FRAGSTATS Workshop

Case Study Exercise #1

Quantifying habitat fragmentation under alternative land management scenarios

The overall purpose of this exercise is to give you basic hands-on experience running FRAGSTATS in the context of a real-world application. The emphasis is on learning to parameterize and run FRAGSTATS and interpret the output. The details of this particular application are less important than learning the mechanics of FRAGSTATS. Note, it is impractical to provide an illustrated example that covers all possible analytical contexts. Each application is unique; the specific decisions made in this exercise are not necessarily transferable to other applications, but the general approach is.

Project Area Description

The project area encompasses roughly 10,000 ha of the Pagosa Ranger District on the San Juan National Forest and lies within the South Central Highlands Section of the southern Rocky Mountains Province in southwestern Colorado. The area lies within the Piedra River watershed and encompasses a portion of the Weminuche Wilderness area west of the continental divide. Elevation ranges from 8,000 ft (m) in the valley bottoms to over 14,000 ft (m) mountain peaks along the continental divide. The geology of the area is quite complex. Parent materials date from ancient Precambrian rocks to recent alluvial deposits. Current landforms were created by a variety of geomorphological processes, including plate tectonics, volcanism, glaciation, and erosion. The climate varies significantly in relation to the pronounced elevational and topographic gradients. Temperatures range from an average high of 73 degrees (c) in July to an average low of -1 degrees (c) in January. Precipitation ranges from a mean of more than 60 in (cm) on the highest peaks to less than 20 in (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. Overall, the project area is typical of the South Central Highlands Section with respect to physiographic variation and landscape patterns.

The Greater San Juan Mountain Area, which includes the project area, supports some 507 native vertebrate species, as well as biotic communities representative of eleven potential natural vegetation types. Thirty-two species of vascular plants and 40 species of vertebrates are listed as endangered, threatened, or species of special concern at the state or federal level. Seven major vegetation types of ecological and economic significance occur within the project area. Each of these types has a unique ecological setting and history, as well as distinctive human impacts and changes since Euro-American 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 (*Pinus ponderosa*) forest interspersed with shrub-dominated stands (Petran chaparral dominated by *Quercus gambelii*). The middle slopes are covered by a mosaic of mixed conifers (*Pinus 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 *Picea engelmannii* and *Abies 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.

The San Juan Mountain country has never been static; there have been continual changes in the climate, geomorphology, vegetation, and human influences at least since the end of the Pleistocene ice age some 14,000 years ago. Rapid and substantial global warming around 14,000 years ago brought about the end of the ice age and caused dramatic readjustments in the distribution and abundance of the biota. Since the end of the Pleistocene, however, the regional climate has been comparatively stable, though there have been important periods of smaller-scale variability–such as the warmer and drier conditions of the altithermal (ca 4,000 years ago), the prolonged droughts that coincided with Anasazi abandonment of the region around 1300 AD, and the Little Ice Age that lasted from about 1600 A.D. until the early 1800s. However, general vegetational zonation and species composition apparently have remained roughly the same throughout the latter half of the Holocene period.

Humans have been present in the South Central Highlands for at least 8,000 years, and have influenced ecosystems and landscape patterns throughout the time of their occupancy. The extent and magnitude of human influences on local ecosystems during the Paleoindian and Archaic periods (ca 8,000 B.C. to 1 A.D.) is poorly understood, but people probably had comparatively minor impact during that early period of low population densities than they had in later periods. There was a dramatic change in the extent, magnitude, and kinds of human impacts during the Anasazi period in southwestern Colorado and northwestern New Mexico (ca 1 A.D. to 1300 A.D.), when human population densities reached levels comparable to those of today and agricultural and subsistence activities affected large areas. Following Anasazi abandonment of the region around 1300 A.D., human populations apparently fell to relatively low levels and remained comparatively low during this period of indigenous settlement until the arrival of large numbers of Euro-American settlers in the mid to late 1800's.

Under the pre-European settlement disturbance regime, landscape dynamics were driven primarily by the patterns of wildfire. The historical fire disturbance regime varied dramatically within the project area. 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; 60 years in the aspen type; and 100-200 years in the spruce-fir type. 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. Although stand replacement fires initiate stand development and maintain a coarse-grain mosaic of successional stages and cover types across the landscape, other disturbance processes, such as landslides, floods, windthrow, insects and disease also play 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.

With the exception of the period of Anasazi occupation between 1-1300 A.D., anthropogenic disturbances probably had a relatively minor effect on landscape structure prior to the early 1900's. Although logging by Euro-American settlers began as early as 1875, the scale and impact of logging increased dramatically in the late 1800's with the advent of railroad logging. Most of the activity was confined to the pine forest at lower elevations. By 1950, essentially all of the old-growth ponderosa pine forests of this region had been exploited and profoundly altered. In contrast, extensive logging at the higher elevations generally began much later in the twentieth century. The first large-scale spruce logging operation began in 1946. 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. The aggressive road-building and logging programs of the 1950s-1980s led to profound changes in the landscape structure of some high elevation areas.

