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# DISSERTATION

# AVIAN RESPONSE TO CHANGES IN LANDSCAPE STRUCTURE IN THE INTER-MOUNTAIN WEST

Submitted by:

Donald O. Hunter

Graduate Degree Program in Ecology

In partial fulfillment of the requirements

for the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Summer 1999

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WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY DONALD OWEN HUNTER ENTITLED AVIAN RESPONSE TO CHANGES IN LANDSCAPE STRUCTURE IN THE INTER-MOUNTAIN WEST BE ACCEPTED AS FULFILLING IN PART THE REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

Committee on Graduate Work

Roger M. Hoffer
Dr. Roger Hoffer, Advisor

Eugene Decker
Professor Eugene Decker

Freeman M. Smith
Dr. Freeman Smith

Curtis H. Flather
Dr. Curtis H. Flather

Dan Binkley
Dr. Dan Binkley

Dan Binkley
Dr. Dan Binkley, Director

# ABSTRACT OF DISSERTATION

# AVIAN RESPONSE TO CHANGES IN LANDSCAPE STRUCTURE IN THE INTER-MOUNTAIN WEST

This research reconciled and compared two, large-area surface cover data sets to examine relationships between temporal changes in landscape structure and trends in breeding bird populations. From remotely sensed data for two time periods approximately 20 years apart, landscape structure was measured within 40 km circles around 45 USGS, Breeding Bird Survey routes in western Montana and northern Idaho. Five landscape structure indices, indicative of fragmentation, were calculated for both time periods in each circle (mean patch size, mean patch size coefficient of variation, edge, percent forest, and mean nearest neighbor distance). For each circle, metrics of change between the two time periods were correlated to individual bird population trend and to current abundance.

Inconsistencies between the surface cover data sets limited the measure of landscape structure changes to only the change in forest cover. Twenty-five bird species had acceptable correlation models of response ($p \leq 0.10$) to the forest component of landscape structure. Patterns of response showed some inter-mountain species were affected by forest fragmentation in similar fashion to eastern species: woodland species, predominantly migrants, exhibited a positive relationship between population trend and mean patch size and percent forest, whereas edge species exhibited a negative relationship. Similar patterns were observed in the spatial abundance of birds, calculated as a current five-year average. Overall, these results showed that useful measures of landscape change can be quantified for a large area, yielding change measures that correlated to a biological response. Results support the approach of blending remote sensing technology with ecology to conduct applied ecological research and as a possible means of monitoring entire ecosystems.

Donald O. Hunter
Graduate Degree Program in Ecology
Colorado State University
Fort Collins, CO 80523
Summer, 1999

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To my sons
Logan, Jesse, and Ben

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# CHAPTER 1. INTRODUCTION

## 1.1 Background

Single species management is giving way to "ecosystem management," an evolving concept that draws theoretical strength from conservation biology and landscape ecology, disciplines that focus more on the interplay of ecological processes than on the autecology of individual species (Soule´ 1986, Forman and Godron 1986). Even though a universally accepted definition of ecosystem management has yet to emerge, conservation agencies, finding single species management too costly and too narrowly focused, are shifting management perspective to the macro-scale, i.e., the ecosystem (FEMAT 1993, USDA 1995, Overbay 1992). Conservation agencies have arbitrarily scaled the term ecosystem to landscapes and regions, usually a watershed or a combination of watersheds and administrative boundaries (USFWS 1995, NPS 1996, Overbay 1992). As the unit area of study has increased, new models of ecosystem analysis have emerged that draw ecological inference from statistical patterns measured over large areas (Brown 1995). The change in perspective from species to ecosystem has brought about a strategic shift in research aimed at understanding and predicting the emerging patterns and trends in ecological systems (USGS, BRD 1996).

In ecology, predictive reliability increases as the spatial and temporal dimensions of research increase, yet resource managers and ecologists tend to make predictions from short-term studies (Wiens 1989a, May 1993). Heretofore, macro-scale studies have been limited by lack of data and computer processing power readily available to the ecologist and natural resource manager (Hirsch et al. 1979, Flather and Hoekstra 1989), and limited also by increased cost and complexity of computer analysis (Edwards et al. 1993). In addition, there is a paucity of good quality, large-area data sets (Opdam 1991). Ecologists and conservation managers would like to see more studies at the macro-scale (Caicco et al. 1995, May 1993, Maurer 1985, Noon et al. 1985, Helle 1986, Hunter 1987, Knight 1987, Ricklefs 1987, Turner 1987, Swanson 1990), but connecting large-area studies with the underlying principles of ecology is an emerging field.

Landscape ecology is founded on the premise that landscape structure, the composition and arrangement of habitat patches, affects ecosystem functions and processes. Changes in structure should produce corresponding changes in ecosystem function and process (Li and Reynolds 1994). The mechanisms of influence from small to large areas, however, are poorly understood and difficult to study. Though not rigorously tested at the landscape level (McGarigal and McComb 1995), several guiding concepts and principles have emerged that provide a theoretical framework for quantitative studies of landscape changes (Forman 1995, Risser et al. 1984, Saunders et. al 1991, Turner 1989). Some of the principles relevant to this research include: (i) changes in pattern and process, over time, are influenced by the land mosaic, (ii) large and small patches are natural and essential to ecological function, (iii) patch shape and size influence ecological function; and, (iv) species extinction rates decrease as patch size increases, or stated differently, species survival increases as its available habitat increases. These are general principles that are system and species dependent.

Brown's (1995 ) introduction of "macroecology" as a sub-discipline of ecology offers some guidance for macro-scale research. He suggests the characterization of statistical pattern observed over large areas offers meaningful inference on species abundance. Further, he contends it is important to discover large system properties without full understanding of underlying process — macroecology deliberately sacrifices detail in search of the larger picture. While the macroecology approach lacks the rigor and power of an experimental approach, it offers greater potential for generality and synthesis, which may be more appealing to land mangers.

In forested systems, fragmentation reduces total forest habitat, increases the number of patches, decreases average patch size, and may increase the distances among patches. Fragmentation implies a change on the landscape where larger patches are in some way reduced to smaller patches that alter the overall landscape mosaic. For example, logging and road construction can turn several large intact forests into many smaller forests. Over time, patches change in size and shape, thus, fragmentation has both a temporal and a spatial influence on organisms. Studies in the eastern and mid-western United States show avian diversity decreases in response to fragmentation (Whitcomb et al. 1981, Ambuel and Temple 1983, Blake and Karr 1984). More telling patterns of avian response to fragmentation have been observed in bird groups associated with the forest edge and forest interior (non-edge). Literature references to these groups often use woodland, forest-interior, and forest interchangeably. Hereafter, 'woodland' is used to mean species that breed inside the forest, away from the forest edge.

In general, forest fragmentation tends to decrease the abundance of woodland species, such as neotropical migrants (Robbins et al. 1992), while increasing the abundance of edge species (Robbins et al. 1989, Robinson and Wilcove 1994, Ambuel and Temple 1983, McIntyre 1995). Woodland species tend to be more specialized than edge species, and have greater sensitivity to predation, brood parasites, competition, and food availability (Robinson et al. 1995). Given these findings, McIntyre (1995) stressed the importance of assessing the impact of fragmentation (in the United States) on neotropical species and Howe (1992) suggests neotropical species are sensitive indicators to environmental change. Note, however, that fragmentation does not necessarily imply habitat loss. Recent studies of fragmentation tend to separate habitat amount from habitat arrangement (Fahrig 1997).

The majority of avian-fragmentation research has taken place in eastern landscapes. Studies in western landscapes are too few and limited (spatially or temporally) to confirm a consistent pattern. McGarigal and McComb (1995) caution that the relationship of eastern bird species to fragmentation has not been fully verified in western species, and Opdam (1991) notes that most avian-fragmentation studies have been conducted in agricultural landscapes, inferring that western species may have evolved in more naturally heterogenous landscapes and may have adapted strategies for coping with fragmented landscapes. Hansen and Urban (1992) stress that life history traits and the trajectory of landscape change may produce geographically unique patterns of response in avian guilds. In comparing similar bird guilds of eastern deciduous forests and the Pacific northwest, they noted greater sensitivity to fragmentation by the eastern guilds. They cautioned that the community dynamics of one system may not extrapolate to another, but add that new approaches are needed that can evaluate past, present, and future landscape dynamics in relation to a broad range of species and guilds.

Avian guilds, defined as a group of species that exploit the same class of resources in similar ways (Root 1967), present one means of simplifying the interpretation of avian response to environmental change. Avian guilds are most often structured around foraging (DeGraff et al. 1985, Verner 1984). Many species have overlapping life histories, adding a subjective dimension to guild assignment, diffusing the value of guilds as an interpretative and predictive tool. Verner (1986) cautions against the use of guild indicators (one species that represents the guild) to assess trends, opting instead for a whole-guild approach. Questioning the value of functional guilds, Szaro (1986) introduces the concept of response guilds, grouping species that respond similarly to habitat perturbations. He contends this grouping is much better suited to the prescriptive practices of conservation agencies, but suggests also that guilds may only be useful as pointers to specific habitat factors that are important to certain species.

Birds have been associated with landscape structure (Lack 1942, Weins 1989, Flather and Sauer 1996, McGargil and McComb 1995, Flather et al. 1992) in both time and space. Local or patch-level ecological processes produce patterns that emerge at higher hierarchical scales, such as the mosaic of heterogenous patches that is observed at local and macro-scales. For birds, patterns may be observed as temporal trends: are local populations going up or down? Or, patterns may be observed as spatial abundance: how are birds distributed among patches? Although birds have been shown to associate with landscape structure, avian responses to change can be troublesome to interpret. Patch occupancy can vary from year to year, yet the population as a whole persists regionally as a metapopulation. In addition, changes in landscape structure that disrupt process and pattern influence the biotic and abiotic elements of an ecosystem in different ways: some species lose habitat, while others thrive on the altered condition. As the landscape mosaic changes, the new landscape may interrupt the movement of organisms, flow of energy, and frequency of disturbance. All of these factors, in turn, affect the dynamics of populations through reproduction, recruitment, competition, abundance, and ultimately, persistence (Soule´ 1986).

Faced with ever-changing landscapes, land managers are calling for more research that moves ecological principle to management practice at the macro or landscape scale, the scale in which they tend to manage. As a tool for land managers, change in landscape structure has greater value when linked to a biological response (Knopf et al. 1988, Szaro 1986, Verner 1984), such as species trend and abundance. This research attempts to link large-area change in landscape structure to avian population trend. The tools, methods, and analytical approaches used are more applied than theoretical so the results may be more meaningful to natural resource managers. In addition, the objectives seek confirmation of major statistical trends.

## 1.2 Objectives

The introduction suggests a relationship between landscape structure and ecosystem function and process. Changes in landscape structure, therefore, might produce observable patterns, relationships, or associations — a biological response — that could be used in ecosystem management as measures of acceptable change. Extant data support a **temporal analysis** that correlates avian population **trend** (~ 20 years) and changes in landscape structure. Also, for a comparative analysis of temporal patterns of correlation to those patterns observed spatially, a **spatial analysis** is feasible that correlates avian population **abundance** (5-year average) and current landscape structure. For example, do the same species or species groups that decline over time due to increased fragmentation also demonstrate lower abundance in more patchy habitat? The principal objective of this research is to examine the relationships between changes in landscape structure over a large area in the inter-mountain west in comparison to avian population trends. Specific objectives included:

1. To reconcile and compare two generally accessible spatial data sets (separated by ~20 years) for use as measures of change in landscape structure. Usefulness will be determined by:
    - the degree to which inconsistencies can be reconciled;
    - whether or not the combination of the two data sets can be used to quantify landscape change.

2. To determine avian patterns of response to changes in landscape structure. Patterns of response of inter-mountain birds will be compared to the patterns reported for eastern birds and bird groups such as:
    - fragmentation has a negative effect on woodland species' population trend;
    - fragmentation has positive effect on edge species' population trend;
    - fragmentation effects are greater for migrant species.

3. To determine whether bird species that respond to temporal changes in landscape structure respond similarly to those species that associate spatially with landscape structure. To check for temporal-spatial relationships, comparisons will be made

between birds that correlate to temporal change and birds that correlate to spatial pattern.

This research is an attempt to find observable patterns of biological response — ecological sensitivity — in a changing landscape. Recognizing that this scale of observation defies a tight experimental approach, methods and analytical procedures were used that help to detect and characterize overt statistical patterns from which meaningful ecological inference might be made.

The current study adds to a very small body of work that attempts to characterize ecological responses to change over large geographic areas. The approach used, blending remote sensing technology and ecology, offers new insights into the usefulness and the problems associated with monitoring large ecosystems. And granting some degree of truth to Saunders et al. (1991), who contended most ecological studies are of little practical use to natural resource managers, the current study used information technology, analysis methods, and software programs that were not theoretical but generally accessible to land management professionals.

# CHAPTER 2. LITERATURE REVIEW

## 2.1 Ecosystem Management and Macroecology

Ecosystem management draws its theoretical underpinning from the ecological subdisciplines of conservation biology (Soule´ 1986) and landscape ecology (Forman and Godron 1986). Even though a universal definition has yet to emerge, ecosystem management has come to be embraced by federal and nongovernment conservation agencies as a practical alternative to single species management (USDA 1995, FEMAT 1993, Overbay 1992), reshaping management practices and creating a focus for applied research (USGS 1996) that covers large areas. Brown (1995) presents "macroecology" as another subdiscipline of ecology which uses statistical pattern analysis as an alternative way of making inference about relationships between organisms and their environment. He concedes the macroecology approach lacks the rigor and power of an experimental approach, but that large-area studies offer greater potential for making generalizations that are more appealing to land managers. Macroecology places value on the study of whole-system properties without full knowledge of underlying process. However, not all researchers are in full agreement as to the functional value or approach to large-area studies (Brown 1995).

Istock and Scheiner (1987), and Knight (1987) describe landscapes as shifting mosaics, possessing properties at different scales. Turner (1990) suggests landscape pattern can be directly related to ecological processes at different scales. Cale et al. (1989) argued that studies of pattern, isolated from process, are not likely to advance ecological theory, while Edwards et al. (1993) contends a focus on ecological minutiae ignores complicating factors at the macro scale and contents that a "rough and ready" answer is better than no answer at all. Turner et al. (1989) narrowed the view of landscape change to that of disturbance vectors, natural or human-caused, that can be modeled for predictive purposes. In an effort to simplify the complexity of landscapes, Urban et al. (1987) present a hierarchical paradigm, suggesting human disturbance tends to re-scale patterns in time and space. Products of these newer disciplines are beginning to find value in prescriptive management practices, such as in determining the size and shape of a reserve (Simberloff and Abele 1982, Blake and Karr 1984). A number of authors use the effects of landscape fragmentation on vertebrate populations as a foundation for prescriptions to improve conservation management practices (Harris 1984, Hunter 1987, Lajeunesse et al. 1995, Lamberson et al. 1992, May 1993, Temple 1986, Murphy and Noon 1992).

## 2.2 Large Spatial Data Sets

In this research, the analytical model uses remotely sensed data to detect temporal changes in habitat structure at the landscape-level. Geographic information systems

(GIS) and statistical software provide the technical foundation for subsequent analysis. Several authors observe that information technology has evolved, becoming more analytically powerful and better suited to landscape-level studies (Wickham and Norton 1994, Wickham et al. 1995, Mladenoff et al. 1993). Meentemeyer and Box (1987) and Reed et al. (1993) advocate the use of GIS and remote sensing for landscape-level studies. Winne (1996) emphasizes the critical importance of change detection, using remote sensing of large areas, as a key element to long-term monitoring, which has been prescribed and needed by conservation agencies. Hess (1994), however, points out the dangers of wide-spread use of unvalidated GIS data and Cherrill and McClean (1995) caution that temporally separated land cover maps may be fraught with observer error. Minns (1992), however, finds GIS and computer tools in general to be integral "holistic" tools, and stresses their potential for gathering and integrating regional environmental data.

### 2.3 Landscape Fragmentation Effects

Practitioners of ecosystem management are directing research aimed at providing conservation agencies with a practical set of guiding principles (Forman 1995, Godron and Forman 1983, Risser et al. 1984, Saunders et al. 1991). Hutchinson (1953) recognized early-on that fragmentation alters the mosaic and its inherent properties, affirming a deeper relevance of pattern. Unfortunately, the nature of landscape-level research, long-term and macro-scale, makes it difficult to test these principles, as noted by McGarigal and McComb (1995). Consequently, universal principles, however, still elude ecologists. Since some taxa thrive quite well in fragmented habitat, disagreement among the scientific community persists as to the true ecological significance of fragmentation (McGarigal and McComb 1995, McCoy and Mushinsky 1994, McIntyre 1995, Weins 1989). As a result, three bird groups are given more attention in avian-fragmentation literature: woodland, edge species, and migrants.

### 2.3.1 Woodland and Edge Species

An impressive literature base of avian studies, primarily in the eastern United States, shows fragmentation—the most studied aspect of landscape change—tends to decrease the abundance of woodland species and to increase the abundance of edge species (Askins and Philbirck 1987, Whitcomb et al. 1981, Ambuel and Temple 1983, Blake and Karr 1984, McIntyre 1995, Opdam 1991, Opdam et al 1993, Robbins 1979). Ambuel and Temple (1983) and Robinson et al. (1995) found fragmentation exposes woodland species to predators and nest parasites. Opdam (1991), Shaffer (1981), and Slobodkin (1986) find metapopulation theory closely tied to fragmentation dynamics, especially in studies on vertebrate response. McGarigal and McComb (1995), however, found that 15 species they studied in the Pacific Northwest did not respond to fragmentation in a way that was consistent with eastern studies. These findings highlight the need for more research in the western states.

McCargil and McComb (1995), and Opdam (1991) caution that the responses noted for woodland and edge species in the eastern United States have not been noted for western species, suggesting ecologists have too quickly embraced the island biogeography work of McArthur and Wilson (1967) as universally applicable to all terrestrial ecosystems. McGarigal and McComb's (1995) study of breeding birds in the Oregon coast range and other similar research (McIntyre 1995, Opdam 1991, Reed et al. 1996b, Merriam and Wegner 1992, Flather and Sauer 1996) helped frame the theoretical background and the need for the current study.

Species diversity appears to be an inconsistent indicator of landscape structure changes. Merriam and Wegner (1992), and Horak (1986) found increased diversity with fragmenation, while McIntyre (1995) noted the opposite response. Hutto et al. (1992) found that in the Rocky Mountain region, clearcuts negatively affect forest-dependent species while they positively affect open forest species; he also notes the inconsistency of traditional guild structures. Looking at 100-year trends, Hejl (1994) found no data linking human-induced changes to west-wide trends for any species in western coniferous forests. She hypothesizes, however, that those species associated with burns, old-growth or snags are less abundant today than 100 years ago as these naturally occurring forest features are less abundant today. Hansen and Urban (1992) stress that life history traits and the trajectory of landscape change may produce unique patterns of response in avian guilds geographically separated. In comparing similar bird guilds of eastern deciduous forests and the Pacific northwest, they noted greater sensitivity to fragmentation by the eastern guilds.

### 2.3.2 Migrant Species

Askins and Philbrick (1987) found long distance migrants are negatively correlated with suburban species. Using Breeding Bird Survey data, Flather and Sauer (1996) found that neotropical migrants were more sensitive as a group to landscape structure than temperate or permanent residents. McIntyre (1995) stresses the importance of assessing the impact of fragmentation (in the United States) on neotropical species, and Howe (1992) suggests neotropical species are sensitive indicators to environmental change that could be used in new approaches to ecological monitoring.

## 2.4 Response Groups

The North American Breeding Bird Survey (BBS) is a unique source of long-term, large-area data on avian population trends (Droege 1990, Hejl 1994). Guilds are a convenient mechanism for partitioning avian response to environmental change. Unfortunately, guild assignment tends to be subjective and inconsistent (Hutto et al. 1992), diffusing the value of guilds as an interpretative and predictive tool. Verner (1984) cautions against the use of guild indicators (one or a few species of a guild) and Szaro (1986) introduces the concept of response guilds, grouping species that respond similarly to habitat perturbations. Szaro (1986) contends response guilds (groups) are better suited to the prescriptive practices of conservation agencies, but suggests also that guilds may be most effective as pointers to the habitat factors important to management decisions.

