Modeling Ecological Restoration Effects on Ponderosa Pine Forest Structure

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Abstract

FIRESUM, an ecological process model incorporating surface fire disturbance, was modified for use in southwestern ponderosa pine ecosystems. The model was used to determine changes in forest structure over time and then applied to simulate changes in aboveground biomass and nitrogen storage since exclusion of the natural frequent fire regime in an unharvested Arizona forest. Dendroecological reconstruction of forest structure in 1876, prior to Euro-American settlement, was used to initialize the model; projections were validated with forest measurements in 1992. Biomass allocations shifted from herbaceous plants to trees, and nitrogen was increasingly retained in living and dead tree biomass over the 116-year period (1876–1992). Forest conditions in 1992 were substantially degraded compared to reference presettlement conditions: old-growth trees were dying at accelerated rates, herbaceous production was reduced nearly 90%, and the entire stand was highly susceptible to high-intensity wildfire. Following an experiment initiated in 1993 to test ecological restoration treatments, future changes were modeled for the next century. Future forest structure remained within the natural presettlement range of variability under the full restoration treatment, in which forest biomass structure was thinned to emulate presettlement conditions and repeated low-intensity fire was reintroduced. Simulation of the control treatment indicated continuation of exceptionally high tree density, probably culminating in stand-replacing ecosystem change through high-intensity wildfire or tree mortality from pathogens. Intermediate results were observed in the partial restoration treatment (tree thinning only); the open forest structure and high herbaceous productivity found immediately after treatment were gradually degraded as dense tree cover reestablished in the absence of fire. Modeling results support comprehensive restorative management as a long-term approach to conservation of key indigenous ecosystem characteristics.

Key words: presettlement, Arizona, Gus Pearson Natural Area, FIRESUM, process model, fire regime, biomass.

Introduction

Restoring ponderosa pine forest ecosystems affected by a century or more of fire exclusion, old-tree harvesting, and livestock grazing is a major challenge. Across the western United States, ponderosa forests have become dense with small, young trees, herbaceous productivity has declined, and forest floor fuels have accumulated. Today forest landscapes support high-intensity wildfires and are increasingly susceptible to large-scale pathogen outbreaks (Covington & Moore 1994*a*). Restoration experiments have begun following several approaches based on thinning, burning, fuel treatments, seeding, and other treatments (Fiedler et al. 1988; Sackett et al. 1996; Covington et al. 1997; Scott 1998). Forecasting long-term effects of alternative treatments through ecological simulation modeling provides useful information to scientists, managers, and the public in making decisions about the nature and priorities of restoration efforts.

Two distinct modeling methods are available: statistical models based on analysis of past system behavior, and process models, often with stochastic elements, which seek to emulate underlying biological or physical processes. Because the fitting of statistical models to readily measurable data can be done with great precision, such models tend to be highly accurate in the near term, as long as the original environmental conditions under which the model was developed are maintained. Several statistical growth and yield models are available for ponderosa pine (e.g., Wycoff et al. 1982; Edminster et al. 1991).

However, because actual biological relationships are modeled, process models are more suitable for analysis of system behavior under changing environmental conditions, over long time periods (when mortality and regeneration processes are modeled), and under disturbance regimes, such as fire, which were not incorporated in the datasets used to develop statistical models. Short-term

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predictive accuracy may be reduced for process models as compared to statistical models, because data for model development may be more difficult to obtain and the variety of complex natural processes may be more difficult to model.

