Simulating Historical Variability in the Amount of Old Forests in the Oregon Coast Range

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Abstract: *We developed the landscape age-class demographics simulator (LADS) to model historical variability in the amount of old-growth and late-successional forest in the Oregon Coast Range over the past 3,000 years. The model simulated temporal and spatial patterns of forest fires along with the resulting fluctuations in the distribution of forest age classes across the landscape. Parameters describing historical fire regimes were derived from data from a number of existing dendroecological and paleoecological studies. Our results indicated that the historical age-class distribution was highly variable and that variability increased with decreasing landscape size. Simulated old-growth percentages were generally between 25% and 75% at the province scale (2,250,000 ha) and never fell below 5%. In comparison, old-growth percentages varied from 0 to 100% at the late-successional reserve scale (40,000 ha). Province-scale estimates of current old-growth (5%) and late-successional forest (11%) in the Oregon Coast Range were lower than expected under the simulated historical fire regime, even when potential errors in our parameter estimates were considered. These uncertainties do, however, limit our ability to precisely define ranges of historical variability. Our results suggest that in areas where historical disturbance regimes were characterized by large, infrequent fires, management of forest age classes based on a range of historical variability may be feasible only at relatively large spatial scales. Comprehensive landscape management strategies will need to consider other factors besides the percentage of old forests on the landscape, including the spatial pattern of stands and the rates and pathways of landscape change.*

Simulación de Variabilidad Histórica en la Cantidad de Bosque Maduro en la Cordillera Costera de Oregon

Resumen: *Desarrollamos un simulador de demografías de clases de edades en paisajes (LADS) para modelos la variabilidad histórica de la cantidad de bosque maduro y bosque sucesional tardío en la cordillera costera de Oregon, USA para los últimos 3,000 años. El modelo simuló patrones espaciales y temporales de incendios junto con fluctuaciones en la distribución de clases de edades del bosque utilizando datos de un número de estudios dendroecólogicos y paleoecológicos existentes. Nuestros resultados indican que la distribución histórica de las clases de edades fue altamente variable y que la variabilidad incrementaba con una disminución en el tamaño del paisaje. Procentajes de bosque maduro simulados fueron generalmente entre 25% y 75% a escala de provincia (2,250,000 ha) y nunca fue menor a 5%. En comparación, los porcentajes de bosque maduro variaron de 0% a 100% a escala de reserva sucesional tardía (40,000 ha). Las estimaciones actuales a escala de provincia para bosque maduro (5%) y bosque sucesional tardío (11%) en la cordillera costera de Oregon fueron más bajas que las esperadas bajo el régimen de incendios históricos simulado, aún cuando se consideró un error potencial en nuestros parametros estimados. Estas incertidumbres limitan, sin embargo, nuestra habilidad para definir con precisión los rangos históricos de variabilidad. Nuestros resultados sugieren que en áreas donde los regímenes de perturbación histórica estuvieron caracter-*

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izados por incendios grandes poco frecuentes, el manejo de clases de edades del bosque basado en un rango de variabilidad histórica puede ser posible solo a escalas espaciales relativamente grandes. Estrategias de manejo de paisaje comprensivas necesitarán considerar otros factores además del porcentaje de bosque maduro en el paisaje, incluyendo los patrones espaciales de bosque y las tasas y vías de cambio del paisaje.

Introduction

The concept of historical variability has emerged as a paradigm for ecosystem management, species conservation, and landscape restoration in western North America (Cissel et al. 1994; Morgan et al. 1994; Swanson et al. 1994). This approach is based on the premise that ecosystems are naturally dynamic and that native species have adapted to disturbance-driven fluctuations in their habitats over past millennia (Bunnell 1995). Because it is not feasible to develop and implement specific management strategies for every species, a logical alternative is to manage for landscape patterns and processes that fall within the historical range of variability. This type of management is essentially a "coarse-filter" approach (Hunter 1990), which assumes that maintaining a range of forest types similar to historical distributions will preserve most of the native species that depend on these habitats. The idea of using historical variability as a basis for forest management also has taken hold outside the United States. The international forestry community recently outlined a set of criteria and indicators for evaluating the sustainable management of temperate and boreal forests in the "Santiago Declaration" of 1995. One of their suggested indicators for assessing ecosystem health was the "area and percent of forest affected by processes or agents beyond the range of historical variability." Applying this indicator of forest health requires new tools and methodologies for quantifying historical variability and making comparisons with current landscape conditions.

Our objectives were (1) to build a model of historical fire regimes that could be parameterized with available data, (2) to use the model to characterize variability in old forests over a range of spatial and temporal scales, (3) to evaluate the sensitivity of model predictions to uncertainties in the parameter estimates, and (4) based on these results, to assess the implications of using the range of historical variability in forest management. We developed the landscape age-class demographic simulator (LADS) as a tool for integrating information from dendroecological and paleoecological studies in order to simulate historical fire regimes and estimate variability in the past proportions of forest age classes. We used the model to estimate historical variability in the amount of old-growth and late-successional forests in the Oregon Coast Range based on a 3000-year record of fire frequencies derived from charcoal fragments in lake sediments (Long et al. 1998). We compared historical trends for landscapes the size of provinces (2,250,000+ ha), national forests (302,500 ha), and late-successional reserves (40,000 ha) to examine how temporal patterns may have varied at different spatial scales.

Pacific Northwest Landscape Dynamics

Forest age classes provide a useful criterion for delimiting patches with similar ecological characteristics. In Douglas-fir (*Pseudotsuga menziesii*) forests of the Pacific Northwest, for example, many structural attributes are correlated with stand age or time since the last stand-replacing disturbance. Young post-fire forests (40–80 years) are characterized by high overstory tree densities and relatively low cover of understory herbs and shrubs, and they may have high levels of residual snags and logs (Spies & Franklin 1991). Mature forests (80–200 years) are distinguished by low levels of snags and down wood compared with younger and older age classes. Old-growth forests (>200 years) are characterized by high densities of large (>100 cm in diameter) Douglas fir, a broad range of tree sizes, a high percentage of trees with broken tops, high densities of shade-tolerant trees, and high levels of snags and down wood.

Wildfire was historically the predominant coarse-scale disturbance in Pacific Northwest landscapes, although other events such as windstorms, disease outbreaks, and volcanic eruptions were locally important (Agee 1993). Fire effects varied with vegetation and weather conditions at the time of the burn. High-severity fires killed most of the trees on a site, resetting it to the beginning of the successional trajectory described above. Lower-severity fires often killed only a portion of the overstory trees, creating multicohort stands that retained some components of old-growth structure. Historical fire regimes exhibited considerable regional variability. In the Central Washington Cascades, for example, the fire history of Mt. Rainier National Park was characterized mostly by high-severity fires, with a natural fire rotation of more than 400 years (Hemstrom & Franklin 1982). In comparison, the drier Central Oregon Cascades had fire rotations on the order of 95–149 years, with a larger proportion of low-severity burns (Morrison & Swanson 1990). The fire regime of the central Oregon Coast Range consisted of a mixture of high- and low-severity fires, with a natural fire rotation estimated at 271 years (Impara 1997).