## 2.5 Temporal and Spatial Association

Wiens (1989a) emphasizes the importance of scale in ecological research, noting scale is too often overlooked, he suggests ecological studies include a multi-scale dimension. Hunter (1987) points out that scale is dependent on the species of interest, while Wiens (1989), Flather and Sauer (1996), and Hansen and Urban (1992) caution that patterns observed at the local scale may not apply regionally. On the other hand, Haila et al. (1987), and vanDorp and Opdam (1987) found local bird community dynamics closely tied to regional dynamics. Opdam (1991) notes a serious paucity of long-term studies and Turner (1987) calls for broadening the spatial and temporal dimensions in landscape studies. Similarly, Freemark et al. (1992, 1995) suggested a spatially hierarchical framework for conservation and management. Wiens (1989) notes that ecologists and resource managers tend to make and test predictions on short-term studies. Hunsaker et al. (1994) suggest more research is needed to help find the appropriate units for linking pattern to an appropriate indicator for monitoring ecological change. Flather and Sauer (1996) suggest understanding the influence of landscape structure on spatial and temporal patterns is key to shaping prescriptive management practices aimed at preserving biological diversity.

Ecologists and natural resource managers are calling for more studies at the macro-level (Caicco et al. 1995; Flather and Sauer 1996, Flather et al. 1992, May 1993, Hirsch et al. 1979, Maurer 1985, Helle 1986, Hunter 1987, Knight 1987, Ricklefs 1987. Turner 1987 and 1990, Swanson 1990, Magnuson 1990). In addition, for avian studies, Noon et. al (1985) suggests research should include a temporal dimension that spans 2-3 average life spans of the target species.

Landscape-avian studies similar to this research have been either short-term and small-area (Ambuel and Temple 1983, Blake and Karr 1984, Haila et al. 1987, Kkroodsma 1982, McCoy and Mushinsky 1994, Strelke 1980, vanDorp and Opdam 1987); or, long-term and small-area (Askins and Philbrick 1987, Horak 1986, Noon et al. 1985, Reed et al 1996a, Reed et al 1996b, Ripple et al. 1991, Robbins 1980); or large-area and short-term (Flather and Sauer 1996, McGarigal and McComb 1995, Robinson et al. 1995, Robbins et al. 1989). Although Peterjohn and Sauer (1994) conducted regional trend analysis across the United States they did not address correlative factors.

Overall, the literature on time-space landscape dynamics was scant, inconsistent, and usually dealt in either one dimension or the other, making it difficult to establish an *a priori* expectation for the current study. Those studies that related to scale and local-regional dynamics appear more relevant to the current study, suggesting a pattern in temporal trend may not manifest a spatially similar pattern.

## 2.6 Summary

In summary, this research draws upon a relatively new body of literature that shifts the focus of study from species to landscape, from underlying process to the importance of pattern. The amount of research on avian response to landscape change is heavily weighted to eastern studies that cite a consistent response to landscape change by woodland, forest edge, and migrant species. The few studies that have taken place in the west suggest woodland and edge relationships may not apply in western landscapes and recent avian guild literature implies response guilds have greater management value than guilds based on behavior or habitat preferences. In addition, several regional studies have cautioned against transferring vertebrate responses across regions, further reinforcing the notion held by some ecologists that insular, continental landscapes may not behave in the same way as island systems and western bird species may not behave similarly to eastern species.

Therefore, this research attempts to characterize a biological response — bird trend — associated with landscape change. Further, the current study adds to a small body of research studies conducted at regional or macro-scales, and by blending remote sensing technology with ecology, it presents a reasonable approach for monitoring entire ecosystems as prescribed in the policies of natural resource management agencies.

## CHAPTER 3. MATERIAL AND METHODS

### 3.1 Study Area Description

The current study is focused on western landscapes primarily due to the information needs of the U.S. Fish and Wildlife Service (FWS) regional office in Portland, Oregon, which has administrative responsibility over the Pacific Northwest. Embroiled in the protection issues surrounding the northern spotted owl (*Strix occidentalis caurina*), the Portland office is responsible for monitoring the landscape changes in western Oregon, Washington, and northern California, about 24 million ha. The regional office partially supported the current study as it relates to ecosystem management and especially new approaches for monitoring landscape change over large areas.

Over the last half century, the landscape of the Pacific Northwest has been altered by logging practices that reduced the stands of mature and old-growth coniferous forests, key habitat for the northern spotted owl. The resulting fragmentation may jeopardize the long term survival of the owl which appears to have sharp critical survival thresholds that are influenced by fragmentation (Lamberson et al. 1991). Until recently, the FWS monitored the owl population by sending biologists into suitable habitat where they would "hoot" for owls. Over 5,500 owl pairs were mapped in the 3-state area using this technique. With many other endangered and threatened species that also need monitoring, it became clear to the FWS that neither the staff nor funds were available for such extensive on-the-ground census surveys. With millions of hectares to monitor and thousands of approved actions that alter the landscape, the FWS recognized the need to use remote sensing in combination with limited ground surveys to take a periodic look at the net change in landscape, then relate that change to a desirable or undesirable condition for certain species. The key to this approach is quantifying landscape change so that net change can be correlated with a biological response that infers either a favorable or unfavorable condition for a given species. Thus, the current study was influenced by the need for an applied approach to quantify landscape change and relate the change to a measurable biological response.

At least two comparable and temporally different surface cover data sets are needed in order to quantify landscape changes. The U.S. Geological Survey produced a continental surface cover data base in the 1970's generally referred to as the Land Use and Land Cover map series. The problem was finding a suitable, recent surface cover data base. It would have been most desirable to locate the current study in the Pacific Northwest, but a survey of available landscape data for this region did not turn up a recent vintage large-area data set that could be differenced against the 1970s surface cover data. While several, large-area remotely sensed surface cover data were available for western states, few had been sufficiently validated (Williams 1996) for use in research. So, further limiting the search for a recent vintage surface cover data set was that it must be comparable with the 1970s data and have sufficient validation. After a thorough Web search and contacting state and federal natural resource agencies throughout the western U.S., only the one data set, the Gap Analysis Program coverage for western Montana and Northern Idaho, was located that was regional, comparable, and had been adequately error-checked. Used in its entirety, this data set established the boundary of the current study, covering 26.2 million ha. in western Montana and northern Idaho (Figure 3.1).

The study area is dominated by Pacific maritime-influenced conifer forests (Peet 1988) that have undergone natural and human-caused changes over the past 100 years. The present landscape mosaic, shaped largely by fire (Hutto et al. 1992, Hutto 1995) is predominantly forests of mixed age classes (Hejl 1995).

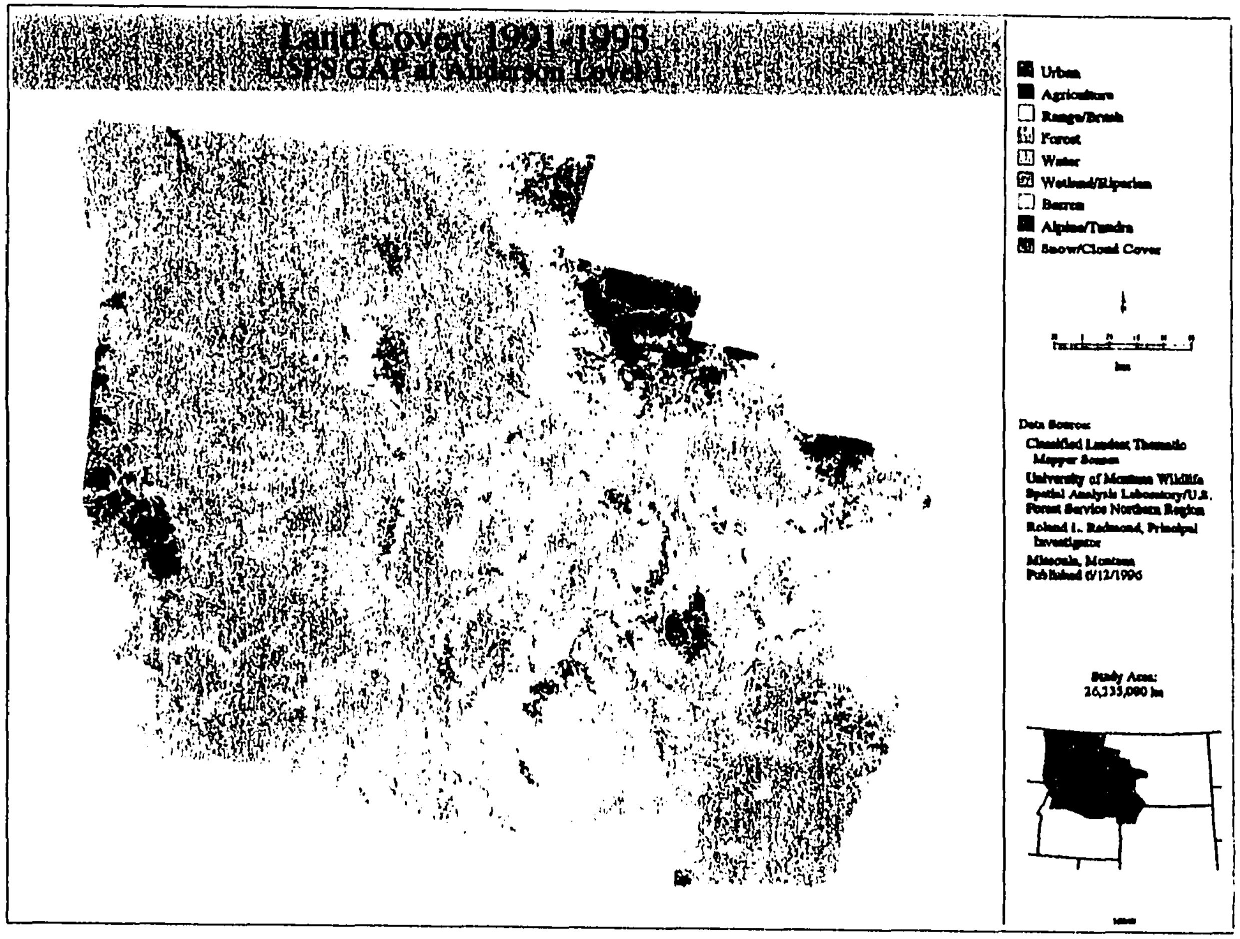


Figure 3.1: The Study Area as Delineated by the GAP Coverage for Northern Idaho and Western Montana (1990s)

### 3.2 Data Description

Three independent data sets were used to examine avian response to change in landscape structure: two geographic data sets from which landscape change could be measured and one biological data set from which a response could be measured. The surface cover data sets had to meet the following criteria: (i) available for two time periods, (ii) coverage at the macro-scale (multiple watersheds), and (iii) similar in terms of classification schemes and quality assessment. The biological response data set had to cover the same temporal and spatial scales. At the time of the current study, the three data sets chosen represented high quality, large-area data sets. Both surface cover data sets reported classification accuracies of 85% or better (Fitzpatrick-Lins 1980, Gopal and Woodcock 1994)). for the Level I classes as described by Anderson et al. (1976), a widely accepted reference for classifying remotely sensed surface cover.

Given the applied aspect of the current study for the FWS, the analysis approach followed a step wise sequence similar to what is presently used to monitor ecosystems. For analysis, some preprocessing was necessary to get the spatial data coverages into comparable formats, whereas the avian response data were already preprocessed. Table 3.1 shows the general flow of data and the sequence of steps followed for the spatial and temporal analysis.

Table 3.1: Data Flow and Steps in Spatial and Temporal Analysis

| Temporal Analysis (1970s-1990s) | Spatial Analysis (1990s) |
|---|---|
| Import Vector Files and Reconcile | Import Raster Files |
| Merge to 9 Classes and Grid GAP & LULC Data to 400 x 400 m, 16 ha cells | Merge to 9 Classes, Grid to 16 ha cells |
| Run Fragstats, Measure change | Run Fragstats |
| Import BBS Trend Data | Import BBS Abundance Data |
| Run Multiple Regressions | Run Multiple Regressions |
| Select Best Model | Select Best Model |

### 3.2.1 Landscape Structure Data

Figure 3.2 shows the 1970s, USGS Land Use and Land Cover (LULC) surface cover data produced from-high altitude aerial photography at scales of 1:60,000 or smaller for the entire study area (USDI, Geological Survey 1987). Coverages are referenced to 1:250,000 or 1:100,000 base maps that have been merged into a "seamless" coverage for the entire United States. Geographic coverages are readily available via a delivery program (Geographic Information Retrieval and Analysis System, GIRAS) supported by the USGS (Mitchel et al. 1977). This data set has been widely used in national and regional studies (Mitchell et al. 1977). For the current study, only the nine broadest classes of land use/land cover (urban or built-up land, agricultural land, rangeland, forest land, water, wetland, barren land, tundra, or perennial snowfields) were used. These are Level I classes as defined by Anderson et al. (1976). Using a point sampling technique and binomial probability theory, 1:250,000 LULC quad maps were randomly sampled and found to meet or exceed accuracies of 85 %. Detailed methods for producing the accuracy assessments for the LULC data are described by Fitzpatrick-Lins (1980).

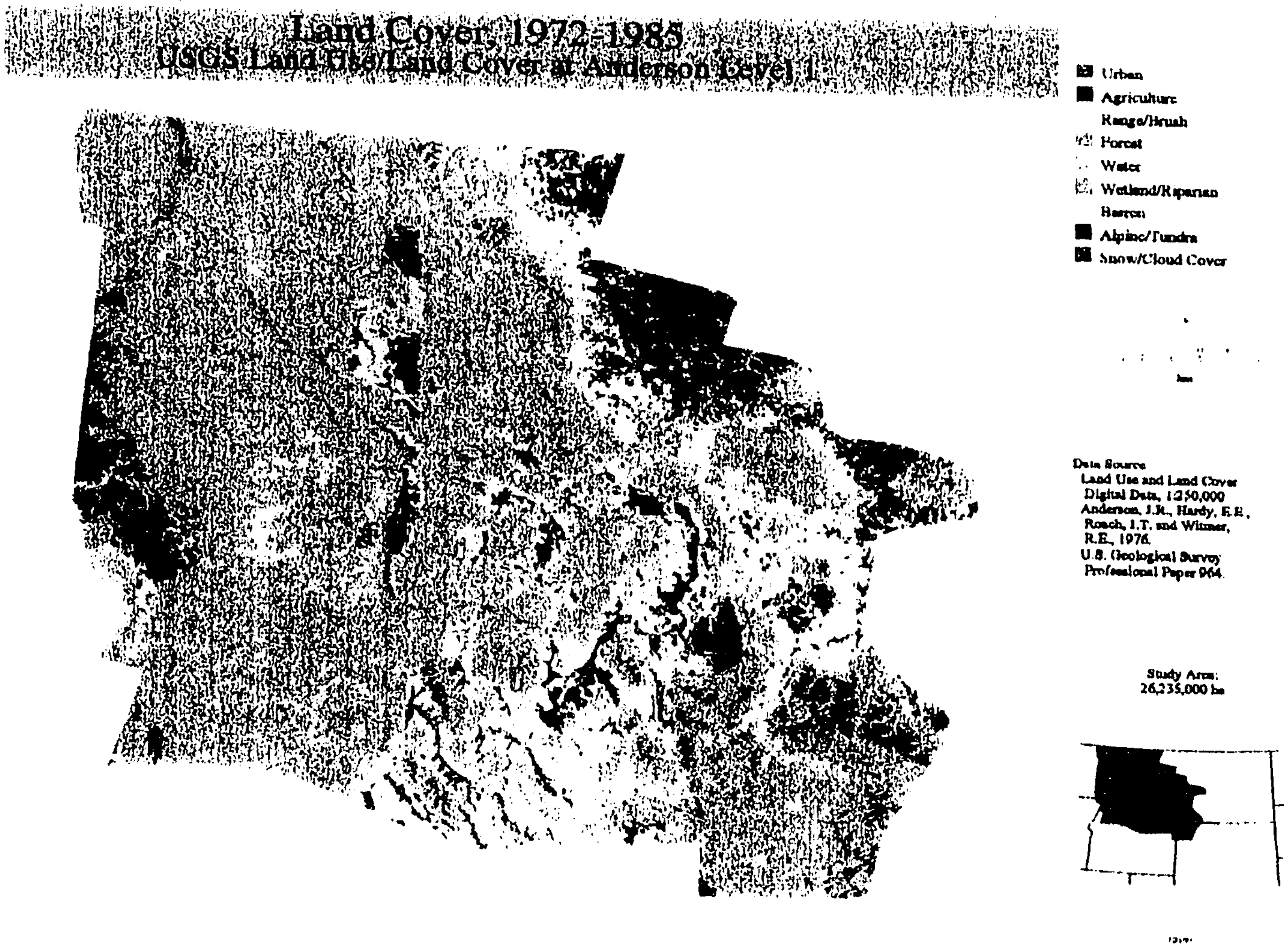


Figure 3.2: Study Area - Northern Idaho, Western Montana, 1970s

The 1990s landscape structure data was derived from a U.S. Forest Service data set that covers 26.2 million ha. in western Montana and northern Idaho (Redmond 1996) (Figure 3.1). These data are also part of the USGS, Gap Analysis Program (GAP) (Caicco et al. 1995). Surface cover classes were derived from 18 Landsat Thematic Mapper scenes in a two-step classification process. Derived from color composites of bands 4,5, and 3, an unsupervised classification performed on all 30 m x 30 m pixels merged into 2 ha. polygons then assigned to ~70 spectral cluster classes. The unsupervised classification was followed by a supervised classification, using 17,854 ground-truth plots scattered throughout the study area to assign cover classes. Of the total ground truth plots, 80% were used as training statistics for assigning cover classes, and 20% held aside for accuracy assessment (overlapping scenes made the number of useful ground-truth plots much higher). The final GIS database contained 58 cover sub-classes, generally following the Anderson et al. (1976) classification scheme, including the nine, broad classes used in the LULC data. The ground-truth plots were subjected to a series of logical and positional checks used to calculate class accuracies which were 85% or better for the nine broadest classes. Accuracy assessment methods used to produce the Idaho-Montana GAP coverages are fully described in Gopal and Woodcock (1994).

### 3.2.2 Avian Response Data

As a response variable to landscape structure change, the North American Breeding Bird Survey (BBS) (Droege 1990) data were selected for the area that matched the spatial and temporal dimensions of the surface cover data sets. The BBS database is administered and maintained by the U.S. Geological Survey and Canadian Wildlife Service, who have been coordinating surveys on avian relative abundance for the U.S. and Canada since 1968 (Droege 1990). BBS monitors over 200 species of North American birds, conducting annual surveys during peak breeding season, usually in June, along more than 3,400 roadside routes. The routes are randomly selected and once established they remain constant for subsequent surveys. Each route, 39.4 km. long, is broken into fifty 0.8 km. intervals where competent observers record seen and heard birds within 0.4 km. for 3 minutes at each stop. Within the study area there were a total of 60 BBS routes as shown in Figure 3.3, of which only 45 had consecutive years of data that matched the time period of the current study.

