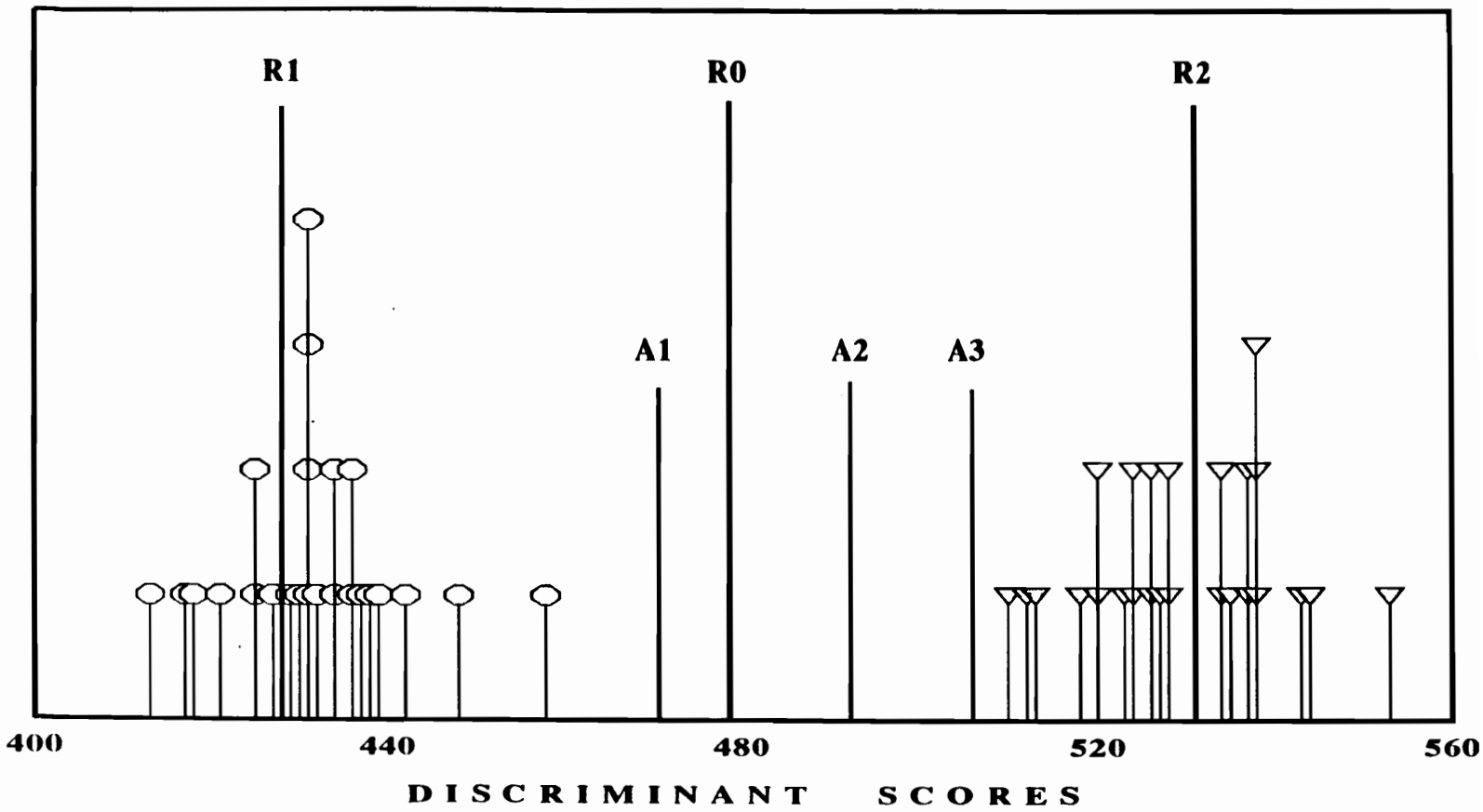


Fig. 17. Projection of landscape maps onto the discriminant function axis. R1 is the multivariate mean for the roadless group, R2 is the multivariate mean for the heavily-roaded group. The discriminant index, R0, is the point along the axis that is exactly halfway between the centers of the 2 groups. The road placement alternatives are identified as A1, A2, and A3.

heavily-roaded group, while the changes in pattern from Alternative **A1** are closest to the roadless landscape group. Alternative **A2** represents a potential landscape modification intermediate to the other 2 alternatives. If the desired management objective is to achieve the least disturbance on the landscape, then the preferred alternatives would be A1, A2, and A3 in that order. Using discriminant analysis can be useful to a resource manager looking for ways to quantitatively evaluate alternatives based on several pattern criteria. Discriminant analysis might have powerful potential for facilitating resource management decision-making.

Reducing the List of Landscape Pattern Expressions

The following discussion, based on the analyses of Chapter 2 and 3, provides a basis for reducing the list of landscape expressions for subsequent analyses.

There are pairs of expressions that appear to yield equivalent information; one could safely use either expression without losing information. The Shannon and Simpson indices, and the number of forest patches and patch density expressions, are such pairs of expressions. The members of these pairs were highly correlated and were rated similarly in the discriminant analyses the majority of the time. Ease of computation is equivalent between the 2 pairs of expressions. It is an arbitrary decision then to drop the Shannon index and number of forest patches expressions and retain the Simpson index and patch density.

Of the highly-correlated roving-window expressions, it appears that omitting spatial diversity and number of different classes, while retaining the binary comparison matrix and interspersion metrics would not result in a significant loss of information regarding landscape pattern. Spatial diversity uses the number of different classes within a 9-pixel window, averaged by dividing the total number of land classes on the map. The binary comparison matrix metric includes the number of different classes with the frequency of those classes in the window, while interspersion values are based on the number of surrounding cells differing from the center pixel.

The Patton and compactness indices are another pair of highly-correlated metrics. These 2 patch shape metrics behaved comparably during modification modeling and often rated insignificant in the discriminant process. It would be useful to retain 1 shape metric for further analysis, however. Patton's index routinely scored slightly higher than compactness in each discriminant analysis; for this reason, Patton's index will be retained and compactness will be removed during further analyses.

The dominance index was consistently ineffective for characterizing landscape modification trends. Dominance and landscape evenness were negatively correlated. Although evenness was an effective descriptor for only 1 landscape modification trial, it will be retained in place of dominance.

With many of the pattern expressions, it is possible to envision the pattern of a landscape from the metric's value alone (i.e., the metric is information-preserving). For example, it would be difficult to envision patchiness values with the relative distance between forest patches and directly apply this information to habitat evaluation. In comparison, mean interpatch distance can be easily related to species-specific habitat evaluation (e.g., an interpatch distance of 900m poses an obstacle for an immigrating salamander but not for a gray fox). The information-preserving quality of mean interpatch distance, along with its efficiency as a pattern discriminator, suggests maintaining this expression and eliminating patchiness from additional analysis.

A similar argument can be made against the continued use of spatial integrity. While many expressions may have the same value derived from different landscapes, the identical spatial integrity values repeatedly described landscapes with very dissimilar patterns. Based on its derivation (Eq. 23), the number of patches is subtracted from the number of forest openings on a landscape. Thus, forested landscapes with a spatial integrity values of -4 may describe 4 forest patches or 20 patches with 16 forest openings. Although spatial integrity was useful when applied to continuously-forested landscapes perforated with several openings, this pattern is also described by fragmentation index II which describes the average distance to non-forest pixels. Because of the disparity in spatial

integrity values and landscape pattern description, this expression will be dropped from further study. There was a problem computing the connectivity index within a GIS environment, centering on how to identify a forest patch and its links to other patches. In this analysis, a patch link was defined as 1 or more diagonal forest pixels connecting 2 forest patches; this identified any diagonal links as single-pixel width connections. To determine contiguous groupings of forest pixels in an image, pixels are assigned to the same forest patch if they touch in any cardinal direction. If diagonal pixels are included in the grouping process, pixels are classified into the same patch if they touch in any possible direction (e.g., N, E, S, W, NW, NE, SW, or SE). Patch connections are no longer identified as links, but are considered part of a forest patch.

The inability to automate a procedure for discriminating between a link and a contiguous pixel is an important problem that needs further study. A sophisticated algorithm is required to override existing grouping functions, allowing patch link identification from any direction using a specified pixel width. Although the connectivity index will be dropped from further analysis, the results presented in Chapters 2 and 3 contribute to the understanding of this expression's behavior. Contiguity, another measure of forest connectedness, will be retained in place of connectivity. This highly effective discriminator is not constrained by the requirement that forest patches or links be identifiable, but uses the location of forest pixels.

SUMMARY

The influence of timber harvest, road expansion, and deforestation and their influence on landscape pattern was examined. Cartographic modeling and the suite of 24 landscape pattern expressions were used to evaluate changes in the spatial patterns of land classes resulting from these modifications. Discriminant analyses were used to assess the effectiveness of landscape pattern expressions as discriminators of landscape pattern change.

The process of cartographic modeling began by observing the differences arising in pattern from variation in timber harvest number, size, and arrangement. Timber harvest schemes with few, large units and those in a clustered arrangement lead to a lower degree of fragmentation than those schemes with several small units or those dispersed across the landscape. Roads tended to have either an invasive or partitioning effect on landscape pattern. Roads located on the periphery of a landscape, or those consolidated close to one another, did not alter landscape pattern as much as roads dispersed across a landscape. The influence of progressive deforestation on changing landscape pattern was cyclical, with metrics changing direction at the half-way point of deforestation. Comparison among the 3 landscape modifications indicated that both timber harvest and road expansion portray patterns similar to the early stages of deforestation.

The change in wildlife habitat associated with forest management practices was discussed within the scope of general landscape modification. Landscape pattern characteristics can be changed from management activity. These activities alter landscape pattern, forest composition, and structure, influencing conditions for species and for communities. Understanding these changes can lead to optimizing the spatial layouts of management activities based on landscape perspectives for accomplishing objectives for specific species. Temporal aspects of such changes are only suggested in this dissertation.

Management activities and natural processes will continue to modify landscape pattern, and the findings of this chapter may provide insights into the consequences of such modification. By understanding the impact of landscape modification, resource managers can direct those activities that will produce the greatest benefit to wildlife.

Discriminant analysis identified metrics that efficiently detected pattern changes resulting from landscape modification. A group of the expressions was found to be highly effective for several of the modification schemes; a few expressions were rarely significant. The list of effective expressions varied for the different landscape modification schemes, providing information on how best to monitor pattern changes using a selection of sensitive metrics.

The list of landscape pattern expressions was reduced based on the discriminant analysis results, duplication of information with other expressions, and their ability to preserve pattern information. The remaining metrics are examined further in Chapter 4, The Influence of Spatial Scale on Landscape Pattern Analysis.

CHAPTER 4

The Influence of Spatial Scale on Landscape Pattern Analysis

INTRODUCTION

A fundamental theme of landscape ecology is how spatial pattern develops following land management, and the consequences of that pattern upon the abundance and dynamics of populations and ecosystems (Levin 1992). Landscape ecology includes the study of the patterns in communities and ecosystems, and the ecological processes that affect those patterns. Species and populations respond to spatial pattern in various ways. For example, some use certain areas for feeding and reproduction, some avoid areas entirely. The structure and dynamics of communities, composed of these populations, are strongly influenced by the variation in patterns over large regions.

The very nature of "landscape" and recent interest in landscape ecology has prompted an awareness of the potential importance of spatial scale. Landscape pattern, its perception, analysis, and description, is influenced by the scale of observation (e.g., a watershed, a state, a region). Ecologists have recognized how the sampling scale can alter the description of species distributions (Weins 1989a). Other authors have expressed interest in scaling effects such as total landscape area and the size of minimum mapping units (Meetenmeyer and Box 1987; Holling 1992; Levin 1992). If spatial scale influences the outcome of pattern analysis, then the resource management and habitat assessment resulting from such analyses may likewise be affected.

The analysis of landscape pattern occurs at several scales: global (e.g., climate changes), continental (e.g., acid precipitation), regional (e.g., landform and elevation), and local (e.g., edaphic factors and rare communities). Landscapes described at 1 scale are not likely to be the same as those described at another scale, and information may be lost as coarse levels of resolution are selected. How to integrate landscape measurements

made at disparate scales (Musick and Grover 1991) and how to extrapolate information from 1 spatial scale to another remain a problem. Little attention has been given to determining appropriate scales of study (Cullinan and Thomas 1992). There is a fundamental need to consider how the scale of examination may limit or add to observable relationships (Allen and Starr 1982; Wiens et al. 1986). Quantitative analysis of these relationships may help clarify these concerns.

The term "scale" is used in many contexts. The cartographic scale of geography is the ratio of distance on a map to distance on the ground. In ecology however, scale often refers to either the space (or time) over which a measurement is made. This concept of spatial scale is characterized by both the total area encompassed by a landscape (context or extent) and the resolution level of the individual units of observation (e.g., precision of measurement or smallness of grain).

Spatial resolution reflects the level of detail inherent in landscape data (i.e., the smallest discernible spatial unit). Image resolution describes the area covered by pixels in raster GISs; for vector GISs, resolution refers to the precision of the spatial coordinate system. I address resolution in this chapter. Herein, I use "fine resolution" to imply small map pixels and "coarse resolution" to refer to large map pixels. These are relative expressions that I attempt to clarify in each situation.

The objective of this chapter is to examine how changes in spatial resolution influence the results of landscape pattern analysis. The analysis will describe the sensitivity and stability of metric behavior using trend curves. First, I explore the changes in pattern metrics as a function of resolution. Second, I examine the variability in metric behavior using semivariograms. Next, I identify how the spatial scale and the underlying pattern of a landscape, in addition to other factors, influences landscape metrics. Finally, the management implications for wildlife habitat evaluation and resource management are discussed.

METHODS

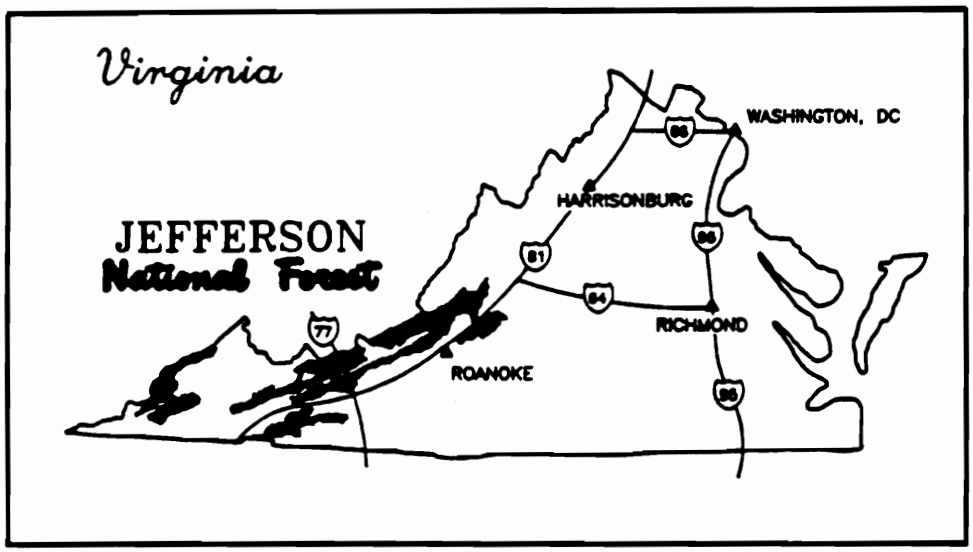
Map Selection and Data

The Craig Creek watershed of the Blacksburg Ranger District, George Washington-Jefferson National Forest, Virginia, was selected as the site for this analysis (Fig. 18). The watershed lies within the Craig Springs, Glenvar, Looney, McDonald's Mill, and Newport 7.5-minute quadrangles (U. S. Geological Survey, 1987). The 45,000 hectare site contains 22 forested compartments (i.e., areas that are designated as official National Forest management units), and is surrounded by both agricultural and urban land classes not within the National Forest. The watershed was selected for analysis because it supported a broad array of landscape conditions, and because there were recent updates of both spatial and forest inventory data on the watershed.

The compartments vary immensely in both the number of vegetation types, the number of land classes per unit area, and in the arrangement of those entities. This variation was used as the criterion for their selection prior to analysis. Compartments with few polygons/compartmental area were classified as "simple"; those compartments with many polygons/compartmental area were classified as "complex". Compartments 1096, 1099, and 1102 were selected to represent simple landscapes. In this group, the average number of polygons per compartment was 9; the mean area of the compartments was 1389 ha. Compartments 1078, 1093, and 1095 were selected to represent complex landscapes. In this group, the average number of polygons was 45; the mean area of the compartments was 1597 ha).

GIS Analysis Process

The watershed was digitized from district maps (scale: 1:24,000) using the process described in Chapter 2. The resulting landscape map based on a 30m pixel (0.09 ha) resolution became the baseline reference map for making comparisons with alternative maps of coarse resolution (pixel size > 30m). Producing the base map at the finest



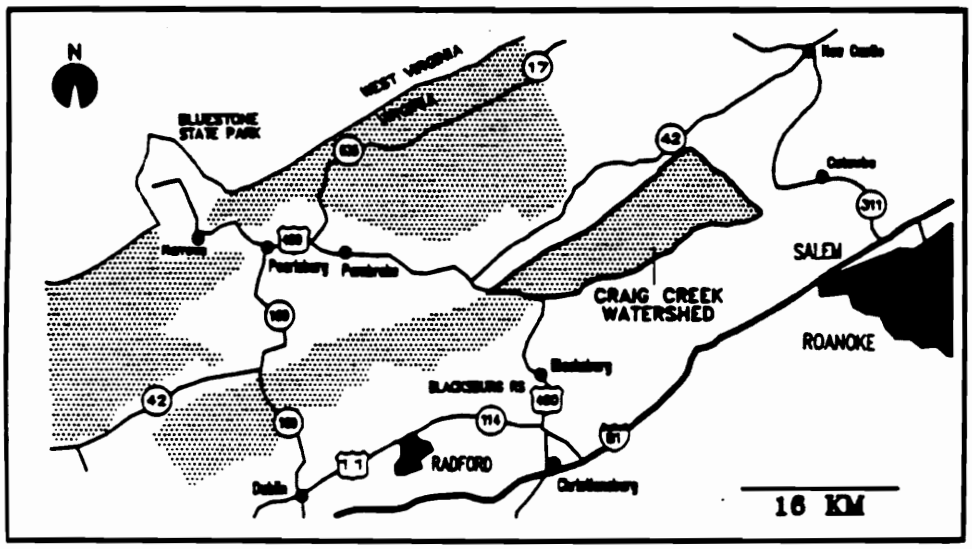


Fig. 18. Location of the forested landscapes used in the pattern and spatial scale analyses.

resolution available made it possible to take area samples at regularly increasing scales during the sensitivity analyses, thus providing a source of data that was virtually continuous over a range of spatial scales.