Humans have dramatically altered these disturbance regimes during the past century. Fire suppression has increased fire-return intervals by an order of magnitude, and timber harvesting on a cycle of 30–100 years has become the dominant disturbance. The higher disturbance frequency has increased the proportion of young forest in Pacific Northwest landscapes and decreased the proportion of old forest habitats. This shift has led to concern that current amounts of old-growth habitat are far below historical levels and that the decline in old-growth forests has threatened populations of native fish and wildlife species (Forest Ecosystem Management Assessment Team 1993).

Previous estimates of historical amounts of old growth have been made for single points in recent time and have used methods that assume steady-state conditions. Using data from a 1933 forest survey, Ripple (1994) estimated that old-growth forest occupied 43% of the Oregon Coast Range landscape during the late nineteenth century and 61% of the landscape before the large fires of the 1840s. Teensma et al. (1991) estimated Coast Range old growth at 40% of the landscape in 1850 and 46% in 1890. In comparison, current estimates of the proportion of old growth in the Siuslaw National Forest range from 5% (Bolsinger & Waddell 1993) to 16% (Congalton et al. 1993). The actual percentage of old growth in the Coast Range as a whole is even lower, because most of the old-growth forest on private lands has already been cut (Ohmann et al. 1988). Although the proportion of old growth has certainly decreased, it is impossible to assess whether the current landscape is actually outside the range of historical variability using estimates based on a single point in time. Better assessments of historical landscape dynamics are needed to provide a baseline for comparison with current conditions and a potential target for restoration activities. These assessments must recognize that the assumption of a steady-state level of old growth is unrealistic and instead quantify a range of possible historical landscape conditions.

Assessing Historical Variability

There are currently two main sources of data on presettlement landscapes in the Pacific Northwest: dendroecological studies based on fire scars and tree ages, and paleoecological studies based on pollen and charcoal from lake sediment cores. Dendroecological studies provide data on the size, frequency, and severity of historical forest fires in the Pacific Northwest (Hemstrom & Franklin 1982; Teensma 1987; Morrison & Swanson 1990; Wallin et al. 1996; Impara 1997). The temporal extent of these studies is limited, however, because evidence of past disturbances is overwritten by more recent events. This erasure problem also makes it difficult to directly infer past distributions of forest structures. Paleoecological studies provide long-term records of fire frequency and species composition (Worona & Whitlock 1995; Long et al. 1998), but these data have a limited spatial resolution and cannot be used to directly infer the proportions of the landscape in different forest structures.

Mathematical models provide a means for integrating disturbance data from retrospective studies with current knowledge of fire behavior and stand dynamics to estimate historical proportions of forest age classes on the landscape. For example, the exponential fire frequency equation (VanWagner 1978) has been applied to estimate the expected percentage of the landscape in one or more age classes for a given stand-replacing fire return interval (Booth 1991; Ripple 1994; Johnson et al. 1995; Lesica 1996). The advantages of the exponential equation are its conceptual and mathematical simplicity. The price of this simplicity, however, is a number of potentially unrealistic assumptions (Boychuk et al. 1997). In addition, the exponential equation produces only a steady-state estimate of the expected proportion of the landscape in a particular age class.

Stochastic simulation models are logical extensions of the classic exponential equation that allow for more complex parameterizations and can be used to estimate temporal variability in the proportions of forest age classes. Our simulation approach is similar to several previous modeling efforts. Benda and Dunne (1997) used a stochastic model of forest age-class dynamics to drive watershed-scale simulations of sediment production in the Oregon Coast Range. The DISPATCH model (Baker et al. 1991) has been used to explore landscape responses to climate change (Baker 1995) and fire suppression (Baker 1992, 1993, 1994). The FLAP-X model (Boychuk & Perera 1997; Boychuk et al. 1997) was used to demonstrate how spatially and temporally autocorrelated disturbances increased temporal and spatial variability in a boreal landscape. To make the LADS model applicable to Pacific Northwest forests, we expanded the methodology of existing landscape-disturbance models to include a fire-severity parameter that allowed us to model both high-severity and low-severity disturbances. We also incorporated a U-shaped function describing flammability as a function of time since fire, which appears to be appropriate for Pacific Northwest forests (Agee & Huff 1987).

Model Overview

Landscape Dynamics

The landscape age-class dynamics simulator represented the forest landscape as a two-dimensional grid of 100-ha cells. Only two state variables were tracked for each grid cell. Stand age (AGE) was the time since the last high-severity fire. Time since fire (TFIRE) was the time since the last low- or high-severity burn. High-severity fires were assumed to kill most of the vegetation in a cell, so the post-fire forest was a single cohort of trees establishing after the fire. Low-severity burns were assumed to kill only a portion of the trees in a cell, so the post-fire forest was a multicohort stand consisting of both pre- and post-fire vegetation. Each patch was classified as a particular structure type as a function of AGE and TFIRE (Fig. 1).

When a fire occurred, it created a disturbed patch on the landscape. Following a high-severity fire, AGE and TFIRE were both reset to 0 for all cells in the patch, causing it to revert to the open class. This class encompassed the period before canopy closure. Both AGE and TFIRE were incremented each year to simulate the successional development of the forest. If cells were not subsequently disturbed, they progressed through the young single-cohort, mature single-cohort, and old single-cohort classes as AGE and TFIRE increased (Fig. 1). Following a low-severity disturbance, TFIRE was reset to 0 for all cells in the patch, whereas AGE remained unchanged. Cells within the patch cell moved into the

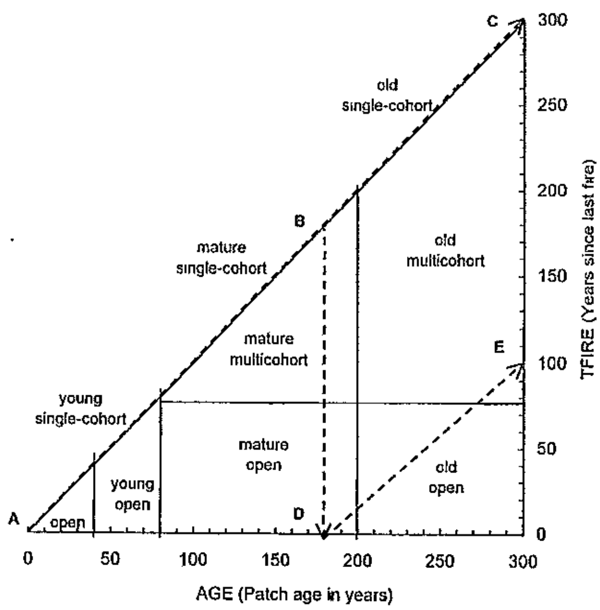

Figure 1. Forest age classes used in the landscape age-class demographics simulator model. Age classes are defined based on patch age (AGE) and time since last high- or low-severity fire (TFIRE). Dashed lines represent successional pathways for two hypothetical patches. Both patches start at A and proceed through the open young single-cohort and mature single-cohort classes. One patch does not experience any subsequent fires and eventually reaches C in the old single-cohort class. The other patch experiences a low-severity fire at B and reverts to the mature open class at D. This patch then passes through the old open class and eventually reaches E in the old multicohort class.

open young, open mature, or open old class, depending on their AGE values. Once TFIRE reached 80, the cells entered either the mature multicohort or old multicohort class, depending on their AGE value. Additional low-severity fires caused cells to again cycle temporarily into one of the open stages and then back to a multicohort class (Fig. 1).