Step 1: Establish Analysis Objectives

Ultimately, the analysis must be guided by well-formulated objectives. Thus, the first step is to establish analysis objectives.

- The goal of this analysis is to evaluate the effects of alternative land management scenarios on the extent and fragmentation (i.e., condition) of habitat for species associated with late-seral coniferous forests. The specific objective is to quantitatively compare the extent and fragmentation of the focal habitat and its variation over time under a natural disturbance regime and several alternative timber harvest regimes. Here, we will consider a single focal habitat: late-seral, mixed-coniferous and spruce-fir forest (hereafter simply referred to as late-seral coniferous forest).

<u>Late-seral coniferous forest</u>.--Late-seral coniferous forest is broadly defined on the basis of overstory composition and structure and includes habitats for a wide range of species; it represents a coarse-filtered habitat evaluation because there is no reference to any particular species. The condition of late-seral coniferous forest is deemed important to the maintenance of biodiversity in this landscape. Due to the prevalence of timber harvesting in the higher elevations over the past 50 years, there is increasing concern over the impact of current and proposed land management activities on species associated with this habitat. It is important to note that catastrophic (i.e., stand-replacing) fires are a natural disturbance process in this landscape and function to alter the extent and spatial configuration of forest cover types and seral stages over time. Thus, this landscape is not static; rather, it is characterized by the dynamic interplay between disturbance and succession. The range of variation in landscape structure under these dynamic conditions provides a reference against which to compare various land management scenarios. In this case, the anthropogenic agent of habitat loss and fragmentation under consideration is timber harvesting. Indeed, timber harvest is the primary anthropogenic disturbance in the high-elevation portion of this landscape.

Step 2: Define the Landscape

Once the analysis objectives have been established, the next step is to define the landscape in a manner that is relevant to the target species/communities. This involves establishing the spatial extent of the analysis (i.e., delineate landscape boundary), establishing a relevant spatial grain for the analysis, and establishing a model of the landscape structure.

<u>Delineate the landscape boundary</u>.–The first step in defining the landscape is to delineate the landscape boundary. This is often a difficult task because the relevant ecological boundaries often do not correspond to the superimposed administrative and/or analysis boundaries. To the extent possible, the extent of the landscape should be meaningful ecologically given the scale at which the target populations/communities operate. For example, the local range of the focal species or of the local population or metapopulation, or the range of the focal community within an ecoregion may be suitable as the basis for delineating the landscape. In many cases, however, there will be other practical considerations that must be taken into account. For example, the landscape extent may have to correspond to a specific project planning area (e.g., timber sale area), a timber or wildlife management unit, a watershed, or an administrative unit (e.g., ranger district or national forest). At a minimum, the scope and limitations of the analysis given these scaling considerations should be made explicit.

For this exercise, we selected a representative area (10,000 ha) on the Pagosa Ranger District of the San Juan National Forest for this exercise. The choice of this landscape extent was based on several factors. *First*, 10,000 ha is sufficiently large to potentially encompass the home ranges of many individuals of several species of concern (i.e., those associated with late-seral coniferous forest). Thus, it is likely that some populations are potentially sustainable within the extent of this landscape. As such, it is likely that habitat changes (in either extent or fragmentation) will have both individual- and population-level implications. And given the overarching management goal of maintaining viable populations of all species, we are most concerned with the impacts of human activities at a scale that affects population processes. *Second*, the extent of this landscape is sufficiently large to incorporate large natural disturbances (i.e., catastrophic fires), except the most extreme events. Thus, the landscape is potentially capable of operating within a dynamic equilibrium that can be characterized. *Third*, this particular landscape is representative of the broader surrounding landscape in terms of land cover patterns. Thus, the results obtained from this experiment should be generalizable to the broader regional landscape. *Finally*, the size of the landscape was selected for processing efficiency; larger landscapes simply take too much time to process.

<u>Establish a model of the landscape structure</u>.–The second step is to establish a digital model of the landscape structure. Here, it is likely that the patch-corridor-matrix model of landscape structure will provide the best means of representing the landscape–and in a manner that lends itself to pattern analysis given the current state-of-the-art and available analytical tools. Specifically, a patch classification scheme should be established for each target species/community, where the patches either represent the focal habitat type under consideration, or other patch types that the target species/community may perceive and respond to

differentially. In some cases, it may be appropriate to designate a matrix patch type consisting of an abundant and highly connected background land cover within which focal habitat patches exist.