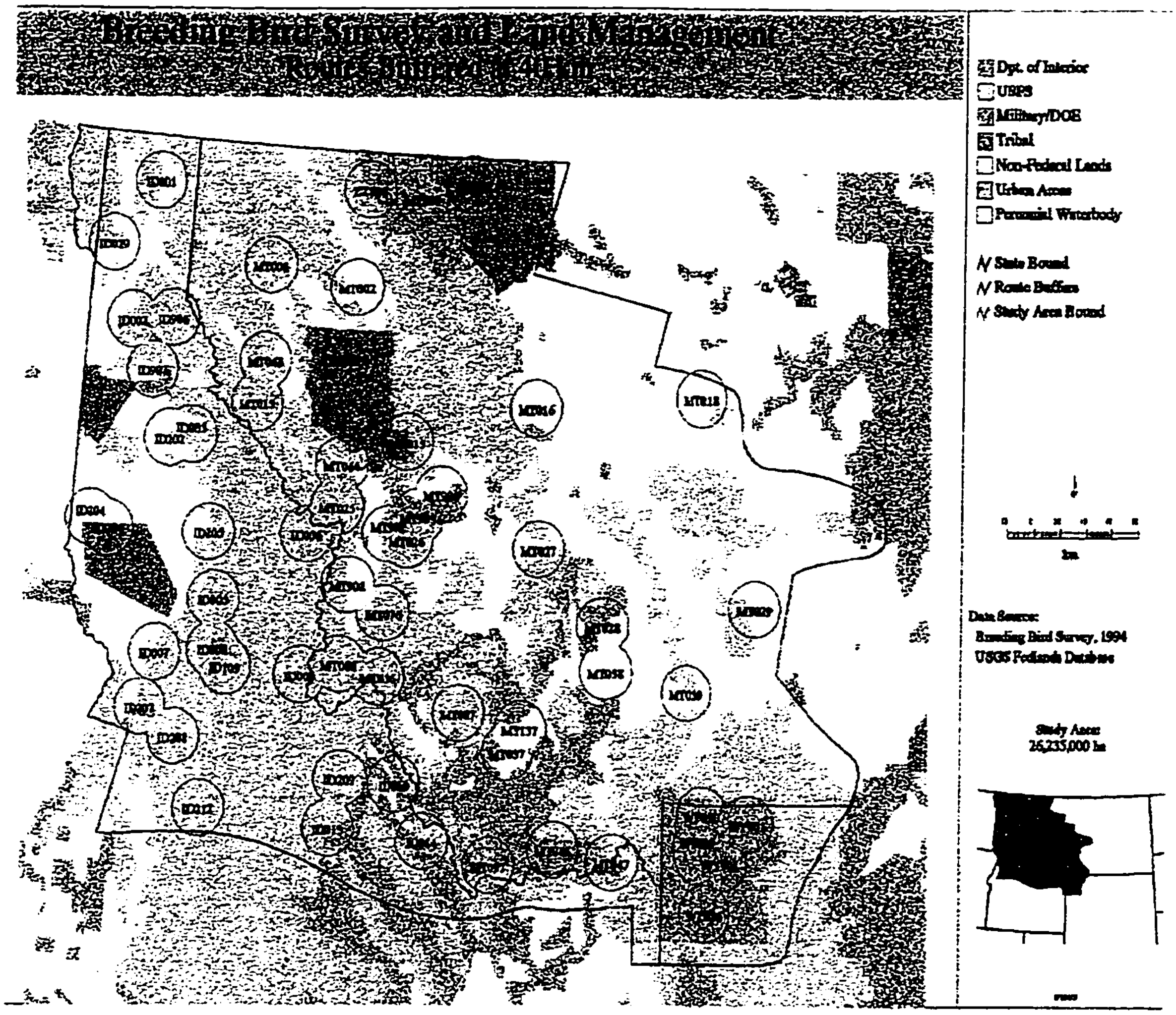


Figure 3.3: BBS Routes in the Study Area (Route IDs Correspond to Fragstats Outputs, Table 3.2)

The underlying premise for using BBS data as a response variable is that certain birds might be sensitive to landscape change which would manifest as a change in population trend. To match the ~ 20 year span on landscape data a ~ 20 year population trend was estimated for each species on each route. Twenty-year trends were computed as the weighted average (weighted to adjust for sampling inconsistency) of annual trend estimates. For the current study's spatial analysis, an estimate of abundance was computed as the 5-year mean bird count adjusted for observer bias. Though BBS is used widely for avian studies, Sauer et al. (1994) notes certain biases exists in the BBS sampling procedure. Sauer and Geissler (1990) provide detailed information on the methods used to calculate BBS abundance and trend, including the procedures used to minimize observer biases and route sampling inconsistencies. For consistency and a simple means of grouping birds, BBS breeding habitat guilds (Peterjohn and Sauer 1994) were used to subset birds into three general groups: 1) woodland, 2) successional scrub (edge), and 3) other (Appendix I). Birds that are sampled poorly by BBS and birds unrelated to the study such as waterfowl, shore birds, and tundra birds were not used.

### 3.3 Reconciling and Comparing Spatial Data Sets

The temporal-spatial dimensions of this research calls for comparing landscape change across time and space. The first step in a comparative study of landscape structure change is to prepare or normalize the two spatial coverages from different time periods so that they cover the same area and have comparable attributes. These coverages are important as they are the source from which metrics of change are derived for independent variables. Ideally, the thematic GIS data sets under consideration would be identical in all aspects except time and net change, having the same sensor type, projection, classification scheme, processing method, scale, and accuracy. If these parameters are identical, the geo-processing steps are straightforward: (i) GIS overlays to cut out study area, (ii) spatial measurements of classes for each time period, and (iii) calculate metrics of change for regression analysis. Some variability between the coverages is acceptable and would not necessarily affect analysis results. For example, software programs can re-project coverages without altering accuracy, and sensor types can differ so long as classification accuracies are acceptable. Irreconcilable differences must be weighed against the effect on the analysis results from which statistical inference is drawn.

The input data sets used for the current study were selected for their availability, documented accuracy assessments, and assumed similarity in methods used to delineate cover classes. Even though different sensors were used, both data sets reported acceptable classification accuracies of 85% at Anderson Level I (accuracies dropped appreciably at Anderson Level II).

An important side note, which emphasizes the importance of metadata in comparative studies, is the lack of source information about the LULC data. An attempt was made to find the original LULC photography. Even though LULC was the largest mapping project ever undertaken by USGS, the source photography could not be located. Some degree of mystery still surrounds the photography as some portion of the images were "classified" for national security purposes. The project took place in the mid-seventies, the height of the cold war era, at a separate facility in Reston, Virginia. Personal contacts assured that the accuracy of the Forest class should be very high as Forest was one of the easiest classes to detect from the photography. USGS staff at Reston and Sioux Falls noted that LULC is USGS's only nation-wide coverage, and as such, its value will increase over time as a historic comparison data set. Staff at USGS intend to follow-up the search for original photography so that LULC can become a "legacy" data set such that USGS can provide single products (e.g. a map, a GIS file, a hard copy map), or the complete set of products, including original photography and metadata. However, at this time, extant literature on the use and quality of the LULC was convincing enough to proceed with the current study.

Both data sets were available in ArcInfo format, a geographic information system commonly used by natural resource management agencies. With the GAP data set setting the geographic extent of the study area, its boundary was used to overlay and extract the LULC data. Once both data sets were in ArcInfo, the next step was to compare themaic map attributes, checking class and sub-classes for consistency. The initial review of both data sets gave the impression the same basic classification scheme was used. On closer examination of the actual data, metadata, historic references, and personal communication, the data sets were found to have a number of inconsistencies that were not readily apparent from primary source materials (expanded on in Chapter 4). Further comparison called for merging sub-classes to the class level (Anderson Level I). Again using the functional programs of ArcInfo, 59 classes were merged to 9 for the GAP data and 37 classes were merged to 9 for the LULC data. For both data sets , thematic maps were prepared for visual inspection and area tables were summarized and put into tables. From the tables, differences for each class were calculated for review prior to further analysis. From this point, Chapter 4 details the processes, comparisons, modifications made to the data sets.

### 3.4 Fragmentation Metrics

To relate landscape change and avian population trend it is necessary to quantify landscape changes between two time periods for the same area in which the avian trend data were collected. Drawing upon the earlier work of Flather and Sauer (1996), an arbitrary zone of influence around each route was set at 40 km., the maximum length of a route. Using GIS, a 19.7 km radius circle (about 1200 $km^2$) was created around the centroid of each route. A circular zone was used for ease of analysis since the actual routes have never been digitized. This extent, ~1,200 $km^2$, was considered biologically generous for most birds, covering an area large enough to encompass the entire 40 km sampling route.

Using the circular scenes as a template, land cover data, already aggregated to Anderson Level I, was extracted from both surface cover data sets (LULC and GAP). For each circle (1970s and 1990s) landscape structure metrics were calculated using Fragstats, a spatial pattern analysis program for quantifying landscape structure (McGarigal and Marks 1995). Fragstats was used for several reasons: (i) ease of use, (ii) compatibility with commonly used GIS programs, and (iii) wide use by land management agencies, especially the U.S. Forest Service. Fragstats is also very comprehensive, producing 59 metrics of landscape structure, many of which are redundant measures (McGarigal and McComb 1995) of the same information.

To reduce the dimensionality of analysis, the selection of independent variables followed a process of eliminating redundant measures, strong covariates, and variables not directly tied to the ecological phenomena underlying the research objectives. In a similar exercise, Ritters et al. (1995) found that 6 landscape metrics out of 55 explained 87% of the variation in landscape change. Keeping with the confirmatory approach of this research, the simplest model of predictor variables was desired. Correlation matrices were examined for the candidate set of variables and they were further reduced by eliminating variables with high collinearity, using SAS calculations of the condition index (computed from the Eigenvalues of the covariance matrix of the independent variables). From 59, the number of independent variables was reduced to five by omitting redundant measures, highly correlated measures, and those measures of lesser ecological bearing on the objectives of the current study. These variables encompass the key, ecologically sensible landscape changes that are most likely to explain the variation in avian population trend. Several extra measures of edge were added to the spatial analysis in order to associate more edge characteristics with avian abundance (Appendix II).

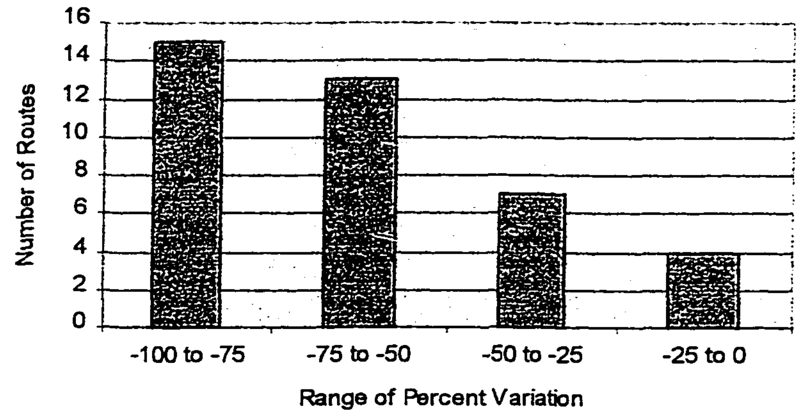
Table 3.2 shows the Fragstats measures that were used in the temporal and spatial regression analysis. As seen in the table, the BBS routes used in the current study represented a full range of forest conditions within the route circles, from almost completely forested to only a small amount of forest. Table 3.3 shows the general condition of the forest in the study area progressed to a more patchy environment. Zero values in Table 3.2 indicate a metric could not be calculated by SAS. For example, in route id212 a value for MFPSCV in the 1970s could not be calculated as the route was characterized by one, large polygon of forest and subsequently also had no value for MNND. The Fragstats outputs for each route correspond to the route circles shown in Figure 3.3.

## Table 3.2: List of Landscape Variables and Fragstats Output Used in Regression Analysis

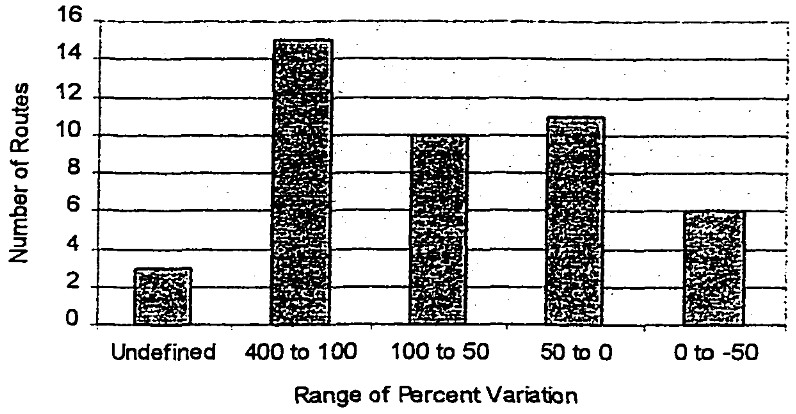
| Route | Mean Forest Patch Size (1970s) $KM^2$ | Mean Forest Patch Size (1990s) $KM^2$ | Mean Forest Patch Size CV (1970s) | Mean Forest Patch Size CV (1990s) | Forest Edge (1970s) meters | Forest Edge (1990s) meters | Percent Forest (1970s) | Percent Forest (1990s) | Mean Nearest Neighbor Distance (1970s) meters | Mean Nearest Neighbor Distance (1990s) meters |
|---|---|---|---|---|---|---|---|---|---|---|
| id001 | 64.94 | 29.09 | 3.11 | 5.15 | 584,423 | 1,344,800 | 83 | 69 | 655 | 496 |
| id002 | 118.66 | 70.97 | 2.81 | 3.59 | 451,135 | 922,000 | 85 | 79 | 447 | 540 |
| id003 | 293.47 | 364.37 | 1.73 | 1.41 | 359,615 | 1,057,600 | 93 | 87 | 400 | 400 |
| id004 | 8.72 | 1.34 | 1.72 | 3.78 | 501,621 | 1,248,000 | 18 | 19 | 919 | 5822 |
| id006 | 125.96 | 153.18 | 2.78 | 2.44 | 375,607 | 1,056,800 | 90 | 85 | 480 | 400 |
| id007 | 242.38 | 24.00 | 1.72 | 6.02 | 236,157 | 1,338,000 | 77 | 74 | 1200 | 465 |
| id008 | 1,219.55 | 1,169.76 | 0 | 0 | 166,027 | 604,800 | 97 | 93 | 0 | 0 |
| id010 | 20.38 | 7.51 | 3.04 | 6.34 | 620,041 | 1,367,600 | 37 | 48 | 1132 | 562 |
| id013 | 17.75 | 9.59 | 5.59 | 7.57 | 970,130 | 1,166,800 | 51 | 49 | 803 | 675 |
| id014 | 11.69 | 3.67 | 2.25 | 4.55 | 536,683 | 1,227,600 | 22 | 32 | 555 | 513 |
| id029 | 61.37 | 55.12 | 3.68 | 3.52 | 583,652 | 1,584,800 | 88 | 75 | 450 | 400 |
| id204 | 4.58 | 0.71 | 0.95 | 3.14 | 261,692 | 744,000 | 8 | 10 | 1550 | 644 |
| id205 | 1,207.98 | 84.89 | 0 | 3.30 | 192,868 | 1,319,600 | 96 | 81 | 0 | 400 |
| id207 | 262.61 | 47.60 | 1.61 | 4.21 | 187,820 | 1,367,600 | 84 | 72 | 2674 | 489 |
| id208 | 232.84 | 119.92 | 1.99 | 2.82 | 348,074 | 970,800 | 93 | 86 | 933 | 400 |
| id209 | 52.57 | 43.07 | 4.19 | 4.65 | 806,890 | 1,319,600 | 84 | 79 | 520 | 443 |
| id212 | 1,163.72 | 356.32 | 0 | 1.41 | 197,480 | 830,800 | 96 | 88 | 0 | 400 |
| id214 | 16.78 | 12.65 | 2.87 | 4.57 | 289,008 | 502,800 | 25 | 39 | 615 | 906 |
| mt001 | 398.83 | 98.84 | 1.38 | 2.99 | 261,470 | 1,385,200 | 95 | 79 | 400 | 400 |
| mt002 | 8.56 | 5.80 | 6.98 | 8.83 | 657,942 | 1,110,000 | 52 | 52 | 706 | 502 |
| mt003 | 8.65 | 3.39 | 3.12 | 8.35 | 303,310 | 1,217,200 | 13 | 33 | 1525 | 537 |
| mt013 | 399.99 | 282.88 | 1.41 | 1.73 | 366,035 | 845,200 | 95 | 90 | 400 | 400 |
| mt014 | 6.84 | 2.72 | 3.49 | 4.50 | 569,002 | 712,000 | 22 | 20 | 679 | 803 |
| mt015 | 138.33 | 60.00 | 2.61 | 3.97 | 384,437 | 1,047,200 | 88 | 81 | 597 | 610 |
| mt016 | 16.94 | 1.06 | 2.42 | 4.48 | 335,178 | 884,000 | 14 | 14 | 861 | 684 |
| mt018 | 5.97 | 1.46 | 4.01 | 7.94 | 328,362 | 1,077,200 | 11 | 21 | 857 | 576 |
| mt025 | 194.38 | 269.00 | 2.20 | 1.73 | 421,576 | 1,047,600 | 93 | 86 | 400 | 400 |
| mt026 | 583.95 | 101.38 | 0.93 | 3.15 | 213,877 | 809,600 | 93 | 89 | 0 | 436 |
| mt027 | 63.15 | 10.30 | 3.29 | 8.15 | 625,145 | 1,485,200 | 65 | 61 | 776 | 490 |
| mt028 | 7.02 | 1.46 | 2.47 | 5.06 | 145,712 | 782,800 | 6 | 13 | 2831 | 603 |
| mt029 | 33.48 | 2.61 | 2.65 | 8.96 | 280,078 | 1,051,600 | 24 | 27 | 1554 | 611 |
| mt036 | 112.67 | 151.52 | 2.33 | 2.44 | 571,077 | 1,130,000 | 90 | 84 | 400 | 423 |
| mt038 | 3.03 | 0.60 | 0.88 | 1.67 | 65,818 | 592,000 | 1 | 7 | 6201 | 704 |
| mt039 | 62.70 | 6.57 | 1.71 | 8.51 | 586,099 | 1,586,400 | 45 | 46 | 400 | 543 |
| mt046 | 4.33 | 1.68 | 3.17 | 7.00 | 592,799 | 1,114,000 | 19 | 23 | 754 | 640 |
| mt047 | 11.17 | 4.98 | 4.81 | 6.75 | 1,013,477 | 1,559,200 | 46 | 43 | 631 | 515 |
| mt076 | 185.44 | 48.54 | 2.22 | 4.66 | 322,197 | 623,200 | 89 | 89 | 593 | 484 |
| mt086 | 77.49 | 51.24 | 3.56 | 4.04 | 533,857 | 1,568,000 | 86 | 73 | 400 | 449 |
| mt137 | 3.92 | 0.76 | 1.51 | 2.83 | 178,169 | 556,400 | 4 | 8 | 1107 | 698 |
| mt900 | 22.71 | 13.44 | 2.52 | 3.98 | 731,148 | 1,272,800 | 38 | 54 | 1063 | 472 |
| wa032 | 34.09 | 122.88 | 4.46 | 1.99 | 126,597 | 797,200 | 95 | 82 | 0 | 400 |
| wy001 | 26.12 | 43.89 | 5.61 | 4.65 | 1,307,220 | 1,162,000 | 71 | 80 | 449 | 421 |
| wy030 | 36.38 | 10.65 | 4.57 | 6.07 | 1,071,046 | 1,606,400 | 66 | 57 | 444 | 474 |
| wy032 | 50.09 | 15.87 | 3.62 | 6.54 | 838,692 | 1,050,000 | 70 | 71 | 418 | 449 |
| wy901 | 88.92 | 37.35 | 3.08 | 4.85 | 596,493 | 825,200 | 78 | 74 | 677 | 577 |