Successional change in western forests under alternative fire regimes has been simulated with the ecological process models SILVA (Kercher & Axelrod 1984) in California and FIRESUM (Keane et al. 1989; 1990) in the northern Rocky Mountains. The models simulate changes in forest species composition and fuel loading at the plot level under alternative fire regimes. They trace their development history to JABOWA (Botkin et al. 1972), a process-based, gap-replacement ecological model, which has been used to analyze ecosystem dynamics in northern hardwoods (e.g., Covington 1981). FIRESUM models overstory and understory ecological change under either user-determined or stochastic fire regimes, incorporating Rothermel's (1972) surface fire behavior model to predict fire behavior. The program can simulate fire effects on mortality, regeneration, and tree density, changes in species composition, forest density, and fuels over several centuries, with implications for fire danger, wildlife habitat, old-growth conservation, and other management issues. Keane et al. (1989; 1990) provide validation data and sensitivity testing for FIRESUM in ponderosa pine and mixed conifer forests. More recently, the gap-replacement and fire modeling components of FIRESUM have been integrated with the detailed mechanistic model FOREST-BGC (Running & Coughlan 1988; Running & Gower 1991) in a new model, FIRE-BGC, to permit successional modeling which incorporates biogeochemical process modeling at highly resolved time steps and large spatial scales (Keane et al. 1996). Because of its complexity and high input requirements, however, FIRE-BGC will require extensive testing and sampling before it can be extended to sites beyond the northern Rockies.

Beginning in 1992, we initiated an ecological restoration experiment in a fire-excluded ponderosa pine forest at the Gus Pearson Natural Area (hereafter GPNA). Three treatments were tested: (1) control; (2) full restoration of ecosystem structure (thinning to restore density, age, size, and spatial distributions of trees) and fire disturbance process (removal of accumulated duff, reintroduction of fire in prescription at four-year intervals); and (3) partial restoration (thinning only, without fire). The treatments and initial results are described by Covington et al. (1997). We selected FIRESUM to predict long-term treatment effects. Our study objectives were to: (1) modify and validate the FIRESUM ecological process model for a southwestern ponderosa pine forest, expanding the model capabilities with biomass and nutrient models; and (2) predict ecological changes at the actual restoration site, comparing and contrasting the long-term effects of alternative treatments.

Methods

The study area is a 4.7-ha experimental restoration site at the Gus Pearson Natural Area, in the Fort Valley Experimental Forest, approximately 15 km NW of Flagstaff, Arizona. Within the GPNA trees larger than 15.2 cm at breast height have been measured at 5- or 10-year intervals since 1920 (Avery et al. 1976 and unpublished data). The study area has gentle topography and a cool, subhumid climate. Mean annual precipitation is 57 cm, with approximately half occurring as snow. The remainder occurs as summer monsoonal rains following the spring/early summer drought. Soils are of volcanic origin, a fine smectitic complex of frigid Typic Argiborolls and Eutroboralfs (Covington et al. 1997). The ponderosa pine structure consists of groups of mature trees, characterized by larger size and yellowed bark, above dense thickets of smaller, dark-barked trees. Understory vegetation includes perennial grasses, primarily *Festuca arizonica* (Arizona fescue), *Muhlenbergia montana* (mountain muhly), and *Sitanion hystrix* (squirreltail), and forbs.

Treatments were applied in 1993–94 as shown in Figure 1. The entire study site was divided into 17 units, each approximately 0.2–0.3 ha in size and each containing examples of presettlement tree groups, postsettlement tree groups, and remnant grassy opening. Units on the eastern and southwestern corners were excluded from the experiment due to the proximity of a highway on the east and previous thinning in the Southwest. The

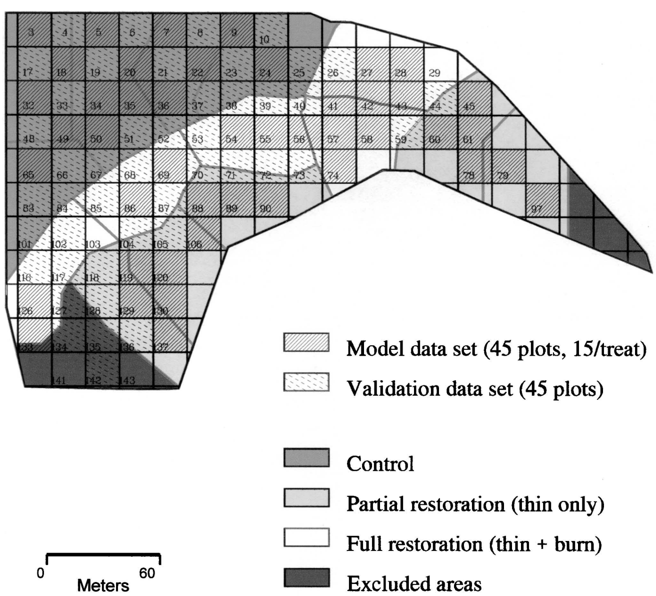

Figure 1. Model and validation datasets from the Gus Pearson Natural Area, Arizona. Numbered squares are the 20 × 20–meter plots used for modeling. Dark lines are unit boundaries (five replicated units of each treatment). Small areas at the eastern and southwestern corners were excluded from the experiment but tree data was applied for model development and validation.

