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Research Article

Cumulative effects of roads and logging on landscape structure in the San Juan Mountains, Colorado (USA)

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Abstract

In the southern Rocky Mountains of temperate North America, the effects of Euro-American activities on disturbance regimes and landscape patterns have been less ubiquitous and less straightforward in high-elevation landscapes than in low-elevation landscapes. Despite apparently little change in the natural disturbance regime, there is increasing concern that forest management activities related mainly to timber harvest and to the extensive network of roads constructed to support timber harvest, fire control, and recreation since the late 1800s have altered disturbance regimes and landscape structure. We investigated the magnitude of change in landscape structure resulting from roads and logging since the onset of timber harvest activities in 1950. We found limited evidence for significant impacts in our study area when all lands within the landscape were considered. The relatively minor changes we observed reflected the vast buffering capacity of the large proportion of lands managed for purposes other than timber (e.g., wilderness). Significant changes in landscape structure and fragmentation of mature forest were, however, evident on lands designated as suitable timberlands. Roughly half of the mature coniferous forest was converted to young stands; mean patch size and core area declined by 40% and 25%, respectively, and contrast-weighted edge density increased 2- to 3-fold. Overall, roads had a greater impact on landscape structure than logging in our study area. Indeed, the 3-fold increase in road density between 1950–1993 accounted for most of the changes in landscape configuration associated with mean patch size, edge density, and core area. The extent of area evaluated and the period over which change was evaluated had a large impact on the magnitude of change detected and our conclusions regarding the ecological significance of those changes. Specifically, the cumulative impact on landscape structure was negligible over a 10-year period, but was notable over a 40-year period. In addition, the magnitude of change in landscape structure between 1950–1993 varied as a function of landscape extent. At the scale of the 228 000 ha landscape, change in landscape structure was trivial, suggesting that the landscape was capable of fully incorporating the disturbances with minimal impact. However, at intermediate scales of 1000–10 000 ha landscapes, change in landscape structure was quite evident, suggesting that there may be an optimal range of scales for detecting changes in landscape structure within the study area.

Introduction

Euro-American activities have altered the disturbance regime of many western forest landscapes, resulting in substantial changes in landscape structure and function (e.g., Baker 1992; Wallin et al. 1996; Baisan and Swetnam 1997; Agee 1999). However, in the southern Rocky Mountains of temperate North America these effects have been less ubiquitous and less straightforward in high-elevation landscapes than in low-elevation landscapes. Despite apparently little change in the natural disturbance regime in these high-elevation landscapes (e.g., Romme and Despain 1989; Bessie and Johnson 1995; Weir et al. 1995; Schmid and Mata 1996), other human activities since the late 1800s have clearly altered disturbance regimes and landscape structure (Hejl et al. 1995; Miller et al. 1996; Reed et al. 1996a,b; Tinker et al. 1997). These activities are related mainly to timber harvest and to the extensive network of roads constructed to support timber harvest, fire control, and recreation.

Of all the novel kinds of disturbances that humans have introduced in high-elevation forests of the southern Rocky Mountains during the last century, roads may be the most ubiquitous and significant long-term legacy of our activities. Roads are unprecedented features in the ecological history of these landscapes (Forman 1995), and potentially affect many ecological processes (Forman and Alexander 1998; Trombulak and Frissell 2000). Roads may increase soil erosion, sedimentation, and landslides (Norse et al. 1986). Roads may influence fire regimes through increased fire ignition as a result of human activities that occur in the transportation corridor (Franklin and Forman 1987), reduced fire size as a result of physical barriers to fire movement (Norse et al. 1986; Covington and Moore 1992), and increased accessibility for fire suppression activities. Roads are linear landscape features that can create high-contrast edges and bisect patches. Consequently, roads can cause greater fragmentation of habitats than the direct loss of habitat from logging activities (Reed et al. 1996b; Tinker et al. 1997). Roads may serve as a conduit for the movement of organisms across the landscape, including the spread of alien weeds and predators (Benninger-Traux et al. 1992), or as a barrier or filter that prevents or impedes the movement of some sensitive species (Forman 1995). Roads may provide a direct source of mortality for organisms, particularly slow-moving terrestrial vertebrates such as many amphibians (Rosen and Lowe 1994; Fahrig et al. 1995). Roads also provide a transportation system for humans, thereby facilitating the spread of potentially disturbing human activities throughout the landscape (Rost and Baily 1979; Lyon and Jensen 1980; McClellen and Shackleton 1988). Despite the ubiquitous nature of roads and their disproportionate influence on landscape structure and function, there is a paucity of research on road impacts in these landscapes (Miller et al. 1996).

Roads are just one factor in the much broader issue of anthropogenic habitat loss and fragmentation, an issue of increasing interest and concern to landscape ecologists, managers, and conservationists (Rochelle et al. 1999; Knight et al. 2000). Forest fragmentation has received considerable research attention in many regions of North America (e.g., Whitcomb et al. 1981; Robbins et al. 1989; Lehmkuhl and Ruggiero 1991; McGarigal and McComb 1995; Schmiegelow et al. 1997; Trzcinski et al. 1999; Villard et al. 1999). However, we are in the earliest stages of understanding the patterns, processes, and ecological significance of forest fragmentation in the southern Rocky Mountain region (Knight et al. 2000). It is not clear, for example, how the native biota responds to anthropogenic changes in landscape patterns caused by logging and road-building. This difficulty is exacerbated because the Rocky Mountain landscapes are inherently very heterogeneous – a result of steep natural gradients in elevation, topography, and substrate – and forests in this region tend to be somewhat patchy even in the absence of human alterations (Hejl 1992).

The concern over road and logging impacts is especially acute for land management agencies that have a mandate to maintain native biodiversity at multiple scales (e.g., genetic, species, and landscape) while simultaneously managing lands for commodity production, recreation, and other objectives. The problem has taken on increased importance under the new paradigm of ecosystem management, which emphasizes sustainable ecosystems and the conservation of whole systems for a variety of purposes (Grumbine 1994; Samson and Knopf 1996; Boyce and Haney 1997; Kohm and Franklin 1997; Vogt et al. 1997). Understanding whether human activities have caused ecosystems or landscapes to move outside their range of natural variation and the ecological impacts of such departures has become a major challenge for land management agencies (Franklin 1997).

The purpose of this study was to determine the extent to which road-building and logging activities during the past half-century have altered the overall landscape structure, and especially the patch structure of mature forests, within a representative portion of the southern Rocky Mountains, where baseline descriptive information of this kind is currently unavailable. In addition, we also sought to assess the degree to which the computed changes in landscape structure were a function of the scale and context of the analysis. It was beyond the scope of this study to deal rigorously with the functional implications of the structural patterns we have documented, but we are addressing those issues in ongoing companion studies.

Methods

Study area

We identified a representative area of high-elevation forests on the Pagosa District of the San Juan National Forest in southwestern Colorado, USA. Initially, we located all lands above 2400 m in order to focus on lands potentially supporting aspen, mixed-conifer, and spruce-fir forests (scientific names given below). The resulting map contained several 'islands' of high-elevation terrain surrounded by lower-elevation terrain in the foothills of the southern and western portions of the District and, conversely, several 'fingers' of lower-elevation terrain projecting up valleys into the high-elevation portion of the landscape. To derive a single, contiguous study area encompassing 228 482 ha and representing the high-elevation landscape of the Pagosa District (Figure 1), we eliminated most of the 'interfingering' area to the southwest.

The study area occupies the approximate geographic center of the San Juan and South San Juan Ranges in the southern Rocky Mountains Province (Blair 1996). It contains a large portion of the Weminuche and South San Juan Wilderness areas west of the continental divide. Elevation ranges from 2400 m in the valley bottoms to over 4200 m on mountain peaks along the continental divide. The study area includes three major subwatersheds of the San Juan River, a major tributary of the Colorado River. The geology of the area is quite complex. Parent materials date from ancient Precambrian rocks to recent alluvial deposits (Campbell and Brew 1996; Ellingson 1996a). Current landforms were created by a variety of geomorphological processes, including plate tectonics, volcanism, glaciation, and erosion (Blair 1996; Brew 1996; Ellingson 1996b). The climate varies significantly in relation to the pronounced elevational and topographic gradients. Temperatures range from an average high of 23 °C in July to an average low of −18 °C in January. Precipitation ranges from a mean of more than 152 cm on the highest peaks to less than 51 cm in the lower reaches of the study area, and usually falls in late summer (July and August) and winter (January through March), although there may be significant local variation (Keen 1996).

Seven major vegetation types of ecological and economic significance occur within the study area. Each of these types has a unique ecological setting and history (Romme et al. 1992; Floyd-Hanna et al. 1996; Jamieson et al. 1996; Somers and Floyd-Hanna 1996; Spencer and Romme 1996), as well as distinctive human impacts and changes since EuroAmerican settlement. At the lowest elevations, the vegetation is dominated by pinon-juniper woodlands (primarily *Pinus edulis* and *Juniperus osteosperma*) and various kinds of grasslands. At the foothills and on tops of broad plateaus and mesas, the vegetation ranges into Ponderosa Pine (*P. ponderosa*) forest interspersed with shrub-dominated stands (Petran chaparral dominated by *Quercus gambelii*). The middle slopes are covered by a mosaic of mixed conifers (*P. ponderosa*, *Psuedostuga menziesii*, *Abies concolor*, *Picea pungens*) and quaking aspen stands (*Populus tremuloides*), broken by occasional meadows and grasslands. The highest elevations contain extensive spruce-fir forests (primarily *P. engelmannii* and *A. lasiocarpa*), subalpine meadows, and treeless alpine communities on the highest peaks. Running through all these types are riparian woodlands and meadows along the borders of perennial rivers and streams.

