

# Chapter 14: Clarifying Concepts

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## Introduction

There are some topics that continue to be raised by managers that were not sufficiently addressed in the first and second edition of U.S. Forest Service General Technical Report GTR 220 “An Ecosystem Management Strategy for Sierran Mixed-Conifer Forests.” All of them concern issues about where and when GTR 220 concepts apply and how these concepts relate to current management practices. The first section discusses the appropriate ecological conditions where GTR 220 concepts might be applied. The second section attempts to clarify the types and potential wildlife uses of defect trees in an effort to make them more recognizable in the field. The third section defines and distinguishes canopy cover and closure, an important distinction because some current management practices are focused on canopy cover targets. Some aspects of GTR 220 conditions, particularly localized wildlife habitat, may be better assessed with canopy closure. The final section examines the link between heterogeneity and ecosystem resilience, a core concept behind GTR 220’s emphasis on increasing variability in managed forests. General Technical Report 220 is a conceptual framework for managing forests and intentionally lacks the specificity that practitioners might sometimes want. We hope that clarifying the following concepts may help elucidate the GTR’s intent without constraining management options.

## Summary of Findings

1. **GTR 220 concepts could be applied to forest types that historically had low-intensity, frequent fire regimes in locations with topographic relief.**
2. **“Defect” structures used by wildlife are often large live or dead trees with decay, irregular bole or crown shapes, broken tops, broomed foliage or hollows created by torn branches** (examples in appendix).
3. **Canopy closure, the percentage of the sky hemisphere obscured by vegetation over a point, should be distinguished from canopy cover, a measure of canopy porosity averaged over a stand.** For some wildlife, including several sensitive species, high variability in canopy closure may be as important as stand average canopy cover.
4. **Heterogeneity in vegetation structure and composition has been strongly linked to ecosystem resilience, but direct empirical evidence is sparse.** Increasing research suggests microclimate variability and refugia from temperature extremes may be one mechanism linking site and vegetation heterogeneity with ecosystem resilience.

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## **Forest Types and Landscapes Where GTR 220 Concepts Apply**

In a broad sense, the GTR 220 strategy could be applied to any low-intensity, frequent-fire forest community within the Western United States. These forests generally are the highest priority for fuels reduction and forest restoration treatments because of their potential to burn at uncharacteristically high severity in the event of a wildfire after decades of fire suppression (Brown et al. 2004). The management concepts hinge on mimicking the variable forest conditions that would have been created by the effect of topography on fire behavior (North et al. 2009; however, see Scholl and Taylor 2010). Areas with no topographic relief may still have experienced variable fire intensities that create a patchy landscape. That variability was probably influenced by weather conditions during the burn or small-scale differences in fuels, making the pattern of the resulting forest conditions difficult to predict. The topographic “roughness” necessary to start directly influencing fire intensity has not been studied in the Sierra Nevada. Working with large, widely distributed fire scar data, research in eastern Washington mixed-conifer forests has started to examine interactions between topography and the scale of fire events (Kellogg et al. 2008, Kennedy and McKenzie 2010). These studies suggest modern fires are larger than historical burns, overriding topographic features that often constrained past wildfires. Contrasts between flatter and more rugged “firesheds” suggest that historically, topography did have a bottom-up influence on fire size, but the “roughness” at which this occurs has yet to be quantified (Kennedy and McKenzie 2010).