treatment area was officially "decommissioned" from the GPNA so that thinning and burning could be carried out (one goal of the treatment was to reduce fire hazard next to the historic research headquarters). The full and partial restoration treatments were randomly assigned to 5 units each in the southern and eastern parts of the study site. The control treatment was constrained to 5 units in the northwestern portion of the site, an area that remained within the Research Natural Area. Both the control and treatment areas crossed the minor elevational and geomorphological gradients in the study site, although the southeastern end of the study site was relatively lower and flatter. Pretreatment soil characteristics (N, P, organic matter) and mature tree characteristics (predawn water potential, basal area growth, diameter, and competition index) were not significantly different between treatments (Stone 1997), strengthening inferences from the experiment.

The FIRESUM model for ponderosa pine/Douglas fir habitats, as documented by Keane et al. (1989), was enhanced or changed for southwestern ponderosa pine forests as follows. Tree biomass and nutrient modules using allometric ponderosa pine equations developed from northern Arizona data were added (W. W. Covington, unpublished data). An herbage module based on overstory–understory relationships in northern Arizona ponderosa pine was incorporated (Solomon 1985; see also Covington & Fox 1991). Several input parameters representing maximal variable levels were changed based on southwestern data, including: maximum age for ponderosa pine = 650 years (Swetnam & Brown 1992); fuel loading by timelag classes, 1 hr = 0.0570, 10 hr = 0.1731, 100 hr = 0.2500 (units kg/m^2, data from GPNA, Covington et al. 1997; Fulé & Covington 1994); herbaceous production (fire group 1) = 0.18 kg/m^2. Different seedling establishment rates were used for fire-exclusion versus frequent-fire modeling, since far more seedlings survive in the absence of fire (White 1985; Savage et al. 1996). Seedling establishment = 0.1 (seedings/m^2, average of data from Heidmann 1988 and Cormier 1990) for fire-exclusion modeling; seedling establishment = 0.005 (seedlings/m^2, data from White 1985 and Mast et al. 1999) for frequent-fire modeling. Mortality risk for ponderosa pine from fire was calculated using data for "light surface fires" and "hot surface fires" from Lindenmuth (1960).

Multiple runs of FIRESUM were made in two different ways. First, each run of the model consisted of 2 stochastic simulations. Second, 10 repetitions were made of each modeling scenario to expand the potential for stochastic variability, for a total of 20 simulations (2 × 10) per plot per scenario.

Two model validation approaches were taken. First, following the validation procedure used by Keane et al. (1989; 1990), data were developed at two points in time for the study area. Trees were measured and mapped in the study area in 1992. Forest structure was reconstructed using dendroecological methods for 1876, the final year of the frequent-fire regime that prevailed prior to Euro-American settlement (Dieterich 1980; Covington et al. 1997; Mast et al. 1999). FIRESUM was initialized with 1876 data and run for 116 years (1876–1992) without fire to estimate the 1992 forest structure. A total of 90 plots, each 400 m^2, were prepared from the Pearson dataset (Fig. 1). Because model runs based on the Pearson data were also needed to assess the effects of changes to model parameters, the dataset was split. Forty-five plots falling within the treatment units (15 per treatment) were used to test changes in model parameters (model data). The remaining 45 plots were set aside as a validation dataset. Model and validation data plots were interspersed.