Prior to European settlement in the late 1800s, landscape dynamics were driven primarily by the patterns of wildfire, which varied dramatically with vegetation type. The median fire interval was only 10–20 years in the lower elevation ponderosa pine type; 20–30 years in the dry mixed-conifer type; 50–100 years in the aspen type; and >100 years in the spruce-fir type (Romme et al. 1998). Many individual stands escaped fire for far longer than the median return interval and some burned at shorter intervals, creating a complex vegetation mosaic at the landscape scale. Under this 'natural' disturbance regime, stand replacement fires initiated stand development and maintained a coarse-grain mosaic of successional stages and cover types across the landscape, although other disturbance processes, such as landslides, floods, windthrow, insects and disease also played a role on a finer scale. In particular, the disturbance regime of individual stands in the later stages of development is dominated by chronic, fine-scale processes that kill individual trees or small groups of trees (Veblen et al. 1989; Lertzman and Krebs 1991; Veblen et al. 1991a, b; Roovers and Rebertus 1993).

Although limited logging by Euro-American settlers began as early as 1875, the scale and impact of logging increased dramatically in the late 1800s with the advent of railroad logging (Pearson 1950). Most of the activity was confined to the pine forest at lower elevations, such that by 1950, essentially all of the old-growth ponderosa pine forests of this region had been exploited and profoundly altered (San Juan Na-

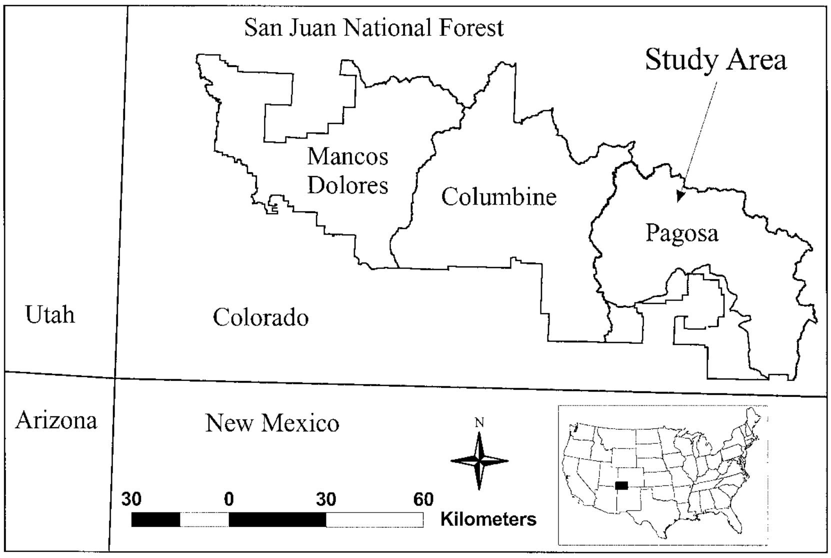

Figure 1. Study area location.

tional Forest 1962; Anonymous 1971). In contrast, extensive logging at the higher elevations generally did not begin until much later in the twentieth century. The first large-scale spruce logging operation began in 1946 (US Forest Service, unpublished data). Logging in spruce-fir and mixed-conifer forests was accelerated dramatically in the 1950s and reached a peak in the 1960s and 1970s. Logging was carried out using clearcutting and a variety of partial cutting methods. Clearcutting was discontinued by 1980 in all but aspen forests because of problems in regenerating clearcut stands.

Data development

We used the Common Vegetation Unit (CVU) and Road data layers from the Integrated Resource Inventory (IRI) database of the San Juan National Forest. The IRI database is managed using the Arc/Info geographic information system (ESRI 1995). The CVU layer was developed from a combination of information sources and processes, including the existing Resource Information System (RMRIS) polygon boundaries, aerial photo interpretation (1993, 1:24 000 natural color), digital image analysis of Landsat Thematic Mapper imagery, and logic written into a C program to delineate and attribute polygons. These digital vector maps contain polygons of relatively homogeneous vegetation composition and structure within a minimum area of 0.4 ha, and coverage is complete for the study area (i.e., there are no unclassified polygons within the study area boundary). Attributes for each polygon include species, percent crown cover, and tree size class, among other items. We reclassified polygons into 13 cover types based on species composition and size class (Table 1). Cover types correspond to the major vegetation types found within the region, as described previously, with a size class modifier to reflect seral stage of development in certain patch types (aspen, spruce-fir, and mixed-conifer). Hereafter, we refer to the large diameter size class as 'mature' forest. We dissolved the coverage on cover type and then in separate analyses eliminated polygons less than 1 ha and 4 ha. Preliminary findings indicated that the choice of resolutions (1 vs. 4 ha) did not influence the results or conclusions. Therefore, we present only the results of analyses based on the 1-ha resolution. The final coverage portrayed the condition of the landscape as it was in 1993.

We used archived timber atlases, timber sale reports, and oral histories to identify all polygons with a timber harvest history (Romme et al. 1998). Few written records exist of logging operations prior to 1950. However, we are confident that little activity occurred in high-elevation forests of the Pagosa District prior to this date, because nearly all of the roads that now exist in high-elevation areas were constructed during the last 50 years. Therefore, we used the year 1950 as a baseline and measured the changes that have occurred since that time. We used the timber harvest records to classify all polygons with a harvest history as either partial cuts or clearcuts based on the per-

Table 1. Cover types used to define landscape structure on the San Juan National Forest, Pagosa Ranger District. Cover types were defined using a combination of dominant overstory vegetation and average tree diameter for forested patch types. See text for scientific names.

Cover Type	Description
Vegetation Class	
Spruce-fir	Overstory vegetation dominated by Engelmann spruce and subalpine fir.
Mixed-conifer	Overstory vegetation dominated by a mixture of coniferous tree species, including Douglas-fir, white fir, ponderosa pine, Engelmann spruce, and subalpine fir, as well as aspen.
Aspen	Overstory vegetation dominated by aspen.
Other forest	Overstory vegetation dominated by trees other than those listed above; typically, ponderosa pine, pinyon pine, juniper, or cottonwood.
Riparian	All lakes, reservoirs, perennial streams, rivers, and wetlands, and the riparian vegetation associated with any of these water bodies and wetlands.
Shrubland	Overstory vegetation dominated by shrubs; typically, gamble oak and sage brush and other associated shrubs.
Nonforest	Nonforested areas, including areas dominated by graminoids and other herbaceous plants, talus slopes, and other barren rock surfaces.
Roads	All types of roads, including primary, secondary, and tertiary roads (including abandoned roads) containing paved, gravel, and dirt surfaces.
Partial cuts	Areas that have been subject to a commercial partial harvest, including shelterwoods and any other harvest method involving partial overstory removal.
Clearcuts	Areas that have been clearcut of overstory trees.
Size Class[1]	
Large	Average overstory tree >22.86 cm (9 in) d.b.h.
Medium	Average overstory tree 12.70–22.86 cm (5-9 in) d.b.h.
Small	Average overstory tree <12.70 cm (5 in) d.b.h.

[1] Size classes apply to the sprucefir, mixed-conifer, and aspen cover types.

centage of the overstory removed. In this manner, we created a second 1993 coverage in which partial cuts and clearcuts were added as two new cover types to the 13 original classes. This map portrayed the condition of the landscape as it was in 1993. In this map, the class ‘large-diameter spruce-fir’ represents natural stands only; partially cut spruce-fir patches are classified separately, even though some still contain some large diameter residual trees.

In many partial cuts, residual canopy cover was great enough that the stands still appeared as large-diameter forests on aerial photos and satellite images, and were so classified in the Forest Service database. Therefore, we also measured landscape structure using the unmodified database, i.e., not treating harvested stands as separate cover types. However, the analyses using new cover types for harvested stands may provide a better picture of actual landscape changes. This is because all of the partially harvested stands do look different from natural stands when one visits them on the ground, and most of the partially logged stands are scheduled for future removal of the residual overstory.

From the timber sale records, we were able to determine when each unit was harvested, as well as the cover type prior to harvest. By projecting back in time, we generated a cover type map for the beginning of each decade from 1950 to 1990. The 1950 map portrays the condition of the landscape as it was in 1950 just prior to the onset of widespread logging in the mid to high elevations of the study area. The 1960 map portrays the impacts of all logging and road-building activities occurring prior to 1960, and so on for each subsequent decade. Hence, each coverage in the sequence represents the cumulative change in landscape structure due to logging activities. Because there were no major fires or other coarse-scale natural disturbances between 1950–93, we assumed that there were minimal cover type changes due to natural succession on unharvested lands.

The Roads data layer was developed using aerial photographs (1:24 000 natural color) and the U.S. Dept. of Agriculture’s Cartographic Feature File (CFF) Code designations to classify road types (e.g., primary, secondary, and tertiary – paved or gravel). We buffered all roads using a distance of 12.5 m to obtain linear polygons 25 m wide for each road. We recognize that not all roads are the same size; they vary from single-lane dirt and gravel forest roads, to large paved roads with significant cleared rights-of-way (Baker and Knight 2000). It follows that all road edges are therefore probably not the same in magnitude and effect. Information identifying the specific type of each road was lacking, so all roads and areas of road influence were treated the same for analysis purposes. The choice of the road buffer width reflects the average width of road types within the study area (US Forest Service, unpublished data), but also represents the minimum width consistent with the resolution of the raster data used to compute certain landscape structure metrics (see below). The importance of roads in quantifying landscape structure was examined by overlaying the road map on the cover type map, and treating roads as another cover type class. We overlaid the 1993 road map on both the 1993 and 1993-with-harvests cover type maps. We used a variety of archival sources of information, primarily oral histories, to determine when each road was constructed. By projecting back in time, we generated a road map for 1950 and overlaid this on the 1950 cover type map.

Roughly 84 percent of the study area presently cannot be logged because of administrative restrictions (e.g., wilderness, research natural areas) or because site conditions make it impractical (e.g., inaccessible, steep slopes) or inappropriate (e.g., nonforested cover). This land is referred to as 'nonsuitable' timberland; all remaining land (37 045 ha) is referred to as 'suitable' timberland. To assess the effects of roads and logging activities on landscape structure within only that portion of the landscape available for logging, we overlaid a map of the suitable timberlands with each of the cover type maps described above and treated all nonsuitable timberlands as background (i.e., ignored it in the analysis of landscape pattern, described below). Because suitable timberland is distributed discontinuously across the landscape, the structure of these landscapes is artificially disjunct. Thus, the landscape metrics described below based on the spatial character of the patches (i.e., patch size, patch shape, core area, and nearest neighbor distance) are somewhat biased by these administrative edges.