For this exercise, our primary interest is the extent and configuration of late-seral coniferous forest. Thus, it was necessary to classify the landscape on the basis of both seral stage and plant community, and in a manner appropriate to the assessment of pine marten habitat suitability, although the details of this classification are not presented here. Briefly, we classified the landscape into broad cover types (plant communities) representing the major vegetation types. Because we are also interested in pine marten habitat, it was necessary to classify the coniferous forest into several types, corresponding to the strong elevation gradient in plant communities (see study area description). Although the non-coniferous forest cover types are not a focus in this analysis, they warrant equal attention because they comprise the ecological neighborhood of all coniferous forest patches and will therefore affect some of the habitat configuration metrics (e.g., similarity index). All forest cover types were further classified into four seral stages according to well-established functional stages of even-age stand development (Oliver and Larson 1990, Spies 1997): (1) stand initiation, (2) stem exclusion, (3) understory reinitiation, and (4) old-growth, shifting mosaic. Forest stands undergo this successional sequence following catastrophic, stand-replacing disturbance (i.e., fire or clearcutting), but the age at which a stand transitions from one stage to another varies among cover types. Thus, there is an element of chance in the speed at which a stand succeeds from one stage to another. In summary, the landscape was classified into a variety of well-established cover types and seral stages which provided a useful initial model of landscape structure sufficient for assessing fragmentation of the two focal habitats.

In addition, although we do not explicitly distinguish a matrix from patches and corridors, it is possible to do so. At one level of resolution, the coniferous forest forms the matrix within which nonforested and deciduous forest patches and road and stream corridors exist. However, given the dynamic nature of the forest (i.e., constantly changing seral stage distribution), and the importance of seral stage to most species, we have chosen not to designate a matrix. Instead, we consider each cover-seral stage as a patch within the landscape mosaic. We can consider streams, riparian zones, and roads as corridors because of their linear form and context, as well as their effects on landscape function; however, because this habitat fragmentation analysis is largely a "structural" analysis, as opposed to explicitly assessing landscape function, it is mute whether we call these features corridors or patches (which happen to be linear in form).

All data for this example were derived from the Integrated Resources Inventory (IRI) and Resources Information System (RIS) database developed and maintained by the USDA Forest Service for the San Juan National Forest. This database contains geographic information on a wide variety of landscape characteristics. We derived our land cover map from a combination of data layers, including the Common Vegetation Unit (CVU) polygon coverage and the Roads and Streams line coverages. The CVU layer itself was developed by the Forest Service from a combination of information sources and processes, including the existing RIS 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. Attributes for each polygon included species, percent crown cover and tree size class, among others. We used information on species composition, size class and stand age to assign each polygon to a cover type and seral stage class. Stand age was either determined directly from stand exams or estimated based on empirically-derived relationships between age and size class of the dominant vegetation for each cover type. Roads and streams of various size classes were superimposed on this CVU-derived cover map to obtain our final cover map.

Based on this initial landscape characterization, and subject to the limitations of the available spatial data in the IRI and RIS database, we subsequently combined cover types and seral stages into four broad classes:

- *Nonforest and low-elevation forest*.–All nonforested cover types (e.g., barren, grass/forb, shrublands, etc.) and low-elevation forest, including pinyon-juniper and ponderosa pine forests.

- *Early-seral coniferous and aspen forest*.–All mixed-conifer (warm, dry and cool, moist) and spruce-fir cover types (with and without aspen) in the stand initiation stage of development and all aspen forest in all stages of development.

- *Mid-seral coniferous forest*.–All mixed-conifer (warm, dry and cool, moist) and spruce-fir cover types (with and without aspen) in the stem exclusion stage of development.

- *Late-seral coniferous forest*.–All mixed-conifer (warm, dry and cool, moist) and spruce-fir cover types (with and without aspen) in the understory reinitiation or old-growth, shifting mosaic stages of development.

Note, in the context of the simulation experiments described below (step 3), we reclassified the landscape in this manner for each time step (10-year snapshots of the landscape). Thus, the simulation operated on the initial landscape definition and was unaffected by this reclassification, only the output landscapes (snapshots) were reclassified for purposes of this habitat analysis. This highlights that in most habitat analyses the landscape will have to be reclassified prior to the analysis, since rarely will the database have been established with reference to a particular habitat definition.

<u>Establish a relevant grain of analysis and data format</u>.–The last step in defining the landscape is to define a relevant grain (or minimum mapping unit) and digital data format. In some cases, these decisions will be guided by technical considerations owing to the source of the data and the data processing software available. In most cases, a raster data format will be desirable because the software for analyzing landscape patterns (see below) is primarily geared to raster formats. Additionally, the grain of the data should represent a balance between the desire for accurate calculations of landscape pattern, computational efficiency, and the desire to scale patterns appropriately for the chosen landscape extent. On the one hand, the grain should be kept as fine as possible to ensure that small and narrow, yet meaningful, features of the landscape are

preserved in the data model. On the other hand, the grain should be increased in relation to the extent so that unnecessary detail is not confounded with the important coarse-scale patterns over large spatial extents.