In Sierra Nevada landscapes with some topographic relief, forest types with a historically frequent, low-intensity fire regime include mixed conifer, ponderosa pine (*Pinus ponderosa* Lawson and C. Lawson), Jeffrey pine (*P. jeffreyi* Balf.), Douglas-fir (*Pseudotsuga menziesii* Mirbel Franco), giant sequoia (*Sequoiadendron giganteum* (Lindl.) J. Buchholz), and combinations of white fir (*Abies concolor* (Gordon & Glend.) Lindl. ex Hildebr), incense cedar (*Calocedrus decurrens* Torr. Florin), California black oak (*Quercus kelloggii* Newberry), and sugar pine (*P. lambertiana*, Douglas) (Sugihara et al. 2006). Many forests, however, at higher elevation or with more mesic conditions such as red fir (*A. magnifica* A. Murray bis), lodgepole pine (*P. contorta* Douglas ex Louden var. *Murryana* (Balf.) Engelm), and western white pine (*P. monticola* Douglas ex D. Don) have different fire regimes (i.e., more infrequent and mixed or high severity), and forest conditions may not have been as strongly influenced by topography. The GTR 220 strategy also is probably not appropriate for some forests in northern California with a high hardwood composition (e.g., >30 percent of basal area) that can occur in forests with

an abundance of tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder), madrone (*Arbutus menziesii* Pursh), or interior live oak (*Q. wislizeni* A. DC.) mixed with Douglas-fir and ponderosa pine sometimes referred to as mixed evergreen. The low densities and gap conditions suggested by GTR 220 for drier conditions might favor a transition of treated areas to hardwood dominance unless the area was repeatedly treated with mechanical thinning or prescribed fire (Stuart et al. 1993).

## Defect Trees

What characteristics make a “defect” tree valuable habitat for wildlife? Although this topic is frequently raised, preferred tree structural conditions for many wildlife species have not yet been specifically defined in the literature for most species. Most of the available research has focused on a few sensitive species such as the fisher (*Martes pennanti*) or spotted owl, or general groups such as bats, small mammals, and cavity-nesting songbirds. Furthermore, the number of studies that occurred in the Sierra Nevada is limited. There is evidence, however, of the preferred use patterns for some well-studied individual and groups of species in western coniferous forests.

Several studies have examined the role of legacy trees; large, old individuals left in a matrix of younger second growth following a past timber harvest or other disturbance. In commercially managed coastal redwoods (*Sequoia sempervirens* (Lamb. ex D. Don) Endl.), Mazurek and Zielinski (2004) found significantly higher diversity and richness of the wildlife they were surveying (bats, small mammals, and birds) in forests that contained some old legacy trees and snags. They suggested the higher diversity might result from the structural complexity offered by legacy trees, particularly the basal hollows produced by fire scarring. Other animals using legacy or old-growth residual trees include northern (*Strix occidentalis caurina*) (Moen and Gutierrez 1997, North et al. 1999) and California spotted owls (*S. o. occidentalis*) (Irwin et al. 2000, North et al. 2000), fisher (Zielinski et al. 2004), southern red-backed voles (*Clethrionomys gapperi*) (Sullivan and Sullivan 2001), northern flying squirrels (*Glaucomys sabrinus*) (Carey 2000), and bats (Pierson et al. 2006). Although most of these studies do not quantify the characteristics of these legacy trees, they note that the trees were often left during earlier timber harvest because of some structural “defect.” The exact habitat value of these trees is unknown, but they probably offer some kind of special substrate that can provide cover and protection from inclement weather and predators.

Trees and snags selected by primary cavity nesters, woodpeckers and nut-hatches, may be particularly important because once vacated, the cavities are used by other birds and mammals (Bull et al. 1997). Several studies have found cavity

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availability can limit the abundance of some of these species in managed forests (Carey et al. 1997, 2002; Cockle et al. 2011; Wiebe 2011). A meta-analysis of the global distribution of tree cavities found that forest management tends to reduce the fungal heart-rots most associated with cavity abundance, thereby increasing reliance on primary cavity-nesters for creating suitable cavities (Remm and Lõhmus 2011).