Table 3.3: Frequency Distribution of Fragstats Output

Mean Forest Patch Size



Mean Forest Patch Size CV



Change in Forest Edge

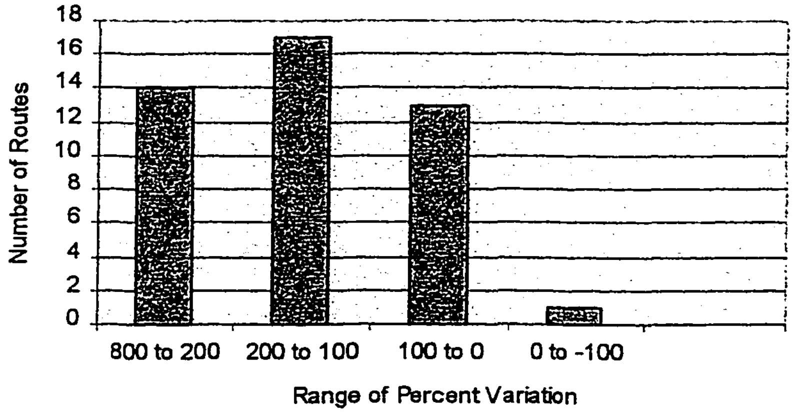
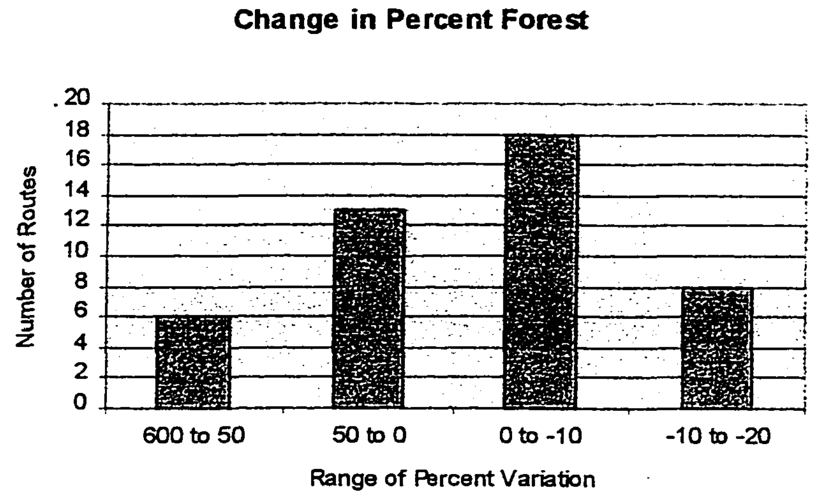
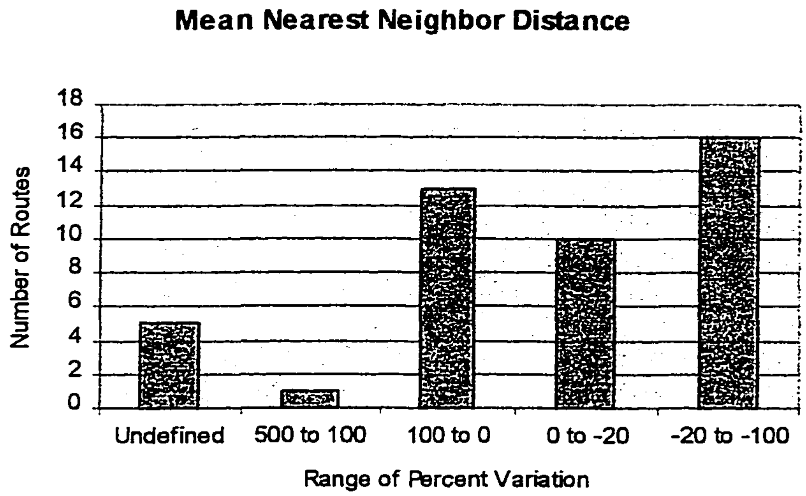


Table 3.3: (continued)


Change in Percent Forest


Mean Nearest Neighbor Distance

Percent Forest measures how much of the route circle is occupied by forest. Though not spatially explicit, the amount of suitable habitat is perhaps the single most important piece of information about the landscape, strongly affecting avian occurrence and abundance (McGarigal and Marks 1995). Over time, it is useful to know how much suitable habit is lost, or conversely how much suitable habitat remains, especially since some species like the northern spotted owl have minimum area requirements.

Mean Forest Patch Size (MFPS) is another key measure of landscape structure, which on a temporal scale, quantifies the progressive change in patch size caused by fragmentation. When the area of observation is constant, a landscape with a smaller MFPS indicates a more fragmented landscape. MFPS, representing the average condition of patch size, is often used as a habitat fragmentation index.

The total amount of edge in a landscape in an important interpretive measure of ecological phenomena (McGarigal and Marks 1995). Unlike the precepts of early wildlife management that viewed edge as an overall good condition for wildlife, edge effects are very species specific. Some species like edge, other species are adversely affected by edge and still others are unaffected by edge. Edge can also provide useful information about habitat suitability. For example, two patches may have the same total area but very different edge lengths, indicating the patch with the longer length is more convoluted with a smaller core area, preferred habitat for an edge species but not so for an woodland species. For the current study, Forest Edge measures the total length of forest edge in the route circle.

Mean Forest Patch Size Coefficient of Variation (MFPSCV) measures the variability in patch size, which is a key indicator of landscape heterogeneity (McGarigal and Marks 1995) that is not captured in MFPS. MFPSCV measures relative variability, that is, variability as a percentage of the mean. This measure is helpful in relating avian trend to variation in patch size. For example, some species prefer a highly varied landscape, patches of differing size, while other species prefer a more uniform landscape of similar sized patches. For the spatial analysis, edge adjacency was also measured for forest next to agriculture, range, urban, and other for extemporaneous analysis.

Mean Nearest Neighbor Distance (MNND) measures the edge-to-edge distance from a patch to the nearest neighboring patch of the same type. An important aspect of fragmentation effects on avian population dynamics is the probability of patch occupancy which is closely tied to patch size and isolation (MacArthur and Wilson 1967). Isolation infers the degree to which a patch is separated from another patch of similar habitat and explained why fragmented habitats often contain fewer birds (Forman 1987). Interpatch distance in avian metapopulations can play a critical role in bird movement, feeding, predation, and breeding behavior.

### 3.5 Data Analysis and Model Selection

The metrics of structure change and avian population trend and abundance were used to run multiple regression analysis for each bird using SAS. An appropriate model (one or more of the independent variables) for each bird species was selected using Akaiki's Information Criteria (AIC) method (Akaike 1969). AIC was preferred over other model selection methods because of its optimal properties and emphasis on parsimony (Burnham and Anderson 1998). AIC selects independent variables for a model based on an information-theoretic optimal tradeoff between reduced bias caused by adding more variables and reduced estimation error caused by including fewer variables. SAS outputs include indices of multicollinearity with suggested thresholds (SAS Institute Inc. 1991). Careful reduction of the independent variables and AIC's method of model selection resulted in no important multicollinearity in the temporal analysis and only a few in the spatial analysis. The adequacy of each model selected by AIC was tested using an F-test of overall model significance and by examining correlation coefficients. Influential points that exceeded thresholds recommended in SAS were flagged but none were removed from the analysis.

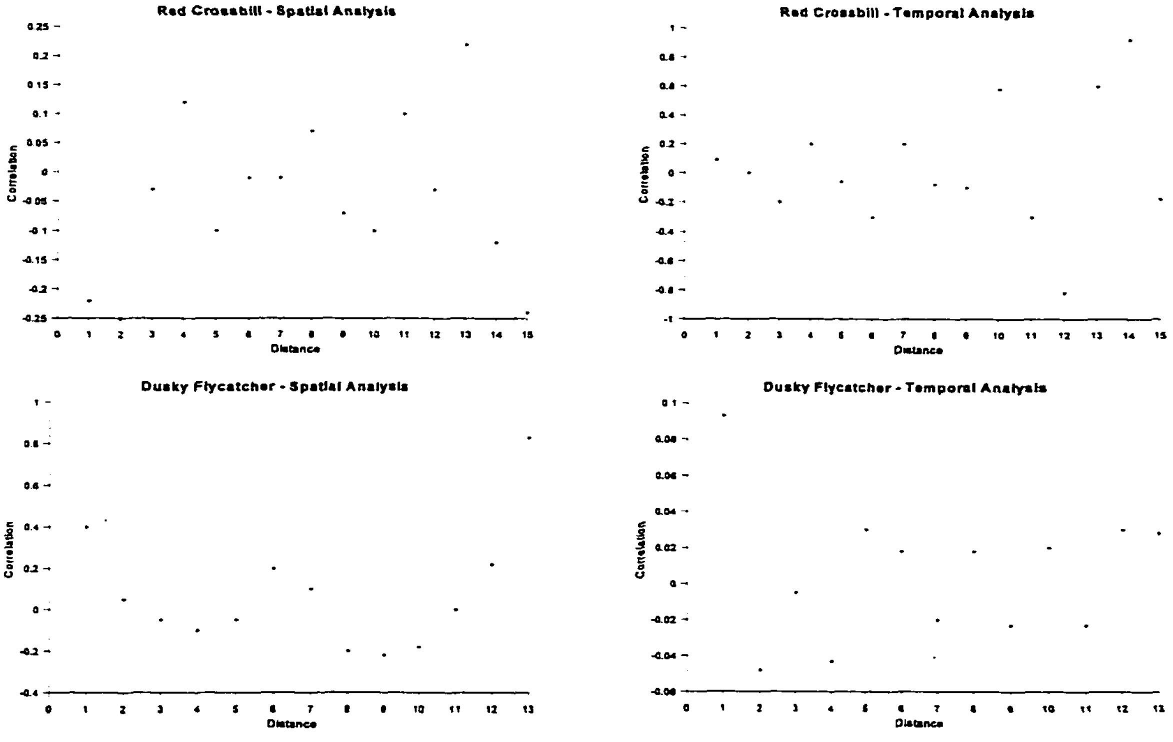
Histograms of the raw Fragstats data and BBS data revealed some degree of skewness in both the dependent and independent data sets, and correlation residuals for many birds showed non-normal distributions based on a Shapiro-Wilk test of normality (SAS Institute Inc. 1991). Transformations of the dependent and independent variables

(Appendix III) improved normality and reduced the number of high influential points without inducing any noticeable non-linearity in the residual plots of the dependent variables.

As noted in the Chapter 2, large-area studies make it virtually impossible to set up an experimental design that controls for all confounding variables. Autocorrelation is a general statistical property that is observed in ecological variables across geographic space (Legendre 1993) which may have a weak or strong influence on correlative analysis. Autocorrelated data violate the statistical assumption of independence, thus increasing the likelihood of misinterpreting statistical results. Generally, two approaches are taken with regard to autocorrelation in spatial data: (i) demonstrate that no significant spatial structure exits or if it does exist, statistically control for it, or (ii) demonstrate a spatial structure exists and include it in the statistical models (Legendre 1993). Given the objective of this research the first option was more appropriate. In addition, there has been no reported autocorrelation in the BBS data.

As a check for the possibility of autocorrelation, residual errors for each model were compared by computing a spatial correlogram (Cliff and Ord 1981). A correlogram graphs correlation values against distance. A test of significance is given for each autocorrelation coefficient. In this case, the correlation coefficients for all pairs of model residual errors within 50 km bands of separation were used. Correlations within each band were tested for significance and the correlograms were plotted and visually inspected for signs of a spatial structure. Figure 3.4 shows example correlograms for the dusky flycatcher and red crossbill. The absence of a pattern in the plots indicate no spatial pattern exists.

Figure 3.4: Example Correlograms for the Dusky Flycatcher and Red Crossbill (absence of pattern in the scatter plots indicates that no spatial structure exits; Distance in 50 km. increments)



### 3.6 Determining Response Groups

Three *a priori* bird groups were identified in the Introduction: woodland, edge, and migrants. The Introduction and objectives also established an anticipated response for each group based on the response of eastern birds or literature reports. An assumption of this research is that regression models will display trends for certain birds and bird groups that confirm anticipated responses of the *a priori* groups and also that other groups may emerge that respond similarly. With multivariate models it is not always easy to separate multidimensional data into meaningful groups. Computers are helpful in their ability to use multivariate methods like ordination and clustering to reduce main trends of variation of multidimensional data into a few, linearly independent sets (Borcard et al. 1992). Two methods were used to determine response groups, (i) visual interpretation (simply looking at the data), and (ii) cluster analysis (using the power of computers to ordinate results).

### 3.7 Temporal and Spatial Comparison

There were no *a priori* assumptions about the relationship between temporal trend and spatial abundance. The principle reason for including a spatial dimension to this research was to see if any birds or bird groups exhibit similar patterns temporally and spatially. As noted in Chapter 2, the literature is scant in this area, with few studies looking at large spatial extent and long time periods. There is however, an interesting temporal-spatial dynamic covered in the literature that lends importance to the comparison presented here. Birds tend to respond differently at different scales of observation; they may become locally extinct at the patch level while persisting at the regional level (Hansen and Urban 1992; Wiens 1989). This suggests that a pattern of temporal trend may not necessarily mean a similar pattern will be observed in species abundance. This would be especially true if the geographic extent is different.

The regression tables for temporal trend were compared to the tables for spatial abundance and evaluated for similarity in model sets and pattern of response to the independent variables. For example, did the same bird or bird groups respond in the same way (positive, negative, or no correlation) to MFPS in the spatial and temporal dimensions. For birds or bird groups with similar temporal and spatial patterns, their BBS guild assignment and migratory status were also compared.

### 3.8 Hardware and Software

Data analysis used hardware and software that closely matched that of a typical natural resource management office with large-area responsibility, such as a U.S. Forest Service supervisor's office or a U.S. Fish and Wildlife Service regional office: a Sun (UNIX) workstation running Solaris 2.5; ESRI Arc/Info v.7.04 and v.7.1.1; Oracle v. 8.0.4.

Fragmentation analysis was conducted with Fragstats 2.0 and statistical analysis used SAS version 6.12 for a personal computer.

# CHAPTER 4. RESULTS AND DISCUSSION

## 4.1 Overview of Results

Results are presented in the same basic sequence in which the analysis occurred. The cornerstone of the current study was the comparability of two, large spatial databases. If these databases were not comparable, further analysis would have been futile. Therefore, the first sections of this Chapter deal with the preprocessing, quality assessment, and reconciling of the spatial data sets. Given temporal change detection was possible, the next step was to produce statistical results from which meaningful ecological inference might be made. The search for statistical patterns used several methods. However, the most revealing was found in visually interpreting the regression tables for each independent variable separately, even though most birds contained several variables in its AIC model. Next, bird groups are discussed, and though the primary focus of the current study was concerned with temporal change and avian trend, the spatial analysis results are presented as it relates to the temporal results.

## 4.2 Comparing Spatial Data Sets

### 4.2.1 Inconsistencies

On close examination, the GAP classification scheme reflected a melding of Anderson classes with GAP programmatic classes, showing the emphasis of the principal client, the U.S. Forest Service. In preparing the GAP data, the USFS wanted the greater accuracy for the Forest and Rangeland (Grassland) components, creating more Forest subclasses and concentrating accuracy assessment around these themes. As a result, most training sites and ground-truth occurred in Forest and Rangeland sites. The Urban class was not ground-truthed, so there would be no way of verifying the accuracy for this class. Thus, the accuracy of the Forest and Rangeland classes are more reliable than the other classes (Redmond 1996). Table 4.1 shows much more attention was given, in terms of sub-class divisions, to Forest and Rangeland in the GAP data versus the LULC data. GAP separated Forest into 25 sub-classes and Rangeland into 15 subclasses versus only three subclasses for each in the LULC data.

Table 4.1: Comparison of GAP and LULC Classification Schemes

| GAP/LULC Codes | Level I Class Names GAP vs LULC | No. Of Sub-Classes - GAP | Level II No. Of Sub-Classes - LULC |
|---|---|---|---|
| 1000/1 | Urban/Urban or Built-up Land | 1 | 7 |
| 2000/2 | Agriculture/Agriculture Land | 1 | 4 |
| 3xxx/3 | Grassland.../Rangeland | 15 | 3 |
| 4xxx/4 | Forest Species/Forest Land | 25 | 3 |
| 5000/5 | Water/Water | 1 | 4 |
| 6xxx/6 | Riparian/Wetland | 7 | 2 |
| 7xxx/7 | Barren, Exposed Rock.../Barren Land | 5 | 7 |
| 8100/8 | Alpine Meadow/Tundra | 1 | 5 |
| 9xxx/9 | Snow, Cloud, Cloud Shadow/Perennial Snow or Ice | 3 | 2 |
| Total | | 59 | 37 |

Further indication that the data sets were not wholly comparable surfaced with area calculations of cover classes for the two time periods (Table 4.2). Side by side comparisons revealed unrealistic changes. For example, it is highly unlikely that in the entire study area urban area decreased by half in the ~20 years separating the two data sets. Nor is it likely that Agriculture decreased by more than 1 million hectares and barren land increased more than three-fold. In selecting the GAP data set for this research, it was assumed the same decision rules were used for defining and delineating cover classes as were used in LULC (Anderson et al. 1976). This was not the case.

In producing the GAP data set, different decision rules were used to define classes. An attempt was made to adjust the Urban, Agriculture and Tundra sets based on the assumption that the LULU was more accurate. As stated earlier, these classes were given little or no accuracy treatment in the GAP data, whereas, the LULC reported 85% or better accuracy. For comparison purposes the GAP data were changed as follows and shown as the adjusted figures in Table 4.2:

- Barren in GAP that was Urban in LULC was changed to Urban
- Barren in GAP that was Agriculture in LULC was changed to Agriculture
- Barren in GAP that was Tundra in LULC was changed to Alpine/Tundra

Table 4.2: Area Comparison of Spatial Data Coverages, Including GAP Adjusted Using Ancillary Data

| Anderson Level I Class | Gap Coverage (1990s) Ha. Original/Adjusted | LULC Coverage (1970s) Ha. |
|---|---|---|
| Urban | 51,000/54,000 | 108,000 |
| Agriculture | 1,785,000/1,812,000 | 2,745,000 |
| Range/Brush | 7,608,000 | 6,890,000 |
| Forest | 14,858,000 | 15,184,000 |
| Water | 322,000 | 306,000 |
| Wetland/Riparian | 281,000 | 124,000 |
| Barren | 1,163,000/843,000 | 269,000 |
| Alpine/Tundra | 75,000/280,000 | 597,000 |
| Snow/Cloud Cover | 174,000 | 9,000 |
| Total | 26,232,000 | 26,232,000 |

Overlaying the LULC Urban class with the GAP coverage, areas classed as Urban in LULC but not classed as Urban in Gap were spread among Range, Forest, Agriculture, Wetland, and Water (Table 4.3). Even with some heuristic changes and cross-checking with other ancillary data such as the U.S. Census Bureau Tiger files, fully reconciling the two data sets with an acceptable degree of accuracy would have been an onerous if not impossible task. Thus, by gross comparison of classes Range and Forest appeared to be reasonably accurate. Since this research emphasizes woodland and forest-edge bird species, Rangeland changes were not considered in the further analysis. Further preprocessing of the two Forest layers for analysis revealed more inconsistencies.

Table 4.3: Urban Areas in LULC That Were Not Classed as Urban in GAP (hectares)

| Range | Forest | Agriculture | Wetland | Water |
|---|---|---|---|---|
| 35,000 | 18,000 | 7,000 | 5,000 | 3,000 |

The issue of clearcut forest areas was not readily discernable from the classification schemes and primary references. On investigating this issue, it was learned that LULC did not delineate forest clearcuts, they choose a definition of "land use" rather than "land cover." GAP mapped clearcut areas as "seedling/sapling,"again adhering to a USFS preference toward a land use definition. Further, LULC defined forest as areas of trees with 10% or greater crown closure, whereas GAP used 15% crown closure to define a forest. In addition, the minimum mapping unit (MMU) for the GAP data is 2 ha. for all classes. On receiving the LULC computer files, an MMU of 4 ha. was noted, and this was assumed to be the MMU for all classes. In fact, the MMU for the Forest class is 16 ha. for the LULC data versus 2 ha. for the GAP data. These findings prompted a detailed look into the all aspects of the Forest data sets (Table 4.4). For GAP, metadata existed and the contractor in Missoula, Montana was contacted directly. LULC on the other hand, had very little metadata and the people directly involved were difficult to locate and those contacted had varying levels of recall concerning many of the specific details about the data. Table 4.4 presents a summary the attributes for each surface cover data set.

