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The effects of 55 years of vegetative change on bighorn sheep habitat in the Sun River area of Montana

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The Effects of 55 Years of Vegetative Change on Bighorn Sheep Habitat in the Sun River Area of Montana

By

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The Effects of 55 Years of Vegetative Change on Bighorn Sheep Habitat in the Sun River Area of Montana (95 pp.)

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I analyzed change in vegetation and bighorn sheep habitat between 1937 and 1992 for an area in the Sun River drainage of Montana. I compared an historic land-cover map comprised of seven land-cover types derived from 1937 aerial photographs to a current land-cover map created by classifying 1992 Landsat TM imagery. Bighorn sheep habitat, for both time periods, was delineated using model rules taken from the literature and a GIS.

The greatest changes in vegetation occurred within grassland and high-density conifer types. Grasslands decreased from 26.2% of the landscape in 1937 to 19.2% in 1992, with 30% of the 1937 grassland converting to high-density conifer. Low-density conifer also declined, with 72.5% converting to high-density conifer. The percentage of high-density conifer occupying the landscape doubled, from 20% to 40%, and the percent of the landscape in high-visibility cover types decreased from 77.5% in 1937 to 54.9% in 1992. Fire suppression efforts, effective since the 1930's, are thought to have contributed to conifer encroachment into grasslands and other open land-cover classes.

Bighorn sheep habitat (all habitat components) dropped from 52.4% of the landscape in 1937 to 34.9% in 1992, a decrease of 33.4%. The summer/fall, winter, and lambing area components of bighorn sheep habitat decreased 31.1%, 39.4%, and 37.1%, respectively. The amount of summer/fall range declined from 36.6% to 25.2%, winter range, fell from 12.8% to 7.8%, and lambing range dropped from 3.0% to 1.9% of the landscape between the two time periods.

Transformations in the landscape between 1937 and 1992, as delineated by the habitat model, imply a potentially serious loss of bighorn sheep habitat. In addition to overall habitat loss, the area of predicted bighorn sheep winter range occurring within patches large enough for bighorn sheep to use (>1.5 km^2), also decreased. Although bighorn sheep populations have increased between 1937 and 1992, these results suggest populations could be compromised if habitat becomes a factor limiting their population size.

Dedication

To Akita-san, who sparked in me a curiosity towards the natural world.

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I owe countless thanks to my friends and colleagues who encouraged and helped me throughout the duration of this project. It is an honor to have been in your company. A few people were especially instrumental in helping me complete this project. Roly Redmond, my advisor, first provided me with the tremendous opportunity to conduct this project and then provided guidance, suggestions, and the awesome computing resources of The Wildlife Spatial Analysis Lab. The entire staff of The Wildlife Spatial Analysis Lab was always helpful; however, Melissa Hart (alias ARC guru) deserves a special thank you for her endless patience in helping me become ARC/INFO literate. Melissa and the 'other-mindeds' could be counted on to encourage me when I was most discouraged, and for many brilliant suggestions. Vita Wright deserves a very special thanks for her tireless support, encouragement, and patience. Her reviews of draft versions of this thesis improved the end result considerably . Thanks also to Chris Winne for writing the Markov model program and to Troy Tady for helping with my image processing needs.

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My parents, Lore and Conrad and my brother Oliver were always there, being supportive of all my endevors. Without their encouragement and support this porject would not have occurred.

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INTRODUCTION

Population persistence is driven by a host of biotic and abiotic factors. Habitat alteration is often the cause of a population's, and eventually a species' decline to endangered status and extinction. To counter anthropogenic habitat alteration, 'nature reserves' have been established over the last few decades. Many of these reserves are extremely important for maintaining species that are sensitive to human disturbance. However, the long-term utility of these areas for conserving sensitive species depends, in part, on the retention of natural disturbance regimes as part of the ecological systems. In the Rocky Mountains of North America fire is the predominant agent of natural disturbance (Arno 1980).

In this thesis I present a change detection analysis of the vegetative characteristics and predicted bighorn sheep habitat, between 1937 and 1992, within a portion of the Sun River drainage of Lewis and Clark National Forest (LCNF) in Montana (Fig. 3). It portrays the consequences of removing fire from an area on the East Front of the Rocky Mountains in Montana, and the significance of fire's role in sustaining one of North America's most magnificent, and sensitive, large mammal species, the bighorn sheep. I begin by offering a brief natural history of bighorn sheep and their habitat characteristics in North America.

Historical Sheep Distribution in North America

Wild sheep were generally thought to have arrived in the New World at the beginning of the Wisconsin glacial period, about 70,000 years ago. However, recent genetic and fossil evidence has pushed this date back to 650,000 years ago (Ramey 1993). Wild sheep colonized western North America during an inter-glacial period (Buechner 1960). Two distinct species evolved when ice sheets isolated northern and southern populations (Cowan 1940). Currently *Ovis dalli* (thinhorn sheep) occupy mountainous terrain in Alaska and northern Canada, and *Ovis canadensis* (bighorn sheep) occupy suitable habitat from southern British Columbia to northern Mexico (Cowan 1940, Buechner 1960). Today, wild sheep are one of the few members of the Pleistocene megafauna remaining in North America.

Prior to the settlement of the western United States by non-indigenous people, bighorn sheep were abundant in most mountain ranges and in the desert canyon country of the southwestern United States. Bighorn sheep were rarely observed on the open prairies, but they did occur in association with buttes, small mountain ranges, and along the bluffs above some major rivers such as the Missouri River in Montana (Couey 1950, Buechner 1960).

Beginning in the 1870's, anthropogenic factors caused bighorn sheep populations to decline. Hide and meat hunters extirpated many of the smaller and more accessible herds. According to the superintendent of Yellowstone National Park, "In the spring of 1875 over 2,000 hides of elk and nearly as many of bighorn sheep were taken out of the park" (in Buechner 1960). Where bighorn sheep populations were able to survive in the face of harvest pressure, competition with domestic sheep, as well as diseases carried by domestic stock, eventually resulted in widespread die-offs (Buechner 1960, Stelfox 1971, Goodson 1982). Where there were once possibly over 2 million bighorn sheep, today fewer than 30,000 remain in North America (Hoefs 1985).

Historic reports of bighorn sheep in Montana date back to the Lewis and Clark expedition in 1806. Based on such accounts, and interviews with local residents and hunters, Couey (1950) compiled maps of former and existing bighorn sheep ranges in Montana. Apparently, bighorn sheep were once present in all the larger mountain ranges, most of the smaller, isolated mountain ranges, and along the Missouri River. By 1946 wild sheep were gone from most of the isolated and smaller mountain ranges east of the continental divide and the bluffs along the Missouri River. In addition, the once large metapopulations of bighorn sheep along the continental divide and in other large mountain ranges of Western Montana had been reduced to small discontinuous remnants (Couey 1950). Since then, bighorn sheep have been reintroduced to several of these former ranges (Janson 1976).

Population Structure

A metapopulation consists of a group of geographically separated sub-populations of conspecifics that are interconnected through immigration, emigration and/or recolonization (Lande and Barrowclough 1987). The bighorn sheep metapopulations that currently exist along the Rocky Mountain Front in Montana is composed of some of the last native (not supplemented with transplanted stock) sub-populations (herds) in the United States (Luikart 1992, Hogg pers. comm.). Unfortunately, even in this seemingly pristine landscape, bighorn sheep sub-populations have become fragmented and isolated. A recent genetic study (Luikart 1992) indicated that little or no flow of genetic material is occurring among six regional sub-populations along the Rocky Mountain Front. These genetic data, derived from mitochondrial DNA, further suggest that a single regional metapopulation existed since the pleistocene. If this is correct, the zoogeographic barriers preventing gene flow among extant sub-populations are of such recent origin (Luikart 1992) that outbreeding depression is unlikely to compromise local adaptations if sub-populations are re-connected.

Inbreeding, one consequence of isolation, has been shown to increase neonate mortality, increase susceptibility to disease, and decrease growth rates (Skiba and Schmidt 1982, and Ralls and Ballou 1983). Therefore, inbreeding may predispose small herds to extinction (Berger

1990). A detailed discussion of the consequences of genetic isolation is beyond the scope of this thesis. However, fragmentation of the metapopulation, genetic isolation, and the small size of some sub-populations have the potential to accelerate the loss of genetic diversity (Luikart 1992). This loss can decrease the long term adaptive potential to resist pathogens (Frankel and Soulé 1981), and compromise the long term viability of the bighorn sheep metapopulation along the Rocky Mountain Front. The potential for inbreeding is positively correlated with the actual distance between sub-populations, and inversely related to sub-population size (Gilpin 1987). Inter-herd migrations by rams of 50 km have been reported, and movements of 15 km are common (Cochran and Smith 1983, Festa-Bianchet 1986). Therefore, sub-populations separated by 15 to 50 km have the potential to form a metapopulation, as long as barriers preventing movements are not present (Dunn 1993).

Maintaining the geographical integrity of a metapopulation may be the only way to insure the long-term viability of long-lived and slowly-reproducing animals (K-selected species) such as bighorn sheep. Not only are the metapopulation's spatial structure, and the dynamics of migration important to the maintenance of genetic variability, but these factors may also play a role in a sub-population's ability prevail in the face of demographic and environmental stochasticity (Shaffer 1987). Fragmentation and isolation can cause demographic constraints that result in the extinction of small sub-populations (Frankel and Soulé 1981). Berger (1990) in his empirical analysis of 122 bighorn sheep herds found that 100% of the populations that contained fewer than 50 individuals went extinct within 50 years, and herds of 50-100 animals were likely to persist for 70 years. In contrast, small herds of less than 50 sheep that were not isolated were less likely to suffer from such demographic effects.

Currently, little is known about the spatial structure of the metapopulation of bighorn sheep along the Rocky Mountain Front in Montana. However, the long term-future of this wildlife resource may depend on our knowledge of the metapopulation dynamics and our ability to maintain adequate gene flow among sub-populations. An understanding of the vegetative dynamics and the landscape attributes associated with bighorn habitat is an important first step in analyzing landscape-level, metapopulation dynamics.

Importance of Fire

Vertebrate distributions are often closely linked with the distribution and abundance of particular plant communities (Morrison et al. 1992). The spatial distribution, pattern, and extent of plant communities influence the metapopulation dynamics of the species that rely on them. As the primary force of disturbance in the Rocky Mountains in Montana, fire has historically played a fundamental role in shaping and maintaining many plant communities (Arno 1980). Today, because fire suppression efforts begun in the early 20^{th} century continue to be remarkably effective, timber harvest has replaced fire as the dominant force in structuring the landscape on managed forest lands. The subsequent increase in mean fire-free interval (MFI) has caused profound effects on fire-prone forest landscapes in the Rocky Mountains of North America, and throughout the world (Trabaud 1987, Sprugel 1991).

Subalpine grasslands on the East Front of the Rockies comprise vital winter range for many ungulate species, including bighorn sheep. Before fire suppression policies became so successful, most of these grasslands came about or were maintained by wildfires (Stelfox 1971, Gruell 1983, Arno and Gruell 1986). As a result of seven decades of effective fire suppression, these seral grasslands are being replaced by Douglas-fir forests, and consequently the amount of ungulate habitat is declining (Cowan 1946, Pfeiffer 1948, Wishart 1958, 1978, Flook 1964, Stelfox 1971,1976, Elliot 1978, Gruell 1983, Arno and Gruell 1986, Wakelyn 1987). Furthermore, many studies show that early stages of forest succession support a greater abundance of ungulates than mature forests (Elliott 1978, Bentz and Woodward 1988, Smith 1988, Arnett 1990), but old-growth forests often support a higher diversity of birds and other species (Harris 1984). Considering bighorn sheep, the encroachment of dense coniferous forest cover not only limits space and light available for desirable forage plants like grasses and forbs, but it also creates visual barriers that reduce the quality of sheep habitat (Risenhoover and Bailey 1985).

Certain types of fire improve ungulate habitat by increasing forage production and nutrient content for up to 20 years post fire (DeWitt and Derby 1955, Dills 1970, Bentz 1981, Hobbs and Spowart 1984, Seip and Bunnell 1985, Cannon et al. 1987, Bentz and Woodward 1988,Smith 1988, Arnett 1990). Furthermore, the numerous large wildfires that burned around the turn of the century contributed to increases in bighorn sheep numbers in the Canadian Rockies, and bighorn sheep ranges originally occurred in areas where fires were frequent (Stelfox 1971). Utilization of bluebunch wheatgrass (*Agropyron spicatum*) by bighorn sheep was greater on a burned site than on unburned site up to four years after the prescribed burning treatment (Peek et al. 1979). Subalpine grasslands that were recently burned supported five times the density and population of Stone sheep (*Ovis dalli stonii*), and had a 75% greater lamb production than herds on unburned ranges (Elliott 1978). Riggs and Peek (1980) found 21% of their bighorn sheep locations in areas that had burned recently. Similarly, Bentz and Woodward (1988) documented a higher density of bighorn sheep fecal pellets on recently burned areas than in nearby unburned sites. During fall, winter, and spring, recent burns were shown to be used in greater proportion than their availability (Arnett 1990). Burned habitat was also shown to be a significant factor in discriminating between used and available habitats for a reintroduced bighorn sheep herd in the Encampment River Canyon in Wyoming (Arnett 1990). Of course, the value of any burned site for bighorn sheep depends on the presence of other important habitat components such as escape terrain.

Habitat Structure and Function

The mountainous terrain in which wild sheep reside around the world is typically open, allowing them to see for long distances. One reason bighorn sheep continue to survive is their success at avoiding predators. Bighorn sheep can detect predators at a great distance due to their excellent eyesight and the distant vistas their open habitat affords them. Bighorn sheep escape predators by fleeing up seemingly vertical cliffs, negotiating terrain impassible to their enemies (Geist 1971). Thus, visibility is an important component of bighorn sheep habitat (Geist 1971, Risenhoover and Bailey 1985, Smith and Flinders 1991b).