A summary of forest structures favored by wildlife in interior forests of the Pacific Northwest focused on five conditions: living trees with decay, hollow trees, broomed trees, dead trees, and logs (Bull et al. 1997). The first three conditions might have been considered “defects” in past silviculture practices focused on stand improvement, and systematically removed. Certainly in some highly managed Sierra Nevada forests, these structures may be more rare than large, old trees as a result of stand improvement management. Bull et al. (1997) suggested identifying these structures by looking for dead and broken tops, large dead branches, wounds, fungal fruiting bodies, cavities, bole bends (where a new leader formed after top breakage), brooms caused by mistletoe, rust fungi, and Elytroderma disease. Bull et al. (1997) also suggested retaining large-diameter, relatively intact (decay classes 1 to 3) snags, and large (>15 in [38 cm] in diameter), long, and if available, hollow downed logs. Example pictures of some of these conditions are presented in the appendix.

The general pattern across these studies is that wildlife use is associated with larger trees that have structural characteristics, which can facilitate a cavity or a platform, enabling nesting, denning, roosting, or resting. Examples of these structures include irregular bole (e.g., hollows, forks, etc.) and crown (multiple or broken tops, platforms, concentrations of dense foliage) shapes. Care will be needed to identify and maintain these structures during mechanical thinning and prescribed fire operations. In some cases where these structures are rare, creating a no-mechanical-entry zone or encircling an area with fire line might be warranted. In some areas, management practices, such as prescribed fire, should be encouraged for their role in actively recruiting these structures.

## Canopy Cover and Closure

Canopy cover is often cited as an important habitat feature for a number of sensitive species associated with old-forest conditions in the Sierra Nevada (Hunsaker et al. 2002). The standards and guides in the 2004 Sierra Nevada Framework (SNFPA 2004) have specific minimum canopy cover targets designed to provide suitable habitat for sensitive species such as the California spotted owl. Some managers concerned that canopy cover would fall below levels in the standards and guides have been hesitant to create gaps in treated stands. Creating a gap will lower canopy closure (a point measure) over the opening, but may not significantly lower canopy

cover (a stand-level average). Canopy closure should be distinguished from canopy cover to accurately assess forest canopy conditions and characteristics that matter to key wildlife species. This distinction is particularly important because, following GTR 220 concepts, treatments are intended to produce tree groups, gaps, and areas with a low density of large trees.

Forest canopies are typically measured at two different scales (i.e., point and stand), with different instruments, to distinguish between two different qualities of canopy structure that create different wildlife habitat features. Jennings et al. (1999) distinguish these two structural qualities by referring to the point measure as canopy closure and the stand-level measure as canopy cover (fig. 14-1).

Canopy closure is a measurement of the percentage “of the **sky hemisphere** obscured by vegetation when **viewed from a single point**” (Jennings et al. 1999) (emphasis added). Closure measures the canopy hemisphere within an angle of view (i.e., a cone) over the sample point. Closure provides valuable information about the understory light, microclimate, and microhabitat environment at a specific location (Nuttle 1997). It is probably most useful for understanding how available light may influence plant composition and growth, and the potential climate conditions and vegetative cover over a specific microhabitat site (e.g., nest site cover to discourage predation). Traditionally, canopy closure has been assessed with a spherical densiometer, “moosehorn,” or hemispherical photograph. Different methods of measurement affect the canopy closure estimation (Paletto and Tosi 2009). As the viewing angle (i.e., the width of the observation cone) increases, closure estimates increase and within-stand variability between point estimates decreases (Fiala et al. 2006). Spherical densimeters use a reflective mirror held in front of the observer, which produces a large viewing angle and hence high canopy closure estimates. Spherical densimeters have large measurement errors at the mid-range of canopy closure owing to this large viewing angle (Cook et al. 1995). The moosehorn and hemispherical photograph reduce this problem because the image is taken straight up and the measurement of canopy closure is typically restricted to 45 to 60° off of vertical. Hemispherical photographs have a benefit over other closure estimates in that computer programs (e.g., the freeware Gap Light Analyzer) can easily and precisely calculate the total direct and diffuse light reaching the point on the forest floor over the course of a year. These light levels are highly correlated with surface microclimate conditions (Bigelow et al. 2011, Ma et al. 2010).