Table 4.4: Comparison of Attributes Between LULC and GAP

| Attribute | GAP | LULC | Analysis Impacts |
|---|---|---|---|
| Base Map | 1:24,000 | 1:250,000 | Geographic precision |
| Metadata | Good | Poor | Processing reliability |
| Forest | 15% crown closure | 10% crown closure | Lessens accuracy |
| Minimum Mapping Unit | 2 ha. | 16 ha. | Must match to largest unit, increases coarseness of analysis |
| Clearcuts | included | excluded | Dampens ecological sensitivity |
| Accuracy | 85% | 85% | Combined equals 72% expected |
| Sensor Type | Satellite | High altitude photography | Satellite = computer processing<br>Photos = manual, then computer processing |
| Data Type | Raster (30mx30m converted to 2 ha. MMU) | Vector (polygons) | Raster is easier to process |

### 4.2.2 Final Comparability

For analysis purposes, the two data sets needed to match as closely as possible so that bird responses could be correlated to comparable landscape data. Therefore, clearcut areas were added back into the GAP data and the GAP Forest class was merged from 2 ha. to 16 ha, using geoprocessing techniques that converted Forest polygons 8 ha or less to the non-Forest class. These conversions increased the coarseness of the independent variables, very likely dampening the overall correlative response of avian trends. However, side by side plots of the Forest data inside the 1200 km$^2$ circles showed distinct differences in the two time periods (70s and 90s), in particular changes in patchiness and total area of forest. Figure 4.1 is an example route that shows the configuration of the forest landscape changed in the twenty-year period, becoming more patchy.

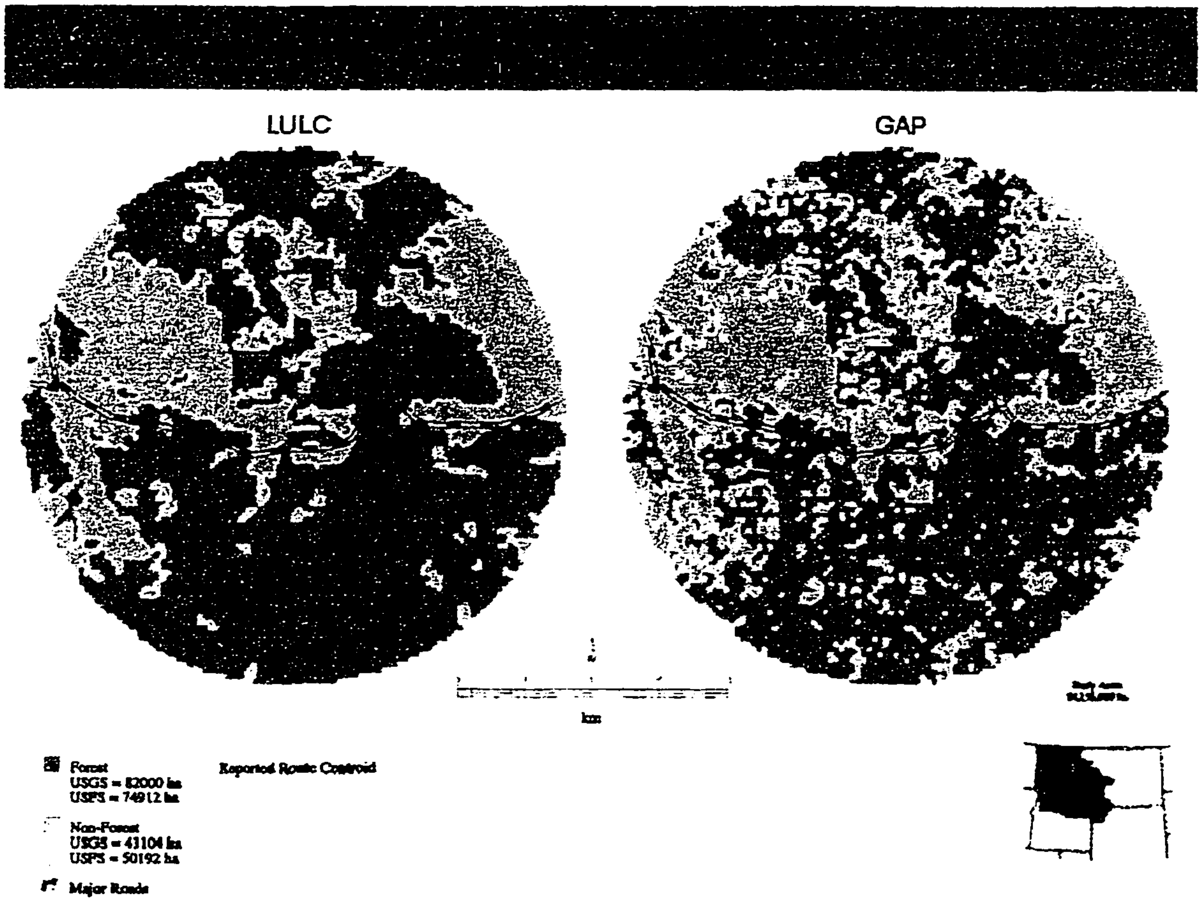


Figure 4.1: Example BBS Route Comparison (1970s-1990s)

There was no way to reconcile the difference in base map scale or the different rules used to define the Forest class (10% verus 15% crown closure). These scale and process differences may cause error in a comparative analysis, compounding the effects of the known error in classification (~15%). However, in the final comparison, many of the differences between GAP and LULC did not necessarily preclude change detection but rather increased the coarseness (grain) of the landscape data. So, while an increase in coarseness of the independent variables may have a dampening effect, it does not mean no effect or that strong relationships cannot be detected in statistical patterns. As a further check of consistency on the Forest layer only, Glacier and Yellowstone National Parks were plotted and cross-matched. Since little change would have taken place inside the Parks, the Forest acreage should have remained relatively constant between time periods. This check also produced better than expected results: Glacier was 81.3 % consistent between time periods and Yellowstone was 86.1 % consistent (Figure 4.2). Comparisons of routes within the parks showed consistency (Figure 4.3), as well. The three sample routes in Yellowstone were highly consistent, whereas, the greatest inconsistency was seen in the Logan Pass route in Glacier. This result is understandable given the Logan Pass area has the greatest topographic relief in the study area, making it more difficult to interpret (with photography or satellite imagery). These comparisons do not shed light on the exact nature of the inconsistencies between the spatial coverages but simply increases confidence that at least the Forest class for both coverages are near the reported accuracy of 85%.

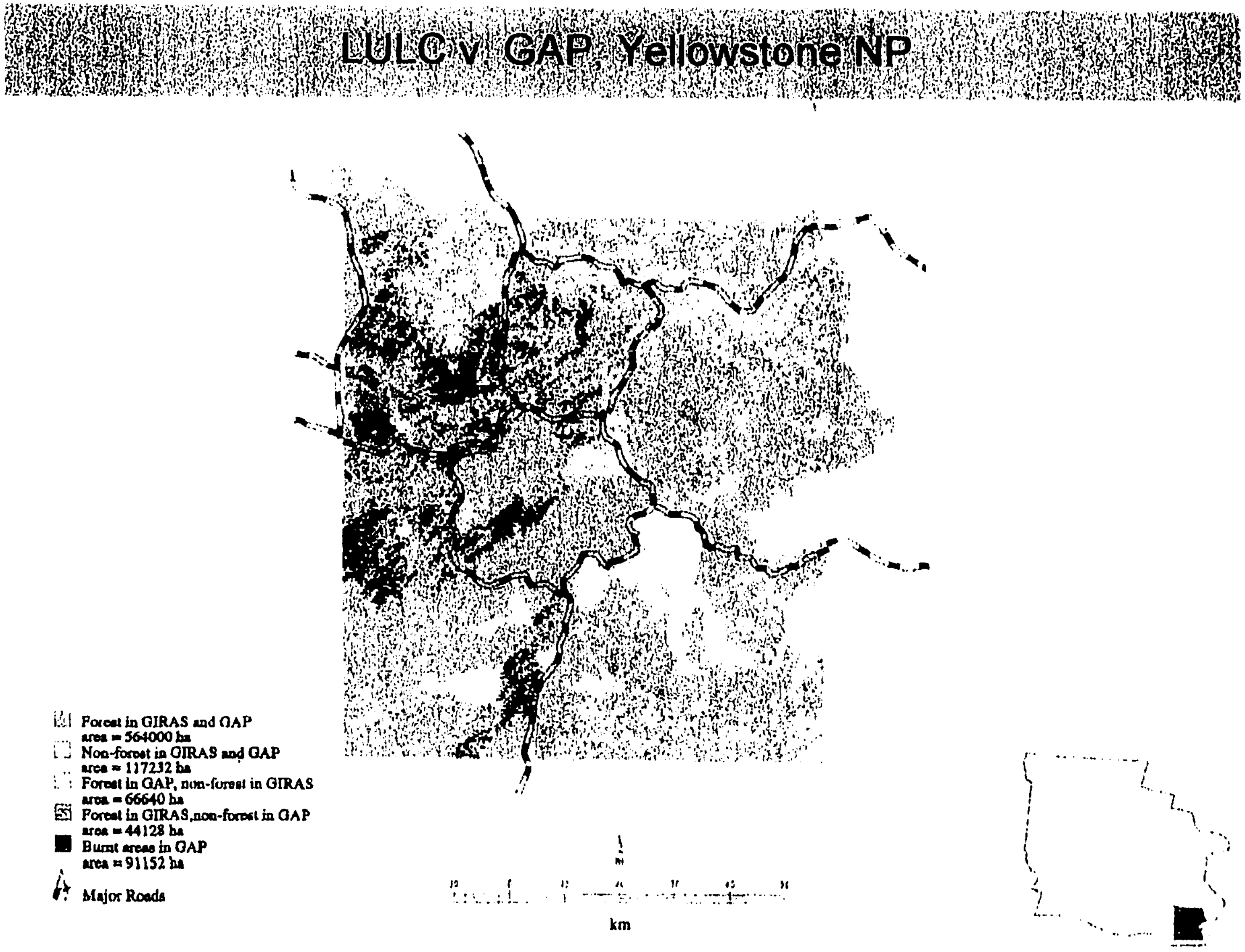


Figure 4.2: Yellowstone National Park, Showing GAP and LULC Forest Class Overlap Areas

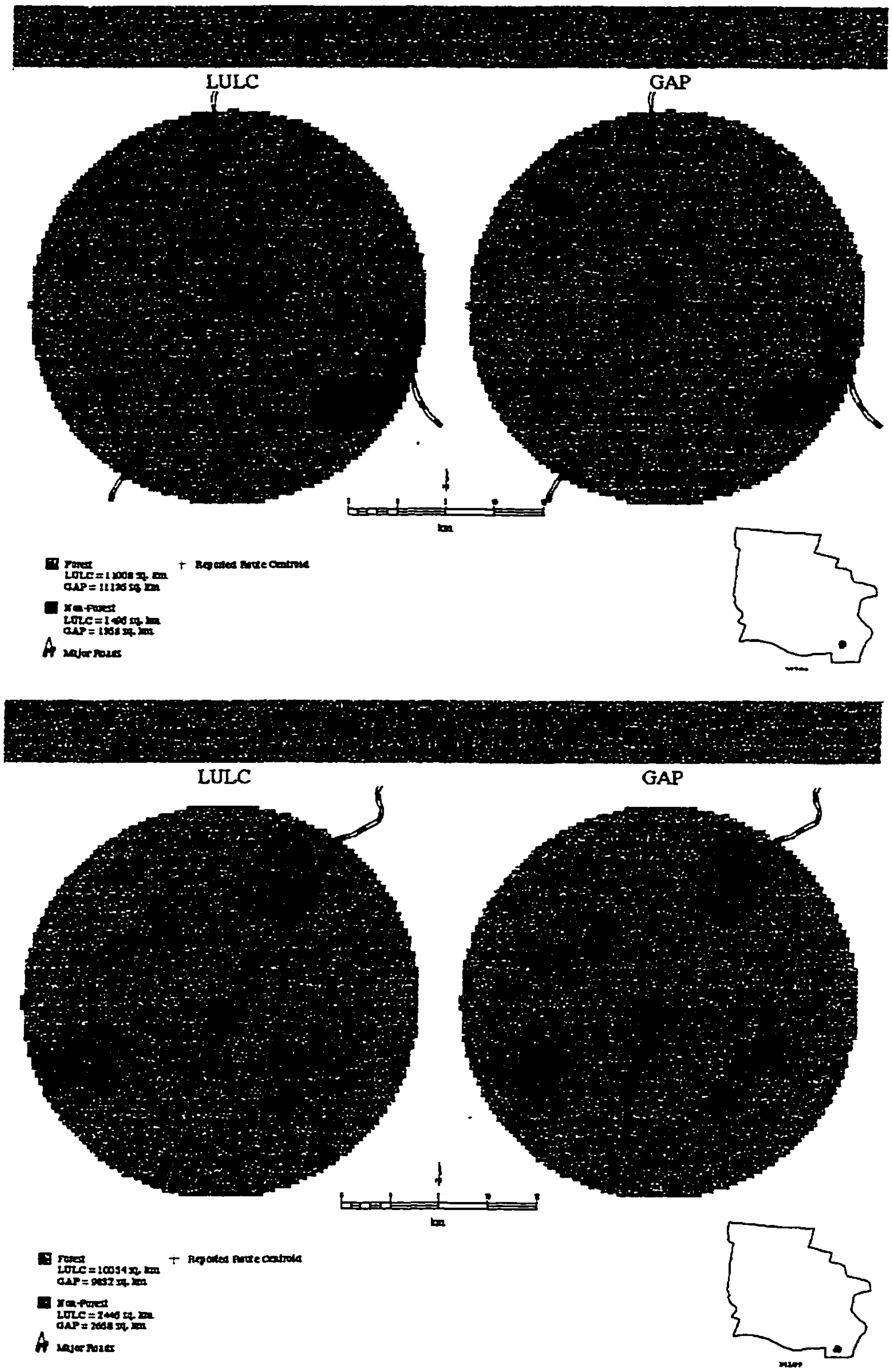


Figure 4.3: BBS Route Comparisons in Yellowstone and Glacier National Parks

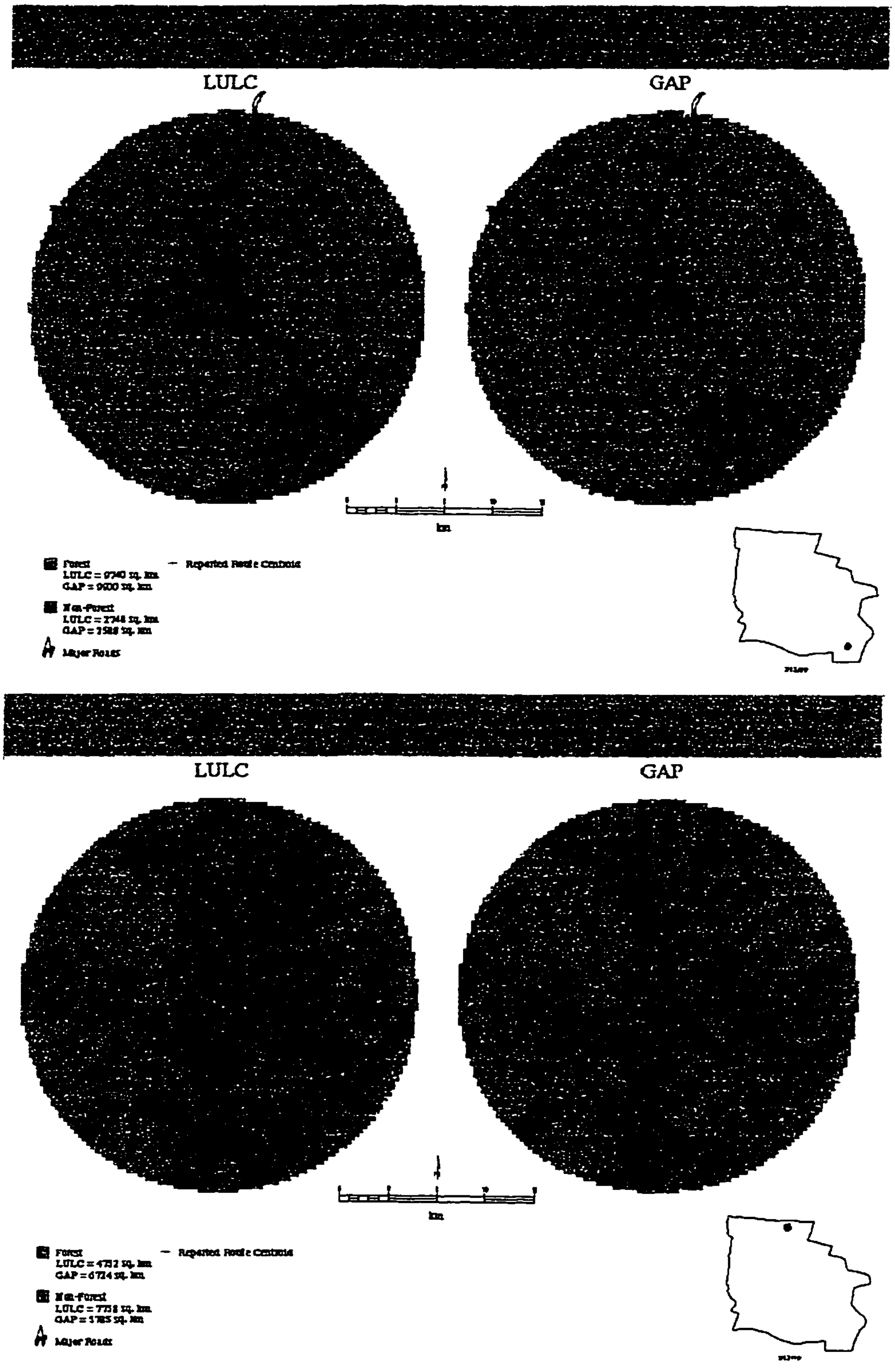


Figure 4.3: (continued)

The decision to proceed with the regression analysis in light of spatial coverage inconsistencies was reinforced by the side by side route comparisons in which routes outside the national parks showed expected patterns of change (Figure 4.1) and routes inside the parks were relatively consisted for forest cover (Figure 4.2 and 4.3). Also, because there is such a paucity of large-area ecological studies it was believed that the current study should continue, using the existing LULC and GAP data sets, especially since the data sets represent the quality of data used routinely for natural resource management. Finally, Brown (1995) notes macroecology studies are not reductionist in nature, but rather such studies knowingly sacrifice detail to see the big picture. This research was not framed on an experimental design that set forth a null hypothesis to either prove or disprove, but rather a confirmatory approach (Burnham and Anderson 1998) that sought to characterize overt statistical patterns that support ecological inference. This doesn't mean poor data are acceptable in research. On the heuristic level, however, it is sensible to assume that the considerable expense and human effort that went into the LULC and the GAP programs would not yield extremely poor quality information, especially for the more notable (in both programs) Forest layer. So, even though the landscape change data increased in coarseness with preprocessing, a net change in Forest cover occurred between the 1970s and 1990s that could be quantified for regression analysis.

### 4.3 Regression Results - Temporal Trend

Regressions models (one or more independent variables) were calculated for 64 bird species. The majority (53%) had Woodland breeding habitat preference, and the balance were equally split (23% each) between Successional Shrub and Other. Declaring statistical significance at $p \leq 0.10$ and $\geq 10.0$ error degrees of freedom, 25 of the 64 total species had acceptable models. These statistical parameters were subjective thresholds chosen at break points in the data that were biologically acceptable yet not so restrictive as to preclude models that would contribute to broad patterns of response. Species not meeting these criteria were deemed inconclusive.

The 25 species with acceptable models were similarly proportioned to the overall sample set in terms of breeding habitat, with 16 species (64%) Woodland, 5 species (20%) Successional Shrub, and 4 species (16%) Other breeding habitat preference (Table 4.5). A bias toward Woodland breeding habitat preference would be expected, given that all the independent variables are measures derived from Forest cover data and most of the routes in the study area are in forests. Ecologically sensible patterns surfaced by looking at individual bird responses for each independent variable, grouped by breeding habitat preference, migration status, and regression coefficients. Groups sorted by a single independent variable should not be interpreted as that variable having the sole influence on the species, because the majority of birds had two or more variables in their model.

|  |  |  |  |  |  | Regression Coefficient |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: |  |  |  |  |  |  | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |

Table 4.5: Temporal Analysis - Acceptable Models Sorted by Breeding Habitat

Guide to Table - Type: W=Woodland, S=Successional Shrub, O=Other; N: Number of Routes with Acceptable Data; MS: Migration Status, A=Long Distance Migrants, B=Short Distance Migrants, R=Residents)

### 4.3.1 Mean Forest Patch Size

Eleven birds species had models that contained Mean Forest Patch Size (MFPS), two birds with a negative relationship and nine birds with a positive relationship (Table 4.6). Eight of the nine birds with positive responses were Woodland birds while the two birds with a negative relationship were Successional Shrub (edge). Thus, in those landscapes with greater increases in MFPS, Woodland birds tend to have higher population trends. The American goldfinch and white-crowned sparrow, edge species, showed the strongest negative relationships with MFPS and the solitary vireo, a woodland species, showed the strongest positive relationship.