The distribution of bighorn sheep herds may also dictated by the amount and arrangement of escape terrain. The distance sheep are willing to travel from escape terrain has significant effects on the total amount of available habitat (Geist 1971, Shannon et al. 1975, Wakelyn 1987). Although the descriptions of escape terrain vary among researchers, it generally consists of continuous steep slopes ranging from 28° to 70°, interspersed with rocky outcrops and/or cliffs greater than 15 m in height and greater than

1.6 ha in size (Geist 1971, Tilton 1977, McCollough 1982, Wakelyn 1987, Arnett 1990). Bighorn sheep minimize their predation risks by remaining close to escape terrain. Risenhoover and Bailey (1985) observed an increase in group size and a decrease in foraging efficiency as sheep's distance from escape terrain increased. Wakelyn (1987) reported, the distance bighorn sheep are willing to wander from escape terrain ranges from 0 to 320 m depending on the study area and season. During lambing, ewes often remain in especially rugged portions of escape terrain. In fact, the availability of steep, rugged sites for lambing can be a limiting factor on lamb survival and subsequent recruitment (Geist 1971, Hogg pers. comm.).

Geist (1971) has observed bighorn rams traveling long distances (>32 km.) to mate with ewes that were not part of their own herds. These rams went out of their way to travel through open terrain, and only occasionally crossed forested areas. In the northern Rocky Mountains, bighorn sheep also use open habitat; however, in many places the land between preferred habitat is currently forested, making it less attractive for sheep to cross when traveling between herds (Geist 1971). Some of these inter-herd corridors may have only recently become forested due to the effects of fire suppression. Gruell (1983) and Arno and Gruell (1986) documented that mid-elevation grasslands along the Rocky Mountain Front are being encroached by Douglas-fir. This vegetative community is important bighorn sheep winter range (Geist 1971, Stelfox 1971).

In Colorado, Wakelyn (1987) showed that encroachment of tall shrubs and forest resulted in fragmentation of bighorn sheep populations and reduced the amount of suitable escape terrain. Risenhoover (1981) found shrub encroachment, due to fire suppression, has caused a 75% decline in the habitat types with high-visibility. This loss of preferred habitat appears to cause sheep to crowd into areas where visibility has not been compromised (Wakelyn 1987). Canopy closure has contributed to the abandonment of at least 17 historic bighorn ranges in Colorado along the East Front of the Rockies (Wakelyn 1987). This recent and continuing closure of the forest canopy may further compromise bighorn sheep habitat and limit their ability to migrate between seasonal ranges and between different subpopulations (Goodson 1980, Wakelyn 1987 and Bailey 1992). The issue of losing habitat and migratory potential due to canopy closure has not been investigated for the Rocky Mountain Front in Montana. However, Gruell (1983) gives vivid pictorial evidence for widespread encroachment of conifers into grasslands throughout the northern Rockies (Figs. 1, 2).

The number of sheep present on winter ranges often exceeds the number present in summering areas (Geist 1971, Shannon et al. 1975), suggesting that members of different herds congregate and intermingle on the winter range. Maintaining migratory corridors to and from winter ranges is essential because herds cannot be nutritionally supported unless suitable habitat for all seasons is accessible. Loss of altitudinal migration between seasonal ranges may be one of the causes leading to the decline of many bighorn herds in Colorado (Shannon et al. 1975, and Goodson 1980). Goodson (1980) suggests using prescribed burning to create pathways of suitable habitat between low-elevation winter range and high-elevation summer range to improve bighorn habitat along the East Front of the Rocky Mountains in Colorado.

Quantitative trends in forest canopy closure, succession and forest encroachment into grasslands can be analyzed from a landscape perspective using satellite imagery in conjunction with historical vegetation data. Projecting such trends into the future may be useful for identifying areas of bighorn habitat that are susceptible to degradation due to anthropogenic changes in the fire regime. Areas of degraded but historically suitable habitatmay also be discernable by such means. Managers who may be responsible for using prescribed fire to improve wildlife habitat can use this information to target areas for treatment with prescribed fire.

Bighorn Sheep Habitat Models

With the advent of geographical information systems (GIS), and the increasing availability of digital geographic data, the creation of spatially explicit, multi-scale, wildlife habitat models has become an attractive tool available to many biologists and managers. Many such models have been created at a variety of scales (Stoms 1994, Hart 1994). It is essential that the limitations, and assumptions of these models are recognized and fully understood prior to using them to drive, management actions.

Smith et al. (1991b, 1992) created a GIS habitat model, and applied it in Zion National Park, to evaluate the park's potential to support a minimum viable population of bighorn sheep. Their biggest obstacle, the lack of digital elevation model (DEM) data, did not allow them to fully evaluate topography which is an essential feature of bighorn sheep habitat. Dunn (1993) used a GIS based bighorn sheep habitat model to evaluate areas for potential reintroduction. This model did not identify habitat. Instead it ranked the suitability of 13 pre-delineated study areas relative to each other and estimated the potential population size that each area could support. Sites that were modeled as being capable of supporting a MVP of 125 sheep, or were close to sites that could, were further considered as transplant sites. To date these are the only published GIS-based bighorn sheep habitat models.

I applied a modification of an existing bighorn sheep habitat model (Smith and Flinders 1991) to vegetation data derived for 1937 and 1992 to conduct a change detection analysis specifically within the context of bighorn sheep habitat. This study will set the stage for future analyses on the effects of vegetative change on bighorn sheep distribution and movement patterns on a regional scale (e.g. the entire Rocky Mountain Front from central Montana to the northern Canadian Rockies). I also offer management recommendations to increase the availability of bighorn sheep habitat and to improve the connectivity between sub-populations of bighorn sheep along the Rocky Mountain Front. This information could lead to an increase in the long term viability of the Rocky Mountain Front metapopulation of bighorn sheep.

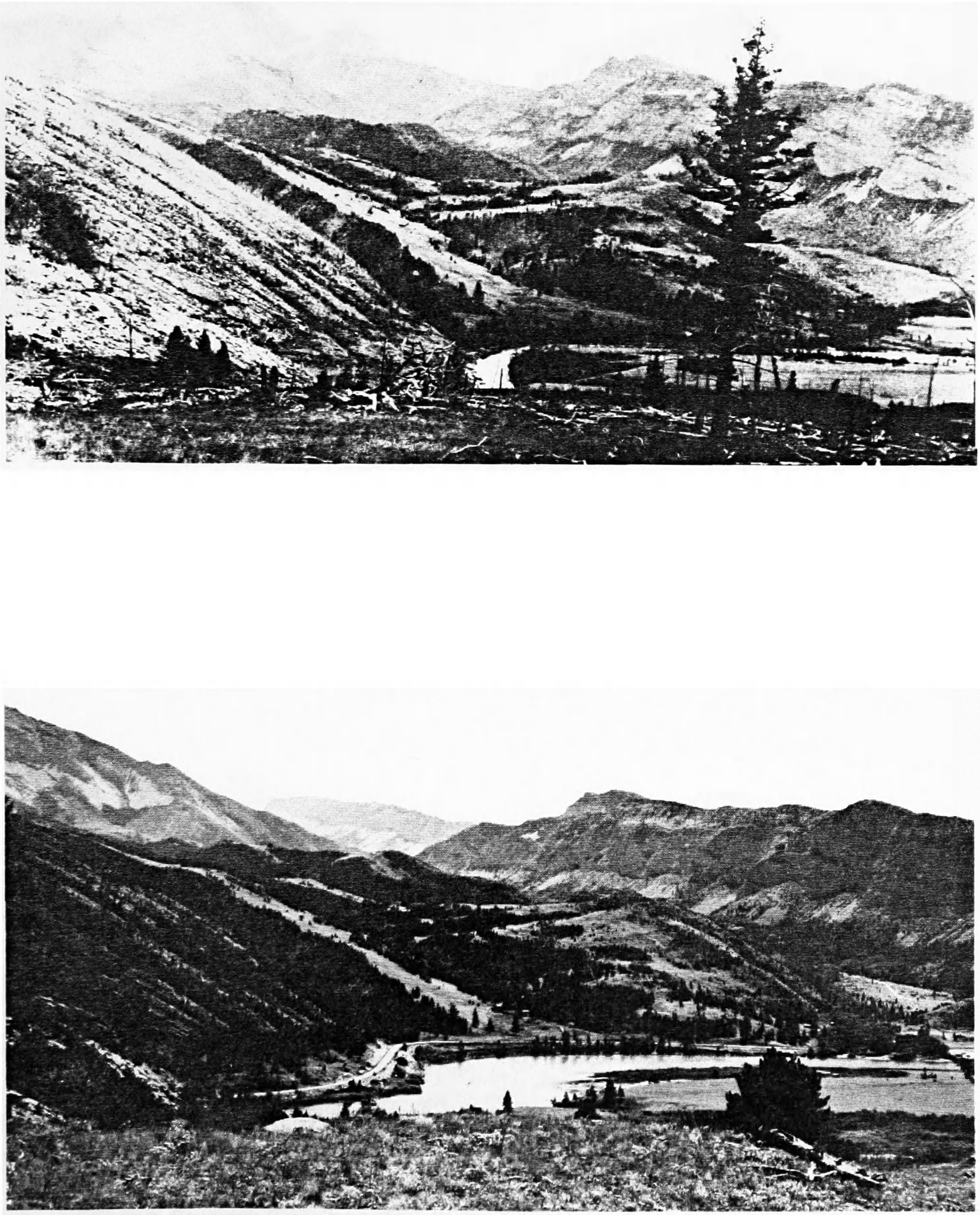

Figure 1. Photographic comparison showing conifer encroachment. Looking southwest across the Sun River towards Home Gulch. Top photograph July 1899 by H.B. Ayres, bottom photo September 1981 by R.F. Wall. From Gruell 1983.

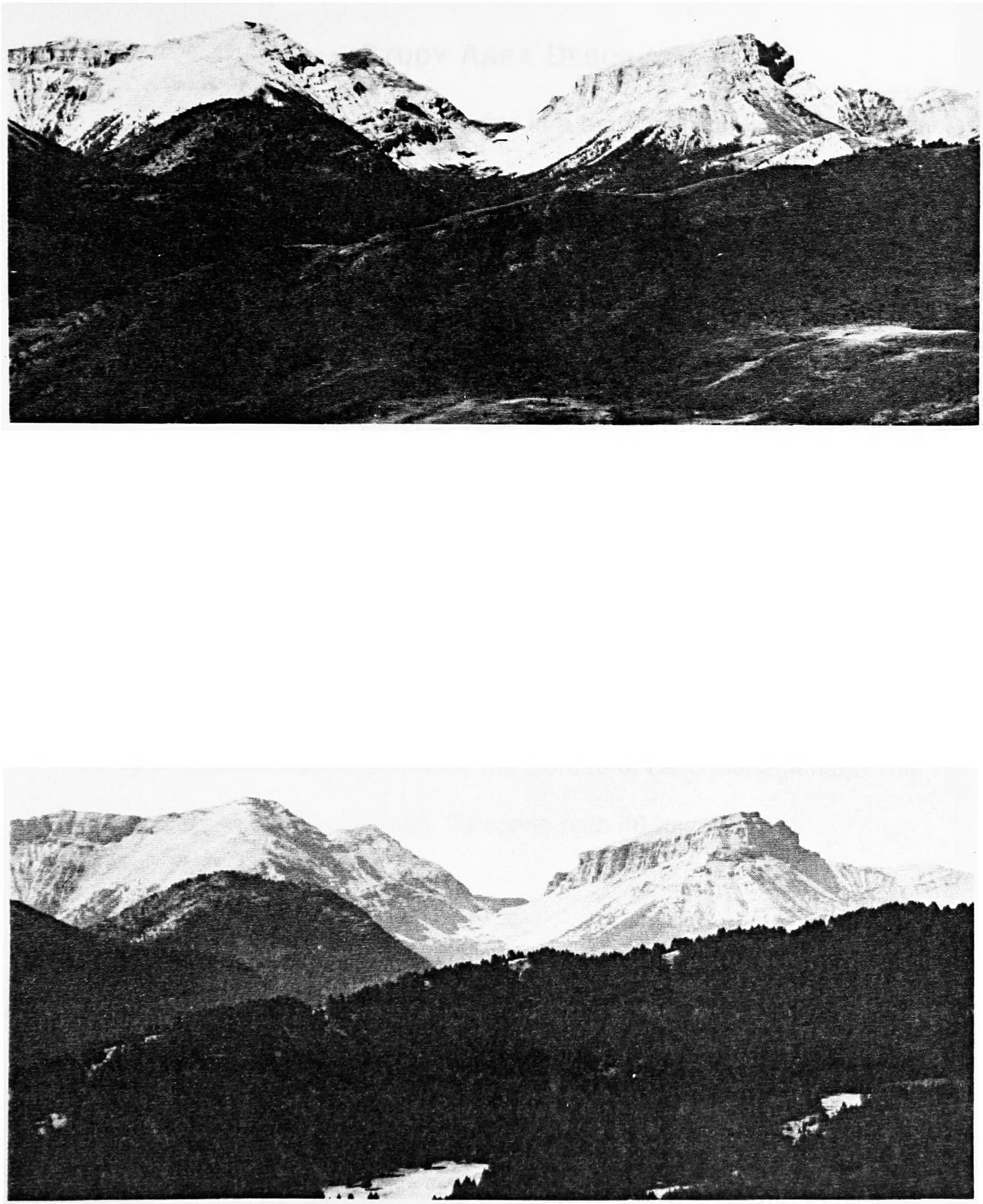

Figure 2. Sawtooth Ridge taken from the foothills east of Castle Reef. Fire-scarred trees indicate wildfire had a significant influence on the landscape prior to settlement. Top photo 1900 by C. Walcott, bottom photo 1981 by G. Gruell. From Gruell 1983.