For this exercise, we chose a grain (cell) size of 25 m for our initial landscape characterization, which allowed us to depict patches as small as 0.0625 ha (1 cell). We chose this grain size in part owing to the resolution of some of the source data (e.g., 30 m resolution of Landsat imagery), but it also reflected a compromise between our desire to accurately represent relevant linear landscape elements such as roads and streams that potentially serve as fragmenting features, on the one hand, and our desire to emphasize coarse vegetation patterns in a computationally efficient manner on the other. In addition, a 25 m cell size allowed us to represent depth-of-edge distances at a resolution of 25 m, which we felt gave us sufficient flexibility in quantifying potential edge effects. Overall, a 25 m cell size seemed a reasonable compromise given these considerations, while at the same time it seemed sufficient to our task of assessing fragmentation of late-seral conifer forest and pine marten habitat at an ecologically relevant scale.

Thus, while a grain size of 25 m was deemed effective for our initial landscape characterization, we believed that 0.0625-ha patches were probably too small to be relevant for most organisms associated with late-seral coniferous forest. More importantly, we recognized that many fragmentation metrics are sensitive to the grain of the data and are particularly affected by the number of patches. In the context of the simulation experiments described below (step 3), we recognized a priori that many extremely small patches would be created by the simulated disturbance processes and that this would have a significant impact on the computed metrics. Thus, we decided to eliminate very small patches less than 0.5 ha in the final land cover and habitat capability maps before computing the fragmentation metrics. Note, given the uncertainty in the impact of different grain sizes on the computed metrics and the uncertainty in the ecological implications of eliminating small patches below an arbitrary minimum patch size, we recognized that it would be wise to resample the final land cover and habitat capability maps at a range of resolutions, bracketing what we believe to be meaningful upper and lower resolutions given our analysis objectives, but we did not due this for this exercise.

A final consideration in defining the landscape is choosing a digital data format. Here, although the original source data was in vector format, consisting of both polygon and line coverages, we elected to convert the land cover maps to a raster (grid) format to facilitate computation of the fragmentation metrics in step 4 below.

Step 3: Establish a Spatio-Temporal Reference Framework

Perhaps the most challenging step is to establish appropriate reference framework for evaluating habitat fragmentation for the target organisms/communities. The difficulty is rooted in the philosophical issue of what constitutes an appropriate reference framework. Under one perspective, the historic range of natural variability is deemed an appropriate reference framework. Unfortunately, there is often little agreement over what constitutes the appropriate period in history to reference. Often it is argued that the period immediately preceding Euro-American inhabitation is best; however, it is argued in opposition that native Americans were present and impacting landscape patterns and processes for thousands of years prior to Euro-American settlement. The contention is rooted in the philosophical issue of what constitutes "natural", and whether natural is even desirable.

Despite these philosophical issues, it is widely accepted that some reference framework is better than none. Therefore, the purpose of this step is to establish some meaningful basis for placing the current and potential future landscape conditions into context. This reference framework may be qualitative or quantitative in nature. For example, it may be sufficient to simply describe in narrative terms the "natural" patterns of variation in habitat conditions over time and space for the landscape under consideration. Where possible, a more quantitative approach is desirable. This may involve the use of retrospective studies of past landscape conditions (e.g., historical reconstructions of landscape patterns and dynamics) or the use of computer simulation models to simulate landscape changes based on the best understanding of the processes that drive landscape change.

In addition to a temporal reference framework (i.e., how habitat conditions vary over time), a spatial reference framework should be established as well. Ideally, this should involve a qualitative or quantitative assessment of habitat conditions over a much broader spatial extent than the landscape under consideration. In other words, what is the broader regional context of the focal landscape. In order to understand the biological consequences of habitat fragmentation, it is necessary to understand whether or not the habitat is both locally and regionally rare and fragmented. Large regional source populations, for example, may offset any local fragmentation impacts. Another strategy is to establish a reference landscape; that is, another landscape that somehow represents a suitable benchmark to compare against the focal landscape. Of course, finding a suitable reference framework is fraught with difficulty, because at the scale of most landscapes under consideration here there are often too many other confounding sources of variation to warrant direct comparison.

As noted previously, the case study landscape is dynamic--constantly changing in structure as a result of the dynamic interplay between disturbance and succession. The range of variation in landscape structure under these dynamic conditions provides a reference against which to compare habitat fragmentation impacts of various land management scenarios. In this case, the anthropogenic agent of habitat loss and fragmentation under consideration is timber harvesting, which is the primary anthropogenic disturbance in the high-elevation portion of this landscape. Thus, to provide a spatio-temporal framework for understanding the impacts of anthropogenic habitat fragmentation, we conducted a simulation experiment using the Rocky Mountain Landscape Simulator (RMLANDS). Specifically, we simulated the following five different disturbance scenarios:

1. Wildfire regime.–In this scenario, the frequency, size and variability of fires represented our best estimate of the fire disturbance regime for pre-Euro-American settlement conditions (circa 1610-1850). The details of this wildfire scenario were described by

Roworth et al. (submitted). Briefly, the mean fire return interval or disturbance rotation period varied from approximately 20 yrs in the low elevation ponderosa pine forest to 300 years in the high elevation spruce-fir forests. The disturbance regime consisted of relatively frequent, small fires in the low elevations and infrequent, large fires in the high elevations. For the purpose of this experiment, we simulated only catastrophic, stand-replacing (i.e., high severity) fires.