Cartographic modeling within Idrisi (Eastman 1992; Version 4.1, Release 1993) was used to model changing spatial resolution. Using the Idrisi CONTRACT command, the baseline map was repeatedly generalized at a sequence of pixel sizes (30m, 60m, 90m, 120m.....420m) using 30m increments. "Generalized" means that an image is reduced by the number of rows and columns while pixel size is enlarged simultaneously. The sampling process used for generalizing a landscape image was as follows. To derive a 60m map image, every second pixel was selected from the first, third, fifth, etc. rows. Because the pixels expand in both width and length during generalization, the second row is omitted entirely and replaced by the new, enlarged pixels selected from the first row. This process continued for row 3, omitting row 4 entirely, and proceeded until the *nth* row was processed. The retained pixels became the basis for the new landscape image, while all other pixels were dropped. Fig. 19 depicts this process as a map is generalized, creating a 60m image. Each succeeding image was derived from the 30m baseline reference map (and therefore was independent of each preceding image). This procedure provided a consistent means of generalizing a map while controlling pixel size.

The landscape images were analyzed at each level of spatial resolution for the 15 expressions of landscape pattern remaining after selections were made as described in Chapter 3. The cartographic modeling and GIS analysis operations were completed using Idrisi functions (Appendix 1) and a Turbo Pascal program that I developed (Appendix 2).

Statistical Analysis

Graphical displays were created for each landscape metric at each level of spatial resolution. Regression analysis (based on the sum of least squares method) was used to generate trend lines illustrating the relationship between each expression and spatial resolution. Trend lines were used solely to provide a useful summary of the data and have

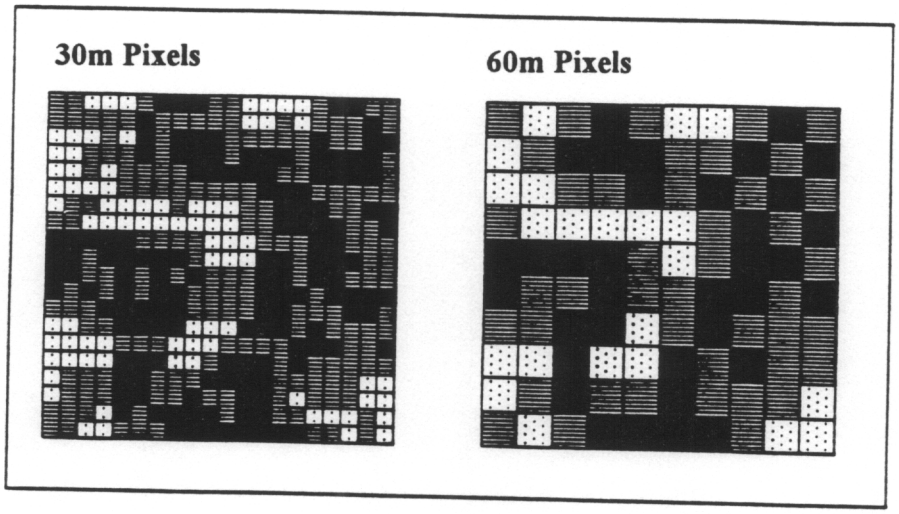


Fig. 19. Schematic illustration showing the increase in pixel size as spatial resolution becomes coarse (Adapted from Turner et al. 1989).

no statistical significance. The plotted lines having the best fit (linear, exponential, logarithmic, or power) were selected on a visual basis. Trend lines were omitted when no single line best summarized the data.

Semivariograms were created for each pattern metric. A semivariogram mathematically expresses the way in which the variance of a property changes as a function of spatial scale; it summarizes the spatial variation in magnitude and general form of the data (Oliver and Webster 1986). This function relates semivariance to the spatial distance between measurements, and provides a concise and unbiased descriptor of the scale of spatial variability (Curran 1988).

Each semivariogram consisted of a plot between spatial scale and the semivariance computed for a particular metric. The semivariance, y_h, is defined as:

$$y_h = \sum_{i=1}^{n-h} (Z_i - Z_{i+h})^2 / 2n$$

where is h is the distance over which y_h is measured, Z_i is the value of an expression taken at resolution level i (e.g., 30m), Z_{i+h} is another measurement taken h levels away (e.g., 60m), and n is the number of observations used in the estimate of y_h. Semivariance is one-half of the mean squared differences between metric values separated by a level h apart. The semivariogram becomes a plot of y_h as a function of h. Semivariance was computed for each expression at 30m intervals from 30m to 420m.

At 30m resolution, the value of a metric is compared to itself and the semivariance is zero. At 60m and beyond, semivariance rises when the comparisons are increasingly different from those obtained at the 30m resolution. This increase continues until the values are so far apart that they are not related to each other and their squared difference becomes equal to the average variance of all samples. The graphical line of y_h levels off (Fig. 20) and becomes flat (the "sill"), representing the maximum level of semivariance

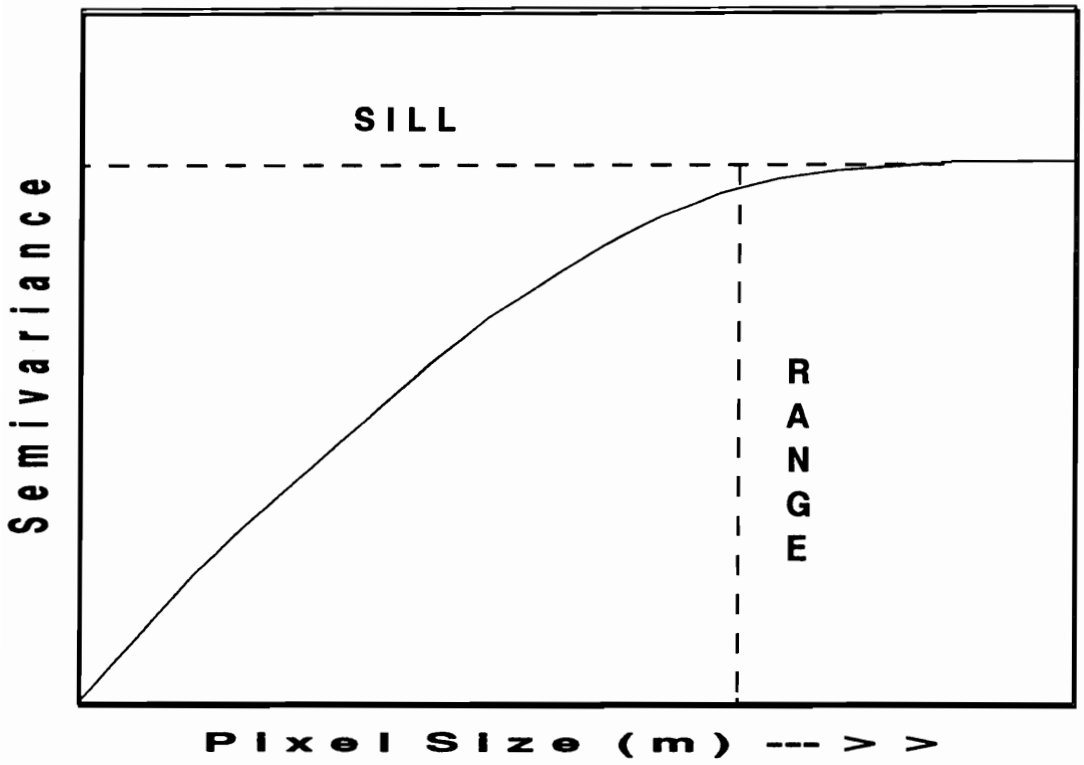


Fig. 20. General form of the semivariogram. The sill represents the maximum level of semivariance observed. The range indicates the place on the x-axis where semivariance reaches approximately 95% of the sill.

observed. The point on the X-axis where semivariance reaches approximately 95% of the sill is called the "range", which serves as an estimate of area similarity (Yost et al. 1982). At distances closer than the range, the values are considered scale-dependent (i.e., they become more alike with decreasing distance between them). Scale-dependence describes the relationship between the magnitude and variability of a process and the scale of measurement. Beyond the range, expression values are considered scale-independent (i.e., the values do not reflect the scale of measurement). The range and the sill contribute to the general form of the semivariogram, which relies on pixel area, the spacing of those pixels, and the pattern expression computed based on landscape characteristics.

RESULTS AND DISCUSSION

The Influence of the Spatial Scaling Process

The spatial scale selected for analysis influences expressions of landscape pattern in a variety of ways. Before these results are presented, insights into the relations of scale to many landscape metrics may be of interest. There were several factors that influenced pattern metrics repeatedly, often in different ways. The first group of factors was related to the underlying pattern of the landscape; these included the number, size, shape, and distribution of land classes. These will be described with specific pattern expressions and in the ensuing discussion section. The second group of factors however, reflected the sampling procedure that was used in the spatial scaling process. These included the sampling intensity and the sample unit sizes of the analysis, and are discussed below.

Sampling Intensity. During the spatial scaling process, each succeeding map image had a reduction in the number of pixels representing the pattern of the original landscape. Each landscape image contained only 25% of the total number of pixels contained in the previous map image. The reduction caused many of the problems

inherent to small sample sizes (e.g., the smaller the sample, the less representative it may become of the sampled landscape). Accuracy (relative to 30m pixels) of many expressions was closely linked to sample size.

At each new level of resolution, a subset of pixels was selected to represent the landscape. As each sample became a smaller and smaller subset, it no longer accurately reflected or represented the original landscape pattern (i.e., there was a lack of similar characteristics of the landscape from which they were drawn).

The variance associated with the estimates of expression means appeared inversely proportional to sample size (O'Neill et al. 1991). Variance increased as sampling intensity decreased with the increase of pixel size. There was a decrease in precision as the pixels become larger. In addition, the reliability of the semivariance estimates decreased as the resolution was reduced. This occurred because the number of observations used to estimate semivariance decreased with increasing distance (Cohen et al. 1990).

The algorithms used to generalize images can be viewed from a perspective of systematic sampling. Each pixel did not have an equal probability of being selected once the resolution level was chosen (e.g., to create an image with 120m pixels, every fourth pixel was selected, regardless of the area it occupied). Fluctuations in pattern descriptions for the new images were dependent upon those pixels retained and their area magnified, and upon those pixels selected for removal. In a very small area, the selection of the initial pixel can have a profound effect on the semivariogram.

Sample Unit Size. As spatial scale was changed and pixel size enlarged, sampling intensity was reduced. The interplay between pixel size and the amount of landscape area it represented at each stage influenced several aspects of landscape description.

When the pixels were small in relation to the size of forest patches (or land class segments), the landscape was sampled with pixels small enough to lie wholly within those patches and not across patch boundaries. The result was that there was little change in both distribution and proportion of patches; landscape pattern description remained

relatively the same as the preceding level. When small pixels were used, the likelihood increased that the patch would be retained into the next resolution level, since patch size exceeded pixel size. As pixel size increased, but remained small enough so that there was a negligible chance that the pixel would cross patch or land class boundaries, a shift in land class distribution occurred with minor changes in the proportions of those land classes.

As pixel size reached and exceeded patch size, retention of patches and land classes became unpredictable. There was either a loss of patches or a coalescence of like-patches with increased pixel size. This loss or gain resulted in large variance which was observed at the coarse resolution levels. With large pixels, the spatial relationships among the new land classes were altered. A marked loss in landscape detail occurred with large pixels and continued as pixel size exceeded the known median patch and land class areas.

The increase in pixel size at coarse resolution levels set the minimum size for land class segments (i.e., the minimum size could not be less than the pixel size). The increase in the minimum size at each new resolution level influenced both the spatial heterogeneity and the patch expressions through either causing a loss or magnifying landscape features. The center-to-center distance between contiguous pixels was doubled with each pixel size increase; the minimum edge-to-edge distance between patches was also established by pixel size (i.e., the interpatch distance between 2 patches cannot be less than the current pixel size, since it defines their separateness).

The relationship between spatial resolution, sampling intensity, and pixel size are presented in Fig. 21. These sampling processes, in conjunction with other factors, contributed consistently to the analyses made of the effects of changing spatial scale. The following discussion attempts to illustrate and re-emphasize these processes.

The Influence of Spatial Resolution on Metric Behavior

·The progression of the evenness and Simpson indices across the range of resolution levels is depicted in Fig. 22. The continuity of these expressions during changes in spatial scale was unlike that of any of the remaining expressions. The Simpson index remained

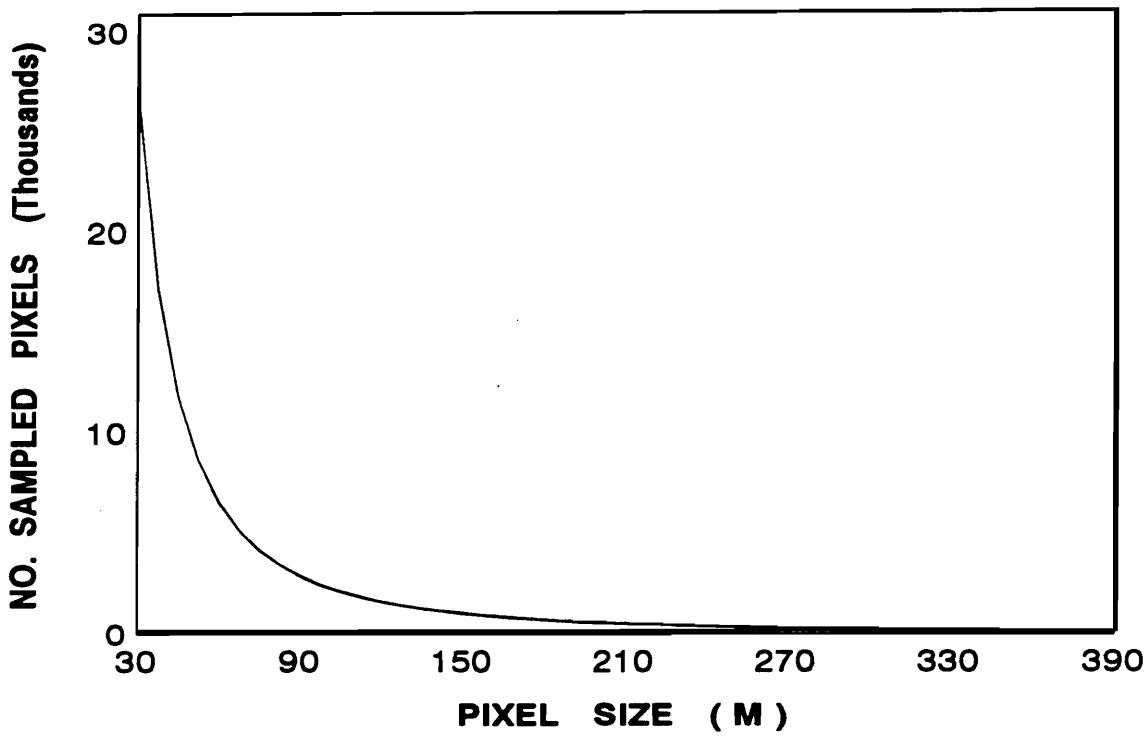


Fig. 21. The relationship between spatial resolution (pixel size) and sampling intensity (number of sampled pixels) within a map.