To assess the effects of spatial scale (specifically, landscape extent) on the magnitude of change in landscape structure, we selected a single timber sale area (i.e., cluster of harvest units) in each of the three major watersheds that comprise the study area (Piedra, San Juan, and Blanco Rivers). For each watershed, we delineated 3 concentric, circular sampling areas using 1.0, 2.0, and 5.0 km radii, roughly centered on the geographic centroid of the harvest units comprising the timber sale (Figure 2). We overlaid these sampling areas on the 1950 and 1993 cover type maps with roads and harvests treated as separate cover types.

Data analysis

We analyzed landscape structure using the program FRAGSTATS (McGarigal and Marks 1995). Structure was quantified using a parsimonious set of metrics representing somewhat independent components of landscape structure (Table 2; Li and Reynolds 1995; Ritters et al. 1995; Hargis et al. 1997). We computed these metrics at both the landscape and class (or cover type) levels. Landscape metrics represent the overall diversity and heterogeneity of the landscape; class metrics, represent the composition and configuration of the landscape relative to each cover type. The latter are often interpreted as fragmentation indices. For purposes of this investigation, we focused our fragmentation analysis on the three major forest cover types that are of particular conservation concern in high-elevation Rocky Mountain landscapes: large-diameter spruce-fir, large-diameter mixed-conifer, and large-diameter aspen; Table 1).

We computed the total core area index using a 50-m edge effect distance. The core area of a patch is the area remaining after removing the area of edge influence, which is defined by buffering the patch with the specified edge effect distance inward from the patch boundary. This edge depth was chosen because empirical studies have demonstrated consistent edge effects approximately two to three tree heights into the surrounding forest habitat in coniferous forests surrounded by clearcut openings (Temple 1986; Chen and Franklin 1990; Chen et al. 1992 and 1993; Vaillancourt 1995). These studies suggest that edge effects may extend much more deeply than 50 m into a patch. Similarly, edge effect distances along roads vary considerably from several meters to several hundred meters (e.g., Baker and Knight 2000; Forman 2000; Forman and Deblinger 2000; Haskell 2000; Trombulak and Frissell 2000). Therefore, a 50-m depth of edge influence is a relatively conservative estimate of edge effect. For comparison, we analyzed the 1950 and 1993 landscapes with and without roads, and with harvests treated as separate cover types, using a 100-m depth of edge influence as well.

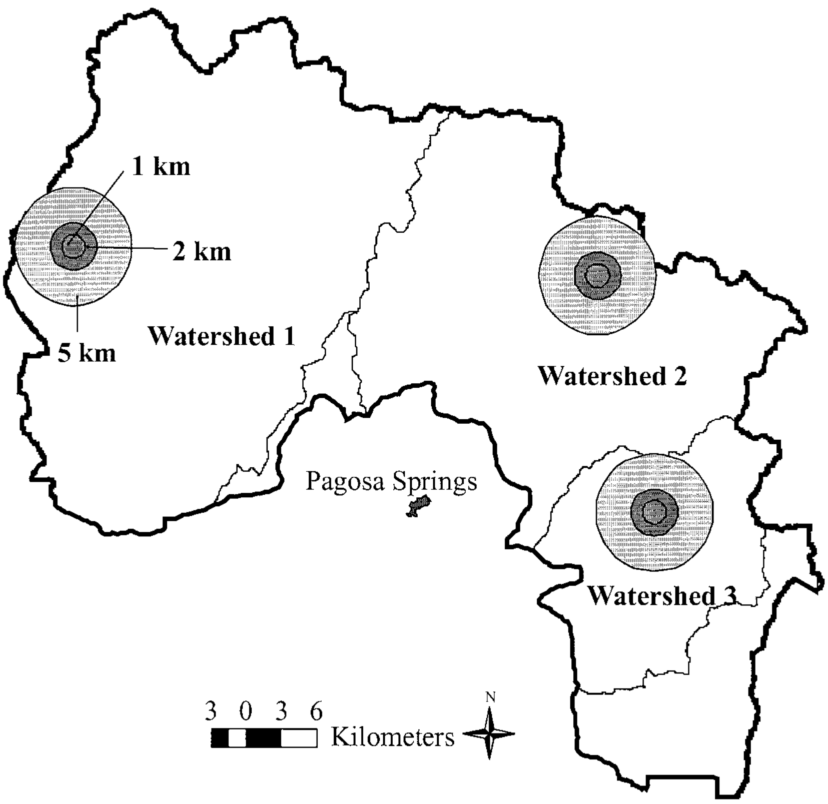

Figure 2. Study area outline (bold line) and the 3 major watersheds. Each watershed contains concentric circular sampling areas (1-, 2-, and 5-km radii) centered on a timber sale area and used to investigate the relationship between landscape extent and landscape structure metrics.

Table 2. Landscape metrics used to quantify changes in (1) landscape structure and (2) the area and configuration of late-seral spruce-fir forest between 1950 and 1993. Metric terminology is consistent with McGarigal and Marks (1995).

Landscape Metric (units)	Description[a]
Percent of landscape (%)	Percentage of the landscape comprised of the corresponding class.
Mean patch size (ha)	Average size of the patches comprising the landscape or class.
Mean patch shape index (none)	Average patch shape complexity for patches comprising the landscape or class; equals 1 when all patches are circular and increases as patches become noncircular.
Contrast-weighted edge density (m/ha)	Density of edge weighted by the degree of structural and floristic contrast between adjacent patches; equals unweighted edge density when all edge is maximum contrast, and approaches 0 when all edge is minimum contrast (see text for details).
Total core area index-50 and -100 (%)	Total percentage of the landscape that is greater than 50 m or 100 m, respectively, from the nearest patch edge.
Mean nearest neighbor (m)	Average distance between patches of the same class based on edge-to-edge distance.
Simpson's diversity index (none)	Simpson's diversity index (Simpson 1949) combines patch richness and evenness into a single index, where the value represents the probability that any two patches drawn at random will represent different patch types.

[a]See McGarigal and Marks (1995) for a more detailed description or visit the Fragstats website (www-unix.oit.umass.edu/~fragstat/).

We computed contrast-weighted edge density by weighting each edge segment by the degree of contrast along that edge. Edge contrast was defined using weights ranging between 0 and 1, with increasing weights representing greater contrast. We defined edge contrast on the basis of floristic and structural differences between adjacent patch types. Specifically, each patch type was assigned a vertical structural class in order of increasing height of the dominant vegetation, including: no structure (e.g., roads), herbaceous (e.g., nonforest), shrub/small trees, medium trees, and large trees. An edge received a weight of 0.2 for each structural class difference along this sequence. Thus, a medium-large tree edge received a weight of 0.2; whereas, a shrubland-large tree edge received a weight of 0.4. In addition, a change in plant community between any combination of nonforest, other forest, riparian, shrubland, aspen, mixed-conifer, and spruce-fir cover types received an additional weight of 0.2. Thus, the edge between a large spruce-fir patch and a nonforest patch received a weight of 0.2 for the plant community change, plus an additional weight of 0.6 for the vertical structural difference, for a total contrast weight of 0.8. As defined, contrast-weighted edge density represents the density of equivalent maximum-contrast edge; that is, the density of edge that would be present if it were all maximum contrast. It is important to note that edge contrast as we defined it may or may not correspond to the degree of edge contrast perceived by any particular organism. Given the impossibility of defining edge contrast from an organism-centered perspective, we chose to define edge contrast based on general differences that are likely to be important to a wide variety of organisms.

We computed mean nearest-neighbor distance at the class level for mature forest cover types by converting vector cover type maps into raster maps using a 25-m cell size, and then computing the nearest patch edge-to-edge distance between nearest neighbors. This cell size ensured that linear road and riparian polygons were not broken up into many small patches as an artifact of the rasterization process. This was reasonable given that most riparian zones (and roads as noted previously) in the study area exceed 25 m in width. In this case, we used the eight-neighbor tracing procedure, which considers any two adjacent pixels to be a part of the same patch even if they only share a common corner, for patch definition. This insured that the rasterization process would not result in the artificial subdivision of patches, and yet also insured that patches would be bounded by linear features such as roads and streams.

Results

Landscape changes: 1950–1993

Landscape composition

Throughout the past 50 years, this high elevation landscape was dominated by mature mixed-conifer and spruce-fir forests (Table 3). These two cover types comprised almost half of the total area and were interspersed with roughly an equal mixture of nonforest (primarily alpine meadows and barren rock surfaces at higher elevations), other forest (primarily ponderosa pine at lower elevations), shrubland (primarily Gambel oak at mid and lower elevations), and aspen forest (at mid elevations). Approximately 2% of the area as mapped was classified as riparian. Despite more than 40 years of logging activities, there has been a paucity of early seral stages in the major forested cover types; in 1993, only 2% of the landscape was comprised of medium and small diameter trees in the spruce-fir, mixed-conifer, and aspen cover types (Table 3).

Despite the fact that almost 10% of the landscape was subjected to some form of timber harvesting – roughly 8.5% of the study area was partially harvested, while less than 1% was clearcut – overall landscape composition changed very little between 1950–1993 (Table 3). When we did not classify partially cut patches as new cover types, roads were the only cover type that changed in proportional coverage by more than two percent, increasing from 1.1% of the landscape in 1950 to 3.3% of the landscape in 1993. Treating partial cuts and clearcuts as separate cover types produced decreases in the amount of natural, mature mixed-conifer forest (4% decrease) and other forest (5% decrease), reflecting the preponderance of timber harvest activities in these cover types; natural, mature spruce-fir and aspen cover types decreased by only 1.5% and 0.9%, respectively.