In contrast, canopy cover is the percentage “of forest floor covered by the vertical projection of the tree crowns” (Jennings et al. 1999). Cover is always measured vertically with a very narrow angle of view that approaches a point. It is a **stand-level measure of canopy porosity** (e.g., how much rain falls directly

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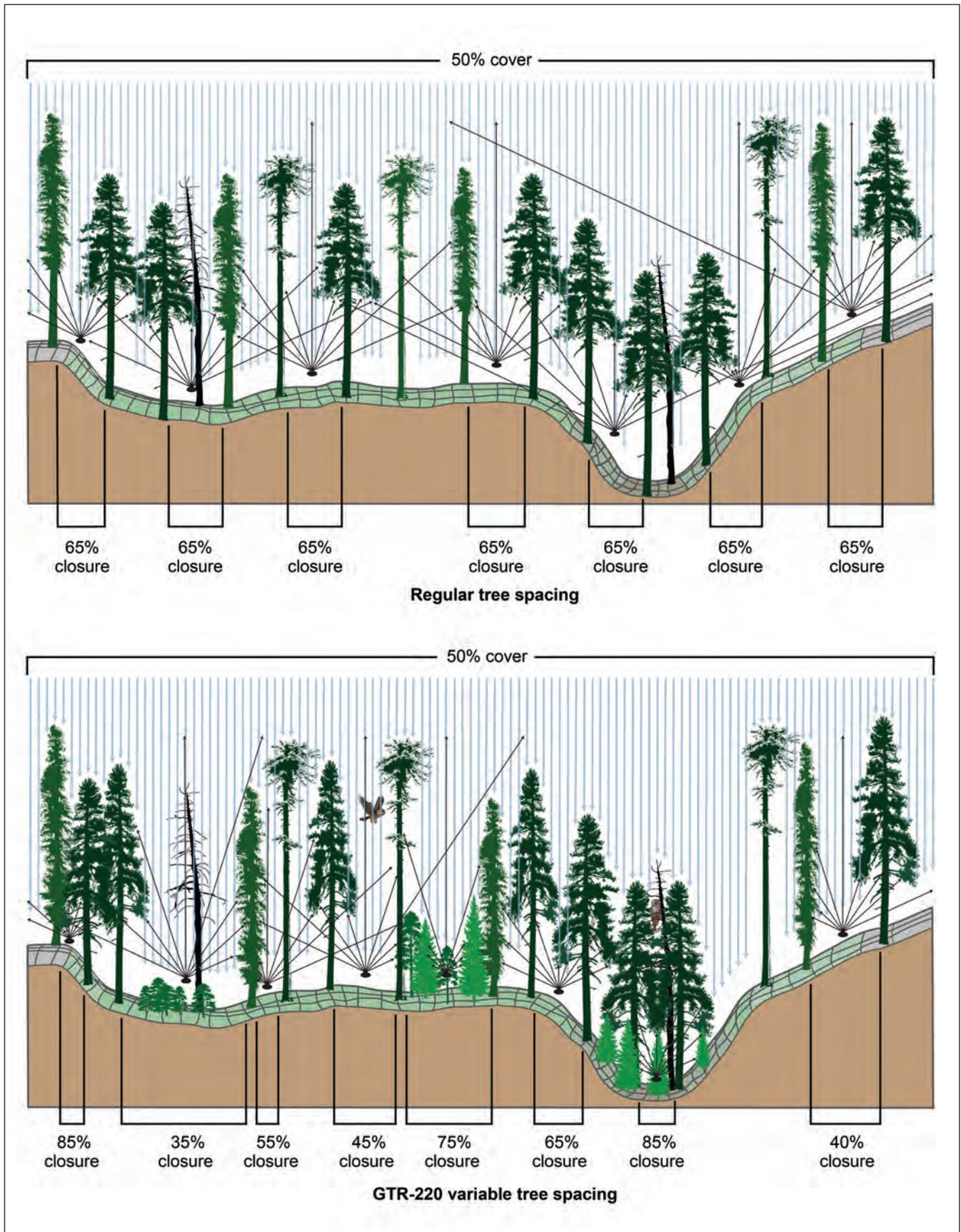