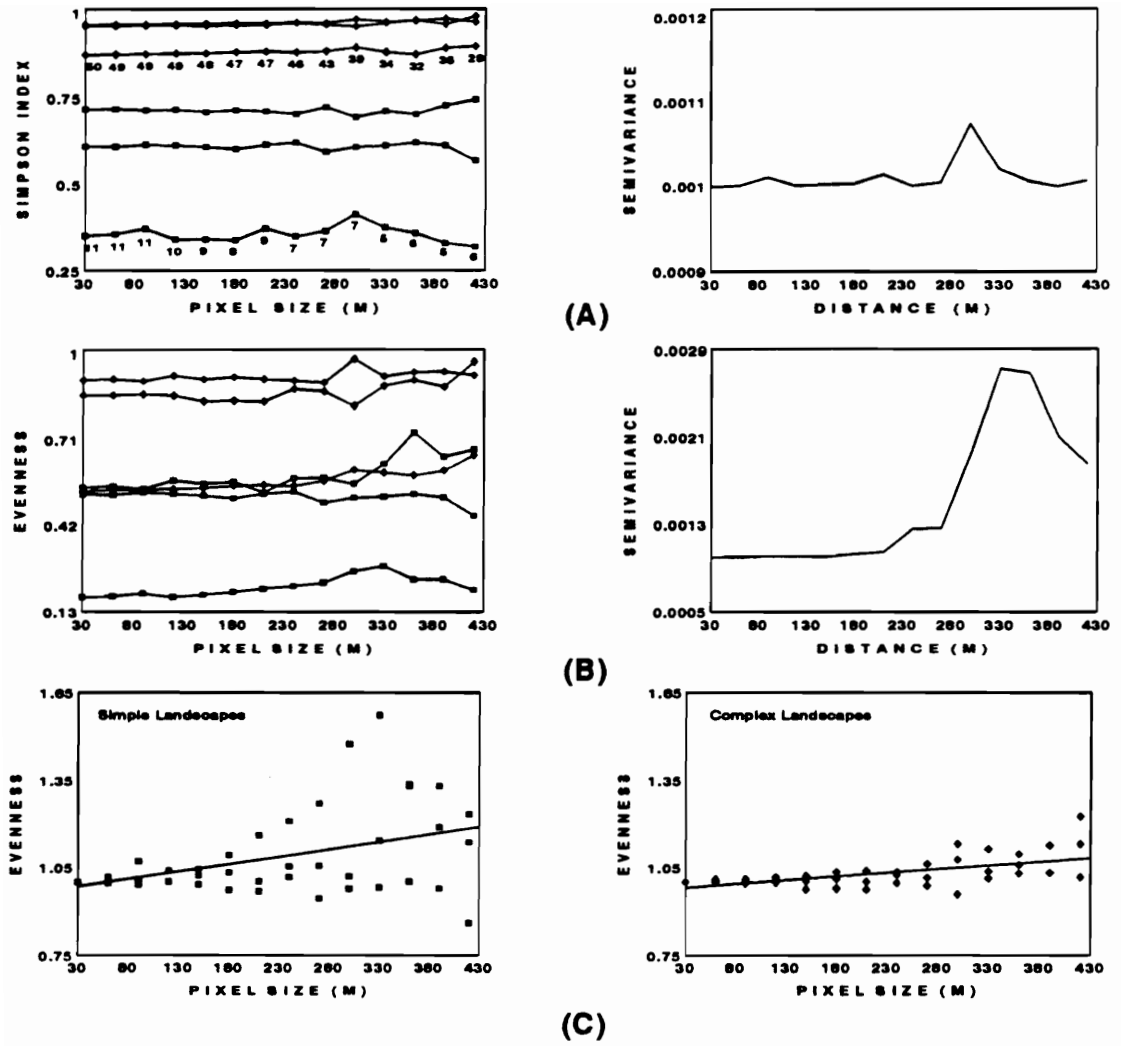


Fig. 22. The influence of spatial resolution on the Simpson and evenness indices. Plot markers identify simple (square) and complex (diamond)landscapes. (A) Simpson values with corresponding semivariogram. The number of land classes present are indicated below selected compartments. (B) Evenness values with corresponding semivariogram. (C) Differences in evenness behavior for simple and complex landscapes. Trend lines are derived from regression analysis. Vertical axes are scaled to 1.0 for direct comparison.

fairly constant over several resolution intervals, with small changes during the 270m - 330m period. Evenness remained stable until a pixel size of 300m and then became quite variable with each reduction in resolution. The values became dependent on the variety of conditions encountered at each change in scale. Large pixel sizes either omitted some land classes or resulted in large proportional changes within the remaining classes.

The areas of the curve which approximate horizontal lines on both semivariograms indicate intervals where the spatial pattern is not scale dependent (i.e., values at 30m and 210m are as similar as those measured at 60m and 90m). These horizontal lines suggest stability of the indices. The abrupt rise in semivariance for evenness following 270m indicates that values measured at these gross resolution levels are becoming more and more different. This marked reversal of slope after reaching a maximum is uncommon (Oliver and Webster 1986); it suggests further fluctuation in evenness values may occur beyond the 420m level.

The vertical axes of Fig. 22C (and subsequent figures where noted) were scaled by dividing metric values (EXP) at each resolution level by the values obtained at the 30m resolution level (EXP_{Nm}/EXP_{30m}). Scaling the axes to 1.0 allowed direct comparisons (i.e., making them unitless) across several different metrics.

The influence of spatial scale on expression behavior was influenced by the degree of landscape complexity existing on a landscape (Fig. 22C). For simple landscapes, the evenness metric becomes unpredictable as pixel size becomes quite large: 1 large pixel contributes to proportional evenness, the next pixel (perhaps very different) may reduce the index value. In contrast, there was a consistent trend for those values computed from the complex landscapes. To understand what is occurring here, it may help to revisit the Simpson graph (Fig. 22A). The number of land classes remaining at each resolution level indicates the influence that existing patterns have on the behavior of these metrics. Increasing pixel size can result in losing 1 land class from a simple landscape with 9 land classes, a loss of 11% (1 / 9). However, the omission of 1 land class from a complex landscape having 45 classes represents a loss of only 2% (1/45). The outcome illustrates

how both the complexity of the initial landscape pattern and the interplay with pixel size influenced the resulting expression response.

Reducing spatial resolution resulted in striking increases for both the interspersion and binary comparison matrix expressions (Fig. 23). Logarithmic regression curves were computed for the relationship between expression behavior and pixel resolution. The interspersion and binary comparison matrix metrics increased at a constant rate (Interspersion = - 8.32 + 2.54 LN Resolution; $r^2 = 0.95$; Binary comparison matrix = -6.79 + 2.14 LN Resolution; $r^2 = 0.95$). Values obtained at 420m levels were approximately 6 to 8 times greater than the original baseline values; a magnitude far surpassing that observed for the Simpson and evenness indices. Interspersion and binary comparison matrix were based on the 9-pixel roving-window which covers proportionally more landscape area with each resolution level. As the pixel size within the window increases, more of the landscape is contained within the window. Thus, fewer windows are needed for examining the entire landscape. Using large pixels reduces the probability of properly embracing the variety on the landscape. The complexity of the landscape also influenced the behavior of the expressions. Heterogeneity values were routinely overestimated on simple landscapes to a greater degree than on complex landscapes.

Both semivariograms depict the "classic" semivariance curve (Curran 1988). Spatial dependency exists at each resolution level until a sill is reached at 390m. Each successive landscape image has a pattern directly dependent on the resolution at which it is measured; the semivariance is a function of scale. The change in semivariogram form beyond 390m appears to be the range beyond which there is no longer spatial dependence (Oliver and Webster 1986).

The reduction in spatial resolution resulted in a marked decline in total forest edge length, convexity (edge-to-area ratio), and Patton's index (Fig. 24). Using pixel count to measure edge length resulted in a cumulative systematic error with each successive resolution level; edge detail was suppressed at each successive coarse level. This tendency is reflected by the exponential regression curves fitted for forest edge

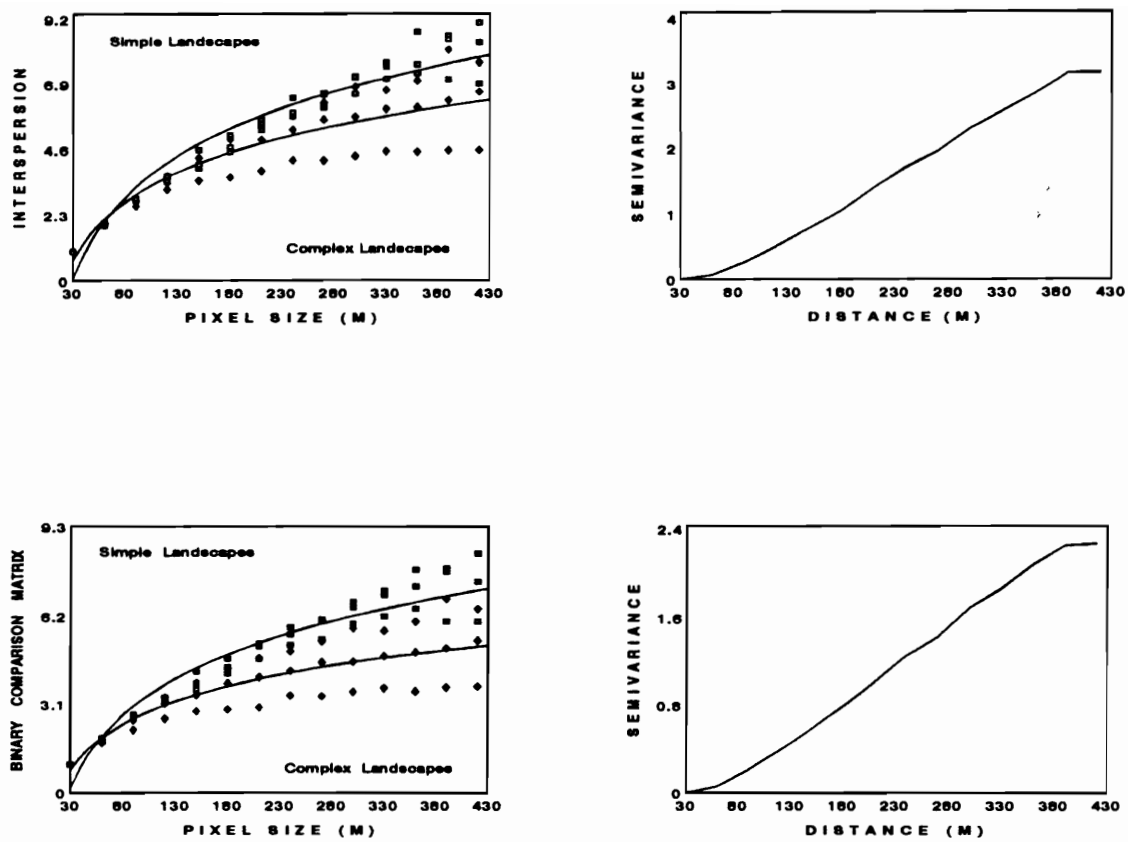


Fig. 23. The influence of spatial resolution on interspersion and binary comparison matrix values and their corresponding semivariograms. Plot markers identify simple (square) and complex (diamond) landscapes. Trend lines are derived from regression analysis. Vertical axes are scaled to 1.0 for direct comparison.

(LN Forest Edge = 0.039 - 0.001 Resolution; $r^2 = 0.973$) and the logarithmic curve computed for convexity (Convexity = 1.60 - 0.156 LN Resolution; $r^2 = 0.971$). The edge length and convexity measured at the coarsest resolution is approximately 50% less than that measured at baseline. The decline in Patton's index with coarse resolution levels was expected (Patton Index = 1.64 - 0.141 LN Resolution; $r^2 = 0.887$). Each change in resolution also reflected the loss of edge detail as patch shapes were repeatedly simplified (recall this expression relates forest edge and area to that of a circle). The larger the pixel, the likelihood increases that elements of edge will be absent in each pixel.

Similarly, the more complex the landscape pattern, the greater the potential for the loss of edge detail as pixel size increases. The rate of information loss from my procedure for all 3 edge expressions became pronounced with the complexity of the landscape. For the Patton index in particular, the more irregularity in original patch shapes, the greater the loss of shape information as forest boundaries were repeatedly smoothed by the scaling process.

Both semivariograms for edge and convexity show a tendency toward a linear trend in semivariance with small scale noise, suggesting weak spatial dependence. Changes in semivariance are almost imperceptible among the 3 smallest pixel sizes; values at 30m are similar to those values measured at 60m and 90m. For forest edge length, the range is not reached within the pixel size range that I tested. The edge semivariogram exhibits "unbounded" variance (Curran 1988). Convexity values after the 360m pixel fluctuate slightly above and below the 0.01 level of semivariance (a possible sill). If this fluctuation continues, these values can be considered spatially independent and will not reflect the scale of measurement. The semivariogram for Patton's index shows both small and large scale dependence (Palmer 1988) with unbounded variance. There are 3 repetitive patterns where semivariance reverses direction with changes in scale. The concave form beginning at 150m in Fig. 24 can also be attributed to local trends or drift in the data (Oliver and Webster 1986). These drifts are dependent upon the pixels selected for removal and the pixels selected for magnification at each resolution level. Both factors repeatedly alter the

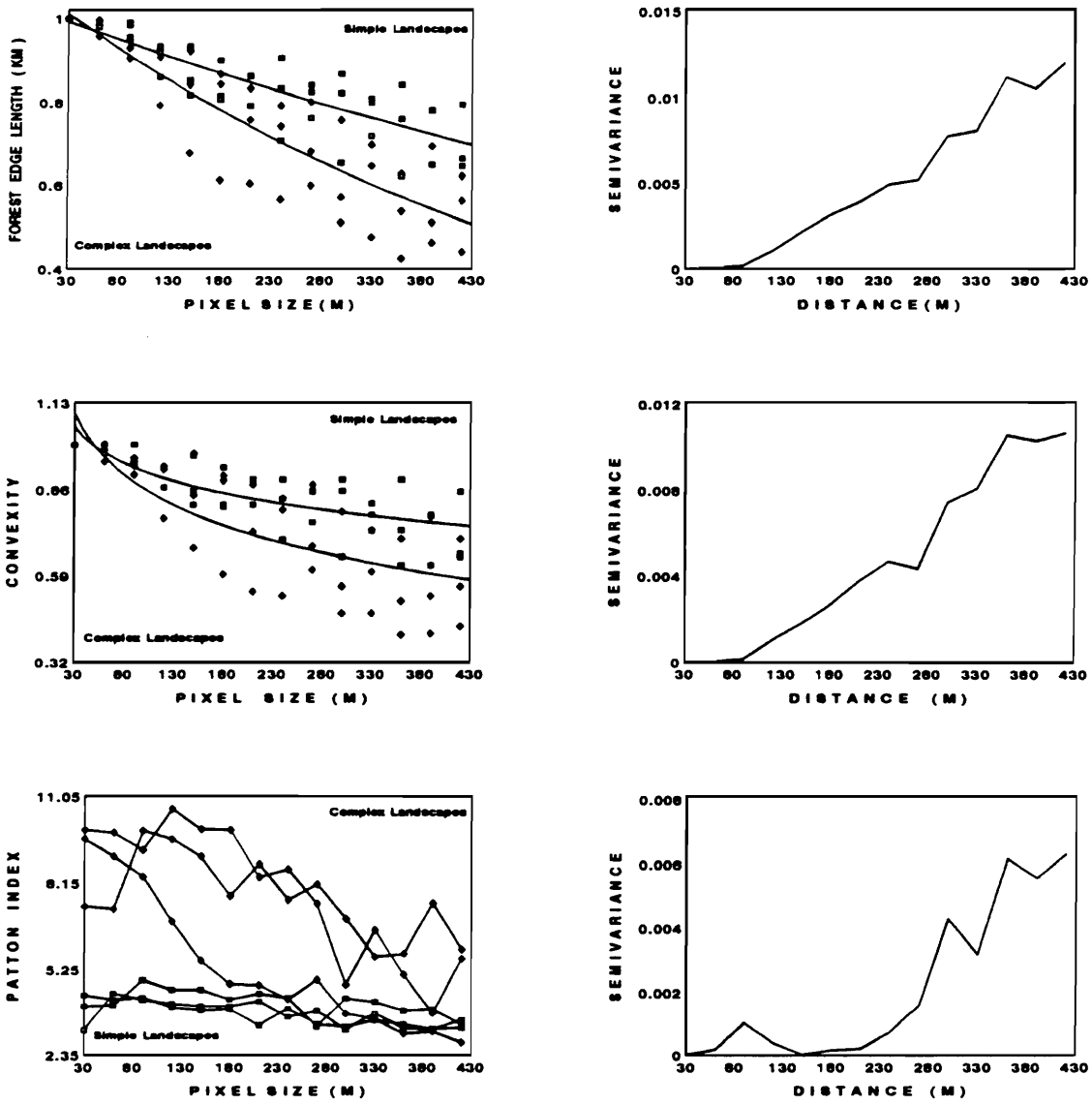


Fig. 24. The influence of spatial resolution on forest edge, convexity, and Patton's index with their corresponding semivariograms. Plot markers identify simple (square) and complex (diamond) landscapes. Vertical axes for edge and convexity are scaled to 1.0 for direct comparison. Trend lines are derived from regression analysis.

shape of the forested landscape.

The progression of fragmentation index I, fragmentation index II, and forest contiguity values across changing spatial scale is depicted in Fig. 25. Fragmentation index I values at the coarsest resolution were over 170 times greater than the index values computed at baseline, the magnitude of which was unsurpassed by any other landscape expression (LN Fragmentation Index I = -6.31 + 1.89 LN Resolution; $r^2 = 0.973$). This index is based on the ratio of the number of polygons to the total number of map units (pixels in this case). With each reduction in resolution, the denominator decreases four-fold, regardless of the number of polygons. When polygon number (the numerator) does change, it is gradual in comparison to the magnitude of change occurring in the denominator. The end result is an almost exponential rise in fragmentation index I values for both simple and complex landscapes. The semivariogram illustrates a form representing non-stationary processes (Oliver and Webster 1986). The curve increases indefinitely; the range for finite variance is not reached at any spatial resolution level examined and there is no indication that it will ever approach the range.

The influence of spatial resolution on the fragmentation index II, which expresses the mean distance to non-forest from each forest pixel, is depicted in Fig. 25. The repeated loss of forest boundary detail and small forest clearings (i.e., those smaller than the current pixel size) results in an increase in metric value with each reduction in resolution (LN Fragmentation Index II = -0.116 + 0.004 Resolution; $r^2 = 0.884$). Application to complex landscapes resulted in a five-fold overestimation, while values computed from simple landscapes were 3 times that of baseline reference (as discussed before, the potential for the loss of edge detail is less for simple landscapes). Values remained unchanged until 90m, rose slightly by 180m, and then increased quickly with succeeding changes in resolution. The semivariogram also shows that fragmentation index II semivariance does not yet reflect the scale of measurement within this interval. The height and spacing of the periods reflect the different landscape patterns emerging during the changes in spatial scale; the distance to non-forest is variable within these periods.