The second validation approach tested the internal consistency and stability of the model. If the presettlement frequent-fire regime had continued after 1876 up to the present, forest structure would be expected to remain relatively similar to the 1876 conditions, in contrast to the substantial increases in tree density and basal area observed under fire exclusion. FIRESUM was initialized with the reconstructed 1876 data (all 90 plots) run for 500 years at a 4-year fire interval. Simulated fire conditions listed in Table 1 corresponded to relatively dry spring and summer burning, the natural fire season in the Southwest (Swetnam & Baisan 1996). Climatic and soils data were summarized from GPNA weather records and Avery et al. (1976).

Finally, using the validated model, we simulated the effects of the three treatments over the next century. The model dataset consisting of 45 plots, 15 plots in each treatment, was initialized in the treatment year (1994) and run till the year 2100 (106 years). Thinning was simulated by setting the input tree list equal to the actual post-thinning forest structure on each plot. For the burning treatment, fires were simulated at 4- and 10-year intervals. Mean density and basal area values in 1994 and 2100 were compared with paired *t*-tests (alpha = 0.05).

Results

Model Verification

Validation test results for the modified FIRESUM model are shown in Table 2. The predicted values were rea-

Table 1. Fire weather and fuel moisture conditions.

Variable	*Value*
Midflame windspeed	10 km/h
Relative humidity	40%
Moisture content: litter	5%
1-hour timelag fuel	6%
10-hour timelag fuel	8%
100-hour timelag fuel	8%

Table 2. Validation test results for the FIRESUM model on two Arizona ponderosa pine datasets (N = 45 plots each) from the Gus Pearson Natural Area. The model was initialized with reconstructed 1876 data and run for 116 years without fire. Model predictions were compared with 1992 measurements. Outputs from multiple stochastic runs (see text) had a coefficient of variation of 10% in density and 6% in basal area.

	Model Dataset			*Validation Dataset*		
Variable	*Observed*	*Predicted*	*Percent Difference (%)*	*Observed*	*Predicted*	*Percent Difference (%)*
Basal area (m^2/ha)	37.6	35.8	4.7	33.7	35.9	6.5
Density (trees/ha)	3603.3	2955.8	18.0	3160.0	2967.8	6.1
Tree biomass (kg/ha)	129600	141200	9.0	129600	140200	9.0
Herbaceous biomass (kg/ha)	70	95.9	37.0	70	101.4	44.9
Litter loading (kg/m^2)	0.314	0.577	83.8	0.314	0.587	86.9
1-hour fuel loading (kg/m^2)	0.042	0.057	35.7	0.042	0.057	35.7
10-hour fuel loading (kg/m^2)	0.123	0.173	40.7	0.123	0.173	40.7
100-hour fuel loading (kg/m^2)	0.265	0.250	5.7	0.265	0.250	5.7

sonably consistent with the contemporary measurements. Basal area predictions were within ±6.5% of observed values for both datasets, but density predictions were 6 to 18% lower than the observed values. The reduced density predictions may be related to the fact that modeled regeneration occurred early during the 1876–1992 simulation period, whereas the actual regeneration pulse at the Pearson Area was delayed until the early twentieth century, especially 1919 (Savage et al. 1996). Because regeneration was simulated earlier, the 1992 modeled density had a longer period of self-thinning. Ponderosa pine self-thinning is slow in the absence of fire (Cooper 1960), but it does occur. Dead post-settlement trees, mostly 1914–1919 origin (Savage et al. 1996; Mast et al. 1999), made up 16.2% of all postsettlement trees in the study area in 1992, providing a minimal estimate of self-thinning rates over approximately 80 years (not accounting for mortality of very small trees which may have decomposed completely by 1992). Differences in aboveground biomass predictions versus observations ranged from 9 (trees) to 45% (herbaceous). Forest floor loadings were more variable, with differences ranging from 6 (100-hr timelag woody fuels) to 87% (litter loading). The higher difference in litter loading may reflect inaccuracies in the modeling of litter production or decomposition rates or may be related to imprecision in litter bulk density data (Harrington 1986). The highest errors in proportional terms occurred for variables with low absolute values, such as herbaceous production. However, the total model error over the 166-year simulation was only ≈30 kg/ha for herbaceous production in absolute terms, a highly accurate estimate for this type of data.