Figure 14-1—The difference between canopy closure and canopy cover in forests treated to produce regular and variable tree spacing. Illustration by Steve Oerding.

on the forest floor) that indicates how much of the forest floor is vertically overtopped with canopy. Canopy cover is usually either directly measured with a densitometer (sighting tube) or indirectly estimated from plot data using the Forest Vegetation Simulator (FVS).<sup>3</sup> Typically, direct densitometer measurements are taken at 100 or more points, sampled along a walked grid, in which the observer records the percentage of points where the sky directly overhead is obscured. Densitometer measurements provide a relatively unbiased estimate of canopy cover for the particular area (i.e., the walked grid) assuming that the stand is well sampled (Korhonen et al. 2006).

Indirect estimates of canopy cover are often made by FVS based on stand exam and plot data. As an indirect estimate, however, FVS assumes a certain amount of crown overlap (Crookston and Stage 1999), but it does not account for spatial variability in tree locations (Christopher and Goodburn 2008). Two stands having the same number, species, and size of trees will have the same FVS calculated canopy cover even if one has regular spacing and the other is mostly open with the trees concentrated in high-density clumps. The FVS will tend to underestimate a stand's actual canopy cover if trees are regularly spaced (i.e., less overlap than the model would predict). The FVS will overestimate a stand's actual canopy cover if trees are clumped (i.e., greater overlap than the model assumes). The FVS estimates should be viewed with caution because the program is unlikely to accurately estimate stand-level canopy cover in stands with a high degree of spatial variability or those with complex canopy structure such as those resulting from application of GTR 220 concepts.

The FVS estimates of stand-level canopy cover do not provide information on whether a stand has points of high canopy closure habitat associated with some sensitive species. For example, Purcell et al. (2009) compared FVS estimates of canopy cover and measures of canopy closure made with a spherical densiometer and a moosehorn in five studies of fisher resting sites. They found substantial differences in the estimates between the three methods. In another comparison, Ganey et al. (2008) evaluated model estimates of canopy cover developed from stand exam data with those derived from in-field densitometer (sighting tube) data at sites used by Mexican spotted owls (*Strix occidentalis lucida*). The model estimates ranged from 50 to 200 percent of field-based measurements, particularly in mesic mixed-conifer forest with multilayered canopies (Ganey et al. 2008).

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<sup>3</sup> Two other remotely sensed methods of estimating canopy cover are not addressed in this discussion; they include using grid overlaps on aerial photographs and light detection and ranging (LiDAR).

Canopy cover measurements alone may not capture aspects of canopy structure important to some sensitive species because it is a stand-level average. Historical forests can provide some inference about canopy conditions that were suitable habitat for these species. A recent study using FVS to derive current and historical canopy cover estimated an average cover of only 22 percent in 1911 compared to current (2005–2007) cover estimates of 28, 42, and 53 percent for the same area in moderate, low, and no fire-severity classes, respectively (Collins et al. 2011). In general, FVS estimates of stand-level average canopy cover would likely be very low if applied to the consistently low tree densities found in reconstruction studies, historical data sets, or photographic records. However, almost all of these sources indicate that trees were grouped in clusters, suggesting that active-fire regime forests had high-canopy closure at many points.

Assessing point-level canopy closure within treated stands may improve assessment of habitat conditions. Using hemispherical photographs, a moosehorn, or a spherical densiometer, closure measurements could be collected with a stratified sampling of gap, low-density, and tree cluster conditions (see suggestions in chapter 9 section “Using FVS to Assess Heterogeneity”). With GTR 220’s goal of producing variable forest structure across a treated area, point-level canopy closure values will be low in gaps and areas with a low density of large trees, and high in tree groups. Canopy closure, and its coefficient of variation, can provide an assessment of canopy heterogeneity within a stand among microhabitat locations. In contrast, canopy cover can provide a mean assessment of stand-level conditions. Canopy cover is best assessed with a densiometer in which at least 100 observations are collected in a systematic sample (i.e., a grid) within an area representative of the stand’s variability. Distinguishing between cover and closure, and the canopy characteristic that each measures and the technique used for canopy estimation, may improve assessments of canopy conditions for wildlife species.