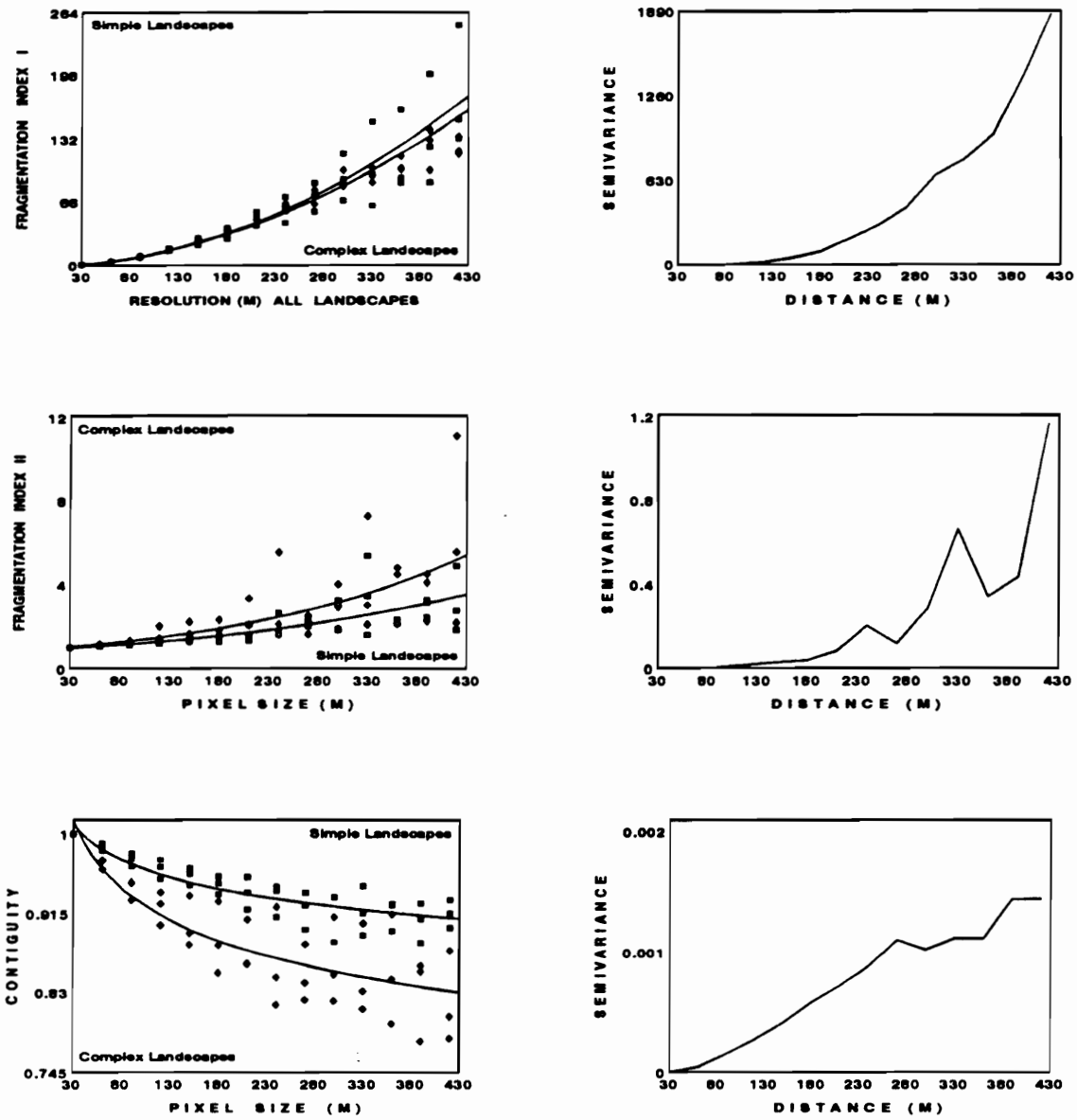


Fig. 25. The influence of spatial resolution on fragmentation I, fragmentation II, and forest contiguity with their corresponding semivariograms. Plot markers identify simple (square) and complex (diamond) landscapes. Trend lines are derived from regression analysis. Vertical axes are scaled to 1.0 for direct comparison.

Changing spatial resolution resulted in a steady decline in contiguity values for all landscapes (LN Contiguity = 0.206 - 0.057 LN Resolution; $r^2 = 0.914$), a decline which was slightly pronounced for the complex landscapes. The mean rate of information loss for simple landscapes was 9%, while that for complex landscapes was 17%. Contiguity values were increasingly variable with each succeeding stage of resolution; much of this variation was dependent upon the distribution of remaining forest pixels. This variability is reflected in the semivariogram. Spatial dependency exists at each resolution level until 270m is reached; each landscape image within the 30m - 270m interval has a contiguity pattern directly dependent on the resolution at which it is measured. Semivariance at 270m changes direction until a range is reached at 390m, beyond which there is no longer spatial dependence (Oliver and Webster 1986).

Fig. 26 illustrates the relationship between spatial resolution and percent forest cover, which describes the proportion of the total landscape that is forested (Forest Cover = 0.991 + 0.0003 Resolution; $r^2 = 0.827$). Expression values remained relatively constant through 90m and then began to increase until the coarsest resolution was reached. The net effect was a mean increase of 10% forest cover. It is evident, however, that this increase was accompanied by much variability at each level. The variability in forest cover was often a function of the distribution of both the forest and non-forest components of the landscape. Forest cover was overestimated for those landscapes representing continuous forest, especially those rather compact in shape (e.g., having little edge detail relative to forest area). The distribution of non-forest within a landscape also influenced the variability observed for the estimation of forest cover. Landscapes with elongated, narrow, or irregularly-shaped non-forest areas (e.g., the shape of riparian areas occurring along a narrow drainage) lost these areas quickly during the scaling process. This loss resulted in an overestimation of forest cover. These narrow areas were routinely missed since pixel selection is across rows (and not down columns) making it easier for the algorithm to skip completely these areas during pixel selection. In contrast, landscapes with wide, compact areas of non-forest (e.g., regularly-spaced fields,

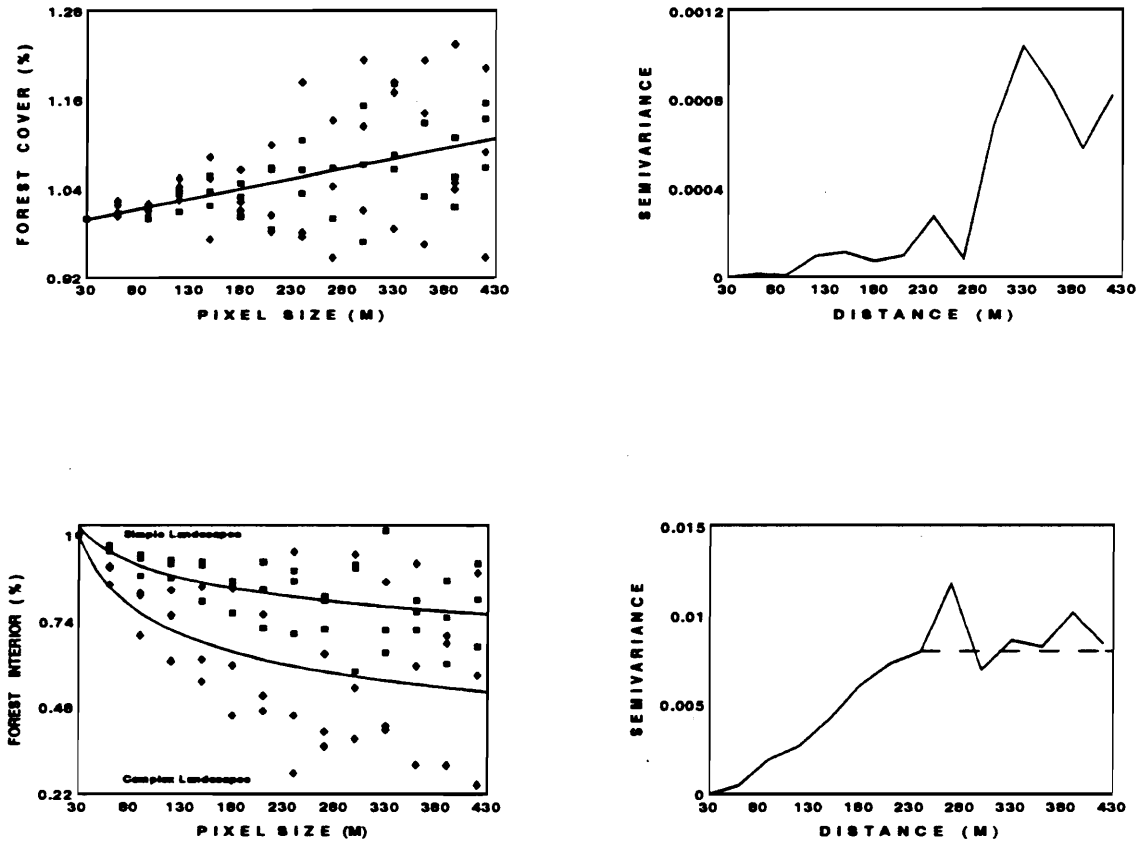


Fig. 26. The influence of spatial resolution on percent forest cover and forest interior with their corresponding semivariograms. Plot markers identify simple (square) and complex (diamond) landscapes. Trend lines are derived from regression analysis. Vertical axes are scaled to 1.0 for direct comparison. The dashed line on the forest interior semivariogram indicates a possible sill.

maintained wildlife openings, and fairly wide areas along roadsteads) often retained these areas during the resolution changes. The amount of forest cover detected was variable.

The variability of forest cover estimation can be clearly seen in the noticeable reversals of slope in the semivariogram. The variable height of the periods reflect the different distribution of forest pixels emerging from each change in resolution. As resolution becomes coarser, semivariance rises. At fine resolution levels (30m - 90m), semivariance does not yet reflect the scale of measurement and can be considered spatially independent (i.e., measuring forest cover at 90m is just as good as 30m).

Fig. 26 also depicts the relationship between spatial resolution and percent forest interior. Percent forest interior was very sensitive to changes in spatial resolution; the net change in expression values over the modeling process was negative (LN Forest Interior = 0.588 - 0.170 LN Resolution; $r^2 = 0.744$). Expression values fell immediately by 60m and continued to drop throughout each level. The steady reduction of forest interior was a function of 3 factors: the loss of edge detail as forested areas were repeatedly simplified, the shrinkage of forested areas (e.g., the radius of a patch decreases as edge detail is eliminated), and the relationship between forest buffer and pixel size. On the latter relationship, the predefined buffer area in this analysis was 1 pixel width. During the scaling process, an increase in pixel size accompanied an increase in buffer area (i.e., the width of a buffer cannot be less than the current pixel width). This combination of changes rapidly reduced the amount of interior forest remaining.

The more complex the landscape pattern, the greater is the potential for the loss of forest interior. At the coarsest resolution, values for simple landscapes were reduced by 25%; losses of interior forest on complex landscapes were 50% of that measured at baseline. Among complex landscapes, the greatest loss of interior forest occurred in locations where several non-forested areas were dispersed throughout a landscape and in those locations where the forest was partitioned by other landuse classes.

The semivariogram depicts a linear scale dependence between 30m - 230m. The semivariance rises with each reduction in resolution. Semivariance values for interior

forest following 230m distance fluctuate slightly above and below the 0.0075 level of semivariance. If the 230m pixel size is the sill, values above this distance can be considered spatially independent and will not reflect the scale of measurement (Palmer 1988).

Fig. 27 illustrates the relationship between spatial resolution and the number of forest patches, mean patch size, and interpatch distance metrics. The variability shown at each resolution level for these expressions was more pronounced than for any other group of expressions. Since each stage of the scaling process was independent of the last (i.e., the number of patches at 90m did not influence the number at 120m), the opportunity for disappearance or coalescence of forest patches varied with each resolution.

The influence on number of forest patches was negative; 45% of the patches were lost within the coarsest resolution level (LN Forest Patches = 0.088 - 0.002 Resolution; $r^2 = 0.858$). The detection of forest patches was suppressed at the coarse levels as pixel size exceeded patch size. Consequently, the rate of patch loss was greater on those landscapes comprised of several, small patches (average 43 ha) than on those landscapes comprised of a few, large patches (average 118 ha). The distribution of forest patches (as expressed by the mean/variance ratio of the Poisson distribution) also proved to be a driving force in the loss of forest patches. Compartments with clumped distributions of forest patches lost those patches at a slower rate than those compartments having fairly uniform, evenly-spaced patch arrangements; however, the number and nature of these compartments limited the extent of such examination. The corresponding semivariogram highlights the reversal of variability that occurs at several levels with a general upward trend; a finite semivariance level was not reached within the resolution levels examined.

The disappearance of forest patches as resolution was reduced significantly influenced the behavior of both mean patch size and interpatch distance (Fig. 27). When pixel size was less than patch size, the likelihood was high that the patch would remain at the next coarse resolution. However, as pixel size reached and exceeded patch size, retention of those patches became unpredictable, resulting in an r^2 of only 0.441 for mean

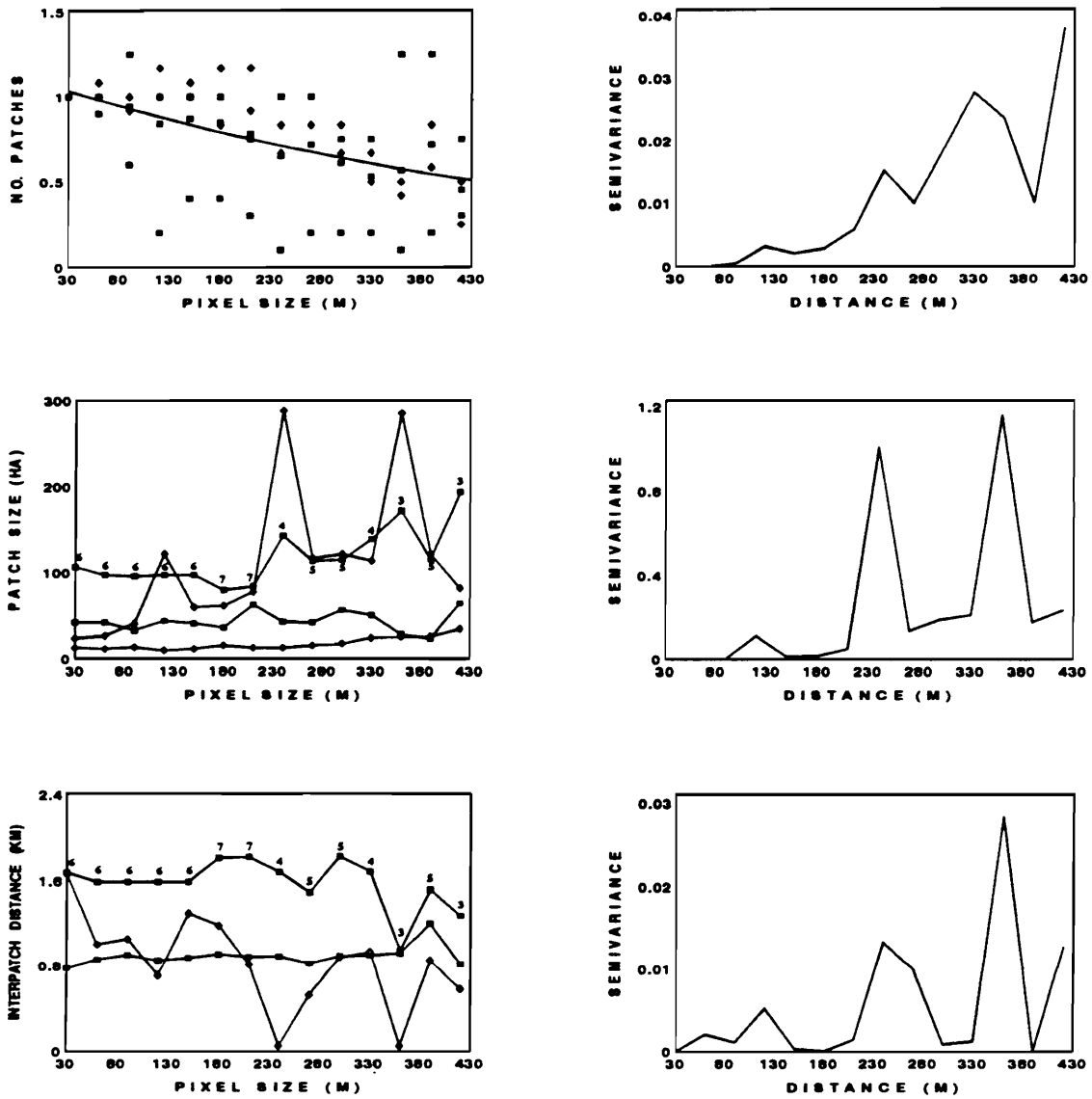


Fig. 27. The influence of spatial resolution on the number of forest patches, mean patch size, and interpatch distance with their respective semivariograms. Plot markers identify simple (square) and complex (diamond) landscapes. Vertical axes for the number of forest patches are scaled to 1.0. The number of forest patches present are indicated below selected compartments.

patch size and an r^2 of 0.184 for interpatch distance. As the resolution decreased, mean patch size could increase (e.g., the area of a selected patch is magnified by the new pixel size) or decrease (e.g., the area of a non-selected patch is replaced by non-forest pixels). Interpatch distance also reflects patch size, expressing the distance from patch edge to patch edge (i.e., the minimum distance between 2 patches is set by the new pixel size). Interpatch distance is also influenced by the distribution of patches. If a non-selected patch is far removed from the others, mean interpatch distance declines. However, if a patch that is not selected is proximal to another patch, mean distance rises. These 2 expressions, along with the number of forest patches, pose serious potential problems concerning their predictability (or meaning) at different spatial scales.

The semivariograms for both mean patch size and interpatch distance show prominent fluctuations of semivariance, suggesting that the pattern expressed by these 2 indices is varying constantly with spatial scale. There are 3 to 5 repetitive patterns with no evidence of either an upward or downward trend in semivariance. The one exception occurred during the 30m - 90m interval where patch size semivariance remained unchanged. The stability of the semivariance suggested that patch size exceeded pixel size for those resolution levels. Semivariance for interpatch distance increased immediately following the 30m baseline level. This progression suggested the influence of patch number and distribution (since patch size remained unchanged) on the variation observed for this expression.

Underlying Landscape Pattern and Spatial Scale Analysis

Landscape pattern is reflected in the number of distinctive classes, their sizes, shapes, and the distances between them. These features uniquely define each landscape. The processes used during map creation have a profound influence on the results of spatial analysis. During map creation, cartographers make decisions about the omission of land classes (e.g., deletion of classes < 10 ha), the combination of land classes (e.g., it may be difficult to represent 50 distinct land classes while retaining map clarity), and the

classification of those classes (e.g., whether a 3 year-old forest stand is classified as abandoned farmland or forest). In this manner, map generalization has occurred prior to any type of spatial analysis.