Study Area Description

The 422.1 km^2 study area lies in a topographically and biologically diverse transition zone between the backbone of the Rocky Mountains, and the Great Plains. On the west side, the study area is bounded by the North and South Fork of the Sun River; on the east side the study area extends 1 km beyond the Lewis and Clark NF boundary. The north and south boundaries are irregular, but generally follow the 47° 45' 00" and 47° 30' 00" parallels (Fig. 3). The center of the study area lies about 100 km south of Glacier National Park and 40 km west of Augusta, Montana . Most of the study area is public land, administered by Lewis and Clark NF. The western portion of the area lies within the Bob Marshall Wilderness Area. A small parcel of land on the eastern edge of the study area is privately owned, and another parcel is administered by the Bureau of Land Management. The study area lies within Landsat TM scene path 40-row 27.

Climate

Climatic conditions vary throughout the area due to rugged topography. Average annual precipitation at the Gibson Dam weather station, located in the center of the study area, is 17.5 inches. Mean temperatures vary from -6.1° C in January to 17.8° C in July. Portions of the area are usually snow-covered from November to April.

Weather patterns generally move in from the west, creating strong down-slope winds. These high winds are a dominant meteorological force in the area. During the winter, these warming “chinook” winds often strip snow off the western and southern aspects of the ridges. This alliance of meteorological and topographic conditions have therefore resulted in some of the best ungulate winter range in Montana. Even during winters of heavy and persistent snowfall, snow-free areas containing sustaining quantities of forage remain available to bighorn sheep and other ungulates (Couey 1950).

Geology

Much of the East Front of the Rocky Mountains in Montana is comprised of parallel ridges and peaks running north and south. These ridges (often called reefs) characteristically have steep cliffs on the east faces and more moderate west facing slopes. Canyons between the ridges ascend to high passes that connect the parallel ridges. Bighorn sheep migrate between their seasonal ranges over these passes (Erickson 1972). These ridges and canyons were formed when Paleozoic limestones and shales (248-590 million year old), were forced up and over younger Mesozoic sediments (65-248 million year old) by the Lewis overthrust, a cenozoic orogeny that occurred 65 million year ago (Deiss 1943). The numerus fossils (corals, and brachipods) exposed on the ridge tops are testimony to the marine origin of these limestones. Subsequent Pleistocene glaciations modified the formations and produced the current diverse topography.

Fire History

Much of the land that currently comprises the Lewis and Clark, and Flathead National Forests was first surveyed and systematically mapped nearly 100 years ago (Ayres 1900). The entire Sun River study area is labeled "old burn" on Ayers' (1900) map; but this may be misleading. Widespread fires occurred in the Northern Rockies in 1889 (Arno and Gruell 1986), and it is likely that wildfires also burned in Sun River area that summer. However, because the effects of wildfire are usually patchily distributed across forest landscapes (Knight and Wallace 1989), it is unlikely that the entire Sun River area had burned just prior to Ayres' visit. More likely, Ayres' map is a crude representation of what was observed in a portion of the area. The most recent significant wildfires recorded in the Sun River area occurred in 1910 and 1919. Since then two major fires occurred adjacent to the study area in 1988. Evidence of recent point ignitions are visible on some of the reef tops in the study area, but these fires each only burned one or two trees.

In 1885, a Great Falls Tribune report (in Picton and Picton 1975) estimated that 10% of the forest in the Sun River area burned every year. Picton and Picton (1975) suggest this is an overestimate, and prefer a 2-3% per year estimate. Arno and Gruell (1986) conducted a fire history analysis using tree ring data from an area south of the Sun River, and found fire intervals to vary between 5 and 40 years (mean = 26 years).

The intensive grazing of domestic livestock, that occurred well into the 20th Century, and continues at a lower level today, decreased the fire frequency in the study area by removing most of the fine fuel (grasses). In addition, since the 1920's, fire suppression efforts have successfully kept major wildfires from occurring within the study area.

Vegetation

Disturbances such as fire, avalanches, and wind-throw occur at a variety of spatial and temporal scales. Long-term climatic cycles affect these disturbance regimes. The vegetative cover of any landscape therefore depends on when, in relation to the disturbance regimes, it is investigated.

Overall, the low elevation canyons between the north and south trending ridges are characterized by shortgrass prairie interspersed with stands of Douglas-fir (*Pseudotsuga menziesii*), snowberry (*Symphoricarpos albus*), and occasional aspen (*Populus tremuloides*) groves. Willows (*Salix spp.*) and cottonwoods (*Populus trichocarpa*) also occur in scattered stands adjacent to the permanent streams that run along the canyon bottoms. The lower slopes above the canyons are often grassy with dense stands of Douglas-fir occurring on northern aspects. The upper slopes are a mosaic of sparse patches of grass, limber pine (*Pinus flexilis*), barren cliffs, scree, and talus slopes. Ridgetops are a mixture of barren areas, patches of grasses and krummholz limber pine and Douglas-fir. Higher elevation mountain sides are often densely stocked with Douglas-fir, lodgepole pine (*Pinus contorta*), and Engelmann spruce (*Picea engelmanni*). Standing burnt timber, evidence of past fires, occurs in several places in the study area. In a few of these areas, trees have not regenerated. These areas are generally small (< 2 ha), with standing burnt trees widely, and sparsely distributed. Consequently, old burns were not discernable on Landsat TM imagery, or on historic (black and white) aerial photographs. Due to these limitations 'old burns' were not a unique vegetation class in this study.

Human History

Picton and Picton (1975) provide a synopsis of the human history of the region. Their material, which comes largely from reports in the Great Falls Tribune, is summarized below.

Prehistoric archeological sites (some of which are now under the waters of Gibson Reservoir) and petroglyphs found in the Sun River Canyon suggest that aboriginal Americans have occupied the Sun River area for thousands of years. Early historical records indicate the area was used by the Flathead-Salish-Kutenai until sometime during the 18th Century when the area was captured by the Blackfeet. Barrett and Arno (1982) present evidence that both accidental and intentional use of fire by Native Americans influenced the vegetation of western Montana. The mean number of years between natural lightning ignitions may have been cut in half by Native American ignited fires (Barrett and Arno 1982).

Meriwether Lewis visited the mouth of the Sun River area in 1805 (near the present location of Great Falls). The first white explorers entered the upper Sun River area in 1854, prospectors followed a few years later and left without discovering gold. The first cattle ranches were established in 1861, and by 1868 an estimated 3,000 head of cattle were grazed in the area. Until their extirpation in 1884, American bison (*Bison bison*) were abundant in the area and occasionally interfered with ranching operations. Native Americans also made life difficult for the early white settlers. However, with the decline of the bison, army raids on Native camps, and a smallpox epidemic among the Blackfoot Indians, white settlers took control over the region in the 1880's.

As the human population of the region increased, the demand for natural resources rose. Lumber required for building the growing towns of Augusta and Choteau in the 1880's came from sawmills located in the Sun River Canyon, 25 km to 35 km to the west. An estimated 100,000 railroad ties were produced to supply the expanding rails that reached Great Falls from the east in 1887. During this era, wildlife populations were sharply reduced by subsistence and market hunting activities. Although, the demand for wood products declined after the railroad was pushed through, the demand for grazing resources for cattle, horses and domestic sheep increased. By 1913, 6,500 head of cattle and 5,500 domestic sheep shared National Forest lands with the native ungulates. Range conditions became “severely overgrazed” by 1925. The resultant lowering of the quantity and quality of forage available to bighorn sheep, and their exposure to domestic sheep precipitated an outbreak of lungworm (*Protostrongylus spp.*) and the associated pneumonia (*Pasteurellosis spp.*) during the winter of 1924-25. Although bighorn sheep hunting had been restricted prior to this epidemic, the die off resulted in this herd gaining management attention from the U.S. Forest Service for the first time.

Hunting bighorn sheep on the East Front of the Rockies was illegal between 1912 and 1952. Beginning in 1953, permits to hunt three-quarter curl rams were issued, resulting in the harvest of between 12 and 52 rams per year. Due to increase in the sheep population and high demands for hunting permits, harvest limits and restrictions on bighorn have been gradually liberalized since 1952 (Erickson 1972, Quentin Kujala pers. comm.). Currently there are no minimum curl restrictions and ewe permits are also issued. Between 1980 and 1984 an average of 42 rams and 39 ewes were harvested in the general area (Montana Fish Wildlife and Parks, unpublished data). It is likely that harvest continues to occur more frequently in the most accessible portions of the winter range (Erickson 1972).

The Sun River Bighorn Sheep Herd

The Sun River bighorn sheep herd is part of a series of bighorn sheep herds that inhabit the Rocky Mountain Front in Montana and may form a single metapopulation (Fig. 4). Although reliable population estimates are not available prior to 1941, the bighorn sheep population declined by an estimated 70% after the winter of 1924-25 due to a pneumonia-lungworm epidemic (Couey 1950). Subsequent die offs occurred in 1927, and in 1936. By 1941, the population in the study area had declined to under 200 individuals (Couey 1950). In 1965 an estimated 265 sheep inhabited the study area (Schallenberger 1966). Possibly due to the removal of domestic sheep from the area, coupled with conservative harvest regulations, the bighorn population grew to 437 animals by 1974 (Andryk 1884). Since 1975, sheep from Sun River have been used to restock nearby areas where wild sheep had been extirpated (Hogg pers. comm.). The herd increased until 1983 when another pneumonia-lungworm complex epidemic ran through the population. Transplanted herds outside of the study area were hit the hardest. The band wintering in the Castle Reef portion of the study area declined by 17%. In 1994 the bighorn population within the study area was estimated to be at least 600 animals (Montana Fish Wildlife and Parks, unpublished data).

METHODS

MAPPING HISTORIC LAND-COVER

Lewis and Clark National Forest inventoried and mapped the land-cover types in most of the Sun River drainage during the early 1930's. Led by R.F. Cooney, these efforts resulted in the production of a 1934 grazing resources map. I digitized the land-cover polygons from an original copy of this map into ARC/INFO and used the map's extent to delineate the study area boundary. Due to cartographic inaccuracies in the 1903 U.S. Geological Service (USGS) Saypo quadrangle, which was used as the base for the grazing resources map, it was impossible to geographically rectify it with the current USGS maps of the area. These cartographic inaccuracies rendered the geographic component of these data unusable for the final analysis. However, the land-cover types defined in the 1934 grazing resources maps were used for my analysis. I also used this historic map to assist with aerial photo interpretation for portions of the study area.

Lewis and Clark NF provided me with a set of black and white, 1:20,000 scale aerial photos taken in 1937 for the entire study area. To my knowledge these are the first set of aerial photographs ever taken for this area. A few of these photographs already had polygons delineated by an unidentified photo interpreter and classified to the same cover types used on the 1934 grazing resources map. I continued this process of delineating land-cover polygons by aerial photo interpretation for the majority of the study area, and classified them to the six cover types described below. I manually transcribed these polygons to orthophotograph quadrangles and digitized them into ARC/INFO, using a minimum mapping unit (MMU) of 0.45 hectares (1 acre). This MMU size corresponds to five 30 m^2 pixels on Landsat TM imagery. In a few cases, vegetation polygons smaller than 1 acre were delineated on the aerial photos. These polygons were later dissolved into the background polygons in which they occurred.

Land-cover Classes

For the purpose of this study, land-cover was lumped into the following six classes:

Grasslands. This class is characterized by native grasses such as Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Agropyron spicatum*), rough fescue (*Festuca scabrella)*, and sedges (Carex spp.). The most abundant forbs included yarrow (*Achillea millefolium*), *Astragalus miser,* and *Ranunculus acris*. Common shrubs are fringed sagewort (*Artemisia fridgida*) and snowberry (*Symphoricarpos albus*), which occasionally occurred in small but dense patches within a grassland mosaic. Total forest cover is less than 15%.

Rocky Reef. Moderate to steep, wind blown, sparsely vegetated, slopes with greater than 50% rock, talus, and/or scree cover characterize this class. Generally, vegetation covers between 20% and 39% of the ground. The most abundant plant is the low-lying shrub kinnikinnik (*Arctostaphylos uva-ursi*). Other shrubs include wild rose (*Rosa spp.*), snowberry, willow, buffaloberry (*Sheperdia canadensis*) and sage *(Artemisia ludoviciana)*. Idaho fescue, *Poa spp.*, sedges, yarrow, goldenrod (*Solidago spp.*), stonecrop (*Sedum spp.*), and *Astragalus miser,* are the most common grasses and forbs in this class. A typical reef has a zone of shrubs at the base above which occur scree, talus slopes, and cliffs. Scattered limber pines may also be present. A few old burns that have not regenerated fall into this class. During winter this class is often cleared of snow by the prevailing winds. Couey (1950) also described a similar land-cover class bearing the same name.

Low Density Conifer. Open conifer stands with a canopy cover between 15% and 39% comprise this class. Pinegrass (*Calamagrostis rubescens*) generally dominates the open understory of these stands. Tree species include pure and mixed stands of Douglas-fir, lodgepole pine, limber pine and Engelmann spruce (*Picea engelmanni*). Open stands of mixed hardwood and coniferous tree species also maybe included in this class.

High Density Conifer. Closed conifer stands with a canopy cover greater than 40%. Tree species include pure and mixed stands of Douglas-fir, lodgepole pine, and Engelmann spruce. Closed stands of mixed hardwood / coniferous tree species are also included in this class.

Aspen. Small stands of quaking aspen comprising least 66% of the canopy cover occur in low-lying moist patches throughout the study area. Understory plants include grasses, and sedges. Lodgepole pine also occurs in some of these stands.

Barren. Exposed rock, scree and/or talus slopes with less than 15% vegetative cover comprise this class.

MAPPING CURRENT LAND-COVER

A 1992 land-cover map was developed using Landsat TM data from scene P40/R27 acquired July 1992, following the methods of Ma et al. (in review). This two-stage digital process (Ma et al. in review) first employed a digital, unsupervised classification to group pixels into spectral classes based on the spectral similarity of bands 3, 4, and 5. The second stage used a supervised classification process to label polygons according to land-cover type based on their similarity to one of the known spectral signatures of land-cover types obtained from ground truth data.