The Introduction noted that considerable evidence suggests forest patch size is strongly correlated with species richness and distribution (Robbins et al. 1989). When area is held constant, a more patchy landscape (lower mean patch size) may be considered more fragmented than one with fewer patches (larger mean patch size), thus mean patch size is often viewed as an index of landscape fragmentation. The results shown in Table 4.6 tend to support a relationship between fragmentation and avian trend that is not inconsistent with eastern studies which suggest Woodland species decline with fragmentation (smaller MFPS).

|  |  |  |  |  |  | Regression Coefficient |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: |  |  |  |  |  |  | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |

Table 4.6: Temporal Analysis - Acceptable Models Sorted by Mean Forest Patch Size

### 4.3.2 Forest Patch Size Coefficient of Variation

Ten birds had models that contained Forest Patch Size Coefficient of Variation (FPSCV); six birds showed a positive relationship, and four birds had a negative relationship to FPSCV (Table 4.7). Unlike MFPS there were no obvious groupings, in that Woodland species showed both positive and negative responses to FPSCV. FPSCV measures the relative variability of patch size about the mean (variability as a percentage of the mean). These results suggest variability in patch size is an inconsistent predictor of woodland bird population trend. The hermit thrush appears strongly influenced by MFPSCV, having higher population trends in landscapes with greater variability in patch size. In this region, area-sensitive species simply may not be limited by patch size variation.

| | | | | | | Regression Coefficient | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: | | | | | | | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |

Table 4.7: Temporal Analysis - Acceptable Models Sorted by Forest Patch Size CV

### 4.3.3 Forest Edge

Of all five independent variables, Forest Edge was included in the least number of models. Only five bird species showed a relationship with edge, three negative and two positive (Table 4.8). The gray jay—a woodland species—showed a very strong positive correlation with edge, while the American goldfinch—an edge species—showed a strong negative relationship. These are not the expected responses of woodland and edge species. This is perhaps another indication of the arbitrary nature of guild assignment in BBS (Peterjohn and Sauer 1994). Thought the gray jay may breed in the forest interior, thus classed Woodland, it frequents the forest edge when foraging. Also, due to the coarseness of the forest data, which became smoother (less total edge), Forest Edge may have lost strength as a predictor variable. It is possible also these birds are more strongly influenced by other environmental factors, or the results are spuriou, due simply to random chance, particular with such a small number of observations.

|  |  |  |  |  |  | Regression Coefficient |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: |  |  |  |  |  |  | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |

Table 4.8: Temporal Analysis - Acceptable Models Sorted by Forest Edge

### 4.3.4 Percent Forest

Again for this variable, Woodland bird responses appear to support patterns found in eastern studies of fragmentation, where larger amounts of total forest in a landscape tend to increase avian population trends. Percent forest was included in the regression models for eight bird species, six showing a positive relationship and two showing a negative relationship (Table 4.9). Except for the brewer's sparrow and mountain bluebird, the birds showing a positive relationship were Woodland species. Mountain bluebirds sometimes nest in forests and Hejl (1994) found them more abundant in clearcut areas. Conversely, the negative relationships to percent forest did not include any Woodland birds. These results suggest that in landscapes with greater measures of Percent Forest Woodland species have higher population trends.

|  |  |  |  |  |  | Regression Coefficient |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: |  |  |  |  |  |  | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |

Table 4.9: Temporal Analysis - Acceptable Models Sorted by Percent Forest

### 4.3.5 Mean Nearest Neighbor Distance

Eight birds included Mean Nearest Neighbor Distance (MNND) in their models, one-half with a negative relationship and one-half with a positive relationship (Table 4.10). All of the positive relationships were Woodland species, whereas the negative relationships contained all three classes. MNND measures the mean distance of one patch to the nearest neighbor of the same patch type, which in this case was forest patch to forest patch. Ecologically, MNND may influence a number of processes such as distance to the same or competing species and isolation from source islands. As mentioned in the Introduction, interpatch distance plays a role in island biogeography theory and metapopulation theory. Theses results are inconclusive for grouping birds, and counterintuitive for the expected response of Woodland birds. It is possible the coarseness of the data decreased ecological sensitivity or these results are spurious.

| | | | | | | | Regression Coefficient | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: | | | | | | | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |

Table 4.10: Temporal Analysis - Acceptable Models Sorted by Mean Nearest Neighbor Distance

## 4.4 Response Groups

As a group, migrants were the majority species in both the temporal and spatial analysis. Of birds with acceptable models, 84% in the temporal analysis and 81% in the spatial analysis were migrants (Table 4.11 and Appendix II) compared to 88% in the overall sample set of 64 birds. Of the nine birds in the temporal analysis with positive relationships with MFPS, four were long-distance migrants, three short-distance, and two residents.

| | | | | | | | Regression Coefficient | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | R$^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Foest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
| Scale: | | | | | | | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |

Table 4.11: Temporal Analysis - Acceptable Models Sorted by Migration Type (A=Long Distance Migrants, B=Short Distance Migrants, R=Residents)

As for grouping birds by response to independent variables, the cluster analysis produced little pertinent results in terms of meaningful ecological groups (Appendix IV). Cluster analysis showed birds had many different patterns of response; each species appears ecologically distinct. Figure 4.4 shows the relative proportion of positive, negative, or no response for each of the three bird groups (Other, Successional Shrub, Woodland) to the five independent variables. In most instances, each of the five independent variables had more birds with no response than those that responded either positively or negatively.

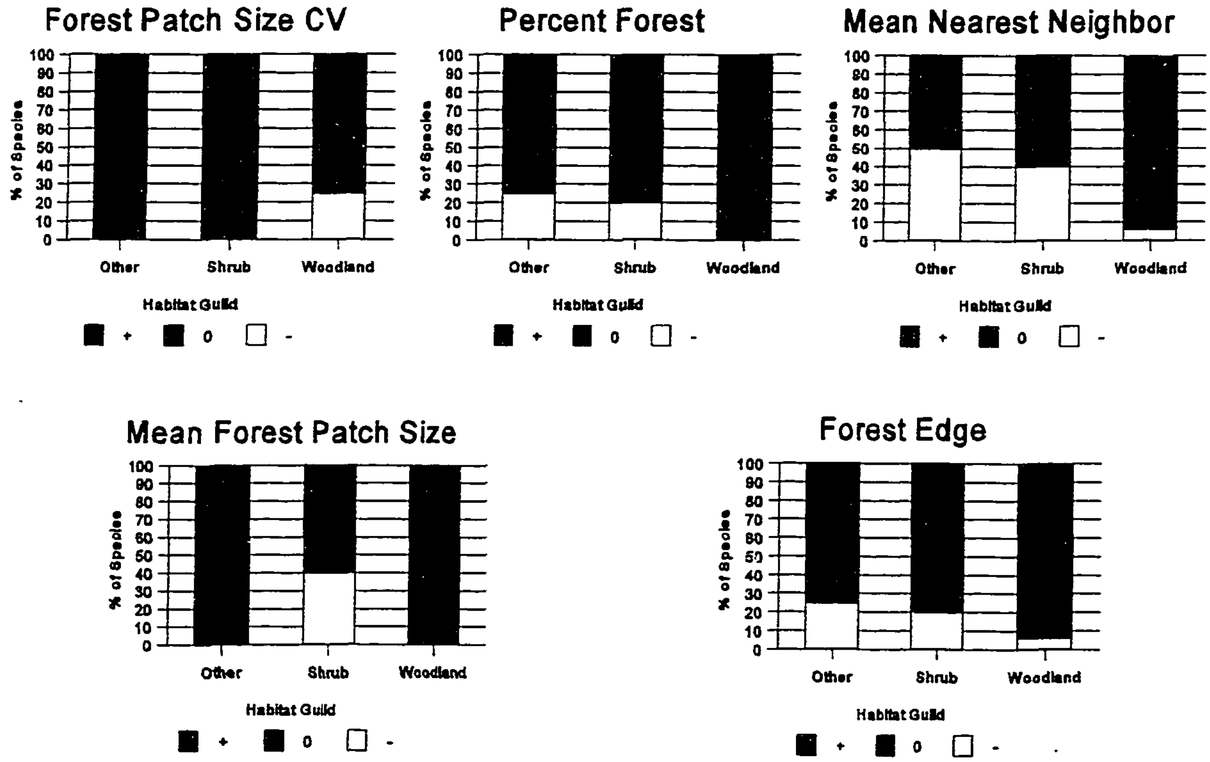


Figure 4.4: Chart of Bird Response For Each Independent Variable (Red=Positive Correlation, Green=No Correlation, Yellow=Negative Correlation)

No new response groups emerged beyond the relationships of woodland species and edge species already reported. From these relationships, however, several birds stand out in their relative strength of response to certain independent variables. The gray jay, a woodland species, exhibited a strong positive relationship with Edge and Percent Forest. The American goldfinch, an edge species, exhibited a strong negative relationship with MFPS and Edge. The white-crowned sparrow, also an edge species, was also negatively correlated with MFPS and Percent Forest. The responses of these birds were not inconsistent in the spatial analysis. These birds might be considered a response group, however, given the sample size for analysis and the coarseness of data used, statistical inference cannot be made with a high degree of confidence.

### 4.5 Temporal - Spatial Analysis

Twenty-one species had significant models in both the temporal and spatial analysis. Eight birds responded similarly, either a positive or a negative correlation, in both the temporal and spatial analysis: American goldfinch, white-crowned sparrow, western tanager, solitary vireo, gray jay, mountain bluebird, western wood-pewee, and barn swallow (Table 4.12). These birds responded to three of the independent variables: MFPS, Forest Edge, and Percent Forest. Moreover, the responses of these birds were ecologically sensible, following literature-based expected patterns of woodland and edge species.

Table 4.12: Birds With Similar Temporal and Spatial Response. Guide to Table - Type: W=Woodland, S=Successional Shrub, O=Other; MS: Migration Status, A=Long Distance Migrants, B=Short Distance Migrants, R=Residents

| Name | Type | MS | Regression Coefficient (+ or -) | |
|---|---|---|---|---|
| | | | Temporal | Spatial |
| **Mean Forest Patch Size** | | | | |
| American goldfinch | S | B | - | - |
| white-crowned sparrow | S | B | - | - |
| western tanager | W | A | + | + |
| solitary vireo | W | A | + | + |
| **Forest Edge** | | | | |
| gray jay | W | R | + | + |
| **Percent Forest** | | | | |
| mountain bluebird | O | B | + | + |
| western wood-pewee | W | A | + | + |
| **Mean Nearest Neighbor Distance** | | | | |
| barn swallow | O | A | - | - |

Some differences surfaced when spatial patterns of response were compared to MFPS. Seventeen of the 41 birds with acceptable models were negatively correlated with MFPS and exhibited no particular grouping with regard to breeding habitat. These results suggest bird abundance for a variety of species in the inter-mountain west decreases in landscapes with greater measures of MFPS . Temporally, Woodland birds have higher population trends in larger patch size, but some birds in all three breeding guilds tend to "live" in greater abundance in more patchy (smaller patches) environments. Similar findings were noted in western Oregon (McGarigal and McComb 1995) in which species abundance was greater in more heterogeneous, fragmented habitat. Hejl (1994) found forest species, in the western states, less abundant associates with clearcuts. Also in the spatial analysis 15 of the 17 species that included MFPS in their model were migrants: this relationship is consistent with the temporal trend results.

The spatial analysis showed no interesting patterns with FPSCV and Edge (Appendix II). Percent Forest, however, tended to parallel the temporal analysis findings in that most Woodland species exhibited a positive relationship. A relationship of landscape and spatial abundance appears to be that Woodland species prefer large amounts of forest while a variety of species, including some Woodland birds, are abundant in smaller patches. These findings also tend to confirm that bird-habitat relationships observed on the temporal scale do not necessarily extrapolate to the spatial scale.

### 4.6 Summary of Results

Comparability between the LULC and GAP spatial data coverages was disappointing, especially since they represented high quality data sets. However, they were comparable enough to derive valid structural change metrics for the "Forest" component of the data set. As a consequence, only the Forest category was used to derive temporal-spatial landscape change metrics that were correlated with avian trend and abundance. Results showed inter-mountain species responded to forest fragmentation in similar fashion to eastern species: woodland species, predominantly migrants, exhibited a positive relationship between population trend and trend mean patch size and percent forest. However, edge species exhibited a negative relationship. When compared to spatial abundance, calculated as a current five-year average, similar relationships were observed with percent forest, but were inconsistent with mean patch size. That is, both woodland and edge species contained models showing a negative spatial abundance relationship with mean patch size. No new response groups emerged in the analysis but several birds exhibited fairly strong relationships to landscape change: gray jay, American goldfinch, and white-crowned sparrow.

# CHAPTER 5. SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

## 5.1 Summary

LULC data for the 1970s and GAP data for the 1990s were compared, reconciled where possible, then used to examine relationships between temporal change in landscape structure and breeding bird population trends for a large area of the inter-mountain west: western Montana and northern Idaho. Inconsistencies between LULC and GAP limited the measure of landscape changes to only the change in forest cover. From the forest layer of LULC and GAP, five indices of landscape change (MFPS, MFPSCV, Edge, Percent Forest, and MNND) were calculated for the areas inside 40 km diameter circles drawn around each of 45 USGS BBS routes in the study area. Within each route circle, metrics of forest change between the two time periods were correlated to BBS population trend and abundance for 64 species of birds.

Of the total birds, 25 bird species had acceptable correlation models of response ($p \leq 0.10$) to the ~ 20-year change in forest. Ecologically, these results repeated patterns noted in avian-fragmentation studies in the eastern United States (Robbins 1979, 1980) where woodland species tended to correlate positively with increases in MFPS, and percent forest while edge species correlated negatively. Similar patterns were observed in the spatial abundance of birds. Looking at each independent variable separate, a majority of birds demonstrated no sensitivity to each landscape change variable at the scale of observation used in the current study. The current study demonstrated that useful measures of landscape change could be quantified for a large area, yielding change measures that correlated to a biological response.

The degree to which the landscape data could be reconciled was disappointing: from nine classes to only one. While these results might reinforce the "shaky" nature of thematic data (Hess 1994), it does not alter the need for using remote sensing to produce such data. How else will it be possible to routinely assess large-area changes? The inconsistencies noted herein were not the fault of the tools. The results of the current study support the approach of blending remote sensing technology with ecology to conduct applied ecological research and as a possible means of monitoring entire ecosystems.

While change detection is a growing field of remote sensing, the current study demonstrated that achieving "comparability" between two different spatial data sets is possible but not without problems—some technical, some procedural, and some administrative. The technical problems are more easily solved, whereas, the other issues— like reaching a consensus on common classification scheme—may take longer as they involve many different organizational entities. Advances in metadata standards and the rapidly evolving National Biological Information Infrastructure hold promise for simplifying many of the onerous steps in this research. With time, more data will go onto the Web where exposure to use will surface data quality and compatibility issues that can be corrected for subsequent use. As this cycle continues, data quality and compatibility should improve, advancing the time when it will be possible for advanced information systems to perform periodic change detection from remotely sensed data for an entire ecosystem or landscape. Once the change is quantified it should be equally possible to automatically link the change to a biological response, thus completing the steps needed for ecosystem monitoring.

The spatial analysis suggests many inter-mountain species are more abundant in landscapes with higher measures of patchiness. This appears to hold true regardless of breeding habitat preference: some Woodland species preferred more patchy environments. This pattern may be due to the differing nature of fragmentation between eastern and western landscapes (Hansen and Urban 1992) and perhaps wide variation in how the term 'fragmentation' is used (Bender et al. 1998). In the east, fragmentation is often associated with a dramatic change in land use, more commonly forests to agriculture or urbanization that results in habitat destruction. In the west, fragmentation is more often associated with roads, fire, and silvaculture practices that do not necessarily change the land use (Hejl 1994). In other words, patches in the east have a higher likelihood of becoming islands that bring area-dependent species closer to local extinction, whereas in the west the patch mosaic remains more or less forested even as patchiness increases. It is possible that in the inter-mountain west landscape patch size, overall, remains well above area-dependent thresholds. Some of the survey routes were almost fully forested in both coverages (Table 3.2). Further compounding interpretation is the wide variation in avian life histories, especially plasticity to move among forest patches of varying sizes (Hansen and Urban 1992) . While not discounting other ecological factors such as predation, food availability, and so forth, each species has its own threshold for patch size, thus making generalization based on avian responses to fragmentation risky. The difficulty lies in knowing when a net increase in patchiness equates directly to critical habitat loss.

That only eight bird species correlated temporally and spatially in a similar fashion is likely a function of the coarseness of the data, suggesting better data for more birds might explain more of the variation in the data. The sample size for analysis was small given that Hejl et al. (1995) reports 215 bird species are found in Rocky Mountain forests (24 species breed outside the United States). The response of these particular eight birds may suggest they have a particular sensitivity at the grain of the study and therefore might serve as a "response guild" that could be monitored as a measure of ecosystem change. Conversely, it may be that these birds are more easily measured by BSS. Bender et al. (1998) found patch size to be an important correlate with vertebrate population density but that the difficulty lies in determining net patch size effects from habitat loss effects. These are very difficult effects to study for a large number of birds over large landscapes. Peterjohn et al. (1995) and Hejl (1994) have found no defensible trends for western species in the past century.

Overall, migrant species were responsive to landscape change than resident species to only a slightly higher degree than their proportion in the sample set. Also, there were no birds with exceptionally high regression coefficients, suggesting as expected that landscape change is only one variable that influences avian population trends. For migrants, this makes sense as they are also influenced by migration and winter survival, factors that have influence at the global scale. Sherry et al. (1995) and Bender et al. (1998) suggest the life histories of migrants make them hardy and more flexible to environmental stressors and habitat change, allowing them to exploit space and time differences more easily than resident species. Flather and Sauer (1996) and Howe (1992) found neotropical migrants more sensitive as a group to change in landscape structure, while Hejl (1994) found almost all species that increased after clearcut harvest were migrants. This research tends to support the notion migrants are sensitive to changes in the landscape to the extent they manifest a measurable response in population trend. The proportion of responses, migrants versus residents, was close to the same proportion in the sample set, suggesting migrants are not necessarily more responsive to landscape change than residents.

## 5.2 Conclusions

A retrospective assessment at the conclusion the current study reveals several factors that weaken the strength of inference from statistical patterns observed: (i) the sample size was small (number of BBS routes and number of birds), (ii) inconsistencies in the landscape coverages increased the coarseness of the predictor variables, (iii) BBS data are coarse, (iv) there are many other obvious influences on avian trend and abundance, and (v) avian populations exhibit wide fluctuations in nature. Still, some expected relationships were observed, strengthening otherwise weak statistical results. Given these caveats, several conclusions can be drawn from the results discussed in Chapter 4:

- Long-term, large-area studies of landscape structure change are possible with comparable data sets. The complexity of interacting processes will always make large-area studies more difficult to interpret.

- Inter-mountain birds species tend to respond similarly to eastern bird species to changes in landscape structure: in those landscapes with greater measures of mean patch size and percent forest, Woodland species tended to have higher measures of population trend while edge species tended to have lower measures.

- Some birds respond similarly to landscape change in both the spatial and temporal dimensions. Though temporal trend and current abundance are not directly related, eight inter-mountain species exhibited similar responses to mean forest patch size, percent forest, and edge.