Modeling for 5 centuries under a continuing frequent-fire regime showed fluctuations in tree density and basal area, but the behavior was relatively stable (Fig. 2). Pulses of tree regeneration were observed at approximately century intervals, affecting tree density (trees/ha) approximately 20 to 30%. Since the seedlings were small, basal area was not affected. This limited variation appears consistent with current understanding of presettlement forest dynamics and shows that the modeled processes are internally consistent.

Changes Since Fire Exclusion

Actual forest density increases during the fire-exclusion period from an average of 60.5 trees/ha and 10.6 m^2/ha basal area in 1876, to 3,098 trees/ha and 34.5 m^2/ha basal area in 1992, (Covington et al. 1997) were closely simulated by the model. Simulated herbaceous production declined as forest density increased (Fig. 3). The actual decline was probably more precipitous, as heavy livestock grazing rapidly removed understory vegetation (Dieterich 1980; also see 1909 GPNA photograph in Covington & Moore 1994*a*) while the model curve responded to the slower increase in overstory density. Simulated tree biomass increased by 73%, from approximately 81,500 kg/ha to 140,700 kg/ha, about 8% higher than the biomass calculated from tree measurements in 1992. Although herbaceous biomass was always a small fraction of tree biomass, the relative change was greater: a decline of 87% from approximately 780 kg/ha in 1876 to 99 kg/ha in 1992. The measured herbaceous production in 1992 averaged even less, 70 kg/ha (Table 2).

Shifts in nitrogen storage paralleled those in biomass. Nitrogen stored in trees increased by 115%, from 1,150 kg/ha to 2,460 kg/ha. The increase in tree N was disproportionately greater than the growth in tree biomass because the new, small trees contributed relatively more N-rich foliage than N-poor wood and bark. Herbaceous N, calculated from average N content for grass and forb species at GPNA (1.17%, average from Harris & Covington 1983 and S. C. Hart, unpublished data),

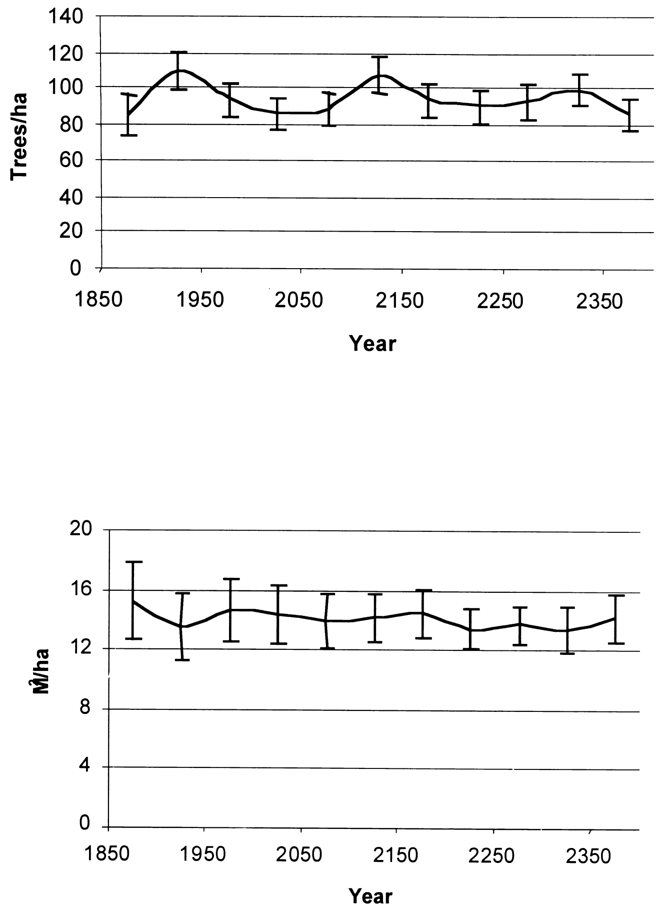

Figure 2. Five hundred year modeling of Gus Pearson Natural Area forest, assuming that fires had continued at 4-year intervals after 1876. Tree density is shown in the top graph and basal area in the bottom graph. Data are the average of 90 plots. Error bars are 1 standard deviation.