## Heterogeneity and Resilience

The Forest Service definition of ecological restoration focuses on reestablishing the resilience or adaptive capacity of ecosystems. This approach is consistent with a recent review of catastrophic shifts in many different ecosystems that suggested maintaining resilience was the best strategy for sustainable management (Scheffer et al. 2001). How then might heterogeneity increase a forest’s resilience? Although it is intuitive that spatial heterogeneity, species diversity, and ecosystem resilience are linked, the connection has been difficult to rigorously test. One study examining the species-rich littoral forests of Madagascar, did find that forest spatial heterogeneity was associated with the resilience and maintenance of high species diversity

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over the last 6,000 years of climatic perturbations (Virah-Sawmy et al. 2009). The foundation of the concept, however, is probably more theoretical, based on the synthesis of decades of ecosystem research.

Holling (1973) was among the first to define ecological resilience as “the capacity of an ecosystem to return to the precondition state following a perturbation, including maintaining its essential characteristics, taxonomic composition, structures, ecosystem functions, and process rates.” Definitions of resilience, however, evolved as many ecologists moved away from Clements’ ideas that ecosystems inherently have a climax or stable state (Clements 1916). A more recent definition by Walker et al. (2004) is “the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks.” In a review of factors associated with loss of resilience and regime shifts in ecosystems, altered disturbance regimes was cited as a common driver (Folke et al. 2004). Recently, Holling (2010) has suggested that there are two aspects to resilience: “the more traditional [engineering] definition concentrates on stability near an equilibrium steady state, where resistance to disturbance and speed of return to the equilibrium are used to measure” resilience. The more ecological definition emphasizes conditions far from any equilibrium, “where instabilities can flip a system into another regime of behavior” and “resilience is the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior.” Ultimately Holling (2010) suggests that ecological resilience hinges on “designing interrelations between people and resources that are sustainable in the face of surprises and the unexpected.”

Beyond its theoretical foundation, empirical plant research has focused on identifying how variable microclimate conditions, driven by topographic heterogeneity, are associated with species resilience to changing climate. Some studies, examining how plant species persisted during past droughts and glacial advances, have identified variability in microenvironments as important in mediating climate change and enhancing species persistence (Suggitt et al. 2011). Microrefugia that support lower minimum temperatures may be particularly important for retaining both cold-adapted and mesophilous taxa (Dobrowski 2011, Scherrer and Körner 2011). The local climate, or topoclimate (Thorntwaite 1953) experienced by individuals is affected by regional advection and local terrain. Several studies have found good estimates of the lowest minimum temperature in a landscape can be made using terrain variables that characterize surface water accumulation (Chung et al. 2002, Daly et al. 2010, Dobrowski et al. 2009, Lookingbill and Urban 2003). Slope, aspect,

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**Variable microclimate conditions, driven by topographic heterogeneity, are associated with species resilience to changing climate.**

and slope shape are strong influences on local microclimate affecting water balance (Dobrowski 2011). These studies support the GTR 220 concept that varying forest conditions in response to topography may be consistent with increasing fine-scale contrasts in microclimate. Such a strategy may compliment one of the mechanisms by which species persist in landscapes as climate conditions change.

In Sierra Nevada forests, resilience, in part, will hinge on the ability of trees to persist under future conditions that may include greater or more frequent drought stress. A consistent pattern associated with the onset of tree mortality is a drop off in annual increment growth. Some studies (Das et al. 2007, 2008; Franklin et al. 1987) have suggested that these decreases eventually cross a threshold after which survival is unlikely. Many studies have documented reductions in tree growth as stem density increases. Drought also decreases annual radial increment growth, and can increase susceptibility to bark beetle attack. Competition from other trees, particularly from high stand density conditions, interact with climate stressors increasing the risk of mortality (Hurteau et al. 2007, Linares et al. 2010). Collectively these studies suggest that stand density reduction and variable forest structure that provides greater microclimate heterogeneity can make trees more resilient to mortality-induced stress. There are still many unknowns in the appropriate scale and mechanisms by which heterogeneity may increase ecosystem resilience.