- Migratory birds appear no more sensitive to landscape variation and change than resident species. More migratory species than resident species exhibited a response to changes in landscape structure, but the proportion was very close to the proportion in the sample set.

## 5.3 Recommendations for Future Research

The results of this research imply several possible topics that may benefit from future research:

- Compare the effects of this research with large-area, temporal change studies with finer grained measures of landscape structure change. Attempts were made to use fine-grained spatial data, but incompatibilities resulted in more coarse-grained data used in the regression analysis. Even though 25 birds correlated with the coarse-grained data, it is likely more birds would have responded given finer grained measures of change in landscape structure. In addition, there is a general paucity of long-term, large-area studies.

- Determine the usefulness of avian "response groups" as a sensitivity measure of ecosystem change, using the species noted in this research with high degrees of response. Resource managers might benefit from a set or group of species whose trend or spatial abundance correlate well to landscape changes. Measuring only a few species in a few locations is obviously easier and less expensive than whole population monitoring over large areas. What is not known are the thresholds or triggers that signal a desired or undesired state approaching. For example, are there patch size thresholds for the American goldfinch, white-crowned sparrow, and gray jay that signal an undesirable condition is developing in the landscape mosaic?

- Streamline the approach of linking landscape changes to biological response, using a successful blend of ecology, remote sensing, and information technology. Even though the current study demonstrated that information technology, including GIS and remote sensing, can be used to support ecological research, data processing and system integration was too complex for routine use by conservation organizations. In the context of this research, several steps required skilled technical support and powerful software programs. Even though the Web is revolutionizing the accessibility of data, there remains much work to do in streamlining a procedure that uses large-area data sets for routine natural resource management or ecological research.

- Produce a more accurate coverage of the LULC data set, using triplicate MSS imagery and aerial photography. The LULC data set is a unique, historic reference for landscape change for the entire United States. The current study revealed inconsistencies between the LULC and GAP data sets that diminishes the usefulness of the LULC data set for ecological research and general use by natural resource agencies. Many of these inconsistencies, especially those pertaining to forest cover, could be resolved by a follow-up interpretation of the study area using archived satellite imagery. For example, the North American Landscape Characterization project provides nation-wide coverage of co-registered, MSS imagery for 1973, 1986, and 1991. Using a triplicate set of these images along with aerial photography for the region, a single interpreter could produce a more accurate coverage of forest cover for the 1970s period.

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# APPENDIX I: Birds Used In The Analysis And BBS Habitat Guild

| AOU | NAME | Habitat Guild | # ROUTES Temporal/Spatial |
|---|---|---|---|
| .06870 | AMERICAN REDSTART<br>*(Setophaga ruticilla)* | W | 11/13 |
| 06560 | AUDUBON'S WARBLER<br>*(Dendroica c. auduboni)* | W | 32/36 |
| 05960 | BLACK-HEADED GROSBEAK<br>*(Pheucticus melanocephalus)* | W | 17/20 |
| 05180 | CASSIN'S FINCH<br>*(Carpodacus cassinii)* | W | 22/21 |
| 04910 | CLARK'S NUTCRACKER<br>*(Nucifraga columbiana)* | W | 14/12 |
| 03940 | DOWNY WOODPECKER<br>*(Picoides pubescens)* | W | 11/11 |
| 04690 | DUSKY FLYCATCHER<br>*(Empidonax oberholseri)* | W | 20/22 |
| 05140 | EVENING GROSBEAK<br>*(Coccothraustes vespertinus)* | W | 13/15 |
| 04840 | GRAY JAY<br>*(Perisoreus canadensis)* | W | 16/17 |
| 03930 | HAIRY WOODPECKER<br>*(Picoides villosus)* | W | 17/16 |
| 04680 | HAMMOND'S FLYCATCHER<br>*(Empidonax hammondii)* | W | 23/28 |
| 07590 | HERMIT THRUSH<br>*(Catharus guttatus)* | W | 15/14 |
| 07380 | MOUNTAIN CHICKADEE<br>*(Parus gambeli)* | W | 24/25 |
| 06750 | NORTHERN WATERTHRUSH<br>*(Seiurus noveboracensis)* | W | 11/11 |
| 05679 | OREGON JUNCO<br>*(Junco h. oreganus)* | W | 31/35 |
| 04050 | PILEATED WOODPECKER<br>*(Dryocopus pileatus)* | W | 11/12 |
| 05330 | PINE SISKIN<br>*(Carduelis pinus)* | W | 32/34 |
| 07280 | RED-BREASTED NUTHATCH<br>*(Sitta canadensis)* | W | 31/35 |
| 06240 | RED-EYED VIREO<br>*(Vireo olivaceus)* | W | 12/13 |
| 04021 | RED-NAPED SAPSUCKER<br>*(Sphyrapicus nuchalis)* | W | 18/19 |
| 05210 | RED CROSSBILL<br>*(Loxia curvirostra)* | W | 19/20 |
| 07490 | RUBY-CROWNED KINGLET<br>*(Regulus calendula)* | W | 25/27 |
| 0300 | RUFFED GROUSE<br>*(Bonasa umbellus)* | W | 9/7 |
| 06290 | SOLITARY VIREO<br>*(Vireo solitarius)* | W | 17/20 |
| 04780 | STELLER'S JAY<br>*(Cyanocitta stelleri)* | W | 13/17 |
| 07580 | SWAINSON'S THRUSH<br>*(Catharus ustulatus)* | W | 22/27 |
| 07540 | TOWNSEND'S SOLITAIRE<br>*(Myadestes townsendi)* | W | 13/15 |
| 06680 | TOWNSEND'S WARBLER<br>*(Dendroica townsendi)* | W | 16/21 |
| 07630 | VARIED THRUSH<br>*(Ixoreus naevius)* | W | 12/15 |
| 07560 | VEERY<br>*(Catharus fuscescens)* | W | 19/20 |
| 06270 | WARBLING VIREO<br>*(Vireo gilvus)* | W | 34/36 |
| 06070 | WESTERN TANAGER | W | 31/34 |

|  |  |  |  |
|---|---|---|---|
|  | *(Piranga ludoviciana)* |  |  |
| 04620 | WESTERN WOOD-PEWEE<br>*(Contopus sordidulus)* | W | 27/29 |
| 07220 | WINTER WREN<br>(*Troglodytes troglodytes*) | W | 8/9 |
| 05290 | AMERICAN GOLDFINCH<br>*(Carduelis tristis)* | S | 17/16 |
| 05620 | BREWER'S SPARROW<br>*(Spizella breweri)* | S | 15/12 |
| 06810 | COMMON YELLOWTHROAT<br>*(Geothlypsis trichas)* | S | 21/21 |
| 05850 | FOX SPARROW<br>*(Passerella iliaca)* | S | 12/13 |
| 07040 | GRAY CATBIRD<br>*(Dumetella carolinensis)* | S | 16/17 |
| 07210 | HOUSE WREN<br>*(Troglodytes aedon)* | S | 23/24 |
| 05990 | LAZULI BUNTING<br>*(Passerina amoena)* | S | 20/20 |
| 05830 | LINCOLN'S SPARROW<br>*(Melospiza lincolnii)* | S | 17/16 |
| 06800 | MACGILLIVRAY'S WARBLER<br>*(Oporornis tolmiei)* | S | 29/34 |
| 07020 | SAGE THRASHER<br>(*Oreoscoptes montanus*) | S | 10/9 |
| 05810 | SONG SPARROW<br>*(Melospiza melodia)* | S | 32/35 |
| 05540 | WHITE-CROWNED SPARROW<br>*(Zonotrichia leucophrys)* | S | 18/16 |
| 04664 | WILLOW FLYCATCHER<br>*(Empidonax traillii)* | S | 31/34 |
| 06850 | WILSON'S WARBLER<br>*(Wilsonia pusilla)* | S | 13/12 |
| 06520 | YELLOW WARBLER<br>*(Dendroica petechia)* | S | 36/36 |
| 06160 | BANK SWALLOW<br>*(Riparia riparia)* | O | 15/14/ |
| 06130 | BARN SWALLOW<br>*(Hirundo rustica)* | O | 28/30 |
| 04750 | BLACK-BILLED MAGPIE<br>*(Pica pica)* | O | 22/24 |
| 05100 | BREWER'S BLACKBIRD<br>*(Euphagus cyanocephalus)* | O | 29/29 |
| 04950 | BROWN-HEADED COWBIRD<br>*(Molothrus ater)* | O | 33/35 |
| 05080 | BULLOCK'S ORIOLE<br>(*Icterus bullockii*) | O | 10/9 |
| 06190 | CEDAR WAXWING<br>*(Bombycilla cedrorum)* | O | 14/15 |
| 06120 | CLIFF SWALLOW<br>*(Hirundo pyrrhonota)* | O | 27/27 |
| 04860 | COMMON RAVEN<br>*(Corvus corax)* | O | 31/34 |
| 04440 | EASTERN KINGBIRD<br>*(Tyrannus tyrannus)* | O | 14/15 |
| 07680 | MOUNTAIN BLUEBIRD<br>*(Sialia currucoides)* | O | 21/20 |
| 04130 | RED-SHAFTED FLICKER<br>*(Colaptes a. cafer)* | O | 37/39 |
| 07150 | ROCK WREN<br>*(Salpinctes obsoletus)* | O | 14/14 |
| 06140 | TREE SWALLOW<br>*(Tachycineta bicolor)* | O | 26/26 |
| 06150 | VIOLET-GREEN SWALLOW<br>*(Tachycineta thalassina)* | O | 21/25 |

# APPENDIX II: Temporal and Spatial Analysis Regression Tables

| NAME | Type | N | MS | R² | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Forest-Edge | Percent Forest | Mean Nearest Neighbor Distance |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Scale | | | | | | | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| AMERICAN REDSTART | W | 11 | | 0.82 | 0.022 | -1.66 | 1.40 | 2.13 | 4.39 | 0.18 | 0.00 |
| AUDUBON'S WARBLER | W | 32 | B | 0.32 | 0.011 | -0.01 | 0.18 | 0.00 | 0.66 | 0.00 | 0.48 |
| BLACK-HEADED GROSBEAK | W | 17 | A | 0.43 | 0.019 | -0.03 | 0.00 | -0.54 | 0.00 | 0.00 | -0.72 |
| CASSIN'S FINCH | W | 22 | B | 0.14 | 0.090 | 0.49 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 |
| CLARK'S NUTCRACKER | W | 14 | R | 0.72 | 0.004 | 0.09 | 1.04 | 1.74 | 0.00 | 0.00 | -0.70 |
| DOWNY WOODPECKER | W | 11 | | 0.05 | 0.506 | -0.09 | -0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| DUSKY FLYCATCHER | W | 20 | A | 0.52 | 0.008 | 0.15 | 0.95 | 1.35 | 0.00 | 0.00 | 0.39 |
| EVENING GROSBEAK | W | 13 | | 0.08 | 0.365 | 0.59 | 0.00 | 0.00 | -1.93 | 0.00 | 0.00 |
| GRAY JAY | W | 16 | R | 0.47 | 0.050 | 0.19 | 0.00 | -0.87 | 2.56 | 0.77 | 0.00 |
| HAIRY WOODPECKER | W | 17 | R | 0.36 | 0.011 | 0.06 | 0.00 | -1.06 | 0.00 | 0.00 | 0.00 |
| HAMMOND'S FLYCATCHER | W | 23 | | 0.19 | 0.125 | 0.18 | -0.52 | -1.26 | 0.00 | 0.00 | 0.00 |
| HERMIT THRUSH | W | 15 | B | 0.39 | 0.053 | -0.14 | 1.02 | 2.24 | 0.00 | 0.00 | 0.00 |
| MOUNTAIN CHICKADEE | W | 24 | | 0.03 | 0.431 | -0.13 | 0.00 | 0.00 | 0.00 | 0.12 | 0.00 |
| NORTHERN WATERTHRUSH | W | 11 | | 0.74 | 0.142 | 0.39 | -1.62 | -3.73 | 4.04 | 1.18 | 2.93 |
| OREGON JUNCO | W | 31 | B | 0.19 | 0.015 | 0.30 | 0.00 | 0.00 | 0.00 | 0.00 | 0.39 |
| PILEATED WOODPECKER | W | 11 | | 0.18 | 0.196 | 0.14 | 0.00 | 0.00 | 0.00 | 0.00 | 0.61 |
| PINE SISKIN | W | 32 | B | 0.21 | 0.033 | 0.26 | 0.00 | 0.00 | 0.00 | 0.26 | 0.60 |
| RED-BREASTED NUTHATCH | W | 31 | R | 0.26 | 0.041 | 0.12 | 0.36 | 1.03 | 0.00 | 0.00 | 0.31 |
| RED-EYED VIREO | W | 12 | | 0.22 | 0.123 | -0.05 | 0.00 | 0.00 | 0.00 | 0.00 | -0.17 |
| RED-NAPED SAPSUCKER | W | 18 | | 0.13 | 0.144 | -0.42 | 0.00 | 0.00 | 1.47 | 0.00 | 0.00 |
| RED CROSSBILL | W | 19 | R | 0.30 | 0.014 | 0.70 | 0.81 | 0.00 | 0.00 | 0.00 | 0.00 |
| RUBY-CROWNED KINGLET | W | 25 | B | 0.14 | 0.069 | 0.27 | 0.00 | -0.40 | 0.00 | 0.00 | 0.00 |
| RUFFED GROUSE | W | 9 | | 0.15 | 0.296 | 0.49 | 0.32 | 0.00 | 0.00 | 0.00 | 0.00 |
| SOLITARY VIREO | W | 17 | A | 0.54 | 0.016 | 0.37 | 1.10 | 1.72 | 0.00 | 0.21 | 0.00 |
| STELLER'S JAY | W | 13 | | 0.19 | 0.141 | 0.24 | 0.37 | 0.00 | 0.00 | 0.00 | 0.00 |
| SWAINSON'S THRUSH | W | 22 | | 0.06 | 0.253 | 0.21 | 0.00 | 0.00 | -0.56 | 0.00 | 0.00 |
| TOWNSEND'S SOLITAIRE | W | 13 | B | 0.26 | 0.075 | 0.43 | 0.00 | 0.00 | -1.24 | 0.00 | 0.00 |
| TOWNSEND'S WARBLER | W | 16 | | 0.13 | 0.179 | 0.01 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| VARIED THRUSH | W | 12 | | 0.81 | 0.011 | 1.12 | -1.36 | -2.72 | -2.51 | 0.26 | 0.00 |
| VEERY | W | 19 | | 0.06 | 0.308 | 0.08 | 0.00 | 0.00 | 0.00 | 0.00 | 0.18 |
| WARBLING VIREO | W | 34 | | 0.04 | 0.259 | 0.00 | 0.00 | 0.00 | 0.00 | 0.07 | 0.00 |
| WESTERN TANAGER | W | 31 | A | 0.10 | 0.086 | 0.23 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 27 | A | 0.29 | 0.004 | 0.06 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| WINTER WREN | W | 8 | | 0.74 | 0.277 | 1.77 | -2.50 | -5.14 | -4.72 | 0.00 | -0.57 |
| | | | | | | | | | | | |
| AMERICAN GOLDFINCH | S | 17 | B | 0.40 | 0.030 | 0.15 | -0.34 | 0.00 | -1.27 | 0.00 | 0.00 |
| BREWER'S SPARROW | S | 15 | A | 0.46 | 0.006 | -0.43 | 0.00 | 0.00 | 0.00 | 0.49 | 0.00 |
| COMMON YELLOWTHROAT | S | 21 | | 0.08 | 0.227 | 0.02 | -0.14 | 0.00 | 0.00 | 0.00 | 0.00 |
| FOX SPARROW | S | 12 | | 0.23 | 0.115 | -0.18 | 0.00 | 0.83 | 0.00 | 0.00 | 0.00 |
| GRAY CATBIRD | S | 16 | | 0.09 | 0.270 | -0.24 | 0.00 | 0.31 | 0.00 | 0.00 | 0.00 |
| HOUSE WREN | S | 23 | | 0.04 | 0.348 | -0.07 | 0.00 | 0.00 | 0.45 | 0.00 | 0.00 |
| LAZULI BUNTING | S | 20 | | 0.19 | 0.167 | 0.04 | 0.94 | 1.97 | 0.00 | 0.00 | 0.00 |
| LINCOLN'S SPARROW | S | 17 | | 0.03 | 0.525 | -0.08 | 0.00 | 0.00 | 0.00 | 0.00 | -0.19 |
| MACGILLIVRAY'S WARBLER | S | 29 | | 0.05 | 0.244 | -0.12 | 0.00 | 0.00 | 0.00 | 0.00 | -0.18 |
| SAGE THRASHER | S | 10 | | 0.65 | 0.083 | -0.39 | -1.65 | 0.00 | -7.01 | 0.98 | 0.00 |
| SONG SPARROW | S | 32 | B | 0.14 | 0.033 | -0.04 | 0.00 | 0.00 | 0.00 | 0.00 | -0.47 |
| WHITE-CROWNED SPARROW | S | 18 | B | 0.58 | 0.006 | -0.21 | -0.19 | 0.00 | 0.00 | -0.18 | -0.68 |
| WILLOW FLYCATCHER | S | 31 | A | 0.12 | 0.059 | -0.34 | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| WILSON'S WARBLER | S | 13 | | 0.13 | 0.223 | 0.50 | 0.00 | -0.81 | 0.00 | 0.00 | 0.00 |
| YELLOW WARBLER | S | 31 | | 0.02 | 0.439 | -0.03 | 0.04 | 0.00 | 0.00 | 0.00 | 0.00 |
| | | | | | | | | | | | |
| BANK SWALLOW | O | 15 | | 0.03 | 0.512 | -0.25 | -0.15 | 0.00 | 0.00 | 0.00 | 0.00 |
| BARN SWALLOW | O | 28 | A | 0.14 | 0.053 | -0.22 | 0.00 | 0.00 | 0.00 | 0.00 | -0.28 |
| BLACK-BILLED MAGPIE | O | 22 | | 0.06 | 0.279 | -0.19 | 0.00 | 0.25 | 0.00 | 0.00 | 0.00 |
| BREWER'S BLACKBIRD | O | 29 | | 0.28 | 0.157 | -0.12 | -0.18 | -0.50 | 0.74 | 0.13 | 0.19 |
| BROWN-HEADED COWBIRD | O | 33 | | 0.07 | 0.143 | -0.13 | -0.14 | 0.00 | 0.00 | 0.00 | 0.00 |
| BULLOCK'S ORIOLE | O | 10 | | 0.82 | 0.013 | -0.04 | 0.00 | -1.04 | 2.14 | 0.00 | 0.56 |
| CEDAR WAXWING | O | 14 | | 0.18 | 0.134 | 0.09 | 0.00 | 0.00 | 0.00 | 0.16 | 0.00 |
| CLIFF SWALLOW | O | 27 | A | 0.33 | 0.060 | -0.13 | 0.60 | 1.53 | 0.00 | -0.24 | -0.49 |
| COMMON RAVEN | O | 31 | | 0.02 | 0.416 | -0.13 | 0.00 | 0.00 | 0.72 | 0.00 | 0.00 |
| EASTERN KINGBIRD | O | 14 | | 0.65 | 0.034 | 0.22 | -0.19 | 0.50 | -2.31 | 0.00 | -0.26 |
| MOUNTAIN BLUEBIRD | O | 21 | B | 0.15 | 0.088 | 0.10 | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 |
| RED-SHAFTED FLICKER | O | 37 | B | 0.01 | 0.657 | 0.03 | 0.00 | 0.00 | 0.00 | -0.03 | 0.00 |
| ROCK WREN | O | 14 | | 0.06 | 0.407 | 0.08 | -0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| TREE SWALLOW | O | 26 | | 0.01 | 0.584 | -0.10 | 0.00 | 0.00 | 0.31 | 0.00 | 0.00 |
| VIOLET-GREEN SWALLOW | O | 21 | A | 0.14 | 0.092 | 0.31 | 0.00 | 0.00 | -0.89 | 0.00 | 0.00 |