Fire Regime

The fire regime for a given simulation was described by three parameters. Natural fire rotation (NFR) was the mean time required to burn an area equal in size to the simulated landscape. Fire extent was expressed as mean fire size (MSIZE). Fire severity (FSEV) was a value between 0 and 1 that indicated the proportion of each fire that was high in severity. To simulate individual fires, the natural fire rotation was converted into the mean number of fires per year on the landscape. This relationship was expressed as MFREQ = TSIZE / (MSIZE × NFR), where TSIZE equals the size of the simulated landscape (Boychuk et al. 1997). The number of fires occurring each year was then modeled as a Poisson random variable with parameter λ computed as $\lambda = 1/\text{MFREQ}$.

Modeling fire occurrence as a Poisson random variable implied that the number of fires occurring in a given year was independent of the number of fires occurring in any preceding year. Thus, the temporal pattern of fire events was random rather than aggregated or dispersed over time. This was a reasonable model for fire occurrence in the Pacific Northwest given the lack of evidence to support the hypothesis of regular fire occurrence (Agee 1993). Fire occurrence was also modeled as a nonhomogeneous Poisson process by allowing λ to vary as a function of time. This approach allowed for long-term fluctuations in the fire frequency as a function of climate change, whereas the short-term patterns of fire occurrence were still essentially random.

Fire size was modeled as a geometric random variable. The parameter of the geometric random variable, p, was related to the mean fire size, MSIZE, as $p = 1/\text{MSIZE}$. Modeling fire size as a geometric random variable assumed that the fire size distribution is negatively skewed, consisting of many small fires but only a few large fires. Evidence from dendroecological reconstructions suggests that the assumption of a negatively skewed fire-size distribution is generally appropriate in Pacific Northwest Douglas-fir forests (Morrison & Swanson 1990; Impara 1997). Fire severity was modeled as a Bernoulli random variable with parameter k = FSEV. For each cell burned, a U(0,1) random variable was generated and compared to k. If the random number was $<k$, the fire in that cell was modeled as a high-severity burn. Otherwise, the fire was modeled as a low-severity burn.

We modeled fire frequency, size, and severity as independent random variables. Some studies suggest that these disturbance elements are correlated, with moist conditions favoring infrequent, large, stand-replacing fires and drier conditions favoring more frequent, smaller, low-severity fires (Morrison & Swanson 1990; Agee 1993). Although this is a reasonable hypothesis, we did not have enough data to allow for reliable estimates of the covariance among these three probability distributions. Rather than introduce additional, questionable parameters into the model, we elected to make the simplifying assumption of independence. We also did not include topographic effects on fire frequency and severity because each grid cell was fairly large (100 ha) and potentially encompassed a wide range of topographic characteristics. Although topographic variability can influence fuel moisture levels as well as the rate and direction of fire movement (Agee 1993), we assumed that these fine-scale effects would average out within the larger cells.

Fire Spread

Each cell had a fire susceptibility value (SUSC) that controlled the probability of fire initiating in a cell and the rate of fire spread through the cell. Fire susceptibility was modeled as a function of time since the last fire (TFIRE). The shape of the fire susceptibility curve was based on research that indicated that rates of fire spread are highest immediately following a fire, reach a minimum at approximately 100 years after a disturbance, and then gradually increase with age (Agee & Huff 1987). The initial decrease in fire susceptibility was modeled as an exponential decay function:

$$\mathrm{SUSC1} = \mathrm{MAXS1} \times \exp(-a \times \mathrm{TFIRE}),$$

where MAXS1 is the value of SUSC1 when TFIRE equals 0, and a controls the rate of decay with increasing TFIRE.

The opposing trend of increasing susceptibility was modeled as a general logistic function:

$$\mathrm{SUSC2} = \mathrm{MAXS2}/(1 + \exp[b - c \times \mathrm{TFIRE}]);$$

where MAXS2 is the asymptotic maximum of SUSC2, b controls the value of TFIRE at which SUSC2 begins to increase, and c controls the rate of increase.

Fire susceptibility was the higher of these two functions at any point in time, expressed as

$$\mathrm{SUSC} = \max(\mathrm{SUSC1}, \mathrm{SUSC2}).$$

In all model runs we used parameter values of MAXS1 = 1.0, $a = 0.025$, MAXS2 = 0.4, $b = 2.5$, and $c = 0.02$. These values were chosen to produce a qualitative approximation of Agee and Huff's (1987) plot of spread rate and flame length as a function of time since fire (Fig. 2).

When a fire occurred, a random cell was selected as the starting point. Probability of selection was weighted such that ignition was more likely to occur in cells with higher susceptibilities. A stochastic cellular automata algorithm (Hogweg 1988) was then used to distribute the fire across the landscape. Fire moved from burned cells into neighboring unburned cells, assuming a four-cell neighborhood. The time required for fire to burn through a cell and into adjacent cells was assumed to be a geometric random variable with mean equal to 1/SUSC. Thus SUSC represented the probability that fire will spread from a burned cell into neighboring, unburned cells during one iteration of the fire spread algorithm. During each iteration, every unburned cell with a neighboring burned cell was assigned a probability of becoming burned. This probability was equal to the intersection of the probabilities of fire burning through its neighboring cells, expressed as

$$P_{\mathrm{burn}} = 1 - \prod_{i=1}^{n} (1 - \mathrm{SUSC}_i).$$

A U(0,1) random variable was generated for each cell with $P_{\mathrm{burn}} > 0$, and the cell was burned if this random variable was $< P_{\mathrm{burn}}$. The algorithm continued iteratively until an area equal to the appropriate fire size was burned. The overall affect of this algorithm was to allow fire to spread most rapidly through cells that had recently burned and to burn a larger proportion of these cells than the other age classes. Fire spread was slowest in cells that were close to 100 years since burning, but it increased after 100 years to a maximum at approximately 300 years since burning (Fig. 2). The actual proportions of cells burned in these different age classes

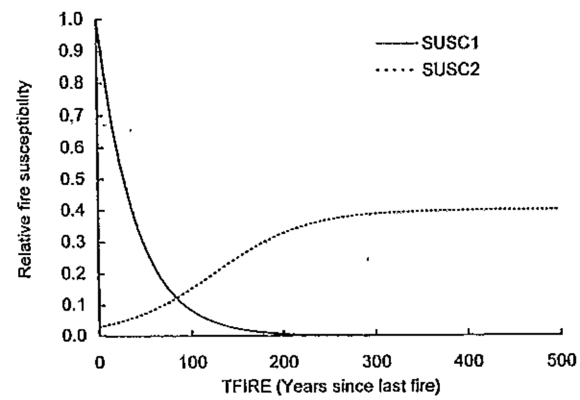

Figure 2. Fire susceptibility function used in the landscape age-class demographics simulator model. Decreasing susceptibility following fire arises from the decay of residual fuels (SUSC1) and is modeled as a negative exponential equation. Increasing susceptibility with age reflects the gradual buildup of fuels (SUSC2) and is modeled as a general logistic equation. Fire susceptibility at a particular time since burning (TFIRE) is computed as the maximum value of SUSC1 and SUSC2.

varied depending on the locations of the fire ignition and landscape patterns at the time of the fire.