Unsupervised Classification

A Landsat TM image is composed of millions of 30 m^2 pixels (band 6 has a 120 m^2 pixel size), each containing seven bands of reflectance data. Reflectance values for each band range from 0 to 255. This allows for 256^7, over 72 quadrillion, possible pixel types when all 7 bands are considered. The unsupervised classification makes these data manageable by classifying pixels into groups based on their spectral similarity within the three bands most useful for resolving ground cover (Horler and Ahern 1986) and that have the least amount of spectral overlap among vegetation types (Ma and Olson 1989). These three bands are also commonly used for displaying false color composites of Landsat TM imagery (Fig. 3). In these visual depictions of Landsat TM data, band 3 (visible red) is assigned to blue, band 4 (near infrared) to red, and band 5 (mid-infrared) to green. The unsupervised process also uses this color model. Therefore, the results of this digital classification simulate manually digitized land-cover patterns and can be visually compared to the false-color composite.

The classification algorithm (Ma et al. in review) begins by creating a color pallet, which is a set of reference pixels that is subsequently used to classify all remaining pixels in the image; both steps are based on Euclidean distance in three dimensional color space. Pixels that are not within the pre-defined Euclidean distance of any of the reference pixels were flagged and added to the color pallet, as reference pixels, for subsequent runs of the unsupervised classification process. For this study the process created 76 spectral classes for Landsat scene P40/R27, which was still unmanageable for field sampling. Therefore, the unsupervised classifications was regrouped. That is, reference pixels in the color pallet were combined, again, based on a new user determined Euclidian distance. The regrouping was adjusted until 33 spectral classes were delineated.

Contiguous groups of pixels belonging to the same spectral class were merged to eliminate groups (regions) smaller than 5 pixels (0.49 ha). I selected this Minimum Mapping Unit (MMU) to match the MMU of the 1937 cover type layer derived from aerial photo interpretation. The merging process (Ford et al. 1993, Guo 1993, Ma 1995) requires as input a similarity matrix for the spectral classes, based on the raw TM data associated with each spectral class. The program then identifies regions that are smaller than the 5 pixel MMU and identified which of the surrounding regions to merge it with, based on the similarity between the neighbor and the region being absorbed.

Collecting Training Site Data

Obtaining botanically and geographically accurate training data (ground truth data) is a critical step in classifying remotely sensed imagery, because the final land-cover map can only be as accurate as the data used to produce it. During the 1994 and 1995 field seasons, U.S. Forest Service (USFS) field crews collected ground-cover data at more than one-thousand plots within the boundary of the Landsat TM scene (Path 40 - row 27 product-93105015-01) used in this study. These plots were located within a sample of the spectral polygons delineated during the unsupervised stage of the classification process. The extent of a Landsat TM scene contains almost 240 7.5' quadrangles; each quad may contain more than a thousand spectral polygons. A representative sample of each spectral polygon type was selected by a computerized algorithm and sampled. The quadrangles to sample were selected based on: 1) the presence of rare spectral classes, 2) a high diversity of spectral classes within a diversity of ELUs 3) the presence of common spectral classes that were under-represented in the rarity and diversity quadrangles, 4) the presence of unique ecological areas based on Forest Service staff knowledge of their districts. In 1995, additional plots were selected based on vegetation classes that were under-represented in the 1994 training data.

The specific location of the plots within a spectral polygon were determined by the field crews, based on locating a representative, homogenous area (micro-sites that were not representative of the polygon were avoided) at least 100 m from the polygon's edge. Plots were 1/10 acre, often circular with a radius of 11.3 m (37 ft). A global positioning system (GPS) provided exact positions for plot centers when terrain and canopy-cover conditions permitted its use. The GPS points were later differentially corrected, using base-station GPS data available at the USFS Region 1 Office in Missoula, with PFINDER software (Trimble Navigation). Data recorded for each plot included numerous fields, and followed the methodology outlined in the ECODATA handbook (U.S.Department of Agriculture: Forest Service 1992). However, the only data that I used in the supervised classification process were the land-cover classification codes shown in Appendix A (U.S.Department of Agriculture: Forest Service 1995), and the canopy closure class.

To improve the classification for my study area and to have enough points to test its accuracy, I collected 140 additional training-site points from within the Sun River Study area. These points were chosen to represent the majority of spectral polygon classes that occurred within the study area with some consideration for accessibility. I recorded dominant ground cover, and canopy closure classes for all plots (see Appendix A for a list). In addition, for 25 plots, visibility was determined using a 1 m^2 target placed 14 m away in four directions. If an average of 80% or more of the target was visible, the point was classified as having high-visibility. Otherwise it was classified as low-visibility (Smith and Flinders 1992). For the remaining plots, visibility was estimated. Whenever possible, I later determined differentially corrected GPS positions for these additional training points.

Supervised Classification

The unsupervised classification was converted into an ARC/INFO grid layer. Attributes including mean TM values for all seven bands, a normalized difference vegetation index modified for Landsat TM data (MNDVI), elevation, slope, and aspect were added to the GIS data base for each region. MNDVI was calculated according to the following formula from Nemani et al. (1993):

$$\mathrm{MNDVI} = \frac{\mathrm{TM4} - \mathrm{TM3}}{\mathrm{TM4} + \mathrm{TM3} + 1} * \left\{ \frac{256}{\mathrm{TM5} + 1} \right\} * 100$$

Mean TM values were obtained from the raw imagery and averaged for the regions delineated by the unsupervised process. Slope, aspect, and elevation were calculated from a digital elevation model (DEM) based on 7.5' USGS data using ARC/INFO. Slope and elevation were averaged for each region. For aspect, a majority value for the region was used. These region attributes represent summary statistics for all the pixels comprising the region. The regional TM values, when displayed as a multispectral image, are considerably smoother than the raw image, and are thus more practical as a basis for a supervised classification.

Training data points were overlaid on this grid, and the region's attributes for TM values and elevation were extracted for each training point. Thus, for every point in the training data set, a vegetation class code (independent variable), seven TM bands, and elevation variables (dependent variables) were present. I did not use the slope and aspect variables in this classification. Box plots of the 'spectral signature' for each vegetation class were constructed (Fig. 5) using DATA DESK (Ver. 4.2, Velleman 1994). To reduce classification errors, outliers identified in the box plots were removed from the training data set of each vegetation class.

The regions (also called "data to be trained") were labeled Unclassified regions were assigned to a vegetation class based on the single point within the training data to which it was closest in Euclidean distance for 8 variables (seven TM bands plus elevation) using the Nearest Member of Group algorithm (Ma et al. in prep.).

A second classification was conducted on the subset of regions that were labeled as any forest class. Training points within these classes were divided into two canopy-cover classes, 15% - 39% and greater than 39% canopy closure. MNDVI ranges for these two canopy-cover classes were determined and subsequently used to classify forested regions into canopy-cover classes.

Regions labeled barren were modified, also using MNDVI, to distinguish the areas that were sparsely vegetated from areas that were truly barren. MNDVI is an index of greenness and its value is proportional to the green leaf area in the pixel being considered (Nemani et al. 1993). MNDVI is also a ratio of refectance values and is therefore not affected by the brightness of an area. For example, north and south sides of a ridge will have the same MNDVI value if their covertype is the same; whereas the values for any given refectance band will be different between the two aspects. Within areas already classified as barren, regions that had an MNDVI of greater than 75 were considered sparsely vegetated and re-labeled as rocky reef.

Accuracy Assessment

Twenty percent of the training points that I collected were not used in the supervised classification process. Instead, these points were used to check the accuracy of the results of the digital classification process. These test points were overlaid on the supervised classification results; and the numbers that were correctly and incorrectly classified were determined using ARC/INFO. To evaluate the classification results in the context of bighorn sheep habitat, the test data and the supervised classification were re-coded into high and low-visibility classes and the accuracy assessment repeated.

COMPARISON OF VEGETATION BETWEEN 1937 AND 1992

The 1992 classes were standardized to the 1937 classes using the scheme outlined in Table 1. The two classifications were then converted to ERDAS GIS files, and the raster version of FRAGSTATS (Ver. 2.0, McGarigal

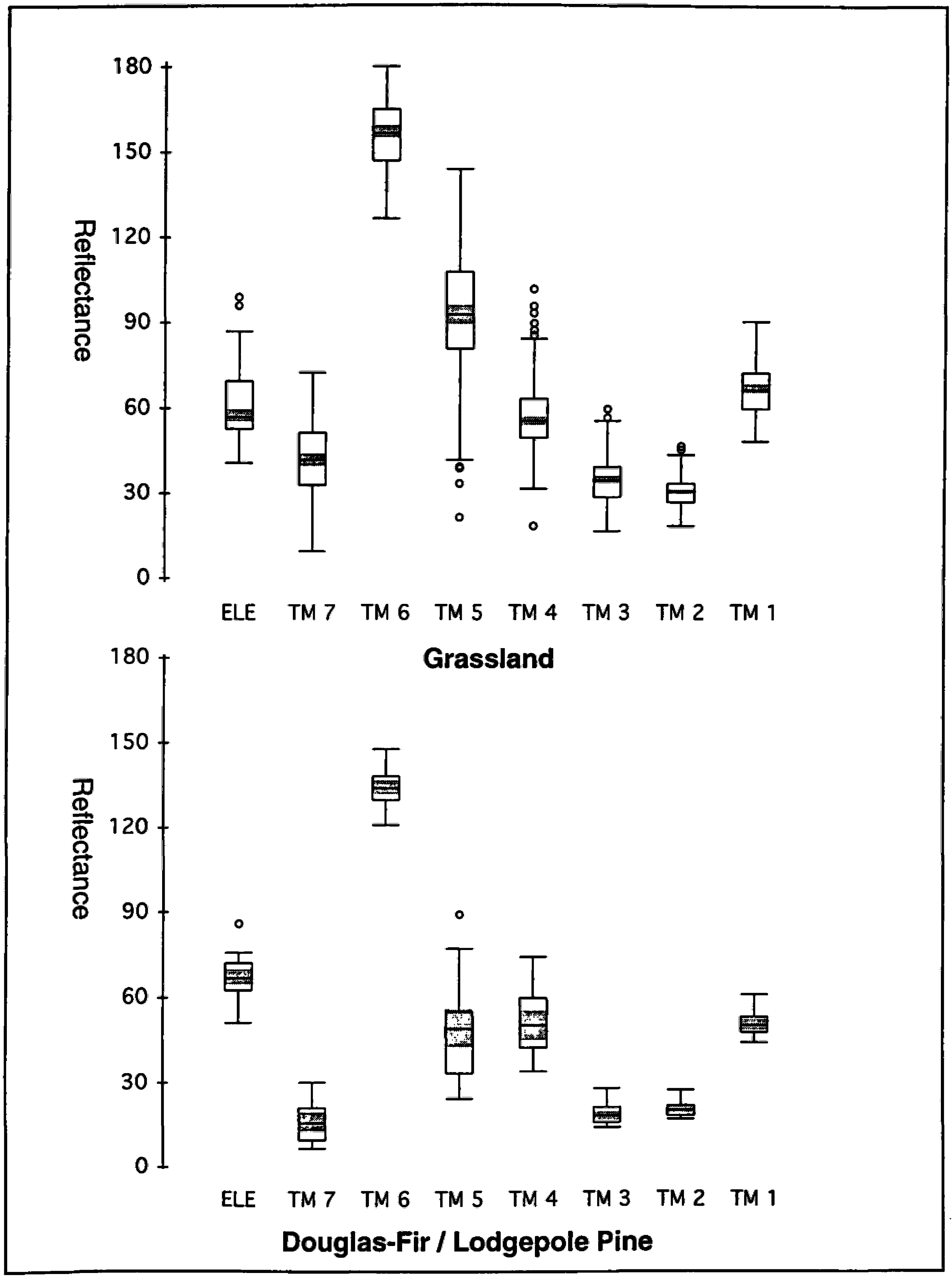

Figure 5. Spectral signatures of grassland and Douglas-fir - lodgepole pine. Reflectance values for Landsat TM bands 1 - 7 with scaled elevation values are depicted for Landsat TM scene P40/R27.

Table 1. Corresponding vegetation classes between the 1937 and 1992 land-cover classifications.

1937 Aerial Photo Classification	1992 Landsat TM Classification
Grasslands	Foothills Grasslands, Disturbed Grasslands Mountain Parklands Grass & Shrubland, Shrublands with less than 69% canopy closure, Alpine Tundra.
High-Density Coniferous Timber	All Needleleaf Forest Mixed Mesic and Broadleaf Forest classes, with greater then 40% canopy closure
Low-Density Coniferous Timber	All Needleleaf Forest Mixed Mesic and Broadleaf Forest classes with open stands with less than 40% canopy closure,
Aspen	Aspen
Rocky Reef	All Barren classes with MNDVI less than 75
Barren	All Barren classes with MNDVI greater than 75

and Marks 1994) was used to quantify their landscape metrics. Metrics were calculated at the individual patch level (individual patches), the class level (all patches for an individual cover-type), and at the landscape level (all patches of all cover-types). DATA DESK was used to analyze the FRAGSTATS metrics. Because at the patch level (considering all the patches within the landscape), patch size, proximity, and nearest neighbor metrics were not normally distributed, Mann-Whitney U tests were used to compare the medians for a selected set of class and landscape level FRAGSTATS metrics. For the formulas used to calculate these metrics see McGarigal and Marks (1994).

COLLECTING BIGHORN SHEEP LOCATION DATA

Surveys for bighorn sheep were conducted during November of 1992 and 1993, and October and November of 1994. Open and timbered areas within Hannan Gulch, Norwegian Gulch, French Gulch, Home Gulch, Blacktail Gulch, Big George Gulch, the north side of Gibson reservoir, the east side of both the North and South Forks of the Sun River, and the east side of Castle Reef from the Sun River to Green Timber Gulch were systematically searched from the ground. Although the aerial locations of bighorn sheep obtained by the Montana Dept. of Fish Wildlife and Parks were not used in the bighorn model evaluation, they were used to gain a sense of where bighorn sheep were likely to occur. The area surveyed is

Table 2. The GIS layers incorporated in the bighorn sheep habitat models. For raster data, cell size is listed rather than scale.