2. <u>Long rotation, dispersed cutting</u>.–In this scenario, 100% of the mixed-conifer and spruce-fir cover was available for harvest on a 300-year rotation period. Maximum harvest unit size was 18 ha (40 acres) and a 400 m buffer was established around each harvest unit within which timber harvest was excluded from the current time step (10 years). In addition, harvest units were maximally dispersed across the landscape in a staggered setting approach.

3. <u>Long rotation, aggregated cutting</u>.–In this scenario, 100% of the mixed-conifer and spruce-fir cover was available for harvest on a 300-year rotation period. Maximum harvest unit size was 18 ha (40 acres) and NO buffer was established around each harvest unit. In addition, harvest units were maximally aggregated. The combination of no buffer and maximum aggregation causes harvest units to merge together (by chance) into larger harvest blocks.

4. <u>Short rotation, dispersed cutting</u>.–Same as scenario 2, but with a 200-year rotation period.

5. <u>Short rotation, aggregated cutting</u>.–Same as scenario 3, but with a 200-year rotation period.

Each simulation consisted of a 200-300 year period of disturbance and succession, depending on the rotation period of the scenario, and was replicated 10 times (although only 1 replicate is used in this exercise). The details of the simulation model are too detailed to present here and are not essential for our purpose. Suffice it to say that under each scenario, disturbances and succession were implemented as constrained stochastic processes in an attempt to mimic real-world patterns of disturbance and succession, while at the same time incorporating an element of chance. The model operates with a 10-year time step and produces an output of the landscape condition (i.e., distribution of cover types and seral stages) at each step.

In all timber harvest scenarios, we used clearcutting exclusively as the silvicultural system. In reality, a variety of other silvicultural systems (e.g., individual tree selection, group selection, shelterwood) are employed, but the use of a single system serves to better illustrate habitat fragmentation differences among scenario. In addition, in all timber harvest scenarios, we restricted harvesting to the mixed-conifer and spruce-fir cover types. This simplified the simulations and facilitated comparisons among scenarios. Clearly, these simulated conditions are not entirely realistic. First, much of the area subject to timber harvesting in the simulations is actually administratively protected in wilderness or research natural areas and therefore has been taken out of the suitable timberland base. Second, we did not allow any wildfire during the timber harvest simulations, despite the fact that wildfires would certainly occur in this landscape regardless of management intended to prevent them. Thus, while these scenarios area not entirely realistic, they serve to highlight and exacerbate the differences in habitat loss and fragmentation patterns resulting independently from these disturbance agents.

There are two important points regarding the simulation modeling approach for generating a reference framework. First, the simulation approach involves using a model to project potential future landscape trajectories based on our current understanding of disturbance and succession processes. This has certain advantages and disadvantages. On the positive side, this approach allows us to quantify landscape conditions accurately and precisely over long periods of time, thereby allowing us to quantitatively describe the range of variation in habitat conditions over meaningful periods of time. This is especially useful because in most landscapes it is not possible to reconstruct historic landscape patterns accurately and over sufficiently long periods of time. On the downside, a model is just that, a simplified "model" of the real world. As such, it is only as good as the data used in the model. More importantly, incorrect specification of model parameters can lead to spurious results and erroneous conclusions. Nevertheless, if interpreted in the proper context and with these limitations in mind, the simulation approach can provide a useful reference framework.

Second, and perhaps more importantly, this approach provides a reference framework that explicitly considers habitat conditions as dynamic. Habitat is not static in any real landscape. Thus, it is inappropriate to use a single snapshot of a landscape as a reference framework, except under special circumstances. Habitat conditions change over time as the landscape undergoes disturbance and succession. Thus, it is more realistic to represent habitat as a range of conditions.

In addition, the project area does not exist in isolation; it has a spatial context, and this broader regional landscape context has an important influence on the function of the landscape. In particular, populations of species associated with the focal habitat (late-seral coniferous forest) do not honor the landscape boundary selected for this analysis; i.e., the landscape is an open system with respect to the flow of organisms. Consequently, it is important to consider the uniqueness of the project area and its potential interaction with the surrounding landscape context. Note, the spatial context of the focal landscape does not affect the structural analysis of habitat conditions within the focal landscape; that is, the measured extent and fragmentation of habitat within the focal landscape is not affected by the character of the surrounding landscape. Rather, the spatial context can affect how we interpret our findings; that is, the ecological implications of particular habitat conditions. In our case, the selected landscape is representative of the broader regional landscape conditions in terms of land cover patterns and disturbance processes. Therefore, it is likely that the habitat patterns we quantify will represent other similarly-sized landscapes within the region.