Running the Model

At the beginning of a model run, the AGE value for each cell was initialized as a geometric random variable with a user-specified mean. We assumed that TFIRE and AGE were equal for all cells on the initial landscape. The simulation then proceeded for a number of model years equal to the user-specified "burn-in" period, which allowed time for the simulated disturbance regime to overwrite the initial landscape. After the burn-in period, the model continued to run for a given length of time, during which summary statistics were computed at user-specified intervals. The basic summary statistics were the percentage of the landscape in each structure type for each model year. The LADS model was coded in the C programming language and can be compiled and run in both PC and UNIX environments. Source code is available upon request from M.C.W.

Methods

We simulated the proportion of old forest in an Oregon Coast Range landscape over the past 3000 years using a time-series of fire frequencies derived from charcoal analysis of lake sediments (Long et al. 1998). We chose this temporal extent because pollen records indicate that the species composition of the Central Coast Range over the past 3000 years was generally similar to that of modern forests, consisting mainly of Douglas fir, with varying levels of red alder (*Alnus rubra*), western hemlock (*Tsuga heterophylla*), and western redcedar (*Thuja plicata*) (Worona & Whitlock 1995). Many of the unique structural characteristics of old forests in the Pacific Northwest arise as a direct result of this species composition. The long life span, high growth potential, and decay resistance of Douglas fir leads to accumulations of large dominant trees, snags, and wood. As dominant trees die from disease or windthrow, shade-tolerant species such as western hemlock and western redcedar establish in and around canopy gaps, producing both horizontal and vertical heterogeneity in the forest canopy. We assumed that if these same species were present in the historical landscape, old forests would have developed the same general structural characteristics they have today.

We converted these fire frequencies into mean fire return intervals using the equation MFRI = 1/MFF, where MFRI is the mean number of years between fire occurrences and MFF is the mean number of fires/year. We then made the assumption that the MFRI for the Little Lake basin was equivalent to the natural fire rotation (NFR) for the study area as a whole. A scarcity of data limited our ability to determine the probability distribution of historical fire sizes. Size estimates were available for some of the large nineteenth-century Coast Range fires, including the Nestucca (120,000 ha), Siletz (325,000 ha), and Yaquina (195,000 ha) burns (Loy et al. 1976). We decided to designate the 325,000-ha Siletz fire as the largest likely fire size and assumed that its size exceeded 99% of all historical Coast Range fires. If fire sizes are geometrically distributed, this is equivalent to specifying a mean fire size (MFS) of 70,600 ha. We estimated a fire severity (FSEV) value of 0.7 using dendroecological data that described the proportions of fire severity classes for four large historical coast-range fires (Impara 1997).

Although these parameter values seemed reasonable given prevailing conceptions of Pacific Northwest fire regimes (Agee 1993), there was still uncertainty associated with them. For example, fire frequency estimates were based on extrapolation from a relatively small (330 ha) watershed (Long et al. 1998). Although comparison with dendroecological data suggested that the charcoal record reflected large, regional fire events more than local fires (Impara 1997), the degree to which this record represented the Coast Range as a whole was still unclear. Our estimation of mean fire size entailed an unknown degree of error because it required an arbitrary assumption about the percentage of fires smaller than the largest historical fire. The estimate of fire severity was also questionable because it was based on measurements from only four historical fires. To assess how much these uncertainties might affect the predicted ranges of historical variability, we carried out six additional sensitivity runs in which we varied each parameter above and below its estimated value (Table 1). We could not precisely quantify the degree of uncertainty in each parameter, so we simply increased and decreased both NFR and MSIZE by a factor of two. We also increased FSEV from 0.7 to its maximum value of 1.0 and decreased it by a similar amount to 0.4. These parameters encompassed what we considered a reasonable

Table 1. Parameter values used in simulating historical variability in the amounts of old-growth and late-successional forest in the Oregon Coast Range.

	*Parameter values**		
Run name	*NFR (years)*	*MFS (ha)*	*FSEV (proportion)*
Base (base)	152–294	70,600	0.7
High frequency (hfreq)	75–147	70,600	0.7
Low frequency (lfreq)	304–588	70,600	0.7
High size (hsize)	152–294	141,200	0.7
Low size (lsize)	152–294	35,300	0.7
High severity (hsev)	152–294	70,600	1.0
Low severity (lsev)	152–294	70,600	0.4

*NFR, natural fire rotation; MFS, mean fire size; FSEV, fire severity.

range of possible values for Pacific Northwest Douglas-fir forests (Agee 1993).

The simulation runs were carried out on a 150 × 150 (22,500 pixel) grid of 100-ha cells, comprising a total area of 2,250,000 ha. Each simulation was allowed a burn-in period of 300 years (approximately twice the initial NFR) at the initial NFR parameter setting. After the burn-in period, each simulation continued for 3000 years, allowing NFR to vary as a function of time. The FSEV and MSIZE parameters were held constant throughout the simulation. Summary statistics were computed at 10-year intervals. These statistics included the percentage of both old-growth forest and late-successional forest on the landscape. Old-growth forests included both the old single-cohort and old multicohort stages, whereas late-successional forests encompassed the old single-cohort, old multicohort, mature single-cohort, and mature multicohort stages (Fig. 1). Because the grain size of our model was fairly coarse (100 ha), we did not attempt to model fine-scale vegetation features such as riparian hardwood stands or shrub patches. Our definitions of old-growth and late-successional forests therefore do not necessarily imply homogeneous areas of old conifer-dominated forest. Instead, these classifications represent a mosaic that encompasses both conifer-dominated patches on ridges and hillslopes and hardwood-dominated patches in riparian areas.

We ran a total of 100 3000-year simulations for each parameter set and used the resulting probability distributions of percent old-growth and percent late-successional forests over time to derive estimates of the ranges of historical variability. For each simulation run, we used different-sized windows to compute summary statistics at the scales of province, national forest, and late-successional reserve. The province-scale window was set at 2,250,000 ha, equal in size to the simulated landscape. This is approximately equal in size to the Oregon Coast Range province. The national forest–scale window was set at 302,500 ha, approximately the size of the Siuslaw National Forest. The late-successional reserve–scale window was set at 40,000 ha, approximating the average size of late-successional and old-growth reserves specified under the Northwest Forest Plan (Forest Ecosystem Management Assessment Team 1993). We chose these particular scales to represent different-sized areas at which policy decisions might be made or management plans implemented in the Coast Range.