While observing how pattern analysis changes as a function of spatial scale, there appeared to be particular features that repeatedly influenced those patterns observed at coarse resolution levels. Turner et al. (1989) also found that the pattern of the landscape influenced metric values during changes in spatial scale. A consideration of these factors may aid in determining the adequacy of coarse resolution levels for describing specific landscape types (Wehde 1982).

The size and shape of land classes influenced their persistence during changes in spatial scale. Compact-shaped classes had a higher probability of being retained during resolution reductions than those land classes occurring as long, narrow delineations (e.g., riparian areas). When the size of forest patches or land classes are larger than the current pixel size, there is little change in the distribution or proportion of land classes, and a high likelihood that the patch will be retained into the next resolution level. When patch size was less than pixel size, the retention of that patch became unpredictable and the spatial relationships between land classes were altered. For example, the loss of small forest inclusions resulted in the overestimation of forest cover and fragmentation index II (distance to non-forest).

The spatial arrangement of land classes and forest patches also influenced their retention during changes in scale. Rare land classes distributed in patchy arrangements disappeared more rapidly than did contiguous classes. Land classes that were clumped disappeared slowly with increasing pixel size. Land classes were often enclosed within a large pixel classified as the same type, resulting in a loss of patches but a gain in patch size. Similarly, land classes that existed as single, continuous units were retained at coarse resolution levels. In contrast, the loss of pattern information was pronounced at coarse resolution levels when 1 or more land classes were scattered as small patches or distributed in a non-continuous manner.

The changes observed during spatial scaling were also influenced by landscape complexity (i.e., the number of land class segments per total landscape area). It has been suggested that the rate of information loss will be greater for complex rather than simple landscapes (Dale et al. 1989); however, this was not always the case in this analysis. The loss of pattern information was more striking on simple than on complex landscapes for each of the spatial heterogeneity expressions examined. The addition or omission of even a single land class on a simple landscape consistently resulted in changes in both the Simpson and evenness indices; both became unpredictable as pixel sizes became larger. For example, spatial heterogeneity increased at 90m as a minor land class increased in areal significance. At 150m however, heterogeneity was reduced as land classes were lost on the same landscape. The interspersion and binary comparison matrix expressions also regularly overestimated the heterogeneity of simple landscapes. These results suggest that scale problems may indeed occur in spatially homogeneous landscapes.

Some of the expressions were not overtly influenced by landscape complexity (or the lack thereof). The rate of pattern loss for fragmentation index I and II, percent forest cover, mean patch size, interpatch distance, and the number of forest patches was influenced by factors other than complexity (e.g., problems in their formulation, the distribution of land classes, and the pixel - patch size relationships).

There were expressions, however, where the rate of pattern loss was greater on complex than on simple landscapes. Forest edge length, convexity, Patton's index, percent forest interior, and forest contiguity were immensely affected by landscape complexity. Complex landscapes quickly lost edge and shape detail with the continual smoothing of land class and forest patch boundaries that occurred during changes in scale. This influenced edge metrics and forest interior (which is based on distance from forest edge). Forest contiguity was affected more in complex than simple landscapes with the shifting distribution of forest and other land classes accompanying these boundary changes.

General Trends in Landscape Pattern Change

Map generalization is a homogenizing process and several trends emerge from this process. A pattern expression may either increase or decrease depending upon the characteristics of a landscape image at a particular resolution. Spatial scaling appears to favor systematically land classes for retention or exclusion on the basis of their area and location (i.e., pixels are retained on the basis of their location rather than their dominance). The sampling process also tends to exclude those land classes due to their characteristic size and shape. If a forest patch or other land class area drops below the resolution of the landscape image, or it is not selected because of the sampling process, the resulting pattern description is modified dramatically.

Landscape features that do remain after the scaling process may be systematically altered in size and shape. The subsequent analysis of these generalized images may result in land class segments being omitted or in areas being exaggerated in relation to their true areas on the ground (Hole and Campbell 1985).

Much of the change observed with change in spatial scale centers on the loss of discernable detail, including the following: Removal of small inclusions and patches results in reassigning those areas to a different land class; enlargement of these features results in overestimating their areas. Local heterogeneity is averaged out at coarse resolutions, masking localized variability in landscape features. Boundary smoothing results in losing edge and shape detail as discussed above; the shift in boundary position distorts land class proportions as edge pixels are transformed from 1 land class to another (Hole and Campbell 1985). Pixel redistribution in turn influences the connectivity of forest and other land classes. The spatial arrangement of land classes is altered, while the spatial relationships between like-pixels are obscured (Chou 1991).

The changes observed in this study may vary with the type of algorithm used for map generalization. However, information will be lost when using graphic generalization (the manipulation of line segments for reducing visual complexity); taxonomic generalization (the removal of boundaries between neighboring pixels that are taxonomically similar);

spatial generalization (a strategy that combines pixel types which occur adjacent to one another); or generalization by sampling (different algorithms based on some sampling scheme such as spatial dominance, uniform sampling, or position points within pixels). With reduction in resolution, error is introduced by discarding different features of the original pattern (Hole and Campbell 1985); each change cannot ensure that landscape features will be represented consistently at coarse resolution levels.

Variability of Pattern Metrics and the Semivariogram

Modeling variance, rather than means, has been suggested as a method to understand the effects of spatial scale on landscape pattern. Mean estimates of pattern average the variation over an area. The timing and magnitude of variance change with spatial scale may provide useful information for landscape analysis. Several statistical approaches are based on the observation that variance increases as transitions are approached in hierarchical systems (O'Neill et al. 1986). If variance is measured as a function of scale, increased variance may indicate the approach of a change in pattern (Turner et al. 1989) with extrapolation possible prior to that change.

The change in variance of an expression associated with changing spatial scale may define the limits within which meaningful extrapolation is possible. (As used here, extrapolation refers to the transfer of the same information from 1 spatial scale to another). The error associated with extrapolation will differ among the expressions selected based on their behavior during changes in resolution. The semivariogram may be valuable in assessing the constancy of an expression. The sill and the range together can serve as a guideline for identifying spatial scales where extrapolation is possible and those intervals where similar scales produce comparable landscape descriptions.

Semivariograms of spatial data have been used for describing the areal variation in regional soils (Oliver and Webster 1986), vegetation structure (Palmer 1988), and digital imagery (Weishampel et al. 1992); for detecting patterns in canopy structure (Cohen et al. 1990); for determining optimal resolution for remotely-sensed plantations (Atkinson et al.

1990); and for many applications in the mining industry (Curran 1988). In this study, the semivariogram has proven to be a useful tool for identifying those resolution levels which influence landscape pattern description.

The semivariogram parameter, $\mathbf{y_h}$, describes the magnitude of an expression's variation while the form of the curve when subjected to changes in **h** provides insight into the nature of that variation (Oliver and Webster 1986). It was useful for identifying metrics which uniformly change direction with scale and those the behavior of which was unpredictable. In addition to identifying the variation inherent in spatial scaling, semivariogram analysis can suggest the resolution of sampling needed to characterize landscape pattern.

The landscape expressions that changed in a steady manner with little variability may be suitable candidates for extrapolation to other scales. The consistency of behavior demonstrated by the binary comparison matrix, interspersion, Simpson, and evenness metrics suggests their potential for extrapolation. Other expressions showing consistent rates of change with bounded variance over partial intervals included forest contiguity, convexity, and forest edge length. It seems reasonable that values for these metrics could be extrapolated well in pixel sizes between 30m and 240m.

The semivariogram identified intervals where the semivariance ($\mathbf{y_h}$) of specific pattern metrics remained unchanged or changed relatively little as **h** changed. These resolution intervals, where semivariance was fairly constant, suggests the potential for extrapolation within that interval and the potential for analyzing pattern at the coarse end of that interval (i.e., by presenting the limits of tolerance for choosing large pixels). For example, the results indicated that landscape description was equivalent at 30m - 90m for forest cover, number of patches, and mean patch size; at 30m - 120m for fragmentation index II; and at 30m - 210m for evenness and Simpson's index. Variation levels outside of these ranges will be too different from neighbors (due to their spatial dependency on scale) to allow the metrics to be equally interpreted at a more coarse resolution.

By using the semivariogram, it was possible to identify expressions that exhibited

great variation during the spatial scaling process. For those expressions demonstrating unbounded variance (e.g., fragmentation index I and II), conclusions cannot be made about 1 scale based on another. The variability observed in Patton's index, percent forest interior, and forest cover values at different levels of resolution indicates that the choice of resolution needs to be carefully considered prior to using these expressions for landscape analysis. The variability observed in both the magnitude and direction for the number of patches, interpatch distance, and mean patch size expressions suggests also that misrepresentation of pattern can occur at coarse resolution levels. The semivariograms suggest that for these metrics the variability (and potential error) associated with extrapolation of information may be great across spatial scales. The pattern suggested by these expressions cannot be generalized directly to produce estimates at large scales.

Examining the semivariograms of landscape metrics demonstrates the importance of scrutinizing variance as a means to discriminate the potential usefulness of expressions at different resolution levels. Understanding the change in variance and the interpretation of expressions as they are influenced by pixel size can be used by resource managers as guidelines to limitations of landscape analysis.

Spatial Scale and Wildlife Habitat Assessment

The literature abounds with papers suggesting the theoretical and conceptual importance of spatial scale (Allen and Starr 1982; Wiens et al. 1986; Holling 1992; Levin 1992; Cullinan and Thomas 1992); however, few studies have examined how spatial scale directly influences wildlife habitat evaluation. Schultz and Joyce (1992) found that changing scale affected the estimation of marten (Martes americana) home range suitability. Computed Habitat Suitability Index (HSI) values for spotted owl (Strix occidentalis) were highest at fine resolution levels, suggesting that large pixels masked small habitat pockets of important value (Laymon and Reid 1986). The usefulness of both species models rose with a decline in pixel size. Stoms (1992) found that generalizing landscape maps using large pixels was accompanied by a decline in predicting species

richness within the Gap Analysis project. Schultz and Joyce (1992) also found that the changes in forest patch size and spatial distribution occurring with coarse resolution levels modified resulting habitat evaluations.

The landscape expressions examined in this chapter are valuable descriptors of wildlife habitat. Their behavior at different spatial scales belies the general importance of examining them for understanding how pattern influences species. Only at the proper resolution do the expressions have potential meaning for the discerning land manager. For example, a metric that overestimates a desirable habitat component at coarse resolution levels would result in an inflated evaluation of habitat potential. Similarly, a metric that underestimates an undesirable habitat component could also result in overrating habitat condition.

Table 14 shows how a resource manager having the knowledge of this chapter may apply it during the evaluation of wildlife habitat at different spatial scales. The several examples presented suggest how failure to consider spatial scale may result in a misleading habitat evaluation. These examples and the other results obtained in this chapter suggest that spatial scale cannot be ignored by resource managers in either pattern analysis or in wildlife habitat evaluation. The choice of scale will have a profound influence on both the strength and the nature of the wildlife-habitat relationships observed. Analyzing landscapes at an inappropriate scale may not correctly describe existing patterns but instead reflect artifacts of the scale of measurement (such as that observed for interpatch distance, mean patch size, and number of forest patches). Whether the relationship between the spatial scale at which landscape parameters are measured matches the scale at which those parameters influence wildlife species is an important question that remains unresolved.

There is no single correct scale for describing a system (Levin 1992). The appropriate scale will depend on the management objective, the species involved, the underlying landscape pattern, and the processes believed to be important. For example, if the objective involves evaluating interior species habitat, the loss of boundary detail

Table 14. The influence of spatial scale on landscape pattern description and potential wildlife habitat evaluation.

Species	Common Name	Key Reference[a]	Landscape Metric	Scale Influence on Metric[b]	Influence on Habitat Evaluation
Canis latrans	Coyote	Thomas et al. 1979	Simpson Index	Increase	Overestimation
Urocyon cinereoargenteus	Gray Fox	Fritzell 1987	Binary Comparison Matrix	Increase	Overestimation
Phasianus colchicus	Pheasant	Yahner 1988	Interspersion	Increase	Overestimation
Accipiter nisus	Sparrowhawk	Hunter 1990	Forest Edge Length	Decrease	Underestimation
Catharus fuscescens	Veery	Hamel 1992	Forest Edge Length	Decrease	Overestimation
Odocoileus virginia	White-tailed Deer	Roller 1978	Patton Index	Variable	Unpredictable
Martes pennati	Fisher	Rosenberg & Raphael 1986	Fragmentation Index I	Increase	Underestimation
Meleagris gallopavo	Eastern Wild Turkey	Gustafson et al. 1994	Distance to Non-Forest	Increase	Underestimation
Martes americana	Marten	Bissonette et al. 1989	Distance to Non-Forest	Increase	Overestimation
Seiurus aurocapillus	Ovenbird	Sweeney & Dijak 1985	Percent Forest Cover	Variable	Unpredictable
Wilsonia citrina	Hooded Warbler	Whitcomb et al. 1981	Percent Forest Interior	Decrease	Underestimation
Dryocopus pileatus	Pileated Woodpecker	Schroeder 1983	Mean Patch Size	Variable	Unpredictable
Streptopelia turtur	Turtle Dove	Van Dorp & Opdam 1987	Interpatch Distance	Variable	Unpredictable
Ursus americanus	Black Bear	Rudis & Tansey 1995	Forest Contiguity	Decrease	Underestimate

[a] Source of observations citing the importance of landscape pattern.

[b] Influence of spatial scale when measured from fine to coarse levels of resolution.

accompanying coarse scales may yield unacceptable results. The conservation of rare habitats may dictate fine resolution levels, whereas monitoring wetland loss for migrating waterfowl may require broad-scale analyses. Different processes will be important at different spatial scales.

Differences in spatial resolution are critically important because they influence the questions that can be addressed, the observations obtained, and how the results may be interpreted. Similarly, patterns of habitat association for many species are dependent on the scale of investigation. In addition, patterns evident at regional scales may be a consequence of processes at local scales; therefore, an appreciation of the local scale is necessary to interpret the broad-scale patterns correctly (Weins 1989b).

The difference among species requirements will also affect the choice of spatial scale. The ability of species to respond to landscape pattern depends on how they view the environment (Weins 1989a). Gray squirrels relate to their environment in a manner different from that of black bear; the territory required by an ovenbird may be too small for barred owl. Species exploit resources within a different range of scales. Large species utilize resources within home ranges encompassing hundreds of kilometers. Small species exploit modest ranges from millimeters to a few kilometers (Holling 1992). The choice of spatial scale for wildlife habitat evaluation should reflect the animal's home range, migration, territorial behavior, seasonal area use, the degree of habitat specialization, and the landscape parameters influencing its lifestyle.

MANAGEMENT IMPLICATIONS

For any landscape analysis, a compromise between spatial scale and the quality of the resulting pattern description must be evaluated. This compromise includes the size of the analysis area and the appropriateness of the available spatial data. Remotely-sensed and cartographic data are available at a wide range of spatial resolution levels, yet there is a tendency to choose fine resolution levels. The detailed landscape data that come with

fine resolution levels comes however, with the high costs of data acquisition, storage, retrieval, processing, and updating.

The resource manager's ability to assess environmental conditions will reflect the resolution of the data used for analysis. The practical difficulties in collecting and analyzing large amounts of data often dictate that resolution be reduced. This decision may also reflect financial considerations or the limitations of data sources. However, analyzing a landscape at coarse levels of resolution as demonstrated in this chapter, will influence pattern accuracy (relative to 30m pixels), ability to distinguish among patterns, and the resulting description of pattern.

In addition to monetary considerations, selecting spatial scale has implications for the adequacy of landscape analysis and the reliability of estimates (Evanisko 1991). The changes that occur as resolution is reduced may not always produce acceptable results in terms of pattern description. Analyzing landscape patterns at coarse resolution levels accompanies the loss of ability to detect localized variability, to differentiate patterns, and to represent accurately boundary detail and land class shape. A resource manager should understand how much pattern detail will be retained during landscape analysis at a specific spatial scale (Figs. 22-27) and then decide whether this detail is sufficient for the species under investigation.

The loss of pattern detail is related to the number of sampled pixels used for describing a landscape (the finer the resolution, the greater number of pixels required). Selecting pixel size presents many of the same problems observed with selecting sample size. Resource managers should be aware that using large pixels results in small samples and all of the problems of induction associated with studies having small samples.

Economic constraints will continue to dictate the amount of sampling and the nature of landscape analyses. An objective should be to maximize the information obtained at acceptable levels of confidence in return for the time and effort invested. Managerial objectives dictate the level of required landscape detail; the selected resolution need not be finer than is necessary for those objectives. Spatial scale is often a last consideration in

many analyses, yet scale directly influences the nature of the final results (Meentemeyer and Box 1987). The results of the analyses reported here indicate that selecting the appropriate resolution may be critical for successful landscape analysis.

SUMMARY

Cartographic modeling and a suite of 15 landscape expressions were used to examine the effects of changing spatial scale (resolution) on landscape pattern analysis. Regression analysis identified trends in the behavior of expressions over changing scale. Using semivariograms, it was possible to observe change in the variability of expressions at different resolution levels. The influence of spatial scale on resulting pattern description was dependent upon the number and types of land classes, their distribution, and the size and shape of those classes. Pixel size selection was suggested as analogous to sampling intensity.

The spatial scale at which landscapes are analyzed influences the observed pattern. Measurements made at different scales may not be comparable. The same landscape expressions differ when spatial scale changes. Pattern analyses at 1 resolution cannot be extrapolated risk-free to another without the knowledge of the scale-dependency of the metrics used. Similarly, resolution must be considered when comparing pattern information obtained at different levels. The potential loss of landscape detail and pattern accuracy at coarse resolution should be considered within the scope of managerial objectives.

The relationship between landscape pattern and the spatial scale used in analyses has important implications for both resource management and wildlife habitat evaluation. Habitat evaluation may be overestimated or underestimated, depending on which metric and which spatial scale is selected. Characterizing the relationship between how landscape pattern is described and the resolution selected may lead to an improved evaluation of wildlife habitat.