Step 4: Parameterize and Run FRAGSTATS (i.e., Quantify Habitat Fragmentation)

The next step is to quantify habitat fragmentation for the landscapes under consideration. This entails selecting a relevant suite of fragmentation metrics that offers a comprehensive, yet parsimonious description of habitat conditions, and then parameterizing and running FRAGSTATS properly. Your task is to complete the steps below.

(1) Create batch file

First, create two batch files: one for the fire scenario and one for the four timber harvest scenarios combined:

- ...\workshop\fragstats\piedra_batch_fire.fbt
- ...\workshop\fragstats\piedra_batch_harvest.fbt

(2) Create class properties file

Next, create the class properties file:

- ...\workshop\fragstats\piedra_classid.fdc

(3) Select a suite of metrics

Next, select a suite of metrics. See the following table for a list of potentially suitable metrics for measuring habitat loss and fragmentation.

Habitat Component	Metric	Description
Extent	Percent landscape	The *percentage of the landscape* (PLAND) comprised of the target habitat.
	Core area percent of landscape	The *core area percentage of the landscape* (CPLAND) comprised of the target habitat.
Subdivision	Number of patches	The *number of patches* (NP) of the focal habitat or, alternatively, *patch density* (PD).
	Clumpiness	The *clumpiness index* (CLUMPY) measures the extent to which habitat is aggregated or clumped.

	Landscape division	*Degree of landscape division* (DIVISION) equals the probability that two randomly chosen places in the landscape under investigation are not situated in the same contiguous habitat patch, and is closely related to the area-weighted mean patch size (AREA_AM).
Geometry	Core area index	The *core area index* (CAI) is basically an edge-to-interior ratio like many shape indices, the main difference being that the core area index treats edge as an area of varying width and not as a line (perimeter) around each patch.
	Shape index	The shape index (SHAPE) is a normalized perimeter-to-area ratio with a square standard (i.e. SHAPE = 1 for a square); alternatively, there are several other shape indices (FRAC, LINEAR, SQUARE, CONTIG)
	Edge density	The *edge density index* (ED) is a simple measure of the total edge per unit area and can be summarized at the class level.
Contrast	Edge contrast index	The *edge contrast index* (EDGECON) is a patch-level measure of edge contrast with abutting neighbors, given as a percentage, and it can be summarized at the class level (TECI)
Isolation	Similarity index	The *similarity index* (SIMILAR) is a patch-level measure of neighborhood similarity. It considers the size and proximity of all like and unlike patches whose edges are within a specified search radius of the focal patch.
	Proximity index	The *proximity index* (PROX) is a patch-level measure of neighborhood isolation. It considers the size and proximity of all like patches whose edges are within a specified search radius of the focal patch.

	Nearest neighbor distance	The *Euclidean nearest neighbor distance* (ENN) is a patch-based measure of neighborhood isolation. At the patch level, it measures the distance to the closest patch of the same class.
Connectedness	Correlation length	The *correlation length index* (CLI) is a measure of the structural continuity or connectedness of the focal habitat based on a measure of the extensiveness of each patch as measured by the *radius of gyration* (GYRATE).
	Connectance	The connectance index (CONNECT) is a measure of the number of functional joinings between patches of the corresponding patch type, where each pair of patches is either connected or not based on a user-specified distance criterion. Connectance is reported as a percentage of the maximum possible connectance given the number of patches.
	Traversability	The *traversibility index* (TRAVERSE) is a functional measure of connectivity based on the idea of ecological resistance. This metric uses a least-cost path algorithm to determine the area that can be reached from each cell in the focal habitat

(4) Create needed ancillary files for functional metrics

Next, depending on the metrics selected above, create the needed ancillary files for the functional metrics.

- *Core area index.–S*ee ...\workshop\fragstats\piedra_edgedepth.csv
- *Similarity index*.–See ...\workshop\fragstats\piedra_similar.csv
- *Edge contrast index*.–See ...\workshop\fragstats\piedra_contrast.csv

(5) Parameterize and run FRAGSTASTS

Finally, parameterize and run FRAGSTATS. Note, the morning session of the workshop was devoted to teaching you how to parameterize and run FRAGSTATS; use this knowledge to complete this step.

Step 5: Interpret the Results (i.e., Characterize the Landscape Relative to Analysis Objectives)

The final step is to interpret the results relative to the objectives of the analysis. Ultimately, this is the most important step because it allows you to apply the knowledge gained from the analysis to improve management. Here the intent is to use the results of the analysis to guide land management decisions. In addition, here the conclusions should be tempered with an explicit discussion of the scope and limitations of the analysis. In particular, the scale of the analysis in relation to the scale of the ecological system (e.g., populations and communities) should be addressed. In addition, the definition of the landscape in terms of the patch classification scheme or schemes employed and the target organisms and/or communities should be discussed. Limitations in the quality of data used to model landscape patterns should be acknowledged as well. Here the intent is not to undermine the results of the analysis, but to provide an honest context for interpreting the reliability of the results.