Table I. Temporal Analysis Regression Table

| | | | | | | Regression Coefficients | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| NAME | Type | N | MS | $R^2$ | p-value (F-test) | Intercept | Mean Forest Patch Size | Forest Patch Size CV | Percent Forest | Forest-Ag Edge | Forest-Other Edge | Forest-Range Edge | Forest-Urban Edge | Mean Nearest Neighbor Distance |
| | | | | | | 1.0e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 | 1.00e+00 |
| AMERICAN REDSTART | W | 13 | | 0.65 | 0.019 | -39.09 | 0.00 | 0.00 | 0.00 | 0.00 | 1.32 | 1.62 | -0.14 | 0.00 |
| AUDUBON'S WARBLER | W | 36 | B | 0.57 | 0.000 | -0.53 | -0.72 | 0.00 | 7.24 | -0.05 | 0.00 | 0.00 | 0.00 | 0.00 |
| BLACK-HEADED GROSBEAK | W | 20 | | 0.04 | 0.374 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| CASSIN'S FINCH | W | 21 | B | 0.32 | 0.008 | 1.60 | 0.00 | 0.00 | 0.00 | -0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| CLARK'S NUTCRACKER | W | 14 | | 0.66 | 0.032 | -4.03 | -4.01 | 0.00 | 21.57 | 0.15 | 0.00 | 0.00 | 0.00 | 1.82 |
| DOWNY WOODPECKER | W | 11 | | 0.79 | 0.002 | 5.10 | 0.00 | 0.00 | 0.00 | 0.00 | 0.53 | -0.86 | 0.00 | 0.00 |
| DUSKY FLYCATCHER | W | 23 | A | 0.51 | 0.003 | 4.24 | 0.00 | 0.00 | 0.00 | 0.00 | 0.59 | 0.00 | 0.06 | -2.56 |
| EVENING GROSBEAK | W | 15 | | 0.85 | 0.006 | -32.38 | -1.76 | 0.00 | 13.52 | -0.15 | 0.54 | 1.97 | 0.00 | -0.93 |
| GRAY JAY | W | 18 | R | 0.41 | 0.020 | 19.44 | 0.00 | 0.00 | 0.00 | 0.00 | 0.97 | -2.21 | 0.00 | 0.00 |
| HAIRY WOODPECKER | W | 17 | R | 0.19 | 0.080 | -12.05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.89 | 0.00 | 0.00 | 0.00 |
| HAMMOND'S FLYCATCHER | W | 28 | A | 0.19 | 0.072 | -6.95 | 0.00 | 0.00 | 2.10 | 0.00 | 0.44 | 0.00 | 0.00 | 0.00 |
| HERMIT THRUSH | W | 16 | B | 0.44 | 0.067 | 9.93 | -4.37 | 0.00 | 18.94 | 0.00 | 0.00 | 0.00 | -0.29 | 0.00 |
| MOUNTAIN CHICKADEE | W | 25 | R | 0.34 | 0.031 | -4.07 | 0.00 | 0.74 | 2.98 | -0.08 | 0.00 | 0.00 | 0.00 | 0.00 |
| NORTHERN WATERTHRUSH | W | 12 | | 0.76 | 0.002 | -1.72 | 0.00 | 1.09 | 0.00 | 0.00 | 0.00 | 0.00 | -0.26 | 0.00 |
| OREGON JUNCO | W | 35 | B | 0.61 | 0.000 | -24.34 | 0.00 | 0.00 | 4.67 | 0.00 | 0.44 | 0.59 | 0.00 | 1.76 |
| PILEATED WOODPECKER | W | 12 | | 0.13 | 0.255 | -11.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.74 | 0.00 | 0.00 |
| PINE SISKIN | W | 34 | B | 0.56 | 0.000 | 0.22 | -1.61 | 0.91 | 9.43 | -0.06 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED CROSSBILL | W | 20 | R | 0.66 | 0.001 | -7.59 | -2.35 | 0.00 | 17.74 | 0.00 | 0.43 | 0.00 | 0.00 | 0.00 |
| RED-EYED VIREO | W | 14 | | 0.72 | 0.089 | -20.21 | -4.58 | -2.25 | 29.60 | 0.12 | 0.87 | 0.00 | 0.00 | 3.60 |
| RED-NAPED SAPSUCKER | W | 19 | B | 0.20 | 0.056 | -0.11 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.08 | 0.00 |
| RED-BREASTED NUTHATCH | W | 35 | R | 0.48 | 0.000 | -6.52 | 0.00 | 0.00 | 3.78 | 0.00 | 0.30 | 0.00 | 0.08 | 0.00 |
| RUBY-CROWNED KINGLET | W | 27 | | 0.08 | 0.155 | 2.31 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | -0.07 | 0.00 |
| RUFFED GROUSE | W | 10 | | 1.00 | 0.007 | -50.01 | -8.63 | 0.00 | 45.76 | -0.28 | 0.85 | 2.53 | -0.15 | 2.92 |
| SOLITARY VIREO | W | 20 | A | 0.48 | 0.004 | -23.74 | 0.37 | 0.00 | 0.00 | 0.00 | 0.00 | 1.52 | 0.00 | 0.00 |
| STELLER'S JAY | W | 17 | R | 0.30 | 0.082 | -13.14 | 0.00 | 0.00 | 0.00 | -0.13 | 0.00 | 0.97 | 0.00 | 0.00 |
| SWAINSON'S THRUSH | W | 27 | A | 0.48 | 0.000 | -3.80 | 0.00 | 0.00 | 5.57 | 0.00 | 0.00 | 0.00 | 0.08 | 0.00 |
| TOWNSEND'S SOLITAIRE | W | 15 | B | 0.71 | 0.009 | -12.75 | -1.11 | 0.00 | 10.65 | -0.07 | 0.00 | 0.00 | 0.00 | 2.00 |
| TOWNSEND'S WARBLER | W | 21 | | 0.20 | 0.139 | -7.71 | 0.00 | 0.00 | 2.40 | 0.00 | 0.57 | 0.00 | 0.00 | 0.00 |
| VARIED THRUSH | W | 15 | | 0.26 | 0.164 | 1.93 | 0.00 | 0.00 | 0.00 | -0.20 | 0.00 | 0.00 | 0.17 | 0.00 |
| VEERY | W | 20 | A | 0.46 | 0.018 | -25.20 | -3.31 | 0.00 | 14.96 | 0.00 | 0.00 | 2.18 | 0.00 | 0.00 |
| WARBLING VIREO | W | 36 | A | 0.47 | 0.000 | -4.81 | 0.00 | 1.27 | 2.81 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| WESTERN TANAGER | W | 34 | A | 0.67 | 0.000 | -9.38 | 1.00 | 1.45 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| WESTERN WOOD-PEWEE | W | 30 | A | 0.32 | 0.016 | 0.20 | -2.91 | 0.00 | 11.41 | 0.00 | 0.57 | 0.00 | 0.00 | 0.00 |
| WINTER WREN | W | 10 | | 0.95 | 0.051 | 5.80 | 0.00 | -2.23 | 6.90 | -0.16 | 0.98 | -1.28 | 0.27 | 0.00 |
| AMERICAN GOLDFINCH | S | 17 | B | 0.27 | 0.031 | 0.42 | -0.80 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| BREWER'S SPARROW | S | 13 | A | 0.53 | 0.153 | -0.29 | 3.00 | 0.00 | -17.41 | -0.30 | 0.00 | 2.08 | 0.00 | 0.00 |
| COMMON YELLOWTHROAT | S | 22 | A | 0.53 | 0.009 | 9.82 | -5.65 | 0.00 | 26.02 | 0.14 | 0.00 | 0.00 | -0.15 | 0.00 |
| FOX SPARROW | S | 13 | | 0.79 | 0.144 | 91.94 | 7.09 | 7.57 | -39.18 | 0.00 | 1.00 | -5.91 | 0.11 | -11.62 |
| GRAY CATBIRD | S | 17 | B | 0.61 | 0.016 | -21.77 | -0.45 | 0.59 | 0.00 | 0.16 | 0.00 | 1.40 | 0.00 | 0.00 |
| HOUSE WREN | S | 24 | A | 0.61 | 0.000 | -17.11 | -0.90 | 0.00 | 0.00 | -0.13 | 0.00 | 1.62 | 0.00 | 0.00 |
| LAZULI BUNTING | S | 21 | | 0.11 | 0.141 | 1.46 | 0.00 | 0.00 | 0.00 | -0.12 | 0.00 | 0.00 | 0.00 | 0.00 |
| LINCOLN'S SPARROW | S | 16 | | 0.37 | 0.125 | 0.77 | 0.00 | 2.00 | -5.79 | 0.00 | 0.00 | 0.00 | -0.37 | 0.00 |
| MACGILLIVRAY'S WARBLER | S | 34 | A | 0.28 | 0.001 | -1.52 | 0.48 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| SAGE THRUSHER | S | 9 | | 1.00 | 0.023 | -71.23 | 7.76 | 2.46 | -48.34 | 0.00 | 2.51 | 4.61 | -0.38 | -8.25 |
| SONG SPARROW | S | 35 | | 0.51 | 0.000 | 9.29 | 0.00 | 0.61 | 0.00 | 0.12 | 0.00 | 0.00 | 0.00 | -2.51 |
| WHITE CROWNED SPARROW | S | 16 | B | 0.46 | 0.019 | 9.55 | -3.46 | 0.00 | 13.56 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| WILLOW FLYCATCHER | S | 34 | A | 0.13 | 0.033 | 6.98 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | -1.42 |
| WILSON'S WARBLER | S | 13 | A | 0.65 | 0.005 | 11.44 | 0.00 | 0.00 | 0.00 | -0.10 | -0.80 | 0.00 | 0.00 | 0.00 |
| YELLOW WARBLER | S | 36 | A | 0.34 | 0.001 | -14.55 | -0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 1.22 | 0.00 | 0.00 |
| BANK SWALLOW | O | 14 | A | 0.60 | 0.006 | 23.01 | 0.00 | -1.36 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | -4.05 |
| BARN SWALLOW | O | 30 | A | 0.48 | 0.012 | -12.55 | 1.05 | 0.48 | -8.54 | 0.00 | 0.66 | 0.93 | 0.00 | -1.91 |
| BLACK-BILLED MAGPIE | O | 24 | R | 0.27 | 0.039 | 5.55 | -1.66 | 0.00 | 6.63 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| BREWER'S BLACKBIRD | O | 30 | B | 0.45 | 0.001 | -9.60 | 0.00 | 0.00 | -3.38 | -0.08 | 0.00 | 1.06 | 0.00 | 0.00 |
| BROWN-HEADED COWBIRD | O | 35 | B | 0.23 | 0.041 | 7.21 | 0.00 | 0.55 | -2.20 | 0.00 | 0.00 | 0.00 | 0.00 | -1.27 |
| BULLOCK'S ORIOLE | O | 11 | | 0.99 | 0.006 | -74.86 | 10.25 | -3.88 | -39.23 | 0.00 | -0.72 | 6.97 | 0.70 | -7.48 |
| CEDAR WAXWING | O | 16 | | 0.16 | 0.123 | -10.42 | 0.00 | 0.00 | 0.00 | 0.00 | 0.84 | 0.00 | 0.00 | 0.00 |
| CLIFF SWALLOW | O | 27 | A | 0.25 | 0.085 | -10.87 | -0.60 | 0.00 | 0.00 | -0.13 | 0.00 | 1.21 | 0.00 | 0.00 |
| COMMON RAVEN | O | 34 | | 0.26 | 0.029 | -19.17 | 0.16 | 0.00 | 0.00 | 0.00 | 0.54 | 0.88 | 0.00 | 0.00 |
| EASTERN KINGBIRD | O | 16 | A | 0.63 | 0.018 | 13.49 | 0.00 | -1.36 | 0.00 | 0.00 | 1.60 | -1.87 | -0.19 | 0.00 |
| MOUNTAIN BLUEBIRD | O | 20 | B | 0.51 | 0.009 | 6.55 | -1.91 | 0.00 | 6.72 | -0.12 | 0.00 | 0.00 | 0.00 | 0.00 |
| RED-SHAFTED FLICKER | O | 40 | B | 0.22 | 0.011 | 1.83 | -0.87 | 0.00 | 4.80 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| ROCK WREN | O | 14 | | 0.09 | 0.294 | 1.53 | 0.00 | 0.00 | -0.99 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| TREE SWALLOW | O | 27 | B | 0.23 | 0.045 | 3.77 | -2.13 | 0.00 | 11.02 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| VIOLET-GREEN SWALLOW | O | 25 | A | 0.33 | 0.080 | -24.77 | 0.00 | -0.75 | 0.00 | -0.12 | 0.80 | 1.30 | 0.00 | 0.00 |

Table II. Spatial Analysis Regression Table

# APPENDIX III: Dependent And Independent Variable Transformations

Table I.1: Transformations of Temporal Data Set

| | |
|---|---|
| Mean Forest Patch Size, 1990s | Log Normal |
| Mean Forest Patch Size, 1970s | Log Normal |
| Forest Patch Size COV, 1990s | SAS SQRT |
| Forest Patch Size COV, 1970s | SAS -SQRT |
| Forest Edge, 1990s | SAS SQRT |
| Forest Edge, 1970s | SAS -SQRT |
| Percent Forest, 1990s | ARCSIN(SQRT) |
| Percent Forest, 1970s | -ARCSIN(SQRT) |
| Mean Nearest Neighbor, 1990s | Log Normal |
| Mean Nearest Neighbor, 1970s | Log Normal |

Table I.2: Transformations of Spatial Data Set

| Variable | Transformation |
|---|---|
| Mean Forest Patch Size | Log Normal |
| Forest Patch Size COV | SAS SQRT |
| Percent Forest | ARCSIN(SQRT) |
| Forest-Agriculture Edge | Log Normal |
| Forest-Other Edge | Log Normal |
| Forest-Range Edge | Log Normal |
| Forest-Urban Edge | Log Normal |
| Mean Nearest Neighbor | Log Normal |

# APPENDIX IV: Cluster Analysis Results

Table 1. Cluster Analysis — Spatial Data (Cluster Groups = 6, 16, 24, 32)

RAW VALUES SORTED BY NAME

| OBS | NAME | CLUSTER1 | CLUSTER2 | CLUSTER3 | CLUSTEER4 |
|---|---|---|---|---|---|
| 1 | AMERICAN GOLDFINCH | 1 | 1 | 1 | 11 |
| 2 | AUDUBON'S WARBLER | 2 | 2 | 2 | 14 |
| 3 | BANK SWALLOW | 1 | 12 | 20 | 28 |
| 4 | BARN SWALLOW | 7 | 15 | 23 | 31 |
| 5 | BLACK-BILLED MAGPIE | 2 | 2 | 2 | 2 |
| 6 | BREWER'S BLACKBIRD | 1 | 8 | 14 | 22 |
| 7 | BROWN-HEADED COWBIRD | 1 | 8 | 15 | 23 |
| 8 | CASSIN'S FINCH | 1 | 1 | 1 | 1 |
| 9 | CLIFF SWALLOW | 1 | 1 | 3 | 4 |
| 10 | COMMON RAVEN | 1 | 1 | 3 | 3 |
| 11 | COMMON YELLOWTHROAT | 8 | 16 | 24 | 32 |
| 12 | DUSKY FLYCATCHER | 1 | 4 | 5 | 6 |
| 13 | EASTERN KINGBIRD | 1 | 7 | 11 | 19 |
| 14 | GRAY CATBIRD | 1 | 1 | 3 | 4 |
| 15 | GRAY JAY | 1 | 7 | 12 | 20 |
| 16 | HAIRY WOODPECKER | 1 | 1 | 1 | 8 |
| 17 | HAMMOND'S FLYCATCHER | 3 | 3 | 9 | 17 |
| 18 | HERMIT THRUSH | 5 | 10 | 18 | 26 |
| 19 | HOUSE WREN | 1 | 1 | 3 | 4 |
| 20 | MACGILLIVRAY'S WARBLER | 1 | 1 | 1 | 1 |
| 21 | MOUNTAIN BLUEBIRD | 2 | 2 | 2 | 2 |
| 22 | MOUNTAIN CHICKADEE | 3 | 3 | 4 | 5 |
| 23 | OREGON JUNCO | 2 | 6 | 13 | 21 |
| 24 | PINE SISKIN | 4 | 9 | 17 | 25 |
| 25 | RED CROSSBILL | 5 | 11 | 19 | 27 |
| 26 | RED-BREASTED NUTHATCH | 3 | 3 | 10 | 18 |
| 27 | RED-NAPED SAPSUCKER | 1 | 1 | 1 | 1 |
| 28 | RED-SHAFTED FLICKER | 2 | 6 | 7 | 12 |
| 29 | SOLITARY VIREO | 1 | 1 | 3 | 3 |
| 30 | SONG SPARROW | 1 | 4 | 5 | 7 |
| 31 | STELLER'S JAY | 1 | 1 | 3 | 3 |
| 32 | SWAINSON'S THRUSH | 2 | 6 | 7 | 13 |
| 33 | TREE SWALLOW | 4 | 5 | 6 | 9 |
| 34 | VEERY | 6 | 13 | 21 | 29 |
| 35 | VIOLET-GREEN SWALLOW | 1 | 1 | 3 | 15 |
| 36 | WARBLING VIREO | 3 | 3 | 4 | 5 |
| 37 | WESTERN TANAGER | 1 | 1 | 16 | 24 |
| 38 | WESTERN WOOD-PEWEE | 4 | 5 | 6 | 10 |
| 39 | WHITE-CROWNED SPARROW | 6 | 14 | 22 | 30 |
| 40 | WILLOW FLYCATCHER | 1 | 4 | 8 | 16 |
| 41 | YELLOW WARBLER | 1 | 1 | 3 | 3 |

Table 2. Cluster Analysis — Temporal Data (Cluster Groups = 6, 16, 24, 32)

RAW VALUES
SORTED BY NAME

| OBS | NAME | CLUSTER1 | CLUSTER2 | CLUSTER3 | CLUSTER4 |
|---|---|---|---|---|---|
| 1 | AMERICAN GOLDFINCH | 1 | 1 | 11 | 12 |
| 2 | AUDUBON'S WARBLER | 1 | 2 | 6 | 7 |
| 3 | BARN SWALLOW | 1 | 1 | 4 | 5 |
| 4 | BLACK-HEADED GROSBEAK | 1 | 9 | 17 | 18 |
| 5 | BREWER'S SPARROW | 7 | 15 | 23 | 24 |
| 6 | CASSIN'S FINCH | 1 | 1 | 2 | 3 |
| 7 | CLIFF SWALLOW | 4 | 12 | 20 | 21 |
| 8 | DUSKY FLYCATCHER | 3 | 5 | 13 | 14 |
| 9 | GRAY JAY | 8 | 16 | 24 | 25 |
| 10 | HAIRY WOODPECKER | 1 | 10 | 18 | 19 |
| 11 | HERMIT THRUSH | 3 | 11 | 19 | 20 |
| 12 | MOUNTAIN BLUEBIRD | 2 | 3 | 8 | 9 |
| 13 | OREGON JUNCO | 1 | 2 | 7 | 8 |
| 14 | PINE SISKIN | 2 | 7 | 15 | 16 |
| 15 | RED CROSSBILL | 1 | 8 | 16 | 17 |
| 16 | RED-BREASTED NUTHATCH | 3 | 6 | 14 | 15 |
| 17 | RUBY-CROWNED KINGLET | 1 | 4 | 12 | 13 |
| 18 | SOLITARY VIREO | 6 | 14 | 22 | 23 |
| 19 | SONG SPARROW | 1 | 1 | 5 | 6 |
| 20 | TOWNSEND'S SOLITAIRE | 1 | 1 | 1 | 1 |
| 21 | VIOLET-GREEN SWALLOW | 1 | 1 | 1 | 2 |
| 22 | WESTERN TANAGER | 1 | 1 | 3 | 4 |
| 23 | WESTERN WOOD-PEWEE | 2 | 3 | 9 | 10 |
| 24 | WHITE-CROWNED SPARROW | 5 | 13 | 21 | 22 |
| 25 | WILLOW FLYCATCHER | 1 | 1 | 10 | 11 |