Temporal trends were summarized for the base run by computing mean values from the 100 independent model runs at each 10-year time step, along with the 5% and 95% quantiles. These quantiles delineated the range within which percent old growth and percent late successional were 90% likely to fall at any point along the timeline. Although the 5% and 95% quantiles do not have any particular ecological significance, they were a reasonable choice for displaying the range of values that were most likely to occur historically. We also computed the range of historical variability over the entire 3000-year simulation for the base run and compared it with ranges of historical variability from the six additional sensitivity runs. We further examined temporal patterns by summarizing "low old-growth periods," defined as periods during which percent old growth remained below 5%, over the 100 base-run simulations. The 5% value was chosen because it is similar to current estimates of old growth in the Coast Range and therefore allowed us to gain a sense of how frequently old-growth levels might have been this low in the past. We computed the percentage of time spent in a low old-growth state, the median length of low old-growth periods, and the maximum length of low old-growth periods.

Results

The mean values of both percent old growth (Fig. 4) and percent late-successional forest (Fig. 5) computed at the province scale exhibited a pattern similar to the 3000-year trend in NFR (Fig. 3). The mean value of percent old growth ranged from a low of 39% 2500–3000 years ago to a high of 55% 1500–2000 years ago (Table 2). Since that time, mean percent old growth declined to approximately 45% over the past 1000 years. The mean value of percent late-successional forest followed a similar pattern but showed slightly less temporal variation, ranging from 66% to 76%. At any given time, mean values of percent late-successional forest were 21–27% higher than mean values of percent old growth.

The amount of temporal variation, reflected in the range of the 5% and 95% quantiles and the variability of the mean values, increased for both percent old growth

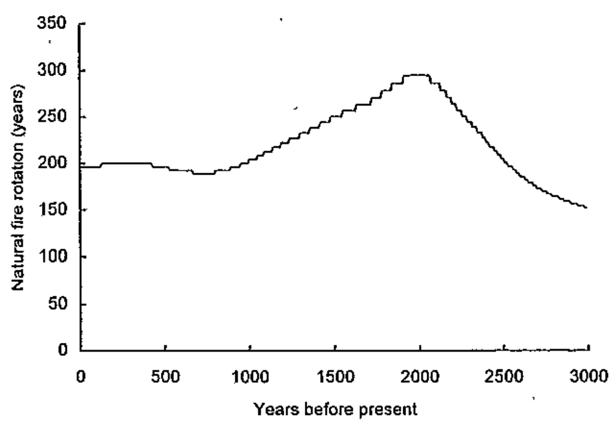

Figure 3. Time series of natural fire rotation for the Oregon Coast Range over the past 3000 years. Values were calculated with fire frequencies derived from a sediment core taken from Little Lake in the Central Coast Range (Long et al. 1998).

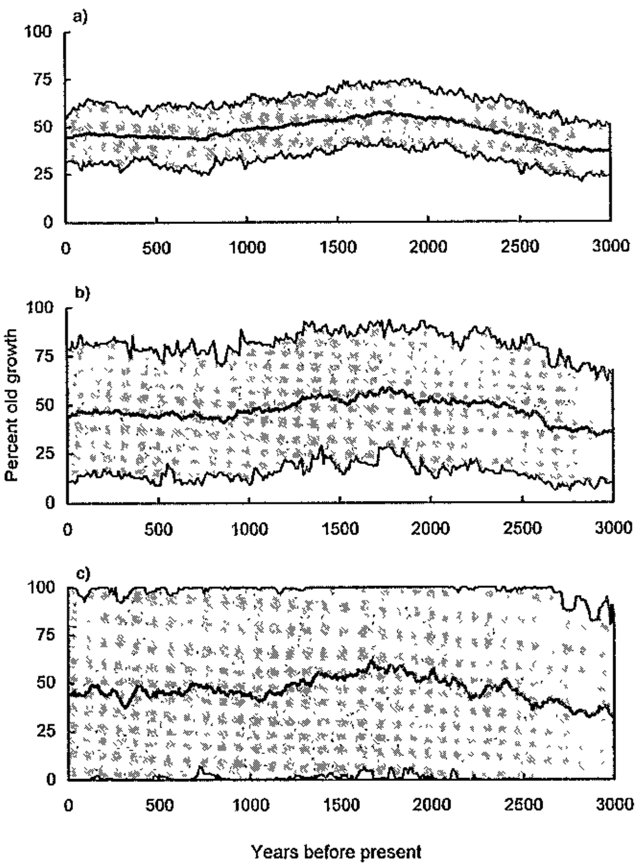

Figure 4. Simulated temporal variability of percent old-growth forest over the past 3000 years in the Oregon Coast Range, summarized at three spatial scales: (a) province scale (2,250,000 ha), (b) national forest scale (302,500 ha), and (c) late-successional reserve scale (40,000 ha). Solid line represents the mean of 100 independent model runs calculated at each time step. Shaded area is bounded by the 5% and 95% quantiles.

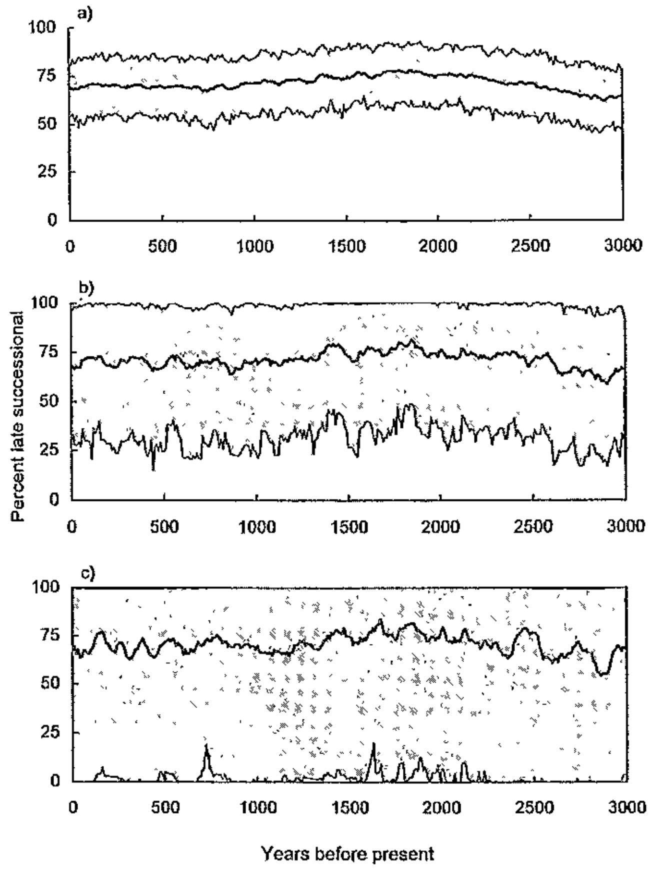

Figure 5. Simulated temporal variability in percent late-successional forest over the past 3000 years in the Oregon Coast Range, summarized at three spatial scales: (a) province scale (2,250,000 ha), (b) national forest scale (302,500 ha), and (c) late-successional reserve scale (40,000 ha). Solid line represents the mean of 100 independent model runs calculated at each time step. Shaded area is bounded by the 5% and 95% quantiles.