Changes in landscape composition were more notable when only suitable timberlands were considered (Table 3). Roughly 34% of the suitable timberlands were partially cut; an additional 3% were clearcut. This resulted in substantial decreases in the proportional coverage of natural, mature mixed-conifer and other forest cover types as before, as well as a 7% and 4% reduction in natural, mature spruce-fir and aspen cover types, respectively. Regardless of how harvested

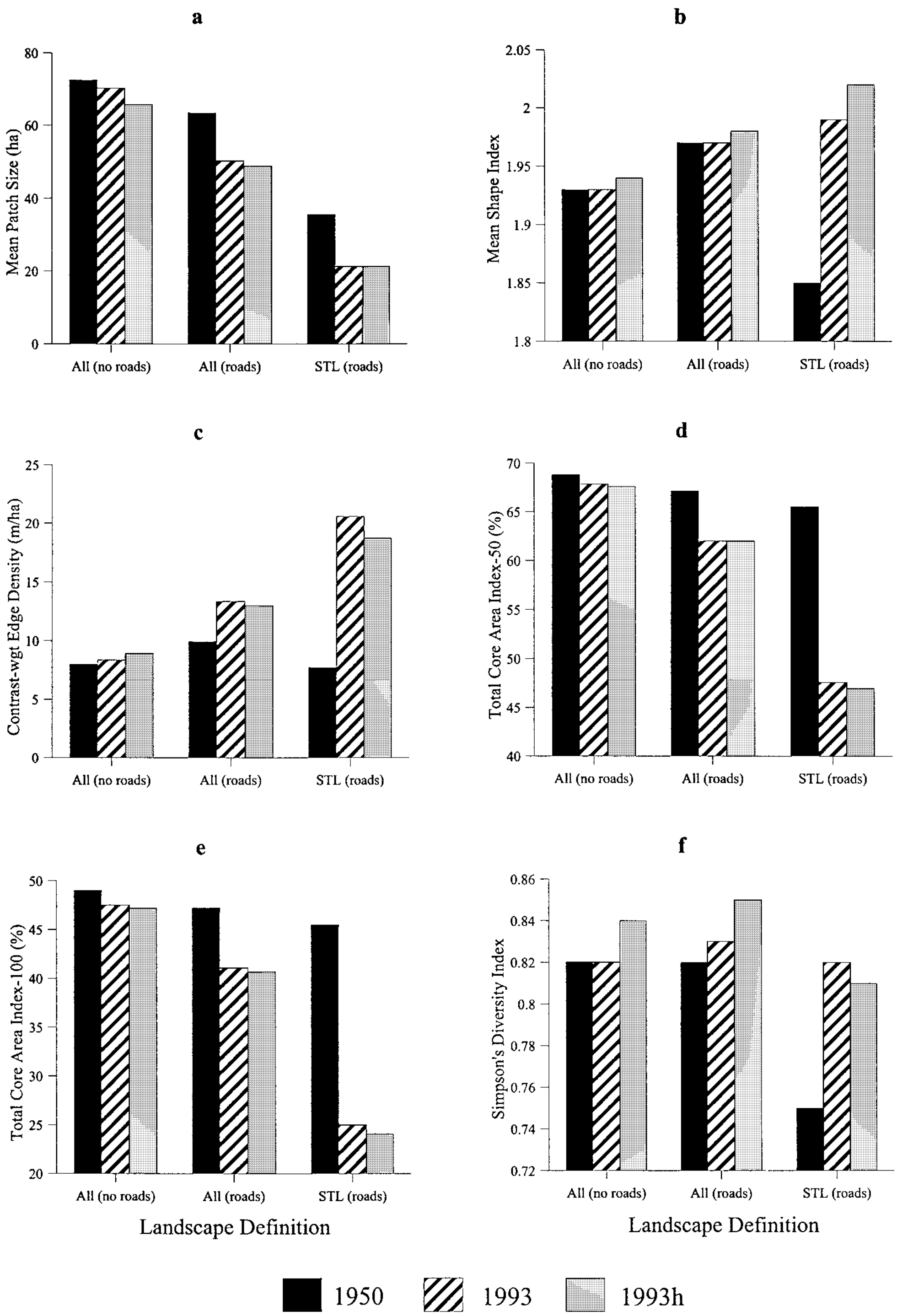

Figure 3. Changes in (a) mean patch size, (b) mean shape index, (c) contrasted-weighted edge density, (d) total core area index-50 (based on 50-m edge influence), (e) total core area index-100 (based on 100-m edge influence), and (f) Simpson's diversity index (see Table 2 for a description of metrics) between 1950 and 1993 for three landscape definitions: (1) all lands within the study area, but ignoring roads; (2) all lands within the study area, including roads (25-m wide buffered lines) as a separate cover type; and (3) suitable timber lands (STL) only, including roads as a separate cover type. In all cases, the 1993 landscape was defined with (1993h) and without (1993) treating harvested areas (clearcuts and partial cuts) as separate cover types.

Table 3. Changes in the percentage of the landscape (% Land) in each patch type between 1950 and 1993 for: (1) all lands and (2) for suitable timberlands only. Roads are included in all landscapes as a separate patch type (i.e., 25-m wide buffered lines). Percent change represents the difference in % Land between 1950 and 1993; bold values represent changes greater than 1 percent. Suitable timberlands include all lands designated by the US Forest Service as 'suitable' for timber harvest; it excludes lands protected administratively (e.g., wilderness, research natural areas) and lands deemed impractical (e.g., inaccessible, steep slopes) or inappropriate (e.g., nonforested cover) for timber harvest.

	All Lands					Suitable Timberlands Only				
Cover Type	1950	1993	Change	1993h	Change	1950	1993	Change	1993h	Change
Spruce-fir large	21.57	20.63	−0.94	20.01	**−1.57**	14.33	10.24	**−4.09**	7.01	**−7.32**
Spruce-fir medium	0.75	0.77	0.03	0.75	0.00	0.06	0.15	0.09	0.07	0.01
Spruce-fir small	0.09	0.09	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00
Mixed-conifer large	28.82	26.90	**−1.92**	24.75	**−4.07**	38.08	30.64	**−7.44**	20.46	**−17.62**
Mixed-conifer medium	0.80	1.13	0.32	0.93	0.13	0.26	1.27	**1.01**	0.36	0.10
Mixed-conifer small	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aspen large	7.67	7.51	−0.16	6.77	−0.90	13.13	12.31	−0.82	9.09	**−4.04**
Aspen medium	0.06	0.11	0.05	0.06	0.00	0.08	0.20	0.12	0.07	0.00
Aspen small	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other forest	12.32	10.51	**−1.81**	7.38	**−4.94**	25.15	19.82	**−5.33**	8.53	**−16.61**
Riparian	1.90	1.91	0.01	1.88	−0.02	0.88	0.96	0.08	0.84	−0.04
Shrubland	11.78	13.66	**1.88**	11.69	−0.09	5.92	12.67	**6.75**	5.82	−0.11
Nonforest	13.17	13.43	0.26	13.05	−0.13	0.78	2.37	**1.59**	0.76	−0.02
Roads	1.06	3.34	**2.28**	3.36	**2.30**	1.34	9.39	**8.05**	9.42	**8.08**
Partial cut	–	–	–	8.56	**8.56**	–	–	–	34.28	**34.28**
Clearcut	–	–	–	0.73	0.73	–	–	–	3.29	**3.29**
Total	100.00	100.00	0.00	100.00	0.00	100.00	100.00	0.00	100.00	0.00

[a] 1993h landscape includes harvested areas (i.e., clearcuts, partial cuts) as separate patch types. The distinction between clearcuts and partial cuts is based on the percentage of the overstory remaining intact after harvest (see text for details).

areas were considered, roads increased in coverage by more than 8%, comprising almost 10% of the suitable landscape in 1993.

Landscape configuration

Not considering roads or harvested areas as separate cover types, there were relatively minor changes in landscape configuration between 1950–1993 based on the suite of landscape metrics considered (Figure 3). For example, mean patch size decreased only 3% from 72 to 70 ha (Figure 3a), and contrast-weighted edge density increased only 4% from an average of 8 m of equivalent maximum-contrast edge per hectare in 1950 to 8.4 m/ha in 1993 (Figure 3c). Treating harvested areas as separate cover types resulted in greater changes, but overall the changes were still relatively minor. However, treating roads as a separate cover type resulted in more notable changes. For example, mean patch size decreased by 23% (Figure 3a), while contrast-weighted edge density increased by 30% (Figure 3c). Despite the significant fragmenting impact of roads (i.e., dividing otherwise contiguous patches and increasing edge), overall landscape diversity, as indexed by the Simpson's index, remained relatively little changed.

Perhaps the most noticeable impact of logging and roads was on the patch size distribution across the entire landscape, especially the dissection of the largest patches (Figure 4). In 1950, the landscape was dominated by several large patches, the largest patch encompassed 20 090 ha. By 1993, timber harvesting alone reduced the largest patch to 16 838 ha, and resulted in an increase in the number of patches from 3153 to 3478. Roads had an even greater effect; increasing the number of patches from 3603 to 4673, and reducing the largest patch to 10 300 ha.

Restricting the analysis to suitable timberlands had a major effect on the configuration metrics and the magnitude of change between 1950–1993 (Figure 3). Overall, suitable timberlands were characterized by much smaller patches and, at least initially, patches with simpler geometric shapes than the landscape as a whole. Throughout the period of investigation, mean patch size on suitable timberlands was less than half

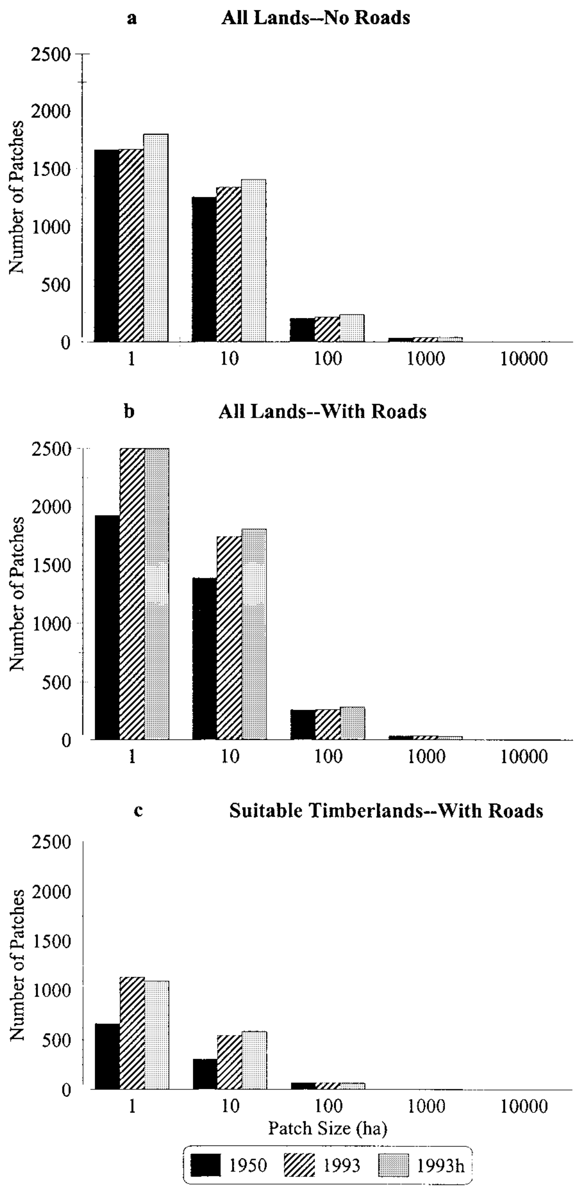

Figure 4. Changes in the frequency distribution of patch sizes between 1950 and 1993 for three landscape definitions: (a) all lands within the study area, but ignoring roads; (b) all lands within the study area, including roads (25-m wide buffered lines) as a separate cover type; and (c) suitable timber lands (STL) only, including roads as a separate cover type. In all cases, the 1993 landscape was defined with (1993h) and without (1993) treating harvested areas (clearcuts and partial cuts) as separate cover types.