Data Layer	Data Type	Source[a]	Scale-Resolution
Topography	raster	USGS 7.5' DEM	30 m
Hydrography	vector	USGS	1:100,000
Roads	vector	USFS from CFFs[b]	1:24,000
Current vegetation	raster	Landsat TM imagery	30 m
Historic vegetation	vector	Aerial Photographs	1:20,000

[a] USGS = U.S. Geological Survey; USFS = U.S. Forest Service
[b] CFF = cartographic feature file.

displayed on Figure 7. Each area was surveyed at least once. Hannan Gulch, Norwegian Gulch, French Gulch and Wagner Basin were surveyed twice, but not in the same year.

Bighorn sheep were observed with 7x binoculars and a 15-60x spotting scope. The sex and number of animals were recorded and their locations were marked on 7.5' USGS topographic maps..

BIGHORN SHEEP HABITAT MODEL

I created a bighorn sheep habitat model using the GIS base layers described below (Table 2). Digital slope and aspect models were derived from 7.5' USGS DEMs using ARC/INFO. Digital hydrography, from USGS 1:100K Digital Line Graphs (DLGs), was used to model distance from water. Vegetation layers were used to model visibility.

The modeled habitat characteristics were developed from literature review specifically for Rocky Mountain bighorn sheep and are described in the introduction and below. I used habitat characteristics associated with visibility, barriers to movement, extent and proximity of escape terrain, elevation and aspect. Summer - fall range, winter range and lambing areas were modeled separately.

Visibility

Visibility is one the most important components of bighorn sheep habitat. Sheep prefer areas with an unobstructed views, but they can also use open forest stands when other habitat conditions are met (Geist 1971, Shannon et al. 1975, Tilton and Willard 1982). I used the vegetation layers (1937 and 1992) to model visibility for the two time periods. I labeled the following classes as having high visibility: barren, rocky reef, grassland, and low density conifer. Training site plots for barren, rocky reef, and grassland all had high-visibility; 80% of the training site plots for low-density conifer plots also had high-visibility. High density conifer and aspen were labeled low-visibility because training site plots for these classes all had low-visibility.

Escape terrain

Although researchers have generated a variety of specific descriptions for escape terrain, they universally agree that wild sheep require it. Steep, broken, rocky terrain characterize escape terrain. Bighorn sheep habitat requirements also include the appropriate juxtaposition and amount of escape terrain. Tilton (1977) determined that sheep did not use escape terrain unless it was at least 1.6 ha in size. DEMs were used to classify steep slopes of 27° to 60° as escape terrain. Because, bighorn sheep rarely venture more than 300 meters from escape terrain (Smith and

Flinders 1991a, and others), areas more than 300 meters from escape terrain were excluded from potentially suitable sheep habitat.

Lambing areas

Digital hydrography was used to identify water sources. Lambing areas were then modeled to include all patches identified above as escape terrain that were greater than 2 ha and less than 1 km from a water source.

Winter range

Areas with suitable visibility and escape terrain (see above) were modeled as winter range if they have a southern to western exposure (135°-270°) and an elevation below 1,974 meters (6,000 feet) (Erickson 1972). Aspect is a important feature of the winter range in the study area due to the strong westerly winds that commonly clear these aspects of snow. Southern aspects are also important for radiant heat dissipation.

Barriers and Winter Range Patch Size

I classified, canals, reservoirs, and areas of low-visibility greater than 120 m wide as barriers to sheep movement (Smith and Flinders 1991b). The roads and trails in the study area receive only light use, and therefore were not considered barriers.

Winter range areas that were within 120 m of each other were combined by reclassifying the intervening areas to winter range. Then, winter range patches less than 1.5 km^2 were eliminated from the habitat model in order to compare the amount of winter range contained within patches larger than 1.5 km^2 between the two time periods.

I delineated general year round habitat, winter range, and lambing areas for each time period. The resulting two ARC/INFO grids were converted into ERDAS GIS layers, and FRAGSTATS metrics for the components of predicted bighorn sheep habitat were calculated for the two landscapes. A subset of these metrics was used to compare sheep habitat between 1937 and 1992.

Model Evaluation

Predicted bighorn sheep habitat for 1992 was overlaid on the GIS point coverage of sheep locations obtained in the field. The percentage of sheep locations not falling within polygons of predicted habitat was calculated and considered as a rough estimate of the model's error of omission.

MARKOV MODEL OF FUTURE LANDSCAPE COMPOSITION

A simple Markov chain matrix projection model (Horn 1975) was used to predict a future landscape composition for the study area based on the vegetative succession that occurred between 1937 and 1992.

Transition probabilities for land-cover classes between 1937 and 1992 were determined by evaluating the landscape models for the two time periods on a cell by cell basis. The proportion of each 1992 land-cover class occurring within the geographic extent of a single 1937 land-cover class was determined. These proportions were determined for all land-cover classes and placed in a transition matrix. The matrix was multiplied by an array containing the proportion of each land-cover class in the 1992 landscape. The results, representing the landscape composition one time period (55 years) in the future, were used in the next iteration of the matrix multiplication. This process was iterated until a steady state of landscape composition was reached.

Results

Accuracy Assessment for the 1992 land-cover classification

Among the 1992 land-cover polygons for which test points existed, the digital classification process correctly classified 61.5% of the polygons for grassland, 80% for low-density conifer, 85.7% for high-density conifer, 44% for rocky reef, 93.8% for barren, and 60% for aspen (Table 3). Rocky reef had the greatest amount of confusion (mis-classification), with 22% of the test points incorrectly labeled as grassland, another 22% labeled as barren, and 11% labeled as high-density conifer. When all land-cover classes were combined into two classes, low and high-visibility, the classification process correctly labeled 91.5% of the high-visibility test points and 85.7% of the low-visibility test points (Table 4). An accuracy assessment was not conducted on the 1937 landscape due to a lack of independent field data to use for test points.

Land-cover Mapping and Comparison

My land-cover mapping efforts for 1937 and 1992 revealed changes in the configuration (Fig. 6) and distribution (Tables 5, 6) of all defined land-cover types in the study area. The percent of the landscape occurring as high-visibility cover types decreased from 77.5% in 1937 to 54.9% in 1992.

Table 3. An accuracy assessment matrix for the 1992 digital classification. Diagonal elements represent correctly classified points.

	Test Data					
Classified data	Grassland	Low-density conifer	High-density conifer	Rocky reef	Barren	Aspen
Grassland	**61.5%** (8)	0	0	22.2% (2)	0	0
Low-density conifer	7.7% (1)	**80.0%** (4)	14.3% (2)	0	0	20.0% (1)
High-density conifer	7.7% (1)	20.0% (1)	**85.7%** (12)	11.1% (1)	0	20.0% (1)
Rocky reef	7.7% (1)	0	0	**44.4%** (4)	6.2% (1)	0
Barren	15.4% (2)	0	0	22.2% (2)	**93.8%** (15)	0
Aspen	0	0	0	0	0	**60%** (3)
Number of test points	13	5	14	9	16	5

Table 4. An accuracy assessment matrix for the 1992 digital classification for high and low visibility land-cover classes.

	Test Data	
Classified data	High-visibility	Low-visibility
High-visibility	**91.7%** (44)	14.3% (2)
Low-visibility	8.5% (4)	**85.7%** (12)
Number of test points	48	14

The greatest changes occurred within the grassland and high-density conifer types. In total area, grasslands decreased from 11,068 ha in 1937 to 8,125 ha with 30.4% of the 1937 grassland becoming high-density conifer in 1992. Low-density conifer also declined in the landscape, with 72.5% converting to high-density conifer (Table 6). The percent of high-density conifer occupying the landscape doubled (Table 5).

Landscape Metrics and Indices

The total number of patches for all land-cover classes increased from 980 patches in 1937 to 5,213 in the 1992 classification (Fig. 6, Tables 7, 8). There were increases in patch density, patch size coefficient of variation, and the interspersion/juxtaposition index for the landscape as a whole. Due to the dramatic increase in high-density conifer (Table 5), the Largest Patch Index (LPI) for the landscape also increased between the two time periods (Table 8). In 1937, the largest single patch was grassland, which alone occupied 6.7% of the landscape. In 1992, the largest single patch in the landscape was high-density conifer which occupied 11.6% of the landscape. There were deceases in the Mean Patch Size (MPS), Mean Nearest Neighbor Distance (MNND), and contagion index for the landscape as a whole (Table 8).

The MPS for grassland and rocky reef classes decreased six and thirty fold, respectively, while the MPS for high-density conifer decreased only two fold. These differences in MPS were significant for all classes except aspen (Table 7). The Mean Nearest Neighbor Distance (MNND) also decreased significantly for all cover types except for low-density conifer and aspen. The LPI decreased for all land-cover classes except high-density conifer, which increased from 6.1 to 11.6.

High-density conifer was also an exception to the trend in Mean Proximity Index (MPI) which decreased for all other land-cover classes. The high-density conifer MPI increased from 1,409.4 to 5,837.1 between the two time periods (Table 7). Changes in MPI were significant for all classes except grassland and aspen. Finally, the interspersion/juxtaposition index increased for all classes between the two time periods.

Bighorn Sheep Locations and Model Evaluation

Field surveys recorded 35 bands of bighorn sheep totalling 593 individual observations. Sheep bands contained between one and sixty-three animals. Overlaying these field locations onto predicted bighorn sheep habitat for 1992 revealed that 85.7% (30/35) of the points fell within patches of predicted habitat (Fig. 7). The 5 points that fell outside of predicted habitat polygons each occurred within 20 m of predicted habitat. The model delineated a number of areas of predicted bighorn sheep habitat in which no sheep were detected.

Bighorn Sheep Habitat Model

The bighorn sheep habitat model delineated 22,136 ha of habitat in 1937 and 14,701 ha in 1992, a decrease of 33.4% (Fig. 8, Tables 9, 10). The percent of the landscape predicted as bighorn sheep habitat (all habitat components) dropped from 52.4% in 1937 to 34.9% in 1992 (Table 9). The summer / fall, winter, and lambing area components of bighorn sheep habitat decreased 31.1%, 39.4%, and 37.1%, respectively. The amount of summer/ fall range fell from 36.6% of the landscape to 25.2%. Winter range, which occupied 12.8% percent of the landscape in 1937, only occurred on 7.8% of the landscape by 1992. Lambing range dropped from 3.0% to 1.9% of the landscape between the two time periods (Table 9).

The number of patches for all types of predicted bighorn sheep habitat combined increased from 1,753 to 2,825. There were also increases in patch density, edge density, contagion, and LPI between 1937 and 1992 for the landscape as a whole (Table 10). The largest single patch in the landscape for both time periods was comprised of unsuitable habitat and increased from 16.8% to 51.6% of the study area between the two time periods. Decreases occurred in MPS, MNND, and the interspersion / juxtaposition index between 1937 and 1992 for the landscape as a whole (Table 10).

Table 5. Percent of landscape in land-cover classes for 1937 and 1992 and the percent change within each land-cover class over time.

Class	1937	1992	Percent Change[1]
Grassland	26.2%	19.2%	-26.7%
High Density Conifer	20.4%	41.4%	+103%
Low Density Conifer	14.4%	5.6%	-61.8%
Rocky Reef	26.3%	16.6%	-36.9%
Barren	10.6%	13.5%	+29.2%
Aspen	0.7%	2.4%	+243%
Water	1.3%	1.3%	No change

[1] {(%(1937) - %(1992)) / %(1937)} * 100

Table 6. Pairwise transformation of land-cover classes between 1937 and 1992, showing changes within each land-cover class. Values represent the percentage of each 1937 land-cover class that converted into the 1992 land-cover class shown in the left most collum.

	1937 landcover					
1992 landcover	Grassland	High density conifer	Low density conifer	Rocky reef	Barren	Aspen
Grassland	41.5% (4127)[a]	5.8% (451)	7.2% (393)	16.8% (1679)	14.1% (565)	23.7% (61)
High density conifer	30.4% (3022)	80.6% (6253)	72.5% (9767)	21.6% (3953)	4.4% (176)	32.6% (84)
Low density conifer	7.1% (705)	3.2% (251)	5.7% (313)	7.0% (699)	2.7% (107)	2.6% (6)
Rocky reef	10% (994)	8.2% (639)	11.3% (614)	33.6% (3348)	16.9% (679)	2.9% (7)
Barren	3.7% (371)	1.2% (94)	2.4% (132)	20.3% (2023)	61.7% (2476)	0.8% (2)
Aspen	6.9% (682)	0.8% (65)	0.5% (28)	0.7% (65)	0.1% (6)	35.5% (91)

[a] hectares

Table 7. A comaprison of class level metrics for land-cover types between 1937 and 1992. HD and LD conifer refer to high density and low density conifer classes respectively.