(Fig. 4) and percent late-successional forest (Fig. 5) as the spatial scale of the analysis decreased. At the late-successional reserve scale, variability was so high that the quantiles encompassed the entire 0–100% ranges for much of the simulation. Individual model runs showed rapid fluctuations in percent old growth at all three spatial scales, with several distinct peaks and troughs occurring within any 1000-year window (Fig. 6).

At the province scale, the mean old-growth percentage over the past 3000 years was 47% in the base run (Fig. 7). Reducing the natural fire rotation by half in the high-frequency run lowered the mean old-growth level by 22% and slightly narrowed the range of variability. Doubling the natural fire rotation in the low-frequency run increased mean old growth by 20%. Changing the mean fire size did not alter the mean old-growth level. Doubling the mean fire size in the high-size run increased the range of variability, whereas decreasing the mean fire size by half in the low-size run reduced the range of variability. Modeling all fires as catastrophic in the high-severity run decreased the mean old-growth level by 7%, whereas reducing fire severity to 0.4 in the low-severity run increased mean old growth by 8% and slightly narrowed the range of variability. Similar patterns were visible at the national forest and late-successional reserve scales. For a given parameter set, variability in the amount of old growth increased with decreasing landscape size, whereas the mean old-growth level remained constant. Similar trends emerged when late-successional forests were considered (Fig. 8). Changes in fire severity did not affect the distribution of late-successional forests, however, because our model assumed that forests reached the late-successional stage 80 years after either a catastrophic or a partial fire (Fig. 1)

Table 2. Simulated ranges of historical variability of old-growth and late-successional forests in the Oregon Coast Range, summarized for 500-year intervals at the province scale.

	Old-growth percentage			*Late-successional percentage*		
Years before present	*5% quantile*	*mean*	*95% quantile*	*5% quantile*	*mean*	*95% quantile*
0–500	30.6	45.5	61.3	53.3	69.8	85.0
501–1000	28.4	45.2	61.6	52.1	69.4	84.4
1001–1500	33.6	50.9	66.8	55.7	73.0	88.3
1501–2000	38.9	55.4	72.9	59.2	76.2	91.2
2001–2500	33.5	49.9	66.4	56.5	73.4	89.0
2501–3000	24.1	39.0	55.2	48.8	65.9	82.5

Landscapes spent more time in a low old-growth state as the spatial scale of analysis decreased. The median and maximum length of individual low old-growth periods also increased with decreasing scale. At the province scale, percent old growth never fell below 5%. At the national forest scale the landscape was in a low old-growth state 0.8% of the time, with a median low old-growth period length of 30 years and a maximum length of 220 years. At the late-successional reserve scale the landscape was in a low old-growth state 12.5% of the time, with a median low old-growth period length of 90 years and a maximum length of 320 years.

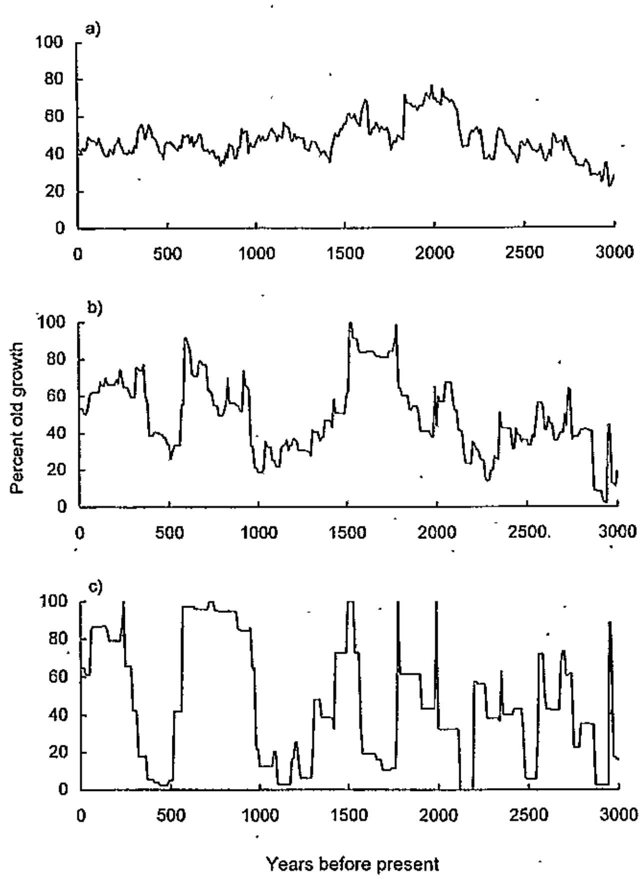

Figure 6. Percent old-growth values over the past 3000 years from a single model run at three spatial scales: (a) province scale (2,250,000 ha), (b) national forest scale (302,500 ha), and (c) late-successional reserve scale (40,000 ha).

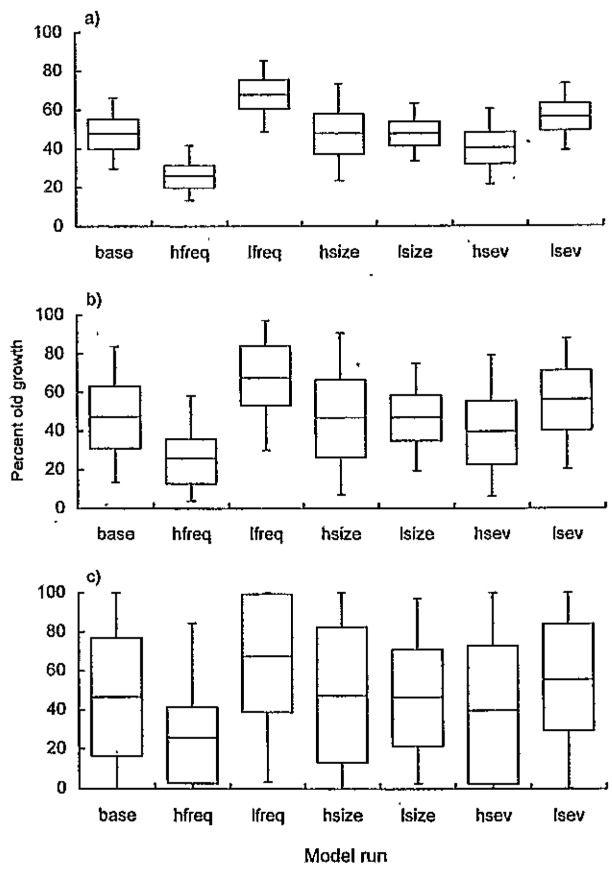

Figure 7. Simulated range of historical variability of percent old growth in the Oregon Coast Range for the past 3000 years summarized at three spatial scales: (a) province scale (2,250,000 ha), (b) national forest scale (302,500 ha), and (c) late-successional reserve scale (40,000 ha). Each boxplot represents one of the model runs described in Table 1. Center lines represent the mean, boxes delineate the 25% and 75% quantiles, and whiskers delineate the 5% and 95% quantiles.