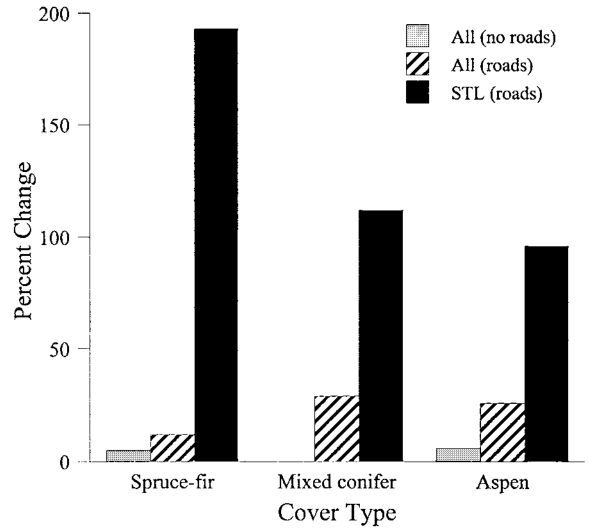

Figure 5. Percent change in contrast-weighted edge density (see Table 2 for a description) between 1950 and 1993 for the mature spruce-fir, mixed-conifer, and aspen cover types under three landscape definitions: (1) all lands within the study area, but ignoring roads; (2) all lands within the study area, including roads (25-m wide buffered lines) as a separate cover type; and (3) suitable timber lands (STL) only, including roads as a separate cover type.

of what it was over the full landscape extent, reflecting the influence of large contiguous tracts of high elevation forest in the natural portion of the landscape (e.g., in wilderness areas). In addition to the finer grain of the patch mosaic on suitable timberlands, these patches underwent much greater reduction in size (roughly 40%) as a result of timber harvesting and road-building activities between 1950–1993 than the landscape as a whole (Figure 3a). As the forest was fragmented, the number of patches increased from 1039 to 1741, and the largest patch decreased from 2156 ha to 1471 ha (Figure 4). The amount of high-contrast edge density increased most dramatically, ranging from 140–167% depending on how harvested areas were classified (Figure 3c). Mean patch shape index increased accordingly due to the additional edges created by roads and harvest units (Figure 3b). Decreasing patch size coupled with increasing patch shape complexity resulted in substantial reductions in the amount of core area (i.e., the portion of the landscape greater than 50 m from the nearest patch edge); total core area index decreased from 66% to 47%, indicating a loss of almost 20% of the interior forest environment on suitable timberlands (Figure 3d). Similarly, assuming a 100-m edge effect, the percentage of the landscape in core area decreased from 46% to 24%. Finally, landscape diversity appeared to increase markedly for two reasons. First, the addition of roads and/or harvested areas as cover types increased patch richness significantly. Second, the conversion of mature forest cover types, which were dominant in areal extent, to other cover types (i.e., early seral stages, other nonforest cover types such as shrublands, or harvest cover types) resulted in a more equitable distribution of area among cover types.

Mature forest fragmentation

During the past 50 years, mature forest constituted the matrix of this high elevation landscape, comprising over 70% of the study area (Table 3). Mature, coniferous forest (both spruce-fir and mixed-conifer) was distributed in large, contiguous patches 2- to 4-fold larger on average than the average patch; in contrast, the mature aspen forest was fragmented into much smaller patches. Despite these differences, the patterns of change in extent and configuration between 1950–1993 were similar for all three mature forest cover types (Figure 5). These patterns were exemplified by the changes in mature, spruce-fir forest (Figure 6). Specifically, the percentage of the landscape in mature, spruce-fir forest decreased relatively little between 1950–1993, slightly more than 1% depending on how harvested areas were treated (Table 3). Accordingly, its spatial configuration (i.e., fragmentation) changed very little as well, even when roads and harvested areas were treated as separate cover types. The only metrics that changed substantially were mean patch size and mean nearest neighbor, which decreased by as much as 20% depending on whether or not roads were treated as a separate cover type.

Not surprisingly, the changes were more notable when considering only suitable timberlands (Figures 5 and 6). The percent of the suitable landscape in natural, mature spruce-fir forest declined from 14% in 1950 to 7% in 1993; mean patch size decreased from 126 ha to 33 ha; and the amount of core area decreased by 27–47% depending on the edge influence distance (50 or 100 m). Despite these rather dramatic changes, the average amount of maximum-contrast edge per hectare increased very little in absolute terms (0.76 to 1.41 m). In addition, the mean nearest neighbor decreased by 28% from 440 m to 318 m. This decrease reflected the subdivision of natural stands by relatively small harvested units, which created several disjunct but nearby natural, mature spruce-fir stands.

Effects of scale

Temporal scale

Changes in landscape structure occurred gradually and cumulatively over the period of investigation. The greatest incremental change occurred during the 1960s and 70s, reflecting a period of intense logging and road-building activity. Almost without exception, landscape structure metrics changed by a trivial amount when evaluating a single decade of change, even when restricting the analysis to suitable timberlands where the changes were most dramatic (Figure 7). However, for some metrics, the magnitude of change was notable when evaluated over several decades. For example, considering only suitable timberlands, mean patch size declined by a few percent each decade between 1950–1990, yet declined by 13% (as a function of the 1950 baseline) when evaluated over the entire period. Similarly, with the exception of a single decade, contrast-weighted edge density increased by 5–20% each decade, yet increased by 126% over the entire 40-year period. These patterns were similar at the class level for each of the mature forest cover types (Figure 8).

Spatial scale

In each of the 3 watersheds where we systematically increased landscape extent, the structure of the landscape varied markedly in relation to extent. Specifically, as the size of the analysis area increased for any particular snapshot of the landscape (e.g., 1993 with roads and harvests as separate cover types), each metric varied in response to the increasing heterogeneity of the landscape mosaic at that broader scale. Given this sensitivity to scale, it is not surprising that the measured change in landscape structure between 1950–1993 at both landscape- and class-levels also varied as a function of extent. Figures 9 and 10 depict the change in each metric (landscape- and class-level, respectively) between 1950–1993 as a function of landscape extent for each of the 3 major watersheds in the study area. Two things are noteworthy in these figures. First, each metric demonstrates a unique behavior in relation to increasing landscape extent. This is not surprising, since each metric quantifies a slightly different component of landscape structure and is therefore responding to a different aspect of the landscape mosaic as extent varies. Second, in most cases, the metric exhibits a peak or threshold within a well-defined spatial scale. For example, at the landscape level (Figure 9), mean patch size, mean