	Grassland		H D Conifer		L D Conifer		Rocky Reef		Barren		Aspen	
Class Statistic	**1937**	**1992**	**1937**	**1992**	**1937**	**1992**	**1937**	**1992**	**1937**	**1992**	**1937**	**1992**
Class Area (ha)	11068	8125.4	8629.7	17410	6063.5	2319.3	11093	6997.3	4462.9	5683.4	285.3	1050.9
Percent of Landscape	26.218	19.3	20.442	41.253	14.363	5.495	26.278	16.580	10.572	13.467	0.676	2.490
Largest Patch Index	6.657	1.652	6.058	11.575	2.074	0.590	5.975	0.351	2.181	2.470	0.091	0.155
Number of Patches	349	1359	149	650	246	921	78	1322	73	592	69	320
Patch Density	0.827	3.220	0.353	1.540	0.583	2.182	0.185	3.132	0.173	1.403	0.163	0.758
Mean Patch Size (ha)	31.714	*5.979	57.918	*26.78	24.648	*2.518	142.22	*5.293	61.136	*9.600	4.135	3.284
Patch Size S. Dev.	200.40	31.776	292.38	275.70	97.372	10.164	332.25	14.454	157.08	58.426	7.109	6.981
Patch Size CV	631.89	531.47	504.86	1029.3	395.05	403.61	233.61	273.08	256.94	608.58	171.9	212.581
Mean Proximity Indx	1419.7	251.32	1409.4	*5837	216.25	*36.75	2829.4	*135.5	392.33	*426.0	23.60	23.618
Mean Nearest Neig	114.92	*81.84	204.54	*76.05	179.80	149.37	197.02	*80.06	327.26	*117.9	317.4	236.323
Nearest Neig S. Dev.	110.05	89.825	291.28	72.881	238.01	149.84	262.28	90.846	371.34	142.64	665.2	492.599
Nearest Neig CV	95.766	109.75	142.41	95.824	132.38	100.31	133.12	113.47	113.47	120.97	209.5	208.443
Interspersion-Jxt.	79.783	85.898	64.330	74.650	65.824	79.550	74.827	76.227	66.093	67.204	26.76	66.321

* differences between the two time periods significant at $P < 0.006$

Table 8. A comparison of landscape indices for land-cover types.

Landscape Level Indices	1937	1992	P value
Largest Patch Index	6.657	11.575	NA
Number of Patches	980	5213	NA
Patch Density (#/100 ha)	2.321	12.352	NA
Mean Patch Size (ha)	43.077	8.096	P < 0.0001
Patch Size CV	473.749	1254.005	NA
Edge Density (m/ha)	44.075	103.360	NA
Mean Nearest Neighbor Dist.	191.217	110.078	P < 0.0001
Nearest Neigh CV	161.440	168.565	NA
Interspersion / Juxtaposition Index	71.542	78.217	NA
Contagion (%)	52.548	43.633	NA

The LPI decreased for all components of predicted bighorn sheep habitat, while the MPI decreased for all the individual components of predicted bighorn sheep habitat. These differences were significant for summer / fall and winter ranges but not for lambing range (Table 9). Most notably, the MPI for modeled winter range declined significantly from 721.9 in 1937 to 192.5 in 1992, a 73% decline. The decrease in the MPI for components of predicted sheep habitat suggests that patches of suitable habitat were more isolated and fragmented in 1992 then they were in 1937.

Mean patch size decreased for all components of predicted bighorn sheep habitat. These decreases were significant, except for summer / fall range. Because bighorn sheep are more likely to use larger patches of habitat (Smith and Flinders 1991b), this trend in MPS could have consequences for bighorn sheep in addition to the loss in total area of predicted habitat. The MNND decreased significantly for all components of predicted bighorn sheep habitat (Table 9). The interspersion/juxtaposition index decreased from 92.1% to 75.2% for the summer / fall range component of predicted bighorn sheep habitat. This index increased for the winter range, and lambing areas.

The changes in total area and structure of predicted bighorn sheep habitat, as indicated by the landscape metrics presented above, suggest that conifer encroachment has caused a decline the quantity of bighorn sheep habitat, and that fragmentation has reduced habitat quality.

Table 9. Percent of the landscape in each bighorn sheep habitat type for 1937 and 1992, and the percent change within each habitat component type.

Habitat Component	1934	1992	Percent Change
Summer / fall habitat	36.6	25.2	-31.1
Winter range	12.8	7.8	-39.1
Lambing area	3.0	1.9	-36.7
Unsuitable habitat	47.6	65.1	+36.8

Table 10. Class metrics for sheep habitat types in 1937 and 1993.

	Summer / fall habitat		Winter range		Lambing areas	
Class Statistic	**1937**	**1992**	**1937**	**1992**	**1937**	**1992**
Class Area (ha)	15465.8	10631.0	5406.2	3274.8	1264.2	795.0
Percent of Landscape	36.6	25.2	12.8	7.8	3.0	1.9
Largest Patch Index	13.825	10.783	2.434	0.666	0.086	0.033
Patch Density	1.554	2.770	1.182	2.087	1.417	1.836
Mean Patch Size (ha)	23.576	9.094	10.835	*3.717	2.114	*1.026
Patch Size Std. Deviation (ha)	272.21	135.92	61.821	16.15	4.53	1.71
Patch Size Coeff. of Var.	1154.60	1494.55	570.56	434.53	214.02	166.44
Mean Proximity Index	8595.76	*5228.4	721.91	*192.53	14.15	9.39
Mean Nearest Neighbor Dist. (m)	74.26	*64.90	90.32	*78.50	148.07	*111.26
Nearest Neighbor Std. Dev.	69.40	61.24	113.52	111.70	199.63	154.80
Nearest Neighbor Coeff. of Var.	93.46	94.36	125.69	142.29	134.82	139.14
Interspersion & Juxtaposition %	92.082	75.23	85.07	85.40	77.49	84.44

* differences between the two time period significant at $P < 0.003$

Table 11. A comparison of landscape indices for predicted bighorn sheep habitat.

Landscape Level Indices	1937	1992	P value
Largest Patch Index	16.838	51.618	NA
Number of patches	1937	3192	NA
Patch Density (#/100 ha)	4.589	7.563	NA
Mean Patch Size (ha)	21.789	13.222	$P < 0.0233$
Patch Size Variation Coefficient	1313.349	3016.861	NA
Edge Density (m/ha)	47.522	62.925	NA
Mean Nearest Neighbor	105.394	79.531	$P < 0.001$
Nearest Neigh Variation Coefficient	133.570	133.602	NA
Interspersion/Juxtaposition	83.204	74.632	NA
Contagion (%)	49.422	54.137	NA

Patches of winter range larger than 1.5 km^2 are more likely to be used by bighorn sheep than smaller patches (Smith and Flinders 1991b). When only considering patches larger than 1.5 km^2 (Fig. 11), the amount of predicted winter range decreased from 8,182 ha in 1937 to 5,839 ha in 1992, a decline of 28.6%. This is less than the 39.1% decrease measured when all patches of winter range are considered indicating that small patches of predicted winter range are dropping out of the landscape faster than larger patches as conifers encroach on high-visibility land-cover types.

Markov Model of Future Habitat Composition

Based on the nature and extent of change since 1937, the Markov projection model predicts continued declines in grassland, low-density conifer, rocky reef, and barren types, and another 15% increase in high-density conifer between 1992 and 2267 (Fig. 10). The proportion of predicted land-cover types reached a steady state after five iterations of the model with high-density conifer occupying 57.3% of the landscape. If these results represent actual future conditions, bighorn sheep habitat will continue to decline as coniferous species encroach into the important high-visibility land-cover classes.

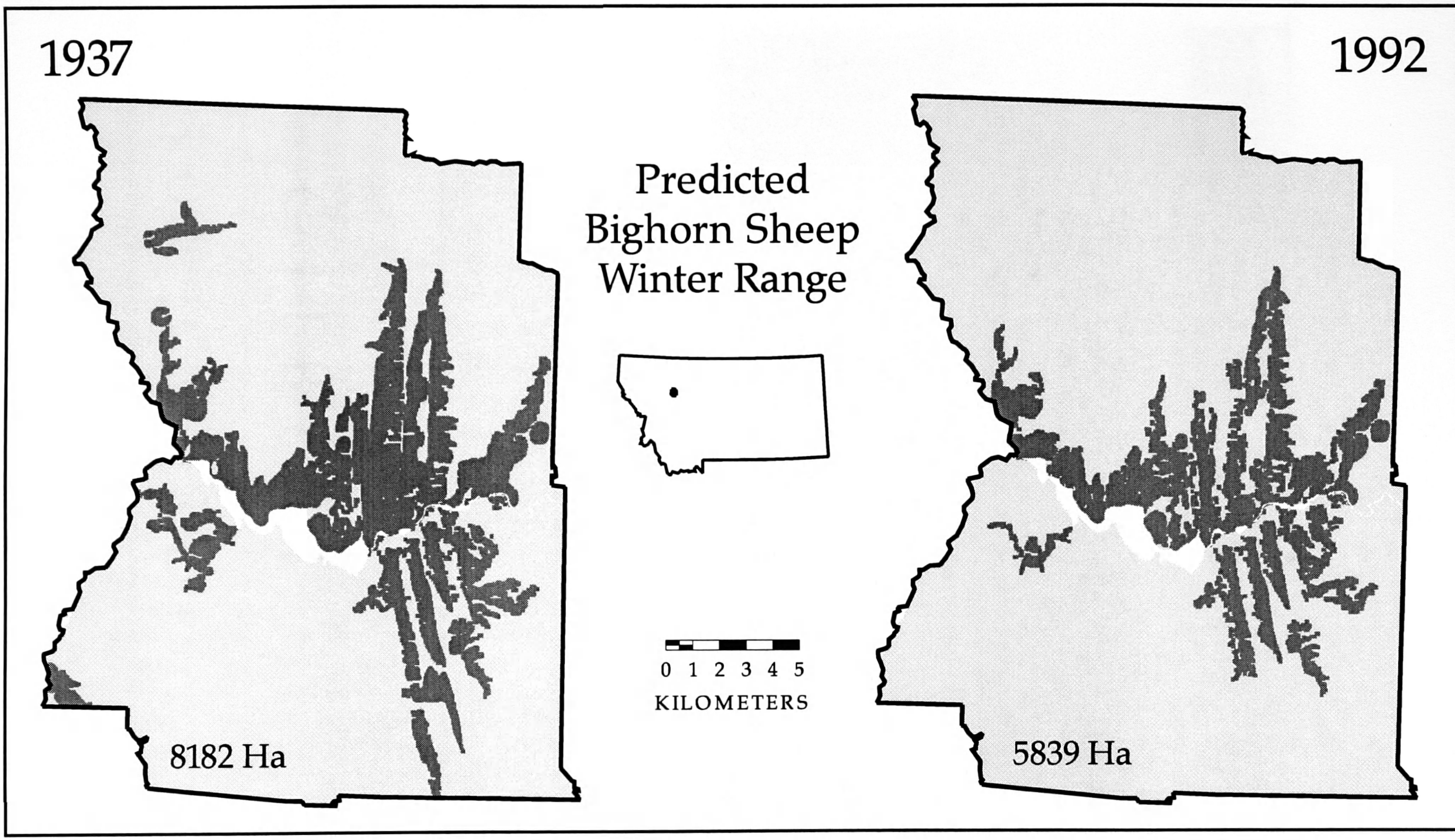

Figure 9. A comparison of predicted bighorn sheep winter range areas displaying patches greater than 1.5 square km.

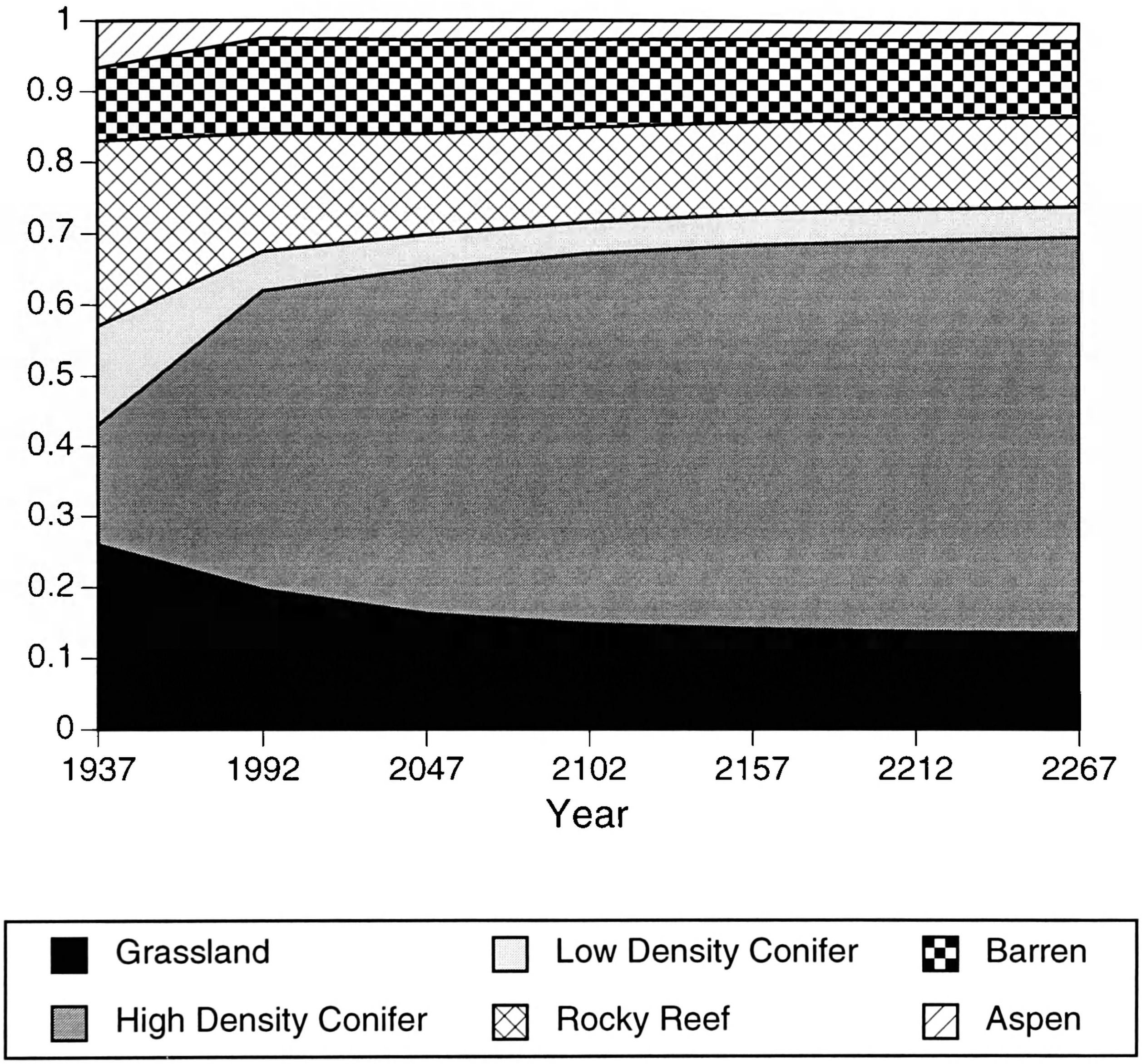

Figure 10. Changes in landscape composition from 1937 to 2267, using a Markov chain projection model. The measured transitions between 1937 and 1992 were used to seed the model.