CHAPTER 5

Conclusion

My major goal in this dissertation was to assess landscape pattern as it relates to forest wildlife resources, developing new approaches for examining how resource management could be enhanced through a landscape approach. I have attempted to show in this research that the analysis of landscape pattern is complex. An extensive literature review suggests that pattern affects the persistence and abundance of wildlife resources. Quantifiable descriptions of pattern provide a component of habitat analyses that can be used to understand species life needs. Many of these components are related; others provide unique information. Landscape pattern is influenced differently by landscape modification and the spatial scale of analysis.

Landscape is a general term; there is uncertainty about its meaning. My preferred definition is an area distinguished by visible structural and functional qualities that are more similar within the landscape than with those of surrounding areas. A landscape may encompass hundreds of square kilometers, such as a large forest, or it may comprise an area of much smaller extent. The definition is dependent upon the context of its use; its meaning is shaped by the ecological processes being discussed and evaluated. A landscape then becomes the working topic for those protecting and managing areas and their resources.

The development of landscape pattern, and the consequences of that pattern on ecosystems and population dynamics, is one of the themes of landscape ecology. A second theme includes interest in the ecological changes that anthropogenic activities may make on the landscape. Both of these themes have become a natural extension of resource management. Resource professionals recognize the influence that the landscape has on species management.

The potential for landscape analysis within resource management rests with its use for learning about landscapes and allowing factors at the landscape scale to become

important for evaluating potential land-use conflicts and solutions. In landscape analysis, off-site factors are included along with site factors in an active quantitative way. The extension to a landscape perspective accompanies a shift in emphasis from monitoring specific management sites to an emphasis on a cumulative approach to land evaluation.

Natural and anthropogenic processes occurring at the landscape level may influence habitat loss and isolation (e.g., subdivision, fragmentation, and attrition). These pattern changes, in turn, alter the heterogeneity, fragmentation, edge, and connectivity characteristics within a habitat. At a local level, these changes may result in eliminating, displacing, or enhancing species and their populations. Understanding how spatial processes alter landscape pattern (and species habitat) may provide resource managers with a basis for making land-use decisions. Managers and land planners can no longer view an ecosystem in isolation from the surrounding landscape or view a region without considering the pattern of landscapes within it (Zonneveld and Forman 1990).

Species respond to pattern in various ways, using certain areas for feeding and reproduction, and avoiding other areas entirely. Relationships have been described for species and their required habitat components (e.g., the U. S. Fish and Wildlife Service Habitat Suitability Index series). The general relationship between a species and its needs can be characterized by:

$$\mathbf{Y = f\,(\,X_1, X_2, X_3, \ldots X_N\,)}$$

where the habitat quality for species (**Y**) is a function of several habitat components (**X**). Often 1 or more of the habitat components for a species represent some aspect of landscape pattern, while the other components represent requisite needs (e.g., food, water, cover, nesting sites). These components vary among species. For example, mean patch size is a component of pileated woodpecker habitat; spatial heterogeneity is important for coyotes; and the distribution of forest and non-forest cover is critical for wild turkey. By altering the distribution and availability of spatial resources, changes in landscape pattern

influence many of the components important for the persistence of species (Merriam 1990). Landscape pattern has important implications, then, for the management of an array of wildlife resources having several different requirements.

Resource managers need to know precisely what pattern is needed for a species. The pattern needed determines the expression, thus the information required. In addition, a manager should have a working knowledge of what exactly is expressed by a metric (Chapter 1), the inputs that an expression is based upon (Table 2), and the expression's relationship to other metrics (Chapter 2). Such a knowledge will aid in selecting the proper analysis expressions.

I have evaluated 24 landscape expressions of landscape pattern. Other expressions exist, and relationships are expected to exist between these expressions as well. Significant correlations existed between each of the spatial heterogeneity metrics (Table 3). Forest edge length was also highly correlated with the heterogeneity group. In contrast, there was a lack of relations within the fragmentation, edge, and connectivity groups (Tables 4 and 5). Metrics that provide equivalent pattern information (e.g., the Shannon and Simpson indices, the Patton and compactness indices, the number of patches and patch density metrics) may lead to double-counting in weighted analyses. Others may show strong interrelations with other expressions (e.g., edge length, interspersion, evenness) but do not duplicate pattern information.

Classifying expressions based upon behavioral properties (Table 6) resulted in all of the spatial heterogeneity expressions grouped into 1 cluster; however, the remaining groups (e.g., fragmentation, edge, and connectivity) were not combined into like-clusters. The 6 clusters of expressions comprising the classification suggest that 7 dimensions of pattern characteristics are relevant to forested landscape analysis: the number, proportion, and arrangement of land classes (including forest cover); the number of patches; patch shape and radius; and forest edge length. The results of the cluster analyses suggest that selecting an expression should be based upon a clear understanding of the nature of information an expression conveys.

Selecting an expression also involves considering its relevance for use in specific environmental monitoring applications. The relationships between selected management activities and changes in habitat components represented by pattern were examined in Chapter 3. The tables and figures presented in that chapter serve as guides for examining how pattern-related habitat components are affected by landscape alteration. For example, if timber harvest promotes interspersion of habitats (Table 7), the influence on gray fox habitat is positive. If a road expansion scheme reduces forest contiguity (Table 10), the net result on black bear habitat is negative. On those landscapes where progressive deforestation has resulted in the loss of forest interior area (Table 12), habitat quality is reduced for the worm-eating warbler, hooded warbler, and Cooper's hawk.

Landscape analysis has resulted in numerous pattern metrics; it is possible for some metrics to have the same value for very different landscapes. These equalities caution against the over-reliance on any single metric and highlights the importance of being aware of the sensitivity and limitations of a pattern metric prior to its application for spatial analysis. In Chapter 3, discriminant analyses identified metrics that were effective discriminators of pattern changes occurring during landscape modification. For evaluating changes arising from timber harvest, recommended expressions include forest contiguity, interspersion, convexity, edge length, Fragmentation Index I, binary comparison matrix, interior forest, spatial diversity, and number of different classes (Table 8). Effective discriminators of pattern differences resulting from changes in spatial arrangement include Fragmentation Index II, number of different classes, spatial diversity, interior forest, connectivity, binary comparison matrix, patchiness, and mean patch size metrics (Table 9). For evaluating the influence that road addition may have on a landscape, recommended metrics include Fragmentation Indices I and II, edge length, interspersion, number of different classes, convexity, patch density, number of forest patches, forest contiguity, and the Shannon index (Table 11). Monitoring pattern changes stemming from progressive deforestation would benefit from the use of the interior forest, forest contiguity, convexity, number of different class, patch size, forest cover, number of forest patches,

and patch density metrics (Table 13). For this last modification, managers should be aware that pattern changes were cyclical in nature, with metrics changing sign at the half-way point of deforestation.

Landscape analysis can illuminate the role of spatial scale for describing pattern (Figs. 22 through 27) and for influencing the outcome of wildlife habitat assessment (Table 14). The choice of spatial scale is directly related to the loss of pattern detail. Once a landscape feature drops below the resolution of the landscape image, the resulting pattern description is modified dramatically. Localized heterogeneity within the landscape is masked, boundary smoothing causes the loss of edge and shape detail, and there is a shift in the spatial arrangement of land classes. Landscape description remains relatively unchanged, however, for some of the metrics during selected intervals of scale (e.g., from 30m - 90m for forest cover, number of patches, and mean patch size; from 30m - 120m for Fragmentation Index II; and from 30m - 210m for the Simpson and evenness indices).

Semivariogram analysis identified the variation inherent in spatial scaling and may suggest the resolution of sampling needed to characterize landscape pattern. Metric values that changed consistently with little variability (e.g., the Simpson, binary comparison matrix, interspersion, and evenness metrics) may be potential candidates for extrapolation to other scales. Other expressions demonstrating consistent rates of change over partial scale intervals included forest contiguity, convexity, and forest edge length. In contrast, the variability observed at different spatial scales for the Patton index, percent forest interior, number of patches, interpatch distance, and mean patch size expressions demonstrates that pattern misrepresentation can occur at coarse resolution levels. For these metrics, the variability (and potential error) associated with extrapolating information may be great across spatial scales.

GIS provided the tool for landscape analysis in this dissertation. This technology has enhanced resource management by facilitating the integration of spatial data with the planning and management of landscapes. The cartographic modeling approach for examining the influence of anthropogenic activities succeeded in identifying potential

landscape conditions. These models also provided the means for examining spatial scale influence. Cartographic models generalize spatial relationships using the location and configuration of landscape elements, permitting the quantitative evaluation of management alternatives. These models allow the processes taking place to be understood and show promise as a technique for cumulative effects analysis.

My research suggests several lines of future study. A natural extension of this study lies in developing cartographic models (and related discriminant analyses) for evaluating potential pattern changes arising from the conversion of river bottoms, swamps, and wetlands to croplands; urbanization; stream channelization and dam construction; wildfire; and virtually any other type of process transforming 1 land-use to another. There is also fertile ground for new applications which quantify several aspects of pattern simultaneously (as depicted in Fig. 17). Such applications could prove useful for rating management alternatives or for classifying landscapes based on several pattern criteria. A final avenue of future study centers on whether the relationships observed between pattern metrics, (and the classification based on their behavioral properties), remains unchanged at different spatial scales.

In summary, increasing concern about global and regional changes will place an emphasis on landscape analysis methods. This concern will encourage advances in the understanding, methods, and techniques of resource management. Spatial analysis, cartographic modeling, and the other methodologies presented in this dissertation can provide a basis for developing ways to monitor environmental influence on landscapes and for assessing potential land-use conflicts. The continued development and application of new and existing spatial analysis techniques will contribute to further awareness of the dynamics of landscape pattern.

A landscape perspective benefits resource management. Pattern plays a role in the conservation of species along with the stability of food resources, access to quality nesting sites, and availability of water during select periods. Anthropogenic activities and natural processes will continue to modify landscape pattern; when these changes influence a

species negatively, an understanding of the landscape can guide resource management towards minimization of those influences. The successful application of landscape analysis will depend on the ability to quantify pattern and relate its relevance to wildlife resources.

LITERATURE CITED

Allen, T. F. and T. B. Starr. 1982. Hierarchy: perspectives for ecological complexity. Univ. of Chicago Press, Chicago, IL.

Alverson, W. S., D. M. Waller, and S. L. Solheim. 1988. Forests too deer: edge effects in northern Wisconsin. Conserv. Biol. 2: 348-358.

Ambuel, B. and S. A. Temple. 1983. Area-dependent changes in the bird communities and vegetation of southern Wisconsin forests. Ecology 64(5): 1057-1068.

Askins, R. A. 1994. Open corridors in a heavily forested landscape: impact on shrubland and forest-interior birds. Wildl. Soc. Bull. 22: 339-347.

Askins, R. A., J. F. Lynch, and R. Greenberg. 1990. Population declines in migratory birds in eastern North America. Current Ornithology 7: 1-57.

Atkinson, P. M., P. J. Curran, and R. Webster. 1990. Sampling remotely sensed imagery for storage, retrieval, and reconstruction. Prof. Geographer 42: 345-353.

Bennett, A. F. 1991. Roads, roadsides, and wildlife conservation: a review. Pp. 99-118 In D. A. Saunders and R. J. Hobbs, Eds. Nature Conservation 2: The role of corridors. Surrey Beatty and Sons, Victoria, Australia.

Beringer, J. J., S. G. Seibert, and M. R. Pelton. 1990. Incidence of road crossing by black bears on Pisgah National forest, North Carolina. Int. Conf. Bear Res. and Manage. 8: 85-92.

Berry, J. K. 1991. Assessing variation, shape, and pattern of map patches. GIS World, August-October issues.

Bissonette, J. A., R. J. Fredrickson, and B. J. Tucker. 1989. American marten: A case for landscape-level management. Trans. N. A. Wildl. and Nat. Res. Conf. 54: 89-101.

Brittingham, M. C. and S. A. Temple. 1983. Have cowbirds caused forest songbirds to decline? BioScience 33: 31-35.

Brody, A. J. and M. R. Pelton. 1989. Effects of roads on black bear movements in western North Carolina. Wildl. Soc. Bull. 17: 5-10.

Burgess, R. L. and D. M. Sharpe, Eds. 1981. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, NY. 310 pp.

Burrough, P. A. 1986. Principles of geographic information systems for land resource assessment. Clarendon Press, Oxford. 193 pp.

Chasko, G. G. and J. E. Gates. 1982. Avian habitat suitability along a transmission-line corridor in an oak-hickory forest region. Wildl. Monogr. 82: 1-41.

Chen, J., J. F. Franklin, and T. A. Spies. 1992. Vegetative response to edge environments in old-growth Douglas-fir forests. Ecol. Applications 2: 387-396.

Chou, Y. H. 1991. Map resolution and spatial autocorrelation. Geographical Analysis 23 (3): 228-246.

Cohen, W. B., T. A. Spies, and G. A. Bradshaw. 1990. Semivariograms of digital imagery for analysis of conifer canopy structure. Remote Sens. Environ. 34: 167-178.

Conner, R. N. and C. S. Adkisson. 1976. Discriminant function analysis: a possible aid in determining the impact of forest management in woodpecker nesting habitat. Forest Science 22: 122-127.

Cullinan, V. I. and J. M. Thomas. 1992. A comparison of quantitative methods for examining landscape pattern and scale. Landscape Ecol. 7(3): 211-227.

Curran, P. J. 1988. The semivariogram in remote sensing: an introduction. Remote Sens. Environ. 24: 493-507.

Dale, V. H., R. Gardner, and M. Turner. 1989. Predicting across scales: Comments of the guest editors of landscape ecology. Landscape Ecol. 3(3/4): 147-151.

Dansereau, P. M. 1957. Biogeography: An ecological perspective. The Ronald Press Company, New York, NY. 394 pp.

Davis, J. C. 1986. Statistics and data analysis in geology. John Wiley and Sons, New York, NY. 646 pp.

Dickson, J. G. and J. Huntley. 1987. Riparian zones and wildlife in southern forests: the problem and squirrel relationships. Pp. 37-39 In J. Dickson and O. Maughan, Eds. Managing southern forests for wildlife and fish. USDA For. Serv. Gen. Tech Rep. SO-65. So. For. Exp. Station, New Orleans, LA.

Dunn, C. P., D. M. Sharpe, G. R. Guntenspergen, F. Stearns, and Z. Yang. 1991. Methods for analyzing temporal changes in landscape pattern. Pp. 173-198 In M. G. Turner and R. H. Gardner, Eds. Quantitative methods in landscape ecology. Springer-Verlag, New York, NY.

Evanisko, F. 1991. Dimensions of scale in landscape analysis. USDA For. Serv. SE Exper. Sta. Gen. Tech. Rept. 74: 81-88. Asheville, NC.

Faaborg, J. 1980. Potential uses and abuses of diversity concepts in wildlife management. Trans. Missouri Acad. Sci. 14:41-49.

Fahrig, L. and G. Merriam. 1985. Habitat patch connectivity and population survival. Ecology 66: 1762-1768.

Falk, J. A. and D. F. Stauffer. 1990. Landscape fragmentation. Pp. 162-170 In Proc. from the Northeast Raptor Management Symposium and Workshop.

Flather, C. H. and R. M. King. 1992. Evaluating performance of regional wildlife habitat models: implications to resource planning. J. Environ. Manage. 34: 31-46.

Forman, R. T. T. and M. Godron. 1981. Patches and structural components for a landscape ecology. BioScience 31: 733-740.

Forman, R. T. T. and M. Godron. 1986. Landscape ecology. John Wiley and Sons, New York, NY. 619 pp.

Franklin, J. and R. T. Forman. 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecol. 1(1): 5-18.

Freemark, K. E. 1988. Landscape ecology of forest birds in the northeast. Pp. 7-12 In R. M. DeGraaf and W. M. Healy, Eds. Is forest fragmentation a management issue in the northeast? USDA For. Serv. Northeastern For. Exp. Sta. Gen. Tech. Rep. NE-140. Radnor, PA. 32 pp.

Freemark, K. E. and H. G. Merriam. 1986. Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. Biol. Conserv. 36: 115-141.

Fritzell, E. K. 1987. Gray fox and island gray fox. Pp. 408-421 In M. Novak, J. Baker, M. Obbard, and B. Malloch, Eds. Wild furbearer management and conservation in North America. Ministry of Natural Resources, Ontario. 1150 pp.

Galli, A. E., C. F. Leck, and R. T. T. Forman. 1976. Avian distribution patterns in forest islands of different sizes in central New Jersey. Auk 93(2): 356-364.

Gardner, R. H., B. T. Milne, M. G. Turner, and R. V. O'Neill. 1987. Neutral models for the analysis of broad-scale landscape pattern. Landscape Ecol. 1: 19-28.

Gates, J. E. and L. W. Gysel. 1978. Avian nest dispersion and fledgling success in field forest ecotones. Ecology 59: 871-883.

Gibson, D. J., S. L. Collins, and R. E. Good. 1988. Ecosystem fragmentation of oak-pine forest in the New Jersey pinelands. Forest Ecol. and Manage. 25: 105-122.

Giles, R. H., Jr. 1978. Wildlife management. W. H. Freeman and Co., San Francisco, CA. 416 pp.

Golley, F. B. 1989. Landscape ecology and biological conservation. Landscape Ecol. 2 (4): 201-202.

Gotfryd, A. and R. I. C. Hansell. 1986. Prediction of bird community metrics in urban woodlots. Pp. 321-326 In J. Verner, M. L. Morrison, and C. J. Ralph, Eds. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press, Madison, WI. 470 pp.

Gustafson, E. J., G. R. Parker, and S. E. Backs. 1994. Evaluating spatial pattern of wildlife habitat: a case study of the wild turkey (Meleagris gallopavo). Am. Midl. Nat. 131: 24-33.

Hamel, P. B. 1992. Land manager's guide to the birds of the South. The Nature Conservancy, Chapel Hill, NC. 417 pp.

Hammett, J. E. 1992. The shapes of adaptation: historical ecology of anthropogenic landscapes in the Southeastern United States. Landscape Ecol. 7 (2): 121-135.

Harris, L. D. 1984. The fragmented forest: Island biogeography theory and the preservation of biotic diversity. Univ. of Chicago Press, Chicago, IL. 211 pp.