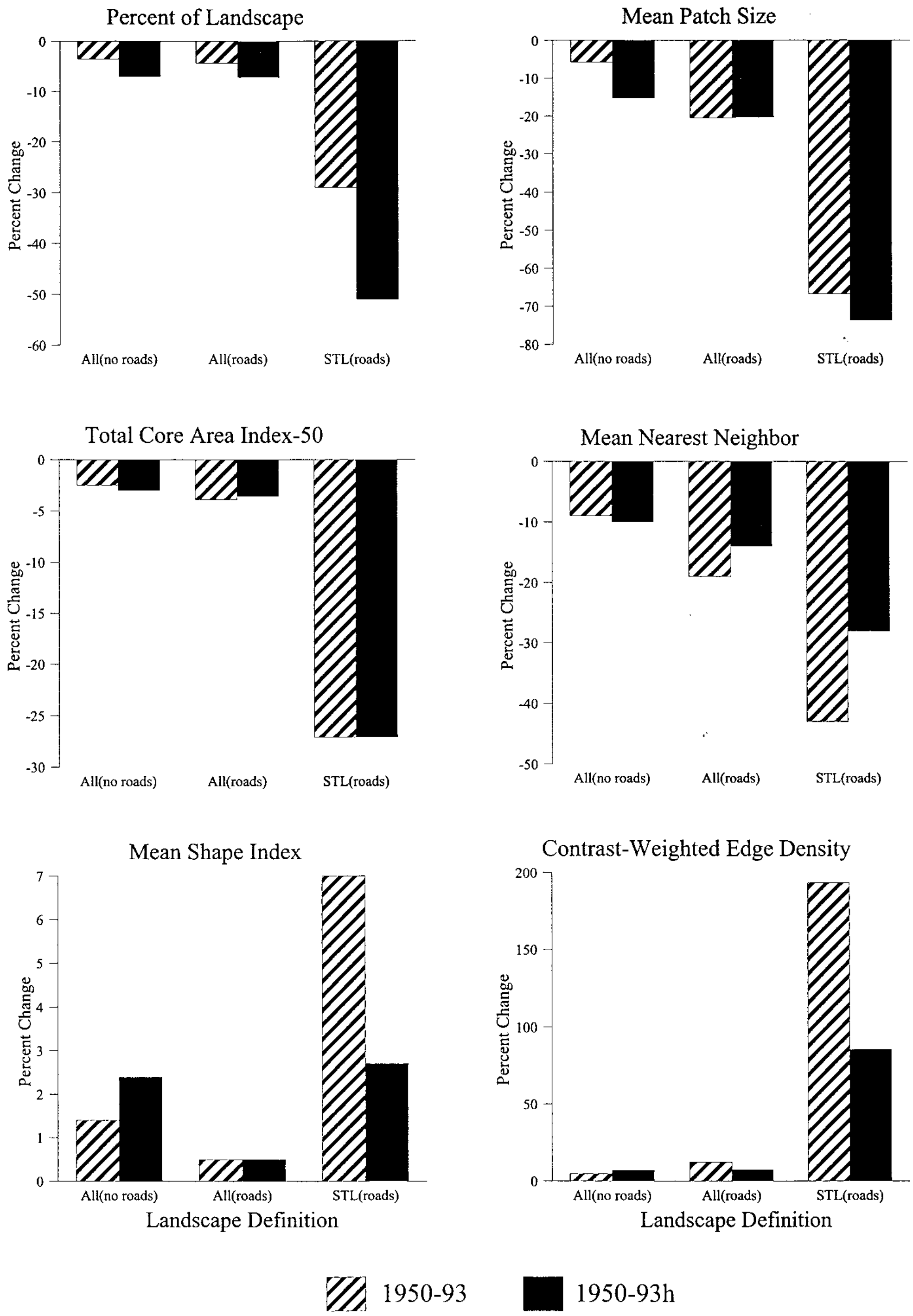

Figure 6. Percent change in several landscape metrics (see Table 2 for a description of metrics) between 1950 and 1993 for the mature spruce-fir cover type under three landscape definitions: (1) all lands within the study area, but ignoring roads; (2) all lands within the study area, including roads (25-m wide buffered lines) as a separate cover type; and (3) suitable timber lands (STL) only, including roads as separate cover types. In all cases, the 1993 landscape was defined with (1993h) and without (1993) harvested areas (clearcuts and partial cuts) as separate cover types.

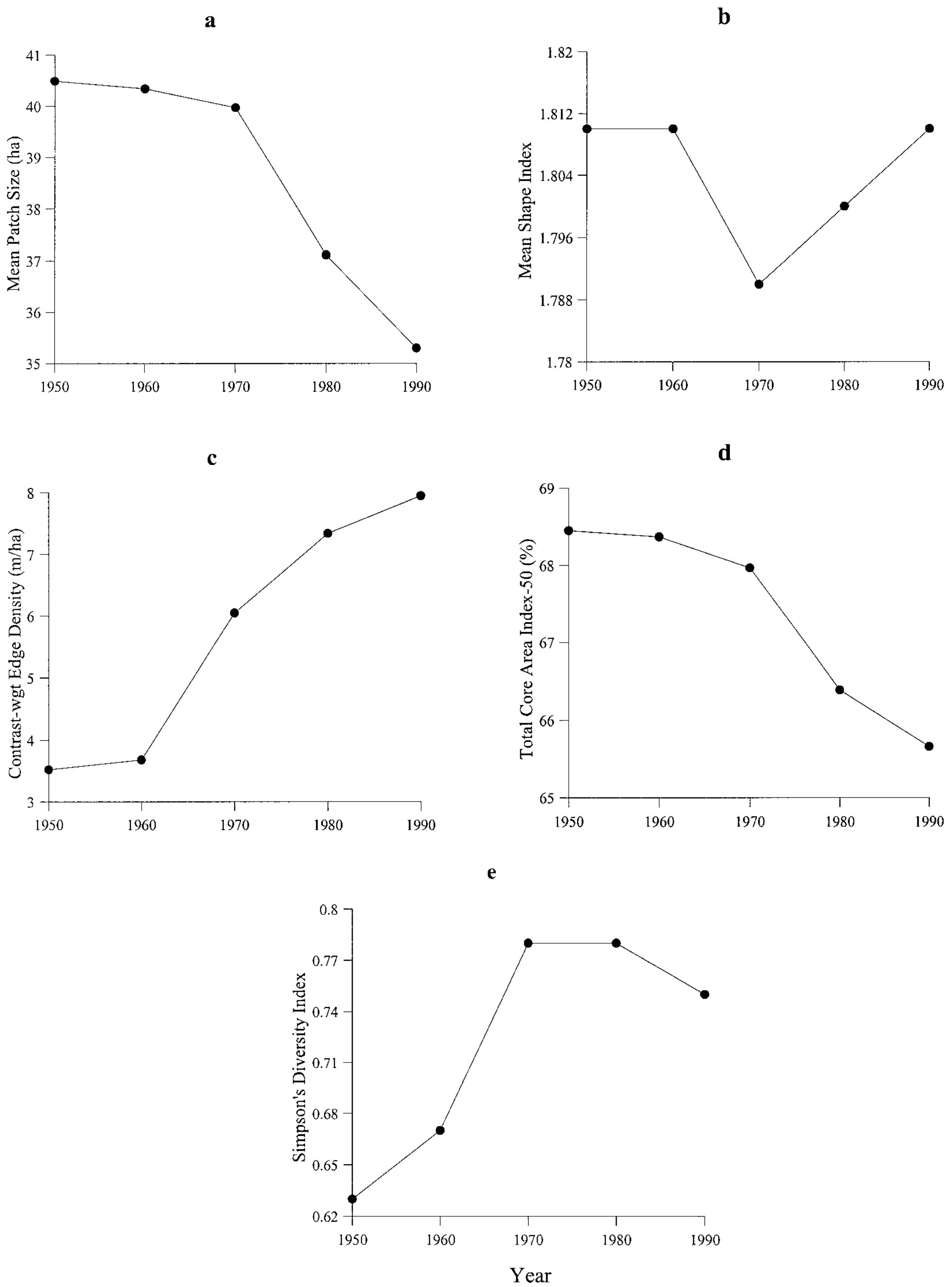

Figure 7. Changes in (a) mean patch size, (b) mean shape index, (c) contrasted-weighted edge density, (d) total core area index-50 (based on 50-m edge influence), and (e) Simpson's diversity index (see Table 2 for a description of metrics) at 10-year intervals between 1950 and 1990 for suitable timber lands (STL) only. Landscapes included harvested areas (clearcuts and partial cuts) as separate cover types; roads were ignored.

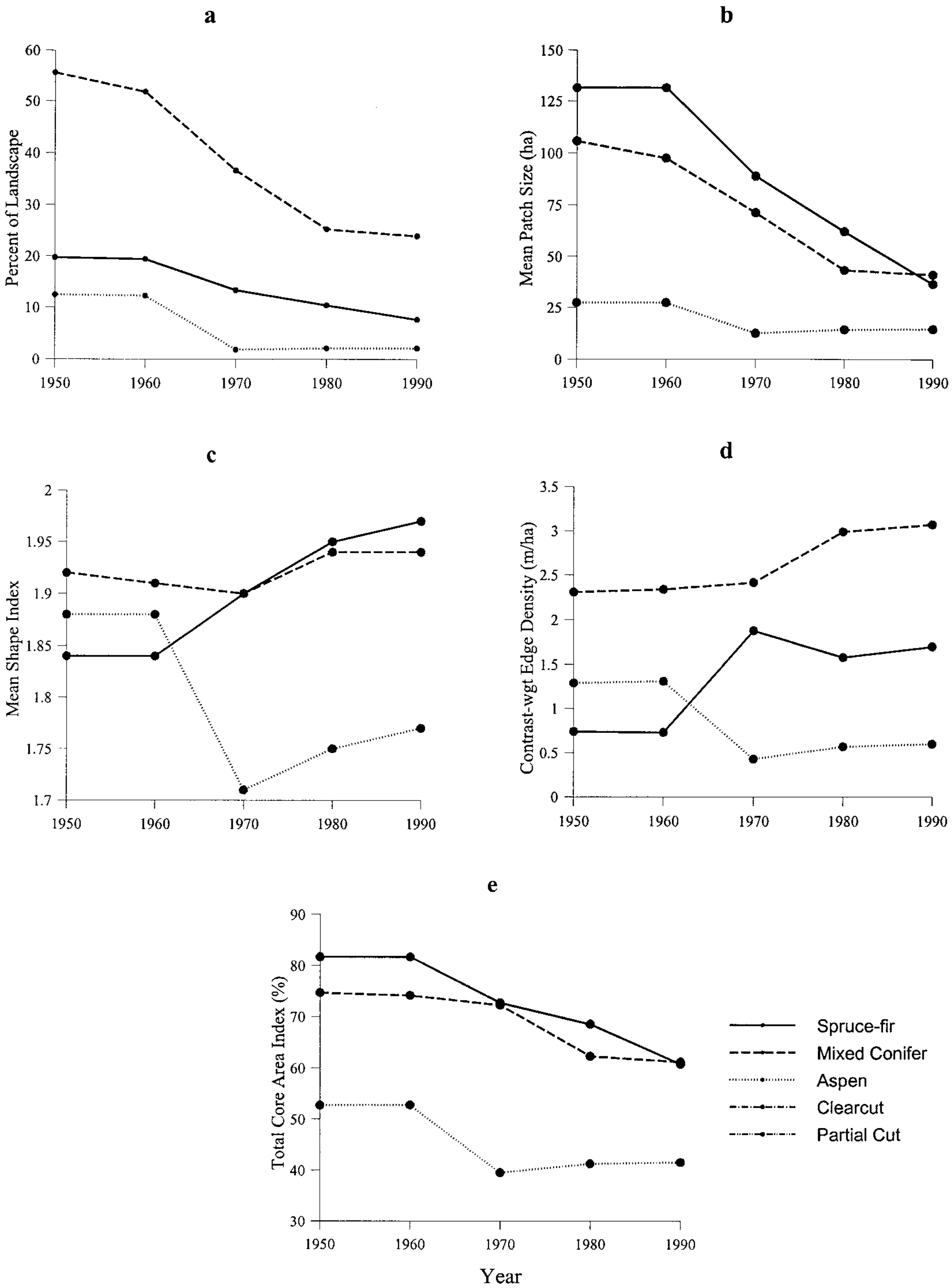

Figure 8. Changes in (a) mean patch size, (b) mean shape index, (c) contrasted-weighted edge density, (d) total core area index-50 (based on 50-m edge influence), and (e) Simpson's diversity index (see Table 2 for a description of metrics) at 10-year intervals between 1950 and 1990 for the mature spruce-fir, mixed-conifer, and aspen cover types on suitable timber lands (STL) only. Landscapes included harvested areas (clearcuts and partial cuts) as separate cover types; roads were ignored.