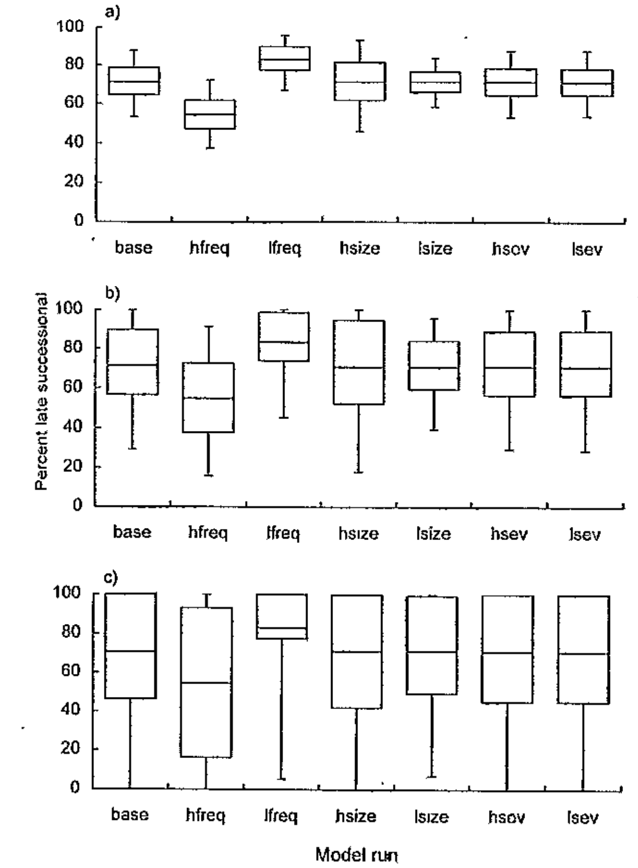

Figure 8. Simulated range of historical variability of percent late-successional forest in the Oregon Coast Range for the past 3000 years, summarized at three spatial scales: (a) province scale (2,250,000 ha), (b) national forest scale (302,500 ha), and (c) late-successional reserve scale (40,000 ha). Each boxplot represents one of the model runs described in Table 1. Center lines represent the mean, boxes delineate the 25% and 75% quantiles, and whiskers delineate the 5% and 95% quantiles.

Discussion

Comparison with Current Conditions

Our model-based estimate of a mean level of 46% old growth in the Oregon Coast Range during the past 500 years (Table 2) is reasonably close to other estimates of old growth in the late nineteenth century: 43% from Ripple (1994) and 40–46% from Teensma et al. (1991). Ripple's (1994) 61% estimate of old-growth area prior to the nineteenth-century fires is close to the 95% quantile estimated by our model (Table 1; Fig. 4) and may have represented an uncharacteristically high percentage of old growth that followed a long, fire-free period. This hypothesis is supported by dendroecological evidence from the central Coast Range, which shows that fire episodes occurring from the mid-seventeenth century through the mid-nineteenth century were mostly small, low-severity burns in the eastern portion of the Coast Range (Impara 1997). Although these comparisons do not constitute a formal validation of the model, the similarity of our results to two independent estimates of historical old growth does corroborate our predictions of historical landscape patterns.

Using data from the report of the Forest Ecosystem Management Assessment Team (FEMAT 1993), we estimated that the current combined federal forested land area in the Coast Range contained 11% old growth and 26% late-successional forests. If nonfederal land has less than 1% old growth and 5% late-successional forests (Ohmann et al. 1988), the entire Coast Range province contains approximately 5% old growth and 11% late-successional forests. These estimates fall far below the 5% quantiles for percent old growth and percent late-successional forest modeled at the province scale (Table 1; Figs. 4 & 5), suggesting only a small probability that levels this low would have arisen under the historical fire regime.

Comparisons of our simulation results with current landscape conditions are valuable from a conservation standpoint but should be interpreted cautiously, given the assumptions used in estimating parameters and the lack of sufficient data for model validation. In the case of the Coast Range, however, our comparison appears relatively robust to potential errors in parameter estimates. Current levels of old-growth and late-successional forests are so low that even halving the natural fire rotation or doubling the mean fire size would not bring them within the estimated province-scale ranges of historical variability (Figs. 7 & 8). Other comparisons might yield less certain conclusions. For example, would restoring the amount of old growth in the Coast Range province to 45% put the landscape within its range of historical variability? This value is close to the mean old-growth level in the base run but would still be below the actual range of historical variability if natural fire rotations were significantly longer than our estimates (Fig. 7).

Although our simulations provide evidence that current old-growth levels in the Coast Range are lower than would be expected under the historical fire regime, the uncertainty in our parameter estimates and their extrapolation to the provincial scale limits our ability to precisely define a range of historical variability. Improved data for parameterization would increase our confidence in the model predictions, although the lack of data on historical landscape patterns would still make it difficult if not impossible to validate the results. Accurate estimates of natural fire rotation are particularly important because errors can produce large shifts in the predicted range of historical variability. Paleoecological studies based on charcoal in lake sediments probably provide the best source of fire frequency data for long-term fire modeling; most dendroecological fire studies, in comparison, encompass only the past 500 years. Additional long-term fire frequency records from other parts of the Coast Range could either confirm our assumption that the charcoal record from Little Lake is representative of the province as a whole or provide the basis for modeling different fire regimes in different portions of the Coast Range.

More fire-severity data would also improve our predictions, although the percentage of older forests is not as sensitive to fire severity as it is to fire frequency. These data could be obtained from dendroecological fire-history reconstructions (Morrison & Swanson 1990; Impara 1997) or from a study of the effects of more recent forest fires (Kushla & Ripple 1997). Accurately modeling fire sizes is also critical because they influence the amount of variation around the mean old-growth percentage. Data on the relative frequency of fires by size class is needed to better estimate the fire-size distribution. One possible approach would be to derive a range of fire sizes from historical forest maps, although recent, large burns may have overwritten evidence of many of the smaller fires. Dendroecological fire-history reconstructions also provide estimates of fire sizes, but the fires are often truncated at the study area boundaries. Although it would be possible to use these fire-size distributions to model disturbance regimes for landscapes the size of the dendroecological study areas, extrapolation to the province scale would underestimate the sizes of large fires. Until better methods are developed to predict fire-size distributions at the province scale, our ability to accurately predict ranges of historical variability will be limited.