Harris, L. D. 1988a. The faunal significance of fragmentation of southeastern bottomland forests. Pp. 126-134 In D. L. Hook and R. Lea (Eds.). Proc. of the Symp. on Forested Wetlands of the Southern United States. Orlando, FL. 168 pp.

Harris, L. D. 1988b. Landscape linkages: the dispersal corridor approach to wildlife conservation. Trans. N. Amer. Wildl. and Nat. Resour. Conf. 53: 595-607.

Heany, L. and B. Patterson. 1986. Island biogeography of mammals. J. Linn. Soc. 28 (1 and 2): 1-271.

Heinen, J. T. and G. Cross. 1983. An approach to measure interspersion, juxtaposition, and spatial diversity from cover type maps. Wildl. Soc. Bull. 11:232-237.

Hobbs, E. R. 1988. Species richness of urban forest patches and implications for urban landscape diversity. Landscape Ecol. 1(3): 141-152.

Hole, F. D. and J. B. Campbell. 1985. Soil landscape analysis. Rowman and Allanheld, Totowa, NJ. 196 pp.

Holling, C. S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecol. Monogr. 62(4): 447-502.

Hoover, S. R. and A. J. Parker. 1991. Spatial components of biotic diversity in landscapes of Georgia, USA. Landscape Ecol. 5(3):125-136.

Hunter, M. L. 1990. Wildlife, forests, and forestry: Principles of managing forests for biological diversity. Prentice Hall, Englewood Cliffs, NJ. 370 pp.

Hunter, M. L., G. L. Jacobsen, and T. Webb III. 1988. Paleoecology and the coarse-filter approach to maintaining biological diversity. Conserv. Biol. 2: 375-385.

Jackson, J. 1987. The red-cockaded woodpecker. Audubon Wildl. Rept. 3: 479-493.

Karr, J. R. and K. E. Freemark. 1983. Habitat selection and environmental gradients: dynamics in the 'stable' tropics. Ecology 64: 1481-1494.

Klein, B. C. 1989. Effects of forest fragmentation on dung and carrion beetle communities in central Amazonia. 1989. Ecology 70(6): 1715-1725.

LaGro, J. Jr. 1991. Assessing patch shape in landscape mosaics. Photogrammetric Engineering and Remote Sensing 57(3): 285-293.

Lauga, J. and J. Joachim. 1992. Modelling the effects for forest fragmentation on certain species of forest-breeding birds. Landscape Ecol. 6(3):183-193.

Laymon, S. A. and J. A. Reid. 1986. Effects of grid-cell size on tests of a spotted owl HSI model. Pp. 93-96 In J. Verner, M. L. Morrison, and C. J. Ralph, Eds. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. Wisconsin

Press, Madison, WI. 470 pp.

Lefkovitch, L. P. and L. Fahrig. 1985. Spatial characteristics of habitat patches and population survival. Ecol. Modelling 30: 297-308.

Leopold, A. 1949. Sand County almanac. Oxford Univ. Press, New York, NY. 295 pp.

Levenson, J. B. 1981. Woodlots as biogeographic islands in southeastern Wisconsin. Pp. 13-39 In R. L. Burgess and D. M. Sharpe, Eds. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, NY. 310 pp.

Levin, S. A. 1992. The problem of pattern and scale in ecology. Ecology 73(6): 1943-1967.

Li, H., J. Franklin, F. Swanson, and T. Spies. 1993. Developing alternative forest cutting patterns: a simulation approach. Landscape Ecol. 8(1): 63-75.

Lovejoy, T., R. Bierregaard Jr., A. Rylands, C. Malcolm, C. Quintela, L. Harper, K. Brown, A. Powell, G. Powell, H. Schubart, and M. Hays. 1986. Edge and other effects of isolation on Amazon forest fragments. Pp. 257-285 In M. Soule, Ed. Conservation biology: the science of scarcity and diversity. Sinauer Associates, Inc. Sunderland, MA. 584 pp.

Ludwig, J. A. and J. F. Reynolds. 1988. Statistical ecology: a primer on methods and computing. John Wiley and Sons, New York, NY. 337 pp.

Lynch, J. F. and R. F. Whitcomb. 1977. Effects of the insularization of the eastern deciduous forest on avifaunal diversity and turnover. Pp.461-490 In Classification, inventory, and analysis of fish and wildlife habitat. USDI Fish. and Wildl. Serv. FWS/OBS 78/76. Washington, DC. 604 pp.

Lynch, J. F. and D. F. Whigham. 1984. Effects of forest fragmentation on breeding bird communities. Biol. Conserv. 28: 287-324.

Lyon, J. G., J. T. Heinen, R. A. Mead, and N. E. G. Roller.. 1987. Spatial data for modeling wildlife habitat. J. of Surveying Engineering. 113 (2): 88-100.

MacArthur, R. and E. Wilson. 1967. The theory of island biogeography. Princeton Univ. Press, Princeton, NJ. 203 pp.

MacClintock, L., R. F. Whitcomb, and B. L. Whitcomb. 1977. II. Evidence for the value of corridors and minimization of isolation in preservation of biotic diversity. American Birds 31: 6-16.

Mandelbrot, B. B. 1983. The fractal geometry of nature. W.H. Freeman and Company, New York, NY. 468 pp.

Matthiae, P. E. and F. Stearns. 1981. Mammals in forest islands southeastern Wisconsin. Pp. 55-66 In R. L. Burgess and D. M. Sharpe, Eds. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, NY.

McNeely, J. A., K. R. Miller, W. V. Reid, R. A. Mittermeier, and T. B. Werner. 1990. Conserving the world's biological diversity. The International Union for the Conservation of Nature and Natural Resources, World Resources Institute, Conservation International, World Wildlife Fund-US, and the World Bank. Gland, Switzerland and Washington, DC. 146 pp.

Mead, R., T. Sharik, S. Prisley, and J. Heinen. 1981. A computerized spatial analysis system for assessing wildlife habitat from vegetation maps. Can. J. Remote Sensing 7: 34-40.

Mech, L. D., S. H. Fritts, G. L. Radde, and W. J. Paul. 1988. Wolf distribution and road density in Minnesota. Wildl. Soc. Bull. 16: 85-87.

Meentemeyer, V. and E. O. Box. 1987. Scale effects in landscape studies. Pp. 15-34 In M. G. Turner, ed. Landscape heterogeneity and disturbance. Springer-Verlag, New York, NY. 239 pp.

Merriam, G. 1990. Ecological processes in the time and space of farmland mosaics. Pp. 121-133 In I. S. Zonneveld and R. T. T. Forman, Eds. Changing landscapes: An ecological perspective. Springer-Verlag, New York, NY. 286 pp.

Milne, B. T. 1988. Measuring the fractal dimension of landscapes. Appl. Math. Comp. 27: 67-79.

Monmonier, M. S. 1982. Computer-assisted cartography: principles and prospects. Prentice-Hall, Englewood Cliffs, NJ. 214 pp.

Murphy, D. L. 1985. Estimating neighborhood variability with a binary comparison matrix. Photogrammetric Engineering and Remote Sensing 51(6):667-674.

Musick, H. B. and H. D. Grover. 1991. Image textural measures as indices of landscape pattern. Pp. 77-103 In M. G. Turner and R. H. Gardner, Eds. Quantitative methods in landscape ecology. Springer-Verlag, New York, NY. 536 pp.

Naveh, Z. and A. S. Lieberman. 1984. Landscape ecology: theory and application. Springer-Verlag, New York, NY. 356 pp.

Noss, R. F. 1983. A regional landscape approach to maintain diversity. BioScience 33: 700-706.

Noss, R. F. and L. D. Harris. 1986. Nodes, networks, and MUM's: preserving diversity at all scales. Environ. Manage. 10: 299-309.

Oliver, M. A. and R. Webster. 1986. Semivariograms for modelling the spatial pattern of landform and soil properties. Earth Surface Processes and Landforms 11: 491-504.

O'Neill, R.V., D. DeAngelis, J. Waide, and T. Allen. 1986. A hierarchical concept of ecosystems. Princeton Univ. Press, Princeton, NJ.

O'Neill, R.V., J. R. Krummel, R. H. Gardner, G. Sugihara, B. Jackson, D.L. DeAngelis, B. T. Milne, M. G. Turner, B. Zygmunt, S. W. Christensen, V. H. Dale, and R. L. Graham. 1988. Indices of landscape pattern. Landscape Ecol. 1(3): 153-162.

O'Neill, R. V., R. H. Gardner, B. T. Milne, M. G. Turner, and B. Jackson. 1991. Heterogeneity and spatial hierarchies. Pp. 85-96 In J. Kolasa and S. T. A. Pickett, Eds. Ecological heterogeneity. Springer-Verlag, New York, NY. 332 pp.

Opdam, P. D., G. Rijsdijk, and F. Hustings. 1985. Bird communities in small woods in an agricultural landscape. Effects of area and isolation. Biol. Conserv. 34: 333-352.

Palmer, M. W. 1988. Fractal geometry: a tool for describing spatial patterns of plant communities. Vegetatio 75: 91-102.

Patton, D. R. 1975. A diversity index for quantifying habitat edge. Wildl. Soc. Bull. 3: 171-173.

Peet, R. K. 1974. The measurement of species diversity. Ann. Rev. of Ecol. and Syst. 5: 285-307.

Pelton, M. 1986. Habitat needs of black bears in the East. Pp. 49-53 In D. Kulhavy and R. Conner, Eds. Wilderness and natural areas in the eastern United States: a management challenge. Stephen F. Austin State Univ., Nacogdoches, TX. 416 pp.

Picton, H. D. 1979. The application of insular biogeographic theory to the conservation of large mammals in the northern Rocky Mountains. Biol. Conserv. 15: 73-79.

Pielou, E. C. 1977. Mathematical ecology. Wiley Publishing, New York, NY. 385 pp.

Ranney, J. W., M. C. Bruner, and J.B. Levenson. 1981. The importance of edge in the structure and dynamics of forest islands. Pp. 67-96 In R. L. Burgess and D. M. Sharpe, Eds. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, NY. 310 pp.

Reese, K. P. and J. T. Ratti. 1988. Edge effect: a concept under scrutiny. Trans. N. Amer. Wildl. and Nat. Resour. Conf. 53: 127-136.

Rice, J., R. D. Ohmart, and B. W. Anderson. 1983. Habitat selection attributes of an avian community: a discriminant analysis investigation. Ecol. Monogr. 53: 263-290.

Ripple, W. J., G. Bradshaw, and T. A. Spies. 1991. Measuring landscape pattern in the Cascades Range of Oregon. Biol. Conserv. 57: 73-88.

Risser, P. G., J. R. Karr, and R. T. T. Forman. 1984. Landscape ecology: directions and approaches. Illinois Nat. Hist. Survey Spec. Publ. No. 2. Champaign, IL. 18 pp.

Robbins, C. S. 1979. Effect of forest fragmentation on bird populations. Pp. 198-212 In Management of northcentral and northeastern forests for nongame birds. USDA For. Serv. North Central For. Exp. Sta., Gen. Tech. Rep. NC-51. St. Paul, MN. 268 pp.

Robbins, C. S., D. K. Dawson, and B. A. Dowell. 1989. Habitat area requirements of breeding forest birds of the middle Atlantic States. Wildl. Monogr. 103: 1-34.

Robinove, C. J. 1986. Spatial diversity index mapping of classes in grid cell maps. Photogrammetric Engineering and Remote Sensing. 52(8):1171-1173.

Robinson, S. and D. Wilcove. 1994. Forest fragmentation in the temperate zone and its effects on migratory songbirds. Bird Conserv. International 4: 233-249.

Roller, N. 1978. Quantitative evaluation of deer habitat. Pp 227-234 In Proc. Pecora IV Symp. Application of Remote Sensing Data to Wildlife Management. Sioux Falls, SD.

Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecol. Monogr. 52: 199-221.

Rosenberg, K.V. and M.G. Raphael. 1986. Effects of forest fragmentation in Douglas-fir forests. Pp. 263-272 In J. Verner, M. L. Morrison, and C. J. Ralph, Eds. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press, Madison, WI. 470 pp.

Roth, R. R. 1976. Spatial heterogeneity and bird species diversity. Ecology 57: 773-782.

Rudis, V. A. and J. B. Tansey. 1995. Regional assessment of remote forests and black bear habitat from forest resource surveys. J. Wildl. Manage. 59(1): 170-180.

Rudnicky, T. C. and M. L. Hunter, Jr. 1993. Reversing the fragmentation perspective: effects of clearcut size on bird species richness in Maine. Ecol. Applications 3(2): 357-366.

Schroeder, R. L. 1983. Habitat suitability index models: Pileated woodpecker. U. S. Fish Wildl. Serv. Biol. Rep. 82 (10.39). Washington, DC. 15 pp.

Schuerholz, G. 1974. Quantitative evaluation of edge from aerial photographs. J. Wildl. Manage. 38: 913-920.

Schultz, T. T. and L. A. Joyce. 1992. A spatial application of a marten habitat model. Wildl. Soc. Bull. 20: 74-83.

Shannon, C. E. and W. Weaver. 1963. The mathematical theory of communication. Univ. of Illinois Press, Urbana, IL. 117 pp.

Simpson, G. 1940. Mammals and land bridges. J. Wash. Acad. Sci. 30: 137-163.

Simpson, E. H. 1949. Measurement of diversity. Nature 163:688.

Small, M. F. and M. L. Hunter. 1988. Forest fragmentation and avian predation in forested landscapes. Oecologia 76: 62-64.

Stauffer, D. F. and L. B. Best. 1980. Habitat selection by birds of riparian communities: evaluating effects of habitat alterations. J. Wildl. Manage. 441: 1-15.

Stoms, D. M. 1992. Effects of habitat map generalization in biodiversity assessment. Photogrammetric Engineering and Remote Sensing 58(11): 1587-1591.

Swanson, F. J., T. K. Kratz, N. Caine, and R. G. Woodmansee. 1988. Landform effects on ecosystem patterns and processes. BioScience 38 (2): 92-98.

Sweeney, J. M. and W. D. Dijak. 1985. Ovenbird habitat capability model for an oak-hickory forest. Proc. Annu. Conf. Southeast Assoc. Fish and Wildl. Agencies 39: 430-438.

Temple, S. A. 1986. Predicting impacts of habitat fragmentation on forest birds: A comparison of two models. Pp. 301-304 In J. Verner, M.L. Morrison, and C. J. Ralph, Eds. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press, Madison, WI.

Temple, S. A. and J. R. Cary. 1988. Modeling dynamics of habitat-interior bird populations in fragmented landscapes. Conserv. Biol. 2(4): 340-347.

Thomas, J. W., Ed. 1979. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. USDA For. Serv. Agric. Handbook No. 553. Washington, D C. 512 pp.

Turner, M. G. 1990. Spatial and temporal analysis of landscape patterns. Landscape Ecol. 4(1): 21-30.

Turner, M.G., R. Costanza, and F. H. Sklar. 1989. Methods to evaluate the performance of spatial simulation models. Ecol. Modelling 48: 1-18.

Turner, M. G., R. V. O'Neill, R. H. Gardner, and B. T. Milne. 1989. Effects of changing spatial scale on the analysis of landscape pattern. Landscape Ecol. 3: 153-162.

Urban, D. L. and H. H. Shugart, Jr. 1986. Avian demography in mosaic landscapes: modeling paradigm and preliminary results. Pp. 273-279 In J. Verner, M.L. Morrison, and C.J. Ralph, Eds. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press, Madison, WI. 470 pp.

Urbanek, J. 1973. Some terminological and conceptual problems of complex landscape research. Pp. 1-5 In Proc. of the Third Intern. Symp. on the Content and Object of the Complex Landscape Research in the Protection and Formation of Human Environment. Bratislava, Czechoslovakia.

Van der Maarel, E. 1975. Man-made natural ecosystems in environmental management and planning. Pp. 263-274 In W. van Dobben and R. Lowe-McConnel, Eds. Unifying concepts in ecology. Junk, The Hague.

Van Dorp, D.and P. F. M. Opdam. 1987. Effects of patch size, isolation, and regional abundance on forest bird communities. Landscape Ecol. 1 (1): 59-73.

Wallin, D. O., F. J. Swanson, and B. Marks. 1994. Landscape pattern response to changes in pattern generation rules: land-use legacies in forestry. Ecol. Applications 4(3): 569-580.

Wehde, M. 1982. Grid cell size in relation to errors in maps and inventories produced by computerized map processing. Photogrammetric Engineering and Remote Sensing 48 (8): 1289-1298.

Weins, J. A. 1986. Spatial scale and temporal variation in studies of shrubsteppe birds. Pp. 154-172 In J. Diamond and T. Case, Eds. Community ecology. Harper and Row, New York, NY.

Weins, J. A. 1989a. Spatial scaling in ecology. Functional Ecology 3: 385-397.

Weins, J. A. 1989b. The ecology of bird communities. Vol. 2. Processes and variations. Cambridge Univ. Press, Cambridge, MA. 316 pp.

Weins, J. A. 1992. Ecological flows across landscape boundaries: A conceptual overview. Pp. 217-235 In A. J. Hansen and F. Di Castri, Eds. Landscape boundaries: Consequences for biotic diversity and ecological flows. Springer-Verlag, New York, NY. 452 pp.

Weins, J. A., C. S. Crawford, and J. R. Gosz. 1985. Boundary dynamics: A conceptual framework for studying landscape ecosystems. Oikos 45: 421-427.

Weishampel, J. F., D. L. Urban, H. H. Shugart, and J. B. Smith, Jr. 1992. Semivariograms from a forest transect gap model compared with remotely sensed data. J. of Vegetation Science 3: 521-526.

Welsh, C. J. and W. M. Healy. 1993. Effect of even-aged timber management on bird species diversity and composition in northern hardwoods of New Hampshire. Wildl. Soc. Bull. 21: 143-154.

Whitcomb, R. F., C. S. Robbins, J. F. Lynch, B. L. Whitcomb, M. K. Klimkiewicz, and D. Bystrak. 1981. Effects of forest fragmentation on avifauna of eastern deciduous forest. Pp. 125-205 In R. L. Burgess and D. M. Sharpe, Eds. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, NY. 310 pp.