## Measuring Resilience

Ecological resilience is an attractive concept, but often lacks measureable indicators. Quantifying resilience will be critical for identifying thresholds of probable concern when adaptively managing forests in the face of climate change (Scheffer et al. 2001). Unfortunately, resilience and adaptive capacity are often described as theoretical constructs rather than measurable indicators of system response to stress or disturbance (Carpenter et al. 2001). Moving from metaphors to measurement of ecological resilience requires clearly defining indicators of stress resilience that can be adapted to the context-specific conditions of different ecosystems.

If resilience is described in a manner similar to engineering resilience (*sensu* Holling 1973, Gunderson 2000), then (1) resistance can be defined as either no reduction or a smaller reduction in ecosystem response to a stress event; (2) recovery is the ability to resume a state relative to the damage experienced during an event; and (3) resilience is the capacity to reach pre-event ecosystem condition. A recent paper presented a quantitative method for assessing resistance, recovery, and resilience (fig. 14-2), in the context of tree radial growth response (Lloret et al. 2011). This approach may have much wider application for measuring and assessing response and resilience of many different ecosystem components (i.e., sensitive

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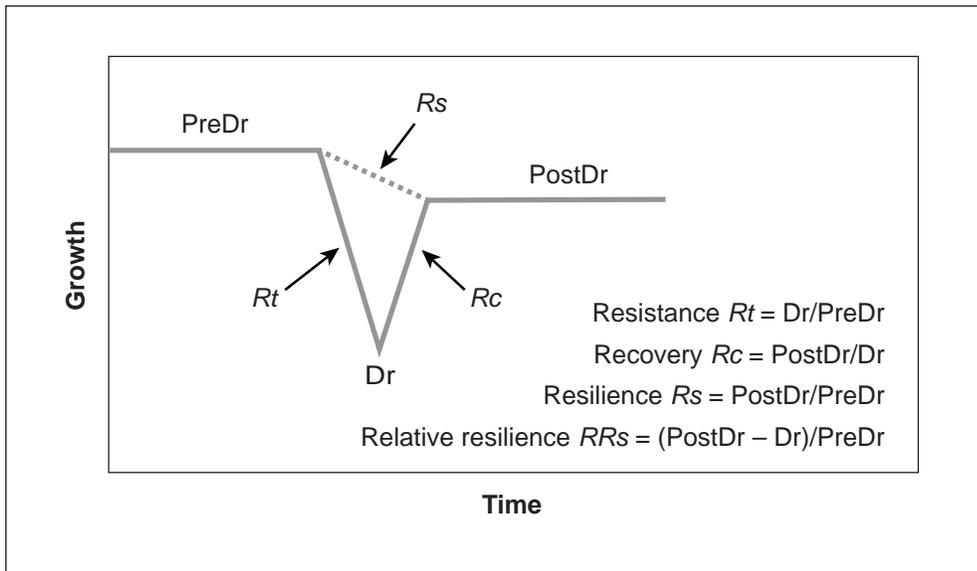


Figure 14-2—Conceptual diagram of resilience components for tree growth response to a drought event (*Dr*), modified from Lloret et al. (2011). Resistance (*R<sub>t</sub>*), recovery (*R<sub>c</sub>*), resilience (*R<sub>s</sub>*), and relative resilience (*RR<sub>s</sub>*) are calculated based on tree growth prior to (*PreDr*), during (*Dr*), and after (*PostDr*) a drought event.

species reproduction, understory plant diversity, tree mortality rates, etc.). Managers might consider using this approach when monitoring and measuring ecosystem response to new forest practices and changing climate conditions.

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