<u>Habitat extent</u>.–As expected, short-rotation timber harvest scenarios resulted in greater habitat loss than long-rotation scenarios, while there were no differences between harvest unit dispersion patterns (aggregated vs. dispersed) for each rotation length (Fig. 24a). Interestingly, under the long-rotation scenarios habitat extent actually increased for a period of about 60 years before eventually declining. This time lag was due to the large cohort of mid-seral stage forest in the current landscape that succeeded to the late-seral stage at a rate that exceeded harvesting in the short term. All harvest scenarios eventually reached a constant equilibrium in PLAND that fell within the 90% range of variation of the wildfire scenario; however, the short-rotation scenarios stabilized at a much lower PLAND than the long-rotation scenarios. Interestingly, all scenarios, including wildfire, resulted in a loss of late-seral coniferous forest over time, suggesting that either the current landscape contains more late-seral coniferous forest than during the reference period (represented here by the wildfire scenario) or that our simulation included too much wildfire. Finally, note that habitat extent under the wildfire scenario fluctuated over time between roughly 7% and 28% of the landscape, reflecting the dynamic shifting mosaic pattern of habitat resulting from the stochastic interplay of disturbance and succession. In contrast, the timber harvest scenarios (as implemented here) resulted in the complete regulation of forest disturbance and succession, which eventually transformed the landscape into one characterized by a constancy in habitat area.

<u>Habitat subdivision</u>.–Although the basic patterns of habitat subdivision under the various scenarios were consistent with our expectations, there were some interesting differences among the three subdivision metrics that revealed subtle differences among scenarios and among the metrics--and highlighted the multi-faceted nature of subdivision as a component of habitat fragmentation. As expected, NP and CLUMPY both revealed that dispersed cutting under either a short- or long-rotation regime resulted in greater habitat subdivision than aggregated cutting. Specifically, the short-rotation, dispersed cutting resulted in substantially greater NP than the other scenarios, while the long-rotation, aggregated cutting resulted in the least NP. Interestingly, NP was comparable for the short-rotation, aggregated cutting and long-rotation, dispersed cutting scenarios. NP increased gradually over time under all timber harvest scenarios

despite the decrease in habitat area, indicating that concomitant habitat loss and subdivision was a universal outcome of all timber harvesting scenarios. NP was well outside and below the range of variation for the wildfire scenario, suggesting that fires caused substantially greater habitat subdivision than timber harvesting. The increased NP under the wildfire scenario was caused by heterogeneous spread of simulated wildfires compared to homogeneous spread of timber harvest units, but it was also a direct consequence of our choice of minimum patch size in the output land cover maps. This highlights the implications of the choices we made in defining the landscape in Step 2. We could have just as easily increased the minimum patch size and decreased the range of variation in NP under the wildfire scenario.

CLUMPY revealed the same basic patterns as NP, but with some interesting additional interpretations. First, the values of CLUMPY (0.85-0.95) indicate that the habitat under all scenarios was more aggregated (i.e., less subdivided) than expected under a spatially random distribution. This is not too surprising as we generally expect vegetation patterns to exhibit some degree of contagion (i.e., aggregation). However, this is also a function of the scale of the data. Recall that as the cell size (i.e., grain) decreases relative to the actual grain of the patch mosaic (i.e., average patch size), the proportion of like adjacencies among cells increases and indicates a greater degree of spatial aggregation. Thus, the measured degree of aggregation can be altered by simply changing the grain of the data without any real-world change in the land cover patterns. This highlights the need to consider this metric (and most other metrics) as comparative measures. Second, in contrast to NP, the CLUMPY trajectories under the timber harvest scenarios largely fall within the range of variation in the wildfire scenario, suggesting that perhaps the most important impact of timber harvesting is not on the level of habitat subdivision but on its temporal dynamic. Specifically, timber harvesting (as simulated) minimized the variation in subdivision over time, whereas wildfire maintained the landscape in dynamic equilibrium.

DIVISION largely mirrored the patterns in PLAND, and thus did a relatively poor job of differentiating dispersed from aggregated cutting patterns. The long-rotation timber harvest scenarios resulted in a temporary decrease in DIVISION, corresponding to the increase in PLAND, followed by a gradual increase in DIVISION until reaching equilibrium at a very high level of division. The values of DIVISION (0.88-0.99) under all scenarios indicate that there was a very high probability that two randomly chosen places in the landscape were not situated in the same contiguous habitat patch. Thus, we can conclude that late-seral, coniferous forest naturally exhibits a high degree of subdivision.