Discussion

Accuracy Assessment

Some overall confusion between land-cover classes should be expected in any broad-scale digital classification. Overlap in the spectral signatures between land-cover classes is due to a number of factors, including: 1) natural variation in a species phenotype that result from physical habitat attributes such as soils, aspect and elevation, 2) variation in the species composition of a land-cover class, and 3) ambiguity in identifying land-cover when gathering training data. On a broader scale, variation in atmospheric conditions across a Landsat scene adds variation to the spectral signatures of land-cover classes. For all the land-cover classes I mapped in this study except rocky reef, the confusion was not excessive.

The low accuracy for rocky reef, 44%, may be due to a number of factors. Rocky reef can be considered a transition or ecotone between grassland and barren rock. Satellite imagery does not always detect enough information to correctly classify ecotonal areas (Anonymous 1991). This ecotone was also difficult to consistently delineate on the 1937 black and while aerial photographs. The spectral signatures, and the actual land-cover classes were similar in many cases, resulting in many areas of rocky reef getting classified as barren in the initial supervised classification. Unfortunately, some confusion in the classification still remained. The confusion between grassland and rocky reef also results from the fact that many areas of rocky reef are grassy while retaining the greater than 50% rock cover. Thus, it is occasionally difficult to determine the appropriate label for a spectral polygon that has elements of both rocky reef and grassland. Patches of conifers occasionally occur within a matrix of rocky reef, which also explains some of the confusion between these two classes. Furthermore, within the USFS field data, rocky reef was encompassed within other land-cover classes, primarily exposed rock and grassland. Removing USFS training data points that conflicted with the ones I collected for rocky reef helped reduce confusion between these classes. Many areas of rocky reef were subsequently separated from barren using MNDVI, an index of greenness, which further improved the accuracy of this class.

When the land-cover classes were merged into the two categories used to model bighorn sheep habitat, high and low-visibility, the accuracy improved greatly, showing that much of the confusion revealed in the land-cover accuracy assessment does not affect the outcome of the bighorn sheep habitat model.

Comparison of Classification Methodology

The number of land-cover types I discerned in the 1930s' black and white photographs, was significantly fewer than those discernable in today's high resolution color aerial photographs, or satellite imagery. I therefore combined the digitally derived land-cover types for 1992 into groups comparable to the 1937 classes. In addition to making cover-types comparable between the two data sources, this increased the accuracy of my 1992 digital classification relative to a scene-wide classification conducted with 19 cover-types. This is because the digital process is well suited to discerning coniferous forest and canopy closure, but is less accurate when attempting to distinguish among different plant species. Because bighorn sheep respond to general land-cover characteristics (high-visibility versus low-visibility vegetation), my grouping of 1992 land-cover classes was appropriate for this analysis. For the purposes of modeling bighorn sheep habitat, these assumptions do not notably alter the results. However, if modeling efforts were directed at other wildlife species, these assumptions may not be appropriate for spatially-explicit habitat models.

Some changes in a few of the land-cover classes between the two time periods may be a result of one or both of the classification processes confusing particular classes. For example, it is unlikely that barren land has changed much during the 55 years between these two landscape 'snapshots'. It is more likely that barren was confused with rocky reef or vice versa during the aerial photo interpretation and/or the digital classification process.

Some of the 1937 land-cover classes were difficult to distinguish using interpretive methods. For example, barren and rocky reef classes were similar in appearance. Similarly, grassland and rocky reef may have been confused (Table 3). The land-cover transition matrix (Table 6) and the landscape composition values may have been affected by the confusion of these pairs of land-cover classes. However, because these are all considered high-visibility classes, this confusion should not effect the results of the bighorn sheep habitat models.

Although I applied the same MMU for the two time periods, some of the differences in the landscape between the two time periods might reflect differences in the way the MMU was applied between the two classifications. Inconsistent application of the MMU during photo interpretation versus consistent application of the MMU during the digital process may have contributed to the dramatic increase in the number of patches of all land-cover classes between the two time periods. The digital process is better at detecting small patches that are embedded within larger ones, leading to an increase in the number of patches between the two process. However, much of this increase is likely due to an increase in the heterogeneity of the landscape. During field work I observed many small stands of conifers that were not present on the 1937 aerial photographs. Furthermore, an increase in land-cover heterogeneity is clearly depicted in many of Gruell's (1983) photographic comparisons of present and historic land-cover. Many stands of coniferous trees existed in the early 1980's where none occurred at the turn of the century.

Changes in Landscape Composition

Since 1937 it appears that coniferous forest has encroached into grassland and rocky reef land-cover types in the study area and thereby reduced the amount of open, high-visibility habitat preferred by bighorn sheep. This phenomenon has been observed in other bighorn sheep ranges throughout the Rocky Mountains (Arno and Gruell 1986, Bentz 1981, Gruell 1983, Picton and Picton 1975, Wakelyn 1987). However, no previous studies have quantitatively measured the extent and magnitude of this encroachment. Although some of the measured landscape changes may be a result of using different methods to derive the two landscape 'snapshots' used in this analysis, the majority of the change in land-cover composition detected in this study represents real landscape change.

Changes in Landscape Structure

Landscape structure, the spatial relationship between landscape components, is also an important aspect of bighorn sheep habitat. Some of the difference observed in landscape structural indices reflect real, on the ground, structural change, whereas others may be due to differences in the classification methods between the two time periods.

The increase in the MPI for high-density conifer, and decrease for the other land-cover classes, suggest that while patches for most classes became more fragmented and isolated from one another, patches in the high-density conifer class became less isolated and fragmented. In other words, small and disjunct conifer stands expanded, became denser (low-density to high-density conversion), and coalesced into larger stands. The increase in LPI for high-density conifer also lends support to my contention that patches of high-density conifer were larger, and closer together in 1992 then they were in 1937.

The decrease in MNND for all land-cover classes can be partly explained by conifer encroachment. When previously continuous patches of open land-cover classes were broken up by invading trees, the MNND decreased as a result of the total number patches increasing within the same area. The decrease in MNND for high-density conifer of 128.5 m was the second greatest change, among land-cover classes for this metric. Notable changes in MNND for barren (210.2 m) and rocky reef (117 m) were also detected. Some of the difference in MNND between barren and rocky reef is also due to methodological difference between aerial photo interpretation and digital image classification. The digital process discerned a greater interspersion of these two patch types than PI. However, some of the change in MNND for rocky reef was no doubt due to conifer encroachment. Rocky reef is particularly susceptible to conifer invasion, based on the 21.6% transformation of rocky reef to high-density conifer. It is unlikely that much of the rocky reef to high-density conifer conversion was due to methodological differences because high-density conifer was never confused with rocky reef based on the results of the accuracy assessment.

The mean patch size for grassland and rocky reef classes decreased six and thirty fold respectively, whereas the MPS for high-density conifer decreased only two fold, further establishing the effects of conifer encroachment on landscape structure. This suggests that areas that were continuous meadows or virtually treeless rocky reef tracts in 1937 were dotted with patches of coniferous forest by 1992. However, the fact that MPS did not increase for high-density conifer implies that some of the change in these landscape structural metrics is due to the aforementioned methodological issues.

A portion of the registered change in landscape structure may be due to differences in the two classification processes already described. Some of the overall increase in the total number of patches and subsequent decrease in MPS and MNND among all land-cover classes may be due to the ability of the satellite sensors (and the digital classification process) to consistently distinguish small land-cover patches. The decrease in MNND for all land-cover classes is at least partially due to the dramatic increase in the number of patches between the two time periods. However, it is unlikely that the variation in the magnitude of the differences in MPS among land-cover types between the two time periods is strictly due to methodological differences. Taken together with the variation in the change for MPI between land-cover classes, it appears that open land-cover types became more fragmented, whereas high-density conifer class underwent a general coalescing of patches as it became more abundant in the study area between 1937 and 1992.

Markov Model of Future Landscape Composition

I assumed the transitions probabilities derived from the change in landscape composition between 1937 and 1992 represent the probabilities of one land-cover class replacing another land-cover class. In addition, I assumed that these probabilities remained constant regardless of how the landscape composition changed during the models progression. A number of factors could result in these assumptions being violated, such as climatic change, population fluctuations of mammal or insect herbivores, and outbreaks of plant pathogens all affect the rate and direction at which succession proceeds and thus can effect the transition probabilities upon which the results of this model depend (Horn 1982).

Because the inputs to the model covered a period during which fires were aggressively suppressed. the results represent a reasonable landscape composition only in the continued absence of the fire-driven disturbance regime that was historically part of this region. Under this condition it is possible that a climax community, or a steady state distribution of land-cover types could be reached. The model also assumed the disturbance regime remained constant, and was the same as the time period during which the transition probabilities were derived. Obviously, if fire once again becomes the dominant disturbance in this landscape, this assumption will be violated, and the land-cover composition might be dramatically different than the results of this model. In reality, given enough time, a fire will occur in the study area despite the current level of fire suppression effort. For example, a major fire occurred just south of the study area in 1988.

Predicted Bighorn Sheep Habitat

The decline in quantity of bighorn sheep habitat components and the change in habitat structure detected in the study area are similar to differences documented in a comparison of presently occupied versus historically occupied but abandoned bighorn sheep ranges in Colorado. Occupied bighorn ranges had significantly more grassland, rocky grassland, and rockland with low shrubs, and less forest than did historically occupied but abandoned ranges (Wakelyn 1987). In three areas sheep had recently abandoned habitat that had more forest cover, more dense, tall, shrubland, less grassland, and less rockland, than adjacent occupied habitat (Wakelyn 1987). Although some habitat conditions differ between the southern Rocky Mountains in Colorado and the northern Rockies of Montana, it is likely that bighorn sheep will eventually respond similarly to analogous changes in landscape composition.

Model Evaluation

The GIS-based bighorn sheep habitat model predicted the location of sheep habitat remarkably well. The fact that no sheep were located more than 20 m from the edge of an area of predicted habitat suggests the model rules were well suited for predicting bighorn sheep habitat in the northern Rocky Mountains (Fig. 7). Many of the areas of predicted sheep habitat that did not contain bighorn sheep point locations may be due to the limited effort available to invest in sheep surveys during this project. The southwest portion of the study area was never visited, because previous surveys conducted by state biologists did not find any sheep there (Quentin Kujala, personal communication). In addition, this portion of the study area contains patches of winter range all smaller than 1.5 km^2. Others parts of the study area only received one visit, which is not enough to exclude the presence of bighorn sheep.

Limitations in the available GIS data also affected the model's results. The 30 m^2 MMU of 7.5' DEMs may have contributed to an over-estimate of escape terrain. Even though I am confident that the vast majority of suitable escape terrain areas were included within 28°-60° slopes delineated by the DEMs, I suspect some of the chosen areas did not contain all the appropriate components of escape terrain. For example, slope delineation could not always distinguish between continuously smooth slopes and ones that are broken by rocky outcrops and cliffs. Even within the steep portions of barren and rocky reef land-cover classes selected by the model, the existence of near-vertical rocky outcrops were not necessarily present.

The GIS-based habitat model selected areas based on generalized characteristics for bighorn sheep habitat. The land-cover data were relatively crude and did not allow for a comprehensive evaluation of the study area. For example, although grasslands are a component of the model, the quality or the forage availability in any particular patch could not be evaluated. Similarly, the ability of patches of rocky reef to produce adequate forage to support bands of sheep is probably quite variable throughout the study area.

Finally, other factors not included in the model, such as the presence of elk or mule deer that compete with bighorn sheep for forage

(Schallenberger 1966, Kasworm et al. 1984), especially during winter, may reduce the value of certain areas as bighorn sheep range. Such situations would not be detectable without additional field work, or having geographically explicit elk and deer population data available. For this reason, it is essential that any area under investigation be carefully evaluated on the ground before any management decisions are made based on model results.

Ability of the Study Area to Maintain Bighorn Sheep

The Sun River drainage of Lewis and Clark NF still contains some of the best bighorn sheep habitat in the Rocky Mountains south of Canada. Although the amount of bighorn sheep habitat in 1992 is clearly less than the amount that was available in 1937, the current population of bighorn sheep in the study area appears to be larger than it was at that time (USDA 1935, Couey 1950). The lack of a numerical response by bighorn sheep to a decline in the amount of suitable habitat may be due to several factors. The bighorn sheep habitat model identifies potentially suitable habitat patches; it does not tell whether a particular patch was utilized by sheep. Although U.S. Forest Service staff conducted wildlife surveys as early as 1935 (USDA 1935), it is impossible to ascertain the geographic extent of their searches. These surveys only list the general location and the number of ungulates observed. It is likely the surveyors only visited areas they felt were likely to harbor large herds of ungulates. Therefore, it is impossible to verify the historic bighorn sheep habitat model, especially for areas that were not visited by survey crews. In many cases, due to this limitation in data availability, it is impossible to tell if modeled habitat patches that suffered conifer encroachment were ever occupied by bighorn sheep. Thus, one possible explanation for the lack of a population response to a 31.3% decrease in the amount of modeled bighorn sheep habitat, is that lost habitat patches were not historically important to bighorn sheep.