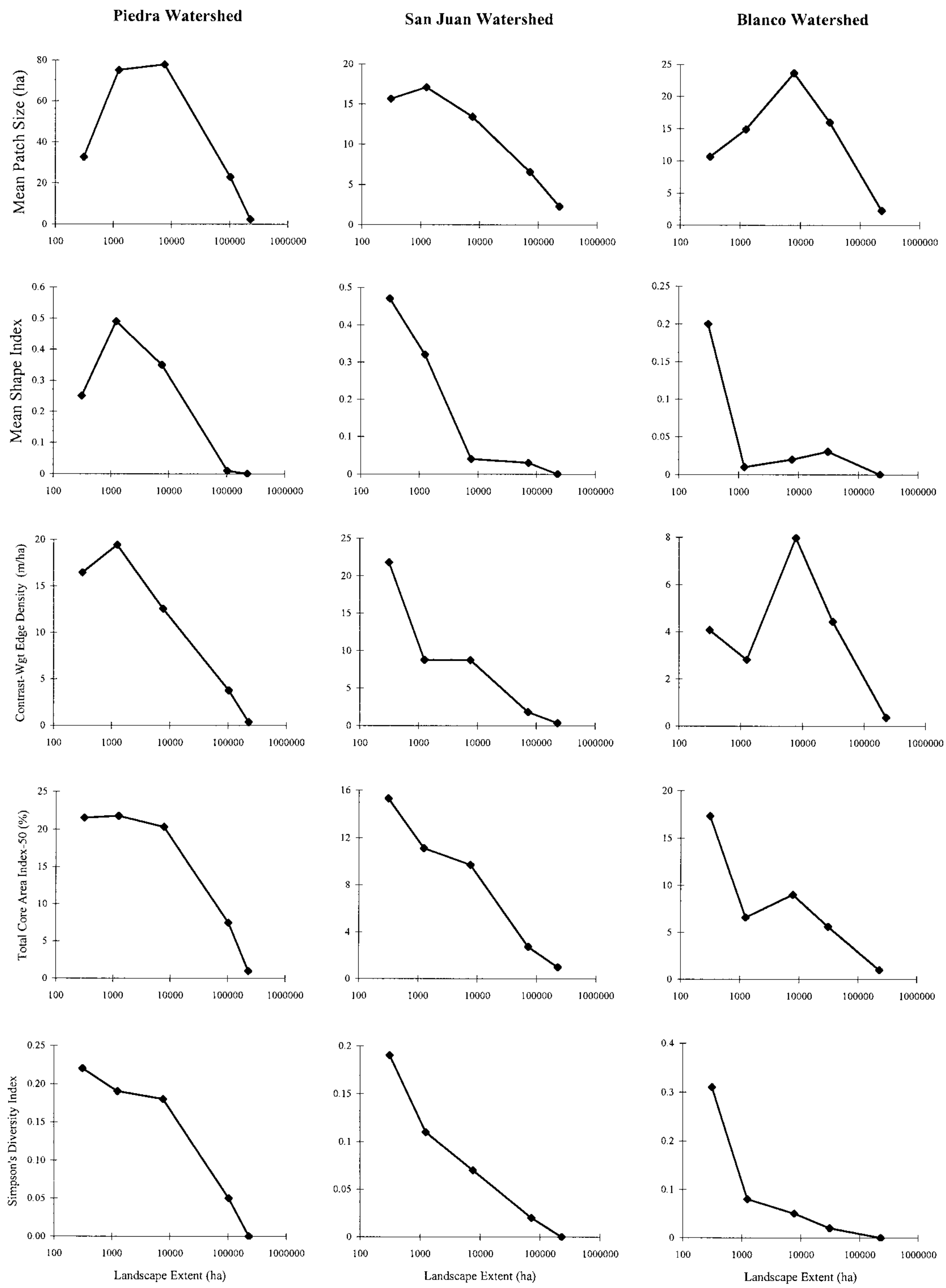

Figure 9. Changes in 5 landscape structure metrics (see Table 2 for a description of each) between 1950 and 1993 in relation to landscape extent. Each figure represents a series of 5 nested sampling areas, including 3 concentric circular areas with roughly 1-, 2-, and 5-km radii nested within a major watershed and the entire study area (228487 ha)(see text for details). Landscapes include all lands within the corresponding extent and buffered roads (25-m wide lines) and harvested areas (clearcuts and partial cuts) as separate cover types. Each point represents the absolute value of the difference in the metric between 1950 and 1993 for the particular landscape extent. Note that the *x*-axis is a log scale and that the *y*-axis range varies among figures.

shape index, and contrast-weighted edge density exhibit peaks in the range of 1000–10 000 ha landscapes in one or more of the watersheds. Similarly, total core area index and Simpson's diversity index both exhibit somewhat of a threshold effect at 10 000 ha in one or more of the watersheds.

Discussion

Cumulative effects of roads and logging on landscape structure

Despite increasing concerns over the cumulative impacts of past forest management activities on landscape structure and the fragmentation of mature forests in high-elevation landscapes of the southern Rocky Mountains, we detected only small changes in landscape structure when all lands within the study area were considered. The proportional coverage of mature spruce-fir forest, for example, declined by less than 1% over the period of study. Even after classifying partially harvested stands with a residual large-diameter overstory as a separate cover type, this cover type declined by only 1.5%. The relatively minor changes in overall landscape structure and fragmentation of mature forest can be attributed to the vast buffering capacity of the large portion of the landscape where logging and road-building cannot occur (ca 84% of the study area).

Much greater and potentially significant changes in landscape structure and mature forest fragmentation were, however, evident on suitable timberlands. Since the 1950s, roughly 10% of the landscape has been logged. Much of the area logged during the early period of study has since been removed from the suitable timber base. Indeed, for the entire San Juan National Forest, there were 480 972 ha designated as suitable for commercial timber use in 1962, but only 151 822 ha so designated in 1992. Therefore, of the lands currently designated as suitable timberlands, approximately 37% have been logged. This management intensity was sufficient to cause rather dramatic changes in landscape structure, especially when harvested areas were classified as separate cover types.

Notably, on suitable timberlands, roughly half of the mature coniferous forest was converted to young stands, shrublands, nonforest, roads and harvested areas, resulting in an absolute reduction in coverage of roughly 25% (Table 3). The increase in shrublands and nonforest can be explained by frequent regeneration failures following clearcut logging in higher elevations. Overall, mean patch size declined by 40%, core area declined by 20–25%, depending on the edge effect distance (50 or 100 m), and contrast-weighted edge density increased 2- to 3-fold. Consistent with the findings from other studies of this type in the western U.S. (Ripple et al. 1991; Baker 1994; Spies et al. 1994; Reed et al. 1996a), these findings suggest that there has been considerable change in landscape structure on suitable timberlands. It should be noted, however, that these impacts were dispersed throughout the landscape, particularly mid and lower elevations, because of the patchy distribution of suitable timberlands within a matrix of lands unsuitable for timber harvest.

As noted by other authors (Reed et al. 1996a,b; Shinneman 1996; Tinker et al. 1997), the main effect of fragmentation in mature coniferous forests may be the conversion of interior forest to edge habitat. In the naturally heterogeneous landscape of our study area in 1950, roughly 65–70% of the landscape existed as interior environment based on a 50-m edge influence. Forest management activities have reduced the proportion of the landscape in interior conditions by roughly 20% on suitable timberlands. More importantly, the induced edges created by clearcuts and roads are significantly different from the inherent edges which result from natural disturbances such as fire and windthrow. Natural disturbances typically leave dead standing and downed trees and irregular, feathered edges, in contrast to the abrupt edges created by roads and clearcuts (Hejl et al. 1995; Reed et al. 1996a,b; Tinker et al. 1997).

Roads have had a much greater impact on landscape structure than logging in our study area. Between 1950–1993, roads increased in proportional coverage 3-fold over the entire landscape and 7-fold on suitable timberlands. In 1993, roads comprised more than 3% of the entire landscape and more than 9% of the suitable timberlands. This represents a direct loss of 3–9% of available habitat for organisms. More importantly, as noted above, roads affect a much greater proportion of the landscape than is indicated by their area alone, and therefore might be considered a keystone landscape element. In our study, the increase in road density between 1950–1993 accounted for most of the changes in landscape configuration associated with mean patch size, edge density, and core area metrics. At the landscape level, for example, the contrast-weighted edge density increased by