<u>Patch geometry</u>.–As expected, the spatial character of habitat patches was effected by both the intensity of timber harvesting and the pattern of harvest units. All timber harvest scenarios resulted in a decrease in TCAI, although only the short-rotation, dispersed cutting scenario caused TCAI to drop below the range of variation estimated for the wildfire regime. Interestingly, the differences between long-rotation scenarios were not discernable for the first 120 years of the simulation, after which TCAI decreased more rapidly for the dispersed cutting scenario. The long-rotation, aggregated cutting scenario maintained the highest TCAI over the duration of the simulation. The long-rotation, dispersed cutting scenario and short-rotation,

aggregated cutting scenario had similar effects on TCAI after reaching equilibrium, similar to the patterns observed for NP and CLUMPY.

Habitat isolation.–As expected, the isolation of habitat patches was affected dramatically by the total amount of habitat in the landscape (PLAND). In general, as PLAND increases, there is an increasing likelihood that a habitat patch will contain more habitat within its neighborhood--and therefore be less isolated. Thus, SIMILAR largely mirrored the patterns in PLAND, with some notable differences. First, due to the pulse of mid-seral, conifer forest succeeding to the late-seral condition during the first 60 years of the simulation, SIMILAR exhibited a two-fold increase for 60 years under the long-rotation scenarios before steadily declining to an equilibrium condition similar to the mean condition under the wildfire scenario. Second, SIMILAR was higher under the dispersed cutting scenarios than the corresponding aggregated cutting scenarios for a considerable portion of the simulation, and this difference was especially pronounced under the short-rotation scenarios. Why did dispersed cutting result in less habitat isolation (i.e., greater neighborhood similarity) than aggregated cutting, a seemingly counterintuitive result? The answer lies in the basis of this particular metric and nicely illustrates why a thorough understanding of each metric is necessary before ecological interpretations can be made.

SIMILAR is a patch-based metric; that is, each patch is evaluated separately, and then (in this case) the area-weighted mean across all patches is taken as the overall landscape metric. For each habitat patch, SIMILAR is computed by summing the area of every other patch within the specified ecological neighborhood (in this case we used 2000 m), where the neighboring patches are weighted by class (i.e., higher weights are given to patches that are more "similar" to the focal habitat patch) and by distance from the focal habitat patch. Thus, a large habitat patch surrounded by dissimilar patches will receive a very low score, whereas a small habitat patch surrounded by several other habitat patches will receive a much higher score. Consequently, as the focal habitat is initially subdivided into disjunct patches, as is the case in the short-rotation, dispersed cutting scenario, as long as the disjunct habitat patches are relatively close together (i.e., within the specified neighborhood distance), the similarity index may actually increase over the case of the more contiguous (undivided) habitat. Hence, while the habitat patches are, in effect, less isolated from other habitat patches, it may be misleading to conclude that habitat isolation overall is less under the dispersed cutting scenarios. Note, this is not a deficiency of this metric, as it is designed to represent the isolation of habitat "patches" from other "patches" of the same or similar class, but rather it illustrates why this particular metric is perhaps best used under certain circumstances, namely, when the habitat is relatively rare (say 0-30% of the landscape) and subdivided into many patches. Under these conditions, SIMILAR will effectively discern clumped distributions of habitat (i.e., where habitat patches are relatively close to each other) from highly dispersed distributions (i.e., where habitat patches are relatively isolated from each in space). These conditions are not reached until 100 (short-rotation) or 200 (long-rotation) years into the simulation.

Habitat connectedness.–As expected, like habitat isolation, the physical continuity of habitat was affected dramatically by the total amount of habitat in the landscape (PLAND). In general, as PLAND increases, there is an increasing liklihood that habitat patches will coalesce into larger, more extensive patches that physically span more of the landscape. Thus, correlation length (GYRATE_AMN) largely mirrored the patterns in PLAND. Long-rotation scenarios maintained a relatively high correlation length during the first 100-200 years of the simulation, but eventually declined to an equilibrium level almost identical to the mean correlation length under the wildfire scenario. Short-rotation scenarios maintained correlation length at current levels for roughly 60 years before declining dramatically and then stabilizing at a relatively low level. Differences between short- and long-rotation scenarios were rather dramatic throughout most of the simulation, indicating that the average distance one might traverse the map and remain within a habitat patch was substantially less under the short-rotation scenarios. In addition, the relationship between dispersed and aggregated cutting patterns was similar to that described above for SIMILAR. Specifically, the dispersed cutting pattern resulted in greater correlation length (i.e., greater habitat continuity) than the aggregated cutting pattern during the early portion of the simulation before the relationship reversed itself. This happened because under the dispersed cutting scenario, initial harvest units were more likely to perforate large, contiguous patches of habitat and therefore not change overall habitat extensiveness; whereas, under the aggregated cutting scenario, initial harvest units were more likely to eliminate large blocks of habitat and thus reduce overall habitat extensiveness.