Ecological and Management Implications

Our analyses indicate that the historical variability of forest age classes depends on the spatial scale of observation or management. The mean fire size relative to the analysis area increased as the spatial scale of the analysis decreased, thereby increasing the variation in old-growth percentages over time. At the late-successional reserve scale, variability was so high that any amount of old growth from 0 to 100% might be considered historically probable. This phenomenon of increasing landscape variability with decreasing analysis scale relative to disturbance size has been observed in other empirical and modeling studies (Baker 1989; Turner et al. 1993; Boychuk & Perera 1997). Under a disturbance regime of large, relatively infrequent fires, a range of historical variability may be definable only for relatively large areas. In the case of the Coast range, management of forest age-class distributions based on historical variability related to fire regimes might not be appropriate or very useful for landscapes much smaller than 300,000 ha. In other landscapes with higher frequency, smaller size, and lower severity of historical fires, temporal variability in the amount of older forests would be lower and management based on historical variability might be feasible at smaller spatial scales.

Defining a range of historical variability to use as a reference is problematic because the amount of variability has changed over time. For example, our simulations indicate that the mean percentage of old growth over the last 1000 years ranged between 45% and 46%, but in the 1000 years before that the range was between 51% and 55% (Table 2). The choice of the period of time to use as a reference will depend on, among other things, similarity of climate, tree species present, and influences of past human populations. Although pollen analyses suggest that forest communities over the past 3000 years generally have been similar to present-day communities, shifts in species dominance indicate that climate has varied over this time period (Worona & Whitlock 1995). Although we might assume that a past climate is broadly similar to the present climate, differences in variability of drought and timing of storms and precipitation could still result in different disturbance regimes and differences in resulting landscape patterns.

The occurrence of conditions outside the range of historical variability at a single point in time is not necessarily a problem from a conservation perspective. What may be more important is the frequency and duration of the conditions outside a particular range. Current species and populations of the Coast Range have probably survived historical periods of low habitat availability by persisting in refugia during periods of low habitat availability. Although these refugia or population fragments are susceptible to extinction from stochastic forces, they frequently serve as source areas for population recovery when habitat increases again (Hunter 1990). But where the frequency or duration of unfavorable habitat conditions exceed the ability of species to recolonize areas following disturbance and succession, species might eventually be extirpated. The longer the habitat stays in a reduced state, the greater the risk to the population. Thus, the frequency or proportion of periods with low amounts of habitat is another criteria that can be applied to landscape management.

Our simulation results indicate that the percentage of old forests in the Coast Range varied from century to century but rarely remained at low levels for an extended period. These patterns suggest that the amount of old growth on the landscape recovered fairly rapidly following large disturbance events, even though reburns may have delayed forest succession on some sites. In the modern Coast Range landscape, by comparison, ownership has compartmentalized disturbance regimes. Approximately 64% of the land is privately owned, with 12% controlled by the state and 24% by federal agencies. On private industrial lands, the historical fire regime has been replaced by regularly timed timber harvests at 30- to 60-year rotations. The natural disturbance regime has thus been supplanted by a chronic disturbance that perpetually maintains these lands in the open and young age classes. These young, managed forests also tend to have less dead wood and a more homogeneous structure than young forests that establish following fire or other natural disturbances (Hansen et al. 1991). Stand-replacing disturbances have been greatly reduced on federal lands, with timber harvests restricted and wildfire limited by fire suppression. In the Siuslaw National Forest, for example, approximately 90% of the land base now lies within reserves for late-successional forests, stream protection, or wilderness areas. If fire suppression continues to be successful, these lands will eventually be dominated by mature and old forests, with few areas in the open or young age classes.

Simply managing for a particular amount of habitat may not ensure species survival if temporal disturbance patterns are not also sustained. Salmonid stream habitat, for example, is influenced by the relative amounts of sediment and large wood. Both of these components will vary over time depending on the relative rates of inputs to and outputs from streams. In the Coast Range, delivery of both sediment and large wood to streams is thought to be most common when watersheds are dominated by early successional vegetation, and landslides are frequent because of low root strength (Reeves et al. 1995). Although delivery of large wood is more common in the early seral stages, it is the late-successional stages when wood grows and accumulates on slopes as live and dead trees. If this scenario is true, then both stages are needed within a watershed over time to maintain fish habitat. Thus, the challenge to management at this scale may be to retain the full range of variability over time rather than just to maintain a particular distribution of age classes.

Other landscape characteristics, such as spatial pattern, can also affect habitat suitability for many species. In the case of the Northern Spotted Owl (*Strix occidentalis caurina*), for example, habitat fragmentation may be as important a factor in species decline as the overall decrease in the amount of old forest on the landscape (Thomas et al. 1990; Ripple et al. 1997). Clearcuts are generally smaller and more regularly distributed than historical fires and have produced a fragmented pattern with smaller patch sizes and higher edge densities than the historical landscape (Spies et al. 1994; Wallin et al. 1996). The structure and microclimate of old-growth forests are altered along clearcut edges (Chen et al. 1992, 1995), so the effective amount of interior old-growth habitat is less than if the same amount of late-successional forest was distributed in a few large patches. In landscapes where there is a spatial relationship between fire regimes and topography, physiographic patterns can serve as a template for landscape management (Cissel et al. 1998). Nevertheless, implementing management strategies to restore historical spatial patterns is likely to be more difficult than simply producing a certain percentage of the landscape in each age class (Wallin et al. 1994).

Conclusions

Our research demonstrates that although it is possible to use simulation models to estimate historical variability in forest age classes based on fire regime data, the predicted range of historical variability will be sensitive to errors in the parameter estimates. The small amounts of data available for model parameterization and validation currently limit our ability to accurately estimate historical landscape variability in the Coast Range and other large landscapes. Given these concerns, managers should exercise caution when incorporating historical variability estimates into land-management decisions.

No current land-management policies in the Coast Range set management goals based on an assumed range of historical variability. FEMAT used estimates of past proportions and ranges of old growth as a reference but not as a specific management target (FEMAT 1993). The Oregon Department of Forestry, in keeping with the Santiago Accord, may utilize the range of historical variability as an indicator of ecosystem health but not necessarily as a management goal (G. Lettman, personal communication). Until we can estimate ranges of historical landscape variability more accurately, it will be difficult to substantiate an argument for their use as precise forest-management goals. Despite these uncertainties, comparisons with historical variability are still useful as general indicators of forest health and the potential to sustain populations of native species. Our study supports assumptions within the Northwest Forest Plan (FEMAT 1993) that declines in the amount of old growth are unprecedented in recent history, and it provides an approximate target for restoring old growth on federal lands. Simply providing areas of habitat similar to historical levels, however, will not necessarily guarantee the survival of associated native species. Other landscape attributes such as the spatial arrangement of habitat types and rates and pathways of landscape dynamics will need to be considered as well.

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