Whitney, G. G. and J. R. Runkle. 1981. Edge versus age effects in the development of a beech-maple forest. Oikos 37: 377-381.

Whittaker, RH. 1972. Evolution and measurement of species diversity. Taxon 21: 213-51.

Wigley, T. and T. Roberts. 1994. A review of wildlife changes in southern bottomland hardwoods due to forest management practices. Wetlands 14(1): 41-48.

Wilcove, D. S., C. McLellan, and A. Dobson. 1986. Habitat fragmentation in the temperate zone. Pp. 237-256 In M. Soule, Ed. Conservation biology: The science of scarcity and diversity. Sinauer Associates, Sunderland, MA. 584 pp.

Yahner, R. H. 1988. Changes in wildlife communities near edges. Conserv. Biol. 2(4): 333-339.

Yahner, R. H. and D. P. Scott. 1988. Effects of forest fragmentation on depredation of artificial nests. J. Wildl. Manage. 52(1): 158-161.

Yost, R. S., G. Uehara, and R. L. Fox. 1982. Geostatistical analysis of soil chemical properties of large land areas. I. Semivariograms. Soil Sci. Soc. Am. J. 46: 1028-1032.

Zonneveld, I. I. and R. T. T. Forman, Eds. 1990. Changing landscapes: an ecological perspective. Springer-Verlag, New York, NY. 286 pp.

APPENDIX 1. Batch program for landscape expression computation.

```
echo off
echo                    LAND Batch Program
echo        Launch Pascal Landscape Program and Idrisi Functions
echo                Computes 24 Landscape Expressions
rem
rem
rem  Call it up by typing     land filename
rem  Converting map and sending as File1.dat to turbo directory
rem
convert x %1 File1 i 1 1
rem
copy c:\Idrisi\clustgrhs\File1.img c:\tp\File1.dat
rem
rem    Grouping and reclassing map as File2.dat to turbo directory
rem **Before doing the reclass below, forest must be classified as "1"
rem **Reclassify agriculture, roads, power lines from 2 - 999
group x %1 n zgrp
scalar x zgrp ztmp 1 1
reclass x i %1 zrecl 2 0 2 999 -9999
overlay x 3 zrecl ztmp zforgrp
convert x zforgrp File2 i 1 1
rem
rem  Sending File2.dat to turbo directory
copy c:\Idrisi\clustgrhs\File2.img c:\tp\File2.dat
rem
cd c:\tp
turbo
cd c:\Idrisi
rem echo.
rem echo                    BCM
rem echo          ---------------------------------------
rem
rem echo Entering Pattern with original map name
rem echo Choose BCM with output filename zbcm
rem echo.
rem pause
pattern x 7 %1 zbcm # Binary Comparison Matrix
rem echo Calling HISTO with zbcm
rem pause
```

```
histo x zbcm .1 1 1
echo Write down BCM mean
pause
rem echo.
rem echo                    INTERSPERSION
rem ECHO            ---------------------------------------
rem
rem echo Entering Pattern with original map name
rem echo Choose CVN - (Interspersion) with output filename zcvn
rem echo.
rem pause
pattern x 6 %1 zcvn # Interspersion
rem echo Calling HISTO with zcvn
rem pause
histo x zcvn .1 1 1
echo Write down CVN mean
pause
rem echo.
rem echo                    RICHNESS
rem ECHO            ---------------------------------------
rem
rem echo Entering Pattern with original map name
rem echo Choose Relative Richness with output filename zrich
rem echo.
rem pause
rem pattern x 1 %1 zrich 3 Richness
rem echo Calling HISTO with zrich
rem pause
rem histo x zrich 1 1 1
rem echo Write down Richness mean
rem pause
rem ECHO.
rem ECHO            NO. DIFFERENT CLASSES  NDC
rem ECHO            ------------------------------------------
rem
rem echo   Entering Pattern with original map name
rem echo   Choose No. Different Classes with output filename zndc
rem echo.
rem pause
pattern x 5 %1 zndc # Number of Different Classes
rem echo Calling HISTO with zndc
rem pause
```

```
histo x zndc 1 1 1
echo Write down NDC mean
pause
echo.
echo.
echo.
echo.
echo.
echo                    SPATIAL DIVERSITY
echo                    ---------------------------------------------------
echo.
echo.
echo                (NDC Mean * No. of Map Cells) - No. Map Cells
echo                -------------------------------------------------------
echo                        (No. of Vegetation Types - 1)
echo.
echo.
echo  IF and ONLY IF No. of Map Cells is not equal to 100 then:
echo  Multiply final answer by 100; then divide by No. of Map Cells
echo.
echo.
pause
echo                    TOTAL FOREST EDGE
echo                    ---------------------------------------------------------
echo
echo  Calling Perim with original map name; choose tabular, sides
echo.
pause
perim
echo  Total Edge  =  ( No. of forest sides  *  Cell Size )
echo.
echo  Forest sides is indicated by category "1" above.
pause
echo.
echo                    PATTON'S DIVERSITY INDEX
echo                    ---------------------------------------------------
echo.
echo  Calling Perim with input file zgrp; output as zperim
echo  Select image; meters.
echo.
pause
perim x zgrp 1 2 zperim
```

```
area x zgrp 1 4 zarea
scalar x zarea z3 3 3.1415
transfor x z3 z4 5
scalar x z4 z5 3 2
overlay x 4 zperim z5 z6
overlay x 3 z6 zrecl zpatton
echo Calling Histo with input file zpatton
pause
histo x zpatton .1 1 1
echo     Weighted Patton's Index =   ( Mean * (df + 1) )
echo     ---------------------------------------------------
echo                        No. of Forest Cells
echo.
pause
echo                    CONVEXITY or PERIMETER : AREA RATIO
echo                    ---------------------------------------------------
echo
echo                                  Total Edge (m)
echo          Convexity =  ----------------------------------------
echo                              Total Forest Area/ha * 10,000
echo.
pause
echo.
echo                        COMPACTNESS
echo                    ----------------------------------
echo
echo Calling CRATIO with zarea & zperim inputs; zcrat1 output
pause
cratio x zarea zperim zcrat1
overlay x 3 zcrat1 zrecl zcrat2
echo Calling HISTO with zcrat2 input
pause
histo x zcrat2 .1 1 1
echo                                      ( Mean * (df + 1) )
echo   Weighted Compactness Index =   ------------------------------
echo                                       No. of Forest Cells
echo.
pause
echo.
echo                        CONNECTIVITY
echo                    ----------------------------------
echo.
```

```
group x %1 y zgrpd
scalar x zgrpd ztmp2 1 1
overlay x 3 ztmp2 zrecl zconn
rem
echo      From the GRAPHICS SCREEN,
echo      No. Forest Groups W/ Diagonals = # graph bars - 1
echo.
pause
histo x zconn 1 1 1
echo No. Corridors  =  No. Forest Patches  -  No. Forest Groups w/Diagonals
echo.
echo.
echo.
echo     Connectivity     =            No. Corridors
echo                          ------------------------------------
echo                          3  *  ( No. Forest Patches - 2 )
pause
echo.
echo                    FRAGMENTATION INDEX - MONMONIER
echo                    ------------------------------------------------------
echo.
echo   Calling HISTO with zgrp
pause
histo x zgrp 1 1 1
echo
echo    Total No. Groups (Forest & all Non-Forest categories) = ( Actual maximum + 1 )
echo.
echo                                   Total No. Groups
echo        Fragmentation Index  =  --------------------------------
echo           Monmonier                No. of Map Cells  - 1
echo.
pause
echo.
echo                    FRAGMENTATION INDEX - RIPPLE
echo                    ------------------------------------------------
echo
pause
reclass x i %1 zopp 2 0 1 2 1 2 999 -9999
distance x zopp zfrag
echo    Calling HISTO with zfrag for mean value
pause
histo x zfrag .1 1 1
```

```
echo                                  ( Mean * (df + 1) )
echo  Fragmentation Index Ripple =  ------------------------------ * (Cell Length)
echo                                   No. of Forest Cells
echo.
pause
echo.
echo                    PROPORTION INTERIOR FOREST
echo                  ------------------------------------------------
echo                 ** NOTE: Coded for 30m buffer
echo.
pause
reclass x i zfrag zinter 2 0 0 1.05 1 1.05 9000 -9999
echo         Calling HISTO with zinter
echo         Choose NUMERIC to get proportion of "1" cells
echo.
pause
histo x zinter 1 2 1
pause
echo.
echo                          CONTIGUITY
echo                  ------------------------------------------
rem
pause
filter x zrecl zfilt 6 1 2 1 2 1 2 1 2 1
overlay x 3 zrecl zfilt zcontig
echo Calling HISTO with zcontig
pause
histo x zcontig .1 1 1
echo                                [  (Mean * (df + 1) )  ]
echo                                [  ------------------ - 1  ]
echo                                [  No. of forest cells   ]
echo              Contiguity =  ------------------------------------------- + 1
echo                                          12
pause
echo.
echo                    SPATIAL INTEGRITY
echo                  ------------------------------------
echo.
echo    Spatial Integrity = (No. holes) + (1 - No. Forest Patches)
echo.
echo.
echo    Holes must be totally contained within a forest patch to count
```

APPENDIX 2. Pascal program for landscape expression computation.

```
Program Landscape (Input, Output);

{This program requires maps converted to integer, ASCII files                }
{Alter CONST vars to handle larger maps and patches                          }
{Currently set at maximum 100 pixel maps, 100 vegetypes, 100 forest patches}

CONST
   File1 = 'File1.dat';
   File2 = 'File2.dat';

VAR
  vegetype : array [1..100] of integer;          {Pixel dimensions}
  B: array [1..10,1..10] of integer;
  cellcount,xcoord,ycoord : array [1..100] of integer;
  p : array [1..100] of real;
                                                 {Patch dimensions}
  edgedist,centerx,centery : array [1..100] of real;
  area : array [1..100] of real;
  radius : array [1..100] of real;
  F,G : text;
  maxtype,m,pixels,sum,nrows,ncols,count,temp,i,n : integer;
  count2,higroup,numgroups,k,forest,row,col,l,j : integer;
  sumdist,dist1,dist2,distance,interpatch,forarea,forest_cover : real;
  maparea,patch_density,mean_patch_size,sumsqrs,devsqrd,stddev : real;
  simpsum,shansum,evensum,simpsons,shannon,dominance,evenness,a : real;
  simp1,simp2,cellsize,patchiness,forest_area_hectares : real;

BEGIN
  For i:= 1 to 100 do                            {*Change these if needed*}
      Begin
      vegetype[i] := 0;
      cellcount[i] :=0;
      xcoord[i] := 0;
      ycoord[i]:= 0;
      End;

  For i := 1 to 100 do
     Begin
     edgedist[i] := 0;
```

```
    centerx[i] := 0;
    centery[i] := 0;
    area[i]:= 0;
    radius[i]:= 0;
    End;

 Write ('What is the number of rows in the original map?   ');
 Readln(nrows);
 Write ('What is the number of columns?   ');
 Readln(ncols);
 Pixels := nrows * ncols;
 Assign (F, File1);
 Reset (F);
 count := 0; maxtype := 0; sum :=0;

 Repeat
      Inc (count);
      Readln(F, m);                          {Read original map}
      vegetype[m] := vegetype[m] + 1;
      If (maxtype < m) then maxtype := m;    {Find largest forest code}
 until (count = pixels);
 Close (F);

 n := 0; simpsum := 0;shansum := 0; evensum := 0;
 For i := 1 to maxtype do
    Begin
       If (vegetype[i] <> 0) then
         Begin
            Inc(n);
            simp1 := vegetype[i] * (vegetype[i] - 1);
            simp2 := pixels * (pixels - 1);
            simpsum := simpsum + (simp1/simp2);
            p[i] := (vegetype[i] / pixels);
            shansum := shansum + ( p[i] * ln(p[i]));
            evensum := evensum + sqr (p[i]);
         End;
    End;

Simpsons := 1 - simpsum;
Shannon := (-1) * shansum;
Dominance := ln(n) - Shannon;
If (ln (n) = 0) then evenness := 0;
```

```
If (ln (n) <> 0) then evenness := ( -1 * ln(evensum) ) / ln (n);
writeln;

Writeln('Number of Map Cells is ',pixels);
Writeln('Number of Vegetation Types is ',n);
Writeln('Shannon-Weaver Index is ',shannon:5:3);
Writeln('Dominance is          ',dominance:5:3);
Writeln('Simpsons Index is      ',simpsons:5:3);
Writeln('Evenness Index is      ',evenness:5:3);
writeln;
Writeln('Press any key to continue'); writeln;
Readln;

{This program portion inputs File2.dat - an image where forest has          }
{been reclassed as 1, nonforest as 0, then GROUP applied.  Result           }
{is then OVERLAY MULTIPLIED with original image to zero out groups}
{representing only zeros, leaving only forest patches with group #s.         }

Write('What is the cell size?  e.g., 30., 60.  ');
Readln (cellsize);
Assign (G,File2);                    {Read in map, find highest group number}
Reset (G);
count2 := 0; higroup := 0; numgroups :=0;

Repeat
    For row := 1 to Nrows do
       Begin
       For col := 1 to ncols do
          Begin
          Readln (G,k);
          Inc (count2);
          B[row,col] := k;
          If (higroup < k) then higroup := k;
          End;
       End;
     Until (count2 = pixels);
Close (G);

forest := 0; numgroups :=0;
For i := 1 to higroup do                          {Finds center and radius of}
   Begin                                                 {each forest patch}
   For row := 1 to nrows do
```

```
        Begin
        For col := 1 to ncols do
           Begin
           If ( B[row,col] = i) then
             Begin
             cellcount[i] := cellcount[i] + 1;
             xcoord[i] := xcoord[i] + row;
             ycoord[i] := ycoord[i] + col;
             End;
           End;
         End;
      If (cellcount[i] <> 0) then
        Begin
        numgroups := numgroups + 1;                    {Finds actual group #s}
        centerx[i] := xcoord[i] / cellcount[i];
        centery[i] := ycoord[i] / cellcount[i];        {Area corrected}
        area[i] := cellcount[i] * sqr (cellsize);        {for cell size}
        radius[i] := sqrt (area[i] / 3.141592654);
        forest := forest + cellcount[i];
        End;
    End;

sumdist :=0; l := 0;                              {Begin calculations on patch distance}
    For i := 1 to higroup - 1 do                  {Distance corrected}
       Begin                  { for cell size }
       For j := i + 1 to higroup do
          Begin
          If ((cellcount[i] <> 0) AND (cellcount[j] <> 0)) then
            Begin
            Inc(l);
            dist1 := sqr ( centerx[i] - centerx[j]);
            dist2 := sqr ( centery[i] - centery[j]);
            distance := ( sqrt (dist1 + dist2) ) * cellsize;
            edgedist[l] := distance - radius[i] - radius[j];
            If (edgedist[l] < 0.0) then edgedist[l] := 0;
            sumdist := sumdist + edgedist[l];
            End;
          End;
       End;

    If (numgroups > 1) then interpatch := sumdist/l;
    If (numgroups = 1) then interpatch := 0;
```

```
    forarea := forest * ( sqr(cellsize));
    forest_area_hectares := forarea/10000;
    forest_cover := (forest/pixels);
    maparea := (pixels * sqr (cellsize)) / 10000;  {in hectares}
    patch_density := numgroups / maparea;
    mean_patch_size := forarea / numgroups;

    {Calculate standard deviation of interpatch distance for index of heterogeneity}
    sumsqrs := 0;
    For i := 1 to higroup do
      Begin
      If (edgedist[i] <> 0) then
        Begin
        devsqrd := sqr (edgedist[i] - interpatch);
        sumsqrs := sumsqrs + devsqrd;
        End;
      End;
    stddev := sqrt (sumsqrs / numgroups);
    If (interpatch <> 0) then patchiness := 100 * (stddev / interpatch);
    If (interpatch = 0) then patchiness := 0; writeln;

    Writeln('Index of heterogeneity or patchiness:  ',patchiness:7:2);
    Writeln('Mean Interpatch Distance:              ',interpatch:6:2);
    Writeln('Proportion of Forest Cover:            ',forest_cover:4:2);
    Writeln('Patch Density/ha:                      ',patch_density:6:3);
    Writeln('Mean Patch Size/ha:                    ',mean_patch_size/10000:6:3);
    Writeln('Number of Forest Patches:              ',numgroups:6);
    Writeln('Total Forest Area/ha:                  ',forest_area_hectares:8:2);
    Writeln('Total Map Area/ha:                     ',maparea:9:1);
    Writeln('Number of Forest Cells:                ',forest:9);
    Writeln;
    Writeln('Press any key to end.....................');
    readln;
END.
```

VITA

Margaret Katherine Trani was born in San Pedro, California, the third daughter of Garmino T. and Louise A. C. (Behling) Trani. After graduation from San Pedro High School, she enrolled in California State University, Long Beach earning her Bachelor of Science in Zoology with a minor in Physical Science. She pursued graduate studies on the ecology of gray fox (Urocyon cinereoargenteus) with Dr. Archie Mossman at California State University, Humboldt, and was awarded the Master of Science degree in Wildlife Ecology. While an instructor and computer laboratory manager at Southern Utah State University, she earned her Bachelor of Science in Computer Science.

Before subsequently enrolling as a doctoral candidate at Virginia Polytechnic Institute and State University, Margaret gained professional experience with the Utah Division of Wildlife Resources, the U. S. Park Service, and the U. S. Fish and Wildlife Service. While in the Department of Fisheries and Wildlife Sciences, she studied the importance of old-growth forest and the relevance of landscape pattern analysis to wildlife resources. She also taught two courses relating to computers in natural resources while at VPI & SU and actively volunteered in environmental education for several area schools and youth organizations.

Throughout most of her doctoral program, she supported herself by working for the U.S. Forest Service in both a wildlife management and computer analyst capacity. During that time, she was also raising her daughter, Jennifer, and son, Joshua through elementary and into middle school. She is married to Paris Eugene Griep.

Margaret Katherine Trani