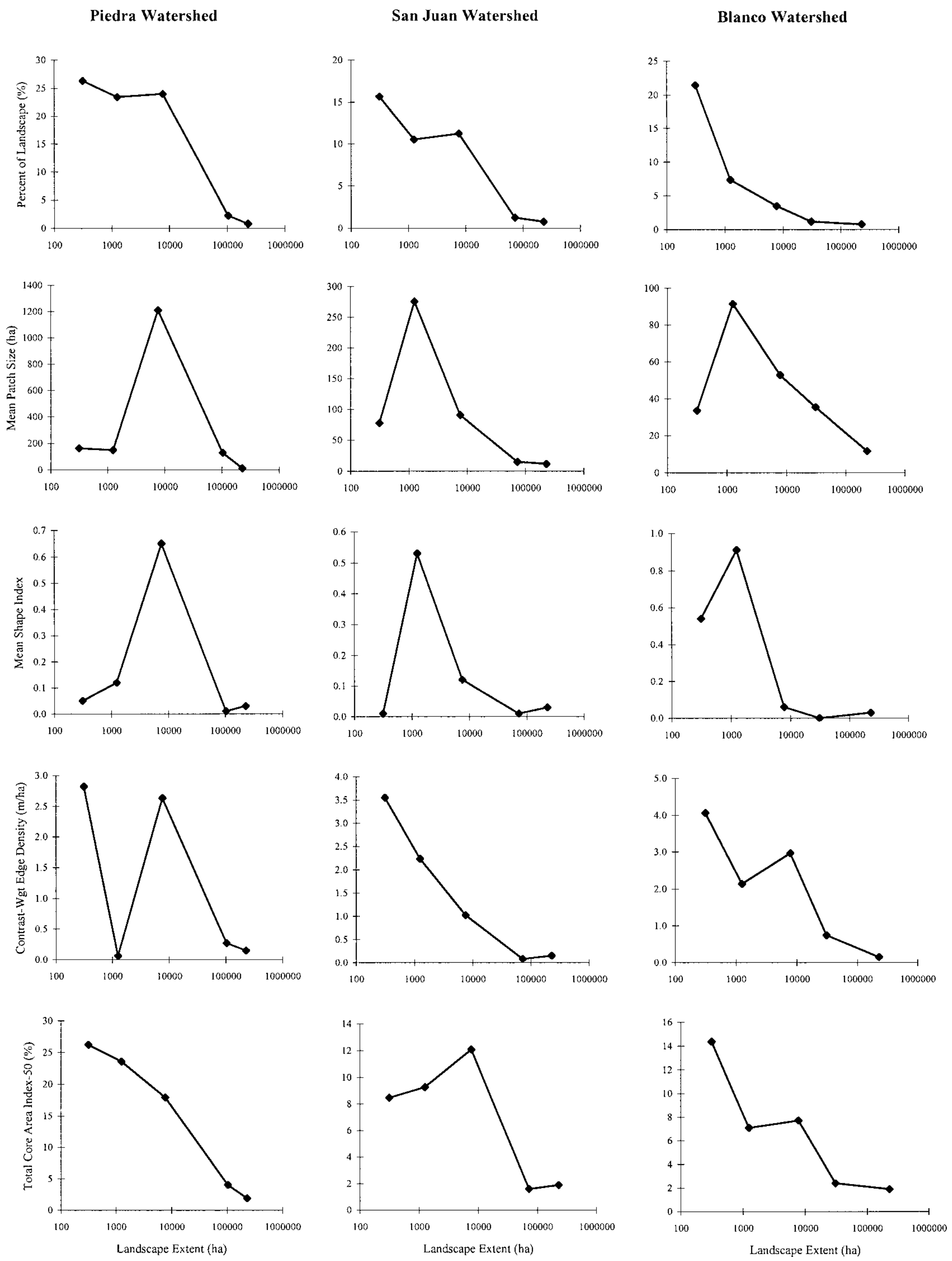

Figure 10. Changes in the area and configuration of late-seral spruce-fir forest based on 5 metrics (see Table 2 for a description of each) between 1950 and 1993 in relation to landscape extent. Each line represents a series of 5 nested sampling areas, including 3 concentric circular areas with roughly 1-, 2-, and 5-km radii nested within a major watershed and the entire study area (228487 ha)(see text for details). Landscapes include all lands within the corresponding extent and buffered roads (25-m wide lines) and harvested areas (clearcuts and partial cuts) as separate cover types. Each point represents the absolute value of the difference in the metric between 1950 and 1993 for the particular landscape extent. Note that the x-axis is a log scale and that the y-axis range varies among figures.

only 4% between 1950–1993 when ignoring roads, but increased 34% when roads were treated as a separate cover type. Similarly, changes in mean patch size were almost exclusively associated with road impacts. These findings are consistent with other studies from the northern Rocky Mountain region. Tinker et al. (1997) documented that roads were a more significant agent of landscape change than clearcuts in several forested watersheds on the Bighorn National Forest in north-central Wyoming. Reed et al. (1996b) also found that the Medicine Bow National Forest in southeastern Wyoming was more highly fragmented from roads than from clearcuts and that road edges may persist longer than natural patch edges or those created by clearcuts.

Potential ecological consequences

In many ways, the overall landscape structure has been changed very little by a half century of logging and road-building, even under the modeling assumptions that maximize the fragmenting effects of human activities (i.e., including roads and treating partially cut areas as a separate cover type). However, some of the more subtle landscape changes of the last 50 years are probably significant concerns for conservation of sensitive interior forest species (Dobkin 1994; Hejl et al. 1995). Logged areas and roads can have a quantitative and qualitative effect on suitable habitat for species that require interior forest environments (Temple and Wilcox 1986; Ruggiero et al. 1994). In addition to a quantitative reduction in suitable habitat, clearcuts and roads may create high-contrast edges that may influence habitat conditions for 10s-100s of meters into the adjacent stand (Murcia 1995) and may block or impede the movement of some species, resulting in population subdivision and increased competition for resources in remaining forest patches (Lovejoy et al. 1986; Saunders et al. 1991). Moreover, roads increase human presence and activities which may be disturbing to sensitive interior forest species. We are currently developing models to evaluate these impacts for a subset of species.

Clearly, the most significant cumulative effects of the last 50 years have been on the areas now designated as suitable for timber production. Given that suitable timberlands comprise only 16% of the landscape, does this have important conservation implications? It may be very important, because lands designated as suitable for timber production in this region may be the more biologically productive areas on relatively gentle terrain. Recent theoretical developments in ecology suggest that heavy exploitation of the most productive ecosystems (i.e., the suitable timber base) may potentially impair the long-term maintenance of native biodiversity (Hansen and Rotella 1999). Many species of plants and animals are now thought to exist as ‘meta-populations’ consisting of numerous small subpopulations that are more or less discrete demographic units, but periodically exchange individuals via dispersal or migration (Pulliam 1988; Gilpin and Hanski 1991; Robinson et al. 1995). In addition, it is thought that some of the subpopulations sustain positive population growth (i.e., in which production of new individuals exceeds mortality in most years) and therefore serve as ‘source’ populations; whereas other subpopulations, in which mortality usually exceeds recruitment, function as ‘sink’ populations. Source populations are presumed to be vital to the long-term persistence of the metapopulation. The practical difficulty in all of this is that we generally do not know which lands provide source habitats and which provide sink habitats. A better understanding of the meta-population structure of representative plants and animals of this region is a high research priority. Until specific information becomes available, it may be a good coarse-filter strategy to conserve habitats that have a high probability of supporting source populations. One such type of habitat for some species may be mature forests growing on productive sites (i.e., suitable timberlands).

Unfortunately, the interpretation of our findings is plagued by an issue that has beset virtually all other studies of landscape change: the lack of a framework within which to interpret landscape structure; that is, a lack of expected values for the range of natural variability in landscape structure. We know that high elevation landscapes in the San Juan Mountains are spatially and temporally dynamic as result of disturbance and succession processes (see Study Area description). Moreover, we know that the natural disturbance regime is dominated by relatively infrequent, large-scale fires that, in large part, control the coarse-grained mosaic of vegetation types and seral stages (Romme et al. 1998). Despite this knowledge, we have not yet quantified these natural dynamic landscape changes (although this is the subject of current investigation), as has been done for a Pacific Northwest landscape (Wallin et al. 1996). Without this frame of reference, it is difficult to know whether the current landscape structure or the changes in structure resulting from roads and logging activities during the past 50 years are within the historic range of variation (last several hundred years) to which native organisms probably are well adapted, or whether they lie outside of the historic range and may represent unprecedented conditions to which the organisms are poorly adapted.

Scaling issues

The pattern detected in any ecological mosaic is a function of scale, and the ecological concept of scale encompasses both extent and grain (Forman and Godron 1986; Turner et al. 1989; Wiens 1989; Moody and Woodcock 1995). Extent and grain define the upper and lower limits of resolution of a study and any inferences about scale-dependency in a system are constrained by the extent and grain of investigation (Wiens 1989). In the analysis of landscape change, spatial scale is defined by the minimum patch size (grain) and the geographic extent of the landscape; temporal scale is defined by the minimum (grain) and total (extent) period over which landscape change is assessed. We cannot detect patterns or changes in patterns beyond the extent or below the resolution of the grain. This has important implications pertaining to the interpretation of our findings.

With regards to temporal scale, the period over which change was evaluated had a dramatic impact on the magnitude of change detected and our conclusions regarding the ecological significance of those changes. For example, mean patch size on suitable timberlands declined by a few percent each decade between 1950–1990, yet declined by 13% when evaluated over the entire period. We concluded that the cumulative change is probably ecologically significant. Had we evaluated change over a 20- or 30-year period, rather than a 40-year period, we would have concluded that the cumulative effects were ecologically insignificant. Furthermore, if we had chosen a 1-year grain and 10-year extent, we would have concluded that the cumulative effects were negligible. Further, we would have reached a different conclusion had we selected the 1960s (period of greatest change) versus the 1980s. Lastly, given the natural landscape dynamics, had we chosen a 10-year grain and 200-year extent (given appropriate data), we would have likely reached an altogether different conclusion.

With regards to spatial scale, the extent of area evaluated (i.e., landscape size) had a dramatic impact on the magnitude of change detected and our conclusions regarding the ecological significance of those changes. For example, mean patch size over the entire landscape declined by only 6% between 1950–1993, yet declined by 72% when evaluated over a 1250 ha landscape centered on a timber sale area. This sensitivity to spatial scale is not surprising given the local variability in landscape structure. Specifically, given the interspersion of reserved lands (e.g., wilderness areas) and managed timberlands, as landscape extent increases outward from any point of origin, there is a strong likelihood of encountering dramatic changes in landscape structure. This suggests that there may be an optimal range of scales for detecting changes in landscape structure within any geographic region.

Clearly, the choice of scale in an investigation is of paramount concern. First, the interpretation of landscape metrics used to measure change is constrained by the scale of the investigation (i.e., the absolute values are scale-dependent). Hence, we cannot compare the values of metrics among investigations using different scales (i.e., caution must be exercised in comparing our results with other published studies). Second, care must be exercised in the choice of time period over which change is evaluated. In our study, we selected the period 1950–1993 because it represents the period extending from the onset of logging and road-building activities to the most recent data available. Lastly, given a particular landscape pattern, there may indeed be an 'optimal' spatial scale for detecting change, but without further study it is too early to say with certainty what that scale is for any particular landscape.

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