Habitat quantity may not currently be a limiting factor to the bighorn sheep population in this study area. When considering historic population estimates, it is important to realize that anthropogenic effects had already been inflicted on the population by the time surveys were conducted in the 1930s. Prehistoric, bighorn populations may have been much larger in size than they were by the time the first wildlife surveys were conducted. At the turn of the century, bighorn sheep populations may have been limited by hunting pressure and disease rather than habitat availability or quality. Today, hunting pressure is much lower than it was at the turn of the century, and bighorn sheep ranges are no longer within domestic sheep grazing allotments. It is therefore possible the population has increased in response to a lower human harvest and less contact with the pathogens transmitted by domestic sheep, and that habitat quality and quantity are not currently limiting population size.

On the scale of hundreds of years, bighorn sheep populations may be much more susceptible to declines in habitat quality and/or quantity then they are in the short term of 5-6 decades. Due to their highly philopatric nature (Geist 1971), a bighorn sheep population may continue to use the same migration routes, even through forest encroachment has compromised the visibility and the amount of forage (Martin and Stewart 1982). A herds's movement patterns may have been set for hundreds of years; and as long as the herd remains viable, migration and movement patterns may be passed down through many generations even if the quality of travel routes has decreased. In the long term, however these herds are likely to become genetically isolated because dispersing bighorns from other herds are unlikely to supplement herds isolated by terrain with low visibility. Mitochondrial DNA analysis (Luikart 1993) indicates bighorn sheep herds on the Rocky Mountain Front may have already experienced some effects of isolation. Were these herds to suffer extinction due to factors such as disease or environmental and demographic stochasticity, it is unlikely the habitat would become recolonized without human assistance to restore the habitat and translocate bighorn sheep.

Suggestions for Future Research

Although I did not investigate the landscape conditions in other bighorn sheep ranges to the north and south of Sun River (Fig. 4), it is likely that the same processes of conifer encroachment into grassland and rocky reef areas are occurring there as well. Although nothing is known about bighorn sheep movements between these and the Sun River population, it is likely that some exchange occurs, or at least did occur in the recent past. A long-term study monitoring bighorn sheep movements should be initiated on the entire Rocky Mountain Front in Montana. Because rams occasionally embark on long distance travels, they should be the focus of this proposed study. Intervening areas between the Sun River and Ear Mountain to the north and Ford Plateau to the south should be evaluated for landscape change.

Management Recommendations

Because bighorn sheep are unlikely to find and/or move to new areas of suitable habitat if their current habitat becomes degraded, it is essential their currently occupied ranges remain in good condition. In addition, to promote genetic exchange, areas between bighorn sheep ranges should be maintained in a condition that is conducive to dispersal. This study and others (Wakelyn 1987) have shown that removing natural wildfire has caused deterioration in an area's utility as bighorn sheep habitat. Because natural wildfires are now rare in the study area, prescribed fire must play an important role in maintaining the quality and quantity of bighorn sheep habitat.

In the wilderness portion of the study area, natural fires should continue to be allowed to burn. Ideally, a natural fire regime could be restored in the majority of the study area which is outside of the wilderness. Because this is probably not realistic due the presence of numerous private structures and other management concerns, the US Forest Service's prescribed fire program for wildlife habitat improvement should be continued.

Most importantly, prescribed fire should be used to kill trees in areas where conifers have encroached into bighorn sheep habitat. Areas of conifer encroachment that are juxtaposed with predicted bighorn sheep winter range areas are illustrated in Figure 11. Big George Gulch and Hannan Gulch are two such areas that are presently used by bighorn sheep. In agreement with the results from this project, the US Forest Service documented a 40% increase in tree cover between 1966 and 1986 in Big George Gulch. The US Forest Service selected both of these areas for range enhancement projects using prescribed fire (USDA 1990, USDA 1990b). Unfortunately, the objectives for both of these projects were not completely met. Of the 65% conifer mortality that was deemed desirable, only 20%-21% occurred. I suggest both of these areas be re-burned in the near future to achieve a higher degree of conifer mortality. Other areas to consider for prescribed fire, are Home Gulch and upper Norwegian Gulch, both of which are used by bighorn sheep in winter. Additional areas where conifers have encroached into previously open areas are identified in Figure 11. All of these sites should be considered for treatment with prescribed fire.

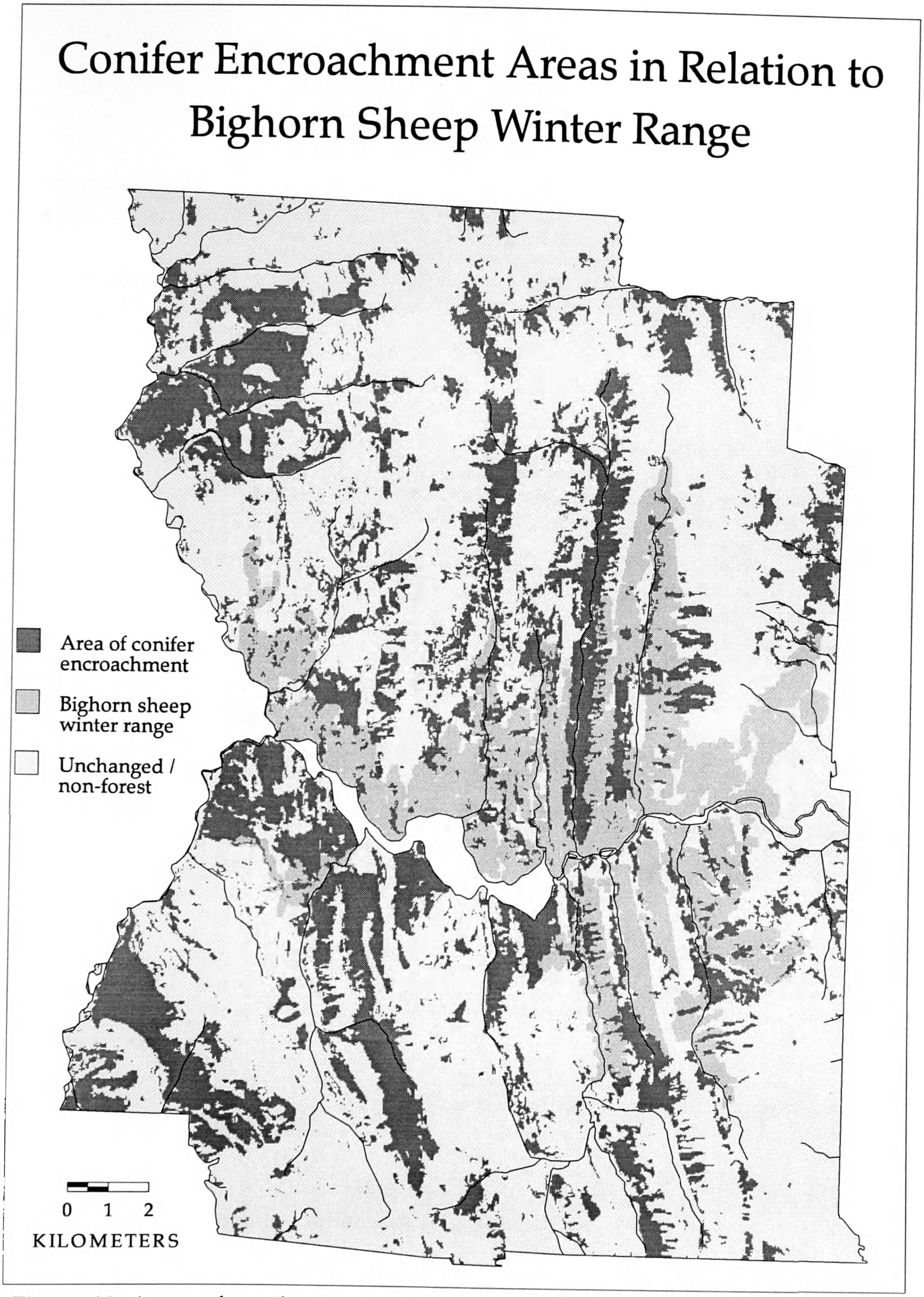

Figure 11. Areas of conifer encroachment, classified as open cover-types in 1937, but as high-density conifer in 1992, shown in relation to bighorn sheep winter range patches larger than 150 ha.

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Appendix A.
Land-cover types and canopy closure classes used to classify Landsat TM imagery

1000 Urban & Developed Land
2000 Agriculture
3101 Foothills Grassland
3102 Disturbed Grasslands
3103 Herbaceous Clearcut
3104 Subalpine Meadow
3201 Mesic Upland Shrubland
3202 Warm Mesic Shrubland
3203 Cold Mesic Shrubland
3301 Curlleaf Mtn Mahogany
3302 Gambel Oak
3303 Skunkbrush Sumac
3304 Bitterbrush
3305 Mountain Big Sagebrush
3306 WY Big Sagebrush Steppe
3307 Basin Big Sage Shrubland
3308 Black Sagebrush Steppe
3309 Silver Sage
3310 Salt-Desert Shrub
3311 Greasewood
3312 Rabbitbrush
3313 Creeping Juniper
3314 Shrub-Dominated Clearcut
3401 Other Shrubland
4101 Aspen
4102 Broadleaf Forest
4201 Engelmann Spruce
4203 Lodgepole Pine
4205 Limber Pine
4206 Ponderosa Pine & Savannah
4207 Grand Fir
4208 Subalpine Fir
4210 Western Red Cedar
4211 Western Hemlock
4212 Douglas Fir
4213 Pinyon-Juniper
4214 Rocky Mountain Juniper
4215 Western Larch
4219 Mixed Alpine Forest
4220 Mixed Subalpine Forest
4221 Mixed Mesic Forest
4222 Mixed Xeric Forest
4223 Douglas Fir-Lodgepole
4224 Burnt Timber Stands
4225 Douglas Fir - Grand FIr
4226 Western Red Cedar-Grand Fir
4227 Western Red Cedar- Hemlock
4228 Western Larch-Lodgepole
4229 Western Larch-Douglas Fir
4301 Mix Needleleaf-Broadleaf
5000 Water
6101 Needleleaf Dominated Rip
6102 Broadleaf Dominated Ripar
6103 Needleleaf-Broadleaf Rip
6104 Mixed Riparian
6201 Grass-Forb Riparian/Wetl
6202 Shrub Riparian/Wetland
6203 Mixed Non-Forest Riparian
7100 Dry Salt Flats
7200 Sandy Areas, Blowouts
7300 Exposed Rock
7400 Barren Tundra
7500 Mines,Quarries,Gravel Pit
7600 Badland Breaks
7700 Clearcut
7800 Mixed Barren Land
7900 Shoreline & Gravel Bars
8100 Alpine Tundra
9100 Perennial snowfields
9101 Permanent Snow
9200 Glaciers
9800 Cloud
9900 Cloud Shadows

Canopy closure classes:

0	No Canopy
1	0-39% Canopy Closure
2	40-69% Canopy Closure
3	70-100% Canopy Closure

Appendix B. Description of FRAGSTATS Metrics (McGarigal and Marks 1994)

Class Area

Class area, expressed in hectares, measures the area of all patches within a given land-cover class. The resolution of area metrics is limited by the minimum mapping units of the landscape being investigated.

Percent of Landscape

Percent of landscape is calculated by dividing the class area by the total landscape area and multiplyed by 100.

Largest Patch Index (LPI)

For a land-cover class, the Largest Patch Index (LPI) is calculated by dividing the area of the largest patch in the class being considered divided by the total landscape area and multiplied by 100 to make it a percent. For the landscape as a whole, LPI is the largest patch in the landscape divided by the total landscape area multiplied and by 100.

Patch Density

For a land-cover class, patch density is the number of patches per 100 hectares in a particular class. For the landscape as a whole, patch density is the total number of patches in the landscape divided by the total landscape area standardized to 100 hectares.

Mean Patch Size (MPS)

At the class level, Mean Patch Size (MPS) is the sum of the areas of all patches in the land-cover class being considered divided by the number of patches in the class. This area metric is expressed in hectares. At the landscape level, MPS is the summed area of all patches in the landscape divided by the total number of patches in the landscape.

Patch Size Standard Deviation

Patch Size Standard Deviation (PSSD) is the population standard deviation of MPS at the class or landscape level.

Patch Size Coefficient of Variation

Patch size coefficient of variation is the PSSD divided by the MPS for the patch type or landscape being investigated.

Mean Proximity Index (MPI)

The mean proximity index (MPI) measures the relative isolation and fragmentation of a particular patch type across the landscape. Within a user specified search radius (5000 m for this study), the index is the mean of the sum of the area of all patches that are in the same class as a focal patch divided by the square of the distance from the focal patch for all the patches in a class. The index has no units and is thus relative, and useful only for comparing different landscapes. This metric differentiates sparse groupings of small habitat patches from landscapes where the patch type being considered forms a cluster of larger patches.

Mean Nearest Neighbor Distance (MNND)

Nearest neighbor distance (MND), at the patch level, represents the distance from a focal patch to its nearest neighbor in the same class. This is calculated as the distance between their nearest edges. For a land-cover class, Mean Nearest Neighbor Distance (MNND) is the sum of all the NNDs for a particular patch type divided by the number of patches in that class. If the proportion of land-cover types are identical in the two landscapes, the MNND should be lower for the landscape comprised of smaller patches. (Gustafson and Parker 1992).

Nearest Neighbor Standard Deviation

The population standard deviation of MNND.

Nearest Neighbor Coefficient of Variation

The population standard deviation of MNND divided by the MNND of the patch type being considered multiplied by 100 to convert it to a percent.

Interspersion and Juxtaposition Index (IJI)

The Interspersion and Juxtaposition Index (IJI) measures patch type interspersion. Landscapes in which patch types are well interspersed will have higher values than those in which patch types are poorly interspersed. See (McGarigal and Marks (1994) for the formula used to calculate this index.

Contagion Index

Contagion measures the spatial dispersion as well as intermixing of different patch types. Landscapes in which patch types are aggregated or clumped will have a higher value than landscape in which patches are will mixed. See (McGarigal and Marks (1994) for the formula used to calculate this index.