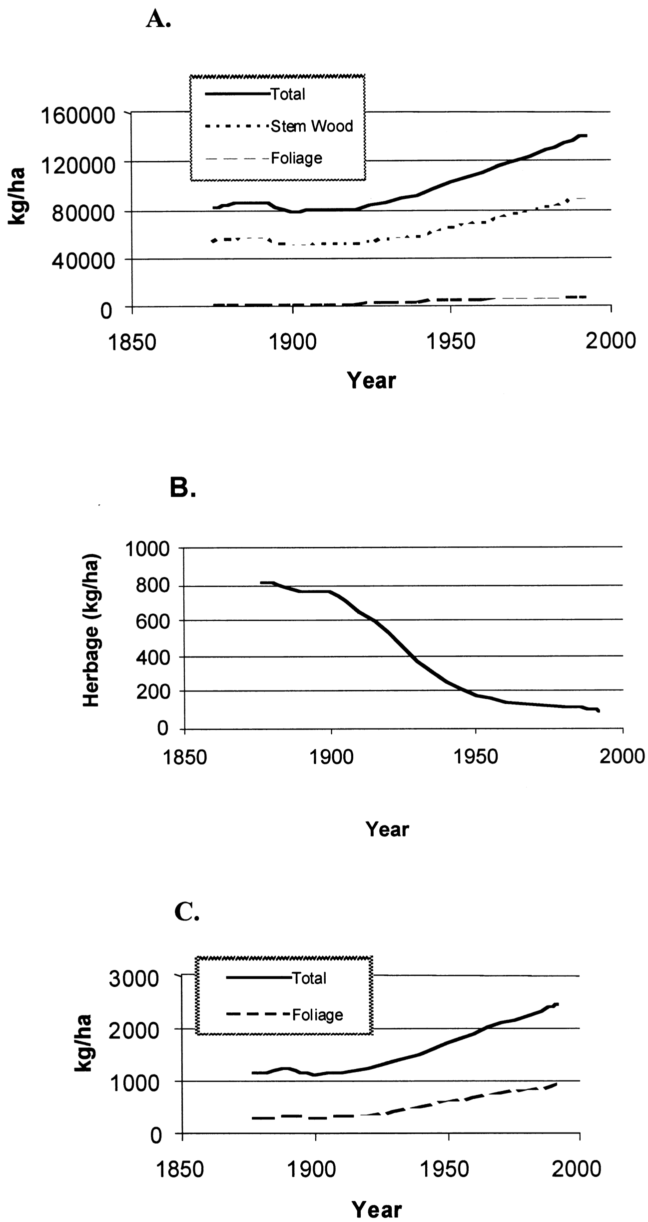

Figure 3. Modeled changes in tree biomass, herbaceous biomass, and tree N content from 1876, the final year of the presettlement frequent-fire regime, through 1992. Data are the average of 90 plots.

dropped from 7.4 kg/ha to 0.9 kg/ha, an 88% decrease. Nitrogen stored in forest floor material, calculated from forest floor N content in Covington and Sackett (1984) and Kaye and Hart (1998*b*), rose from 70 kg/ha to 1,340 kg/ha, an 1,800% increase. From all aboveground sources, modeled N storage rose from approximately 1,230 kg/ha in 1876 to 3,800 kg/ha in 1992. Increased vegetation and fuel biomass therefore represented a 209% increase in total N.

Effects of Restoration Treatments

The control treatment (Fig. 4) showed a gradual decline in tree density due to self-thinning in the extremely dense stands ($p < 0.001$), averaging 3,098 trees/ha at the start of the simulation period. Basal area increased slightly over most of the modeled period of 106 years, but the change was not statistically significant ($p = 0.210$). Herbaceous production remained at minimal levels throughout the modeled century (70–80 kg/ha). Nitrogen allocation remained primarily in trees and forest floor biomass.

The full restoration treatment—thinning, fuel treatment, and repeated burning at 4-year intervals—maintained a relatively constant density, but basal area increased slightly over the modeled period. Neither change was statistically significant ($p = 0.519$ and 0.748, respectively). The basal area increase may reflect growth expected from the relatively young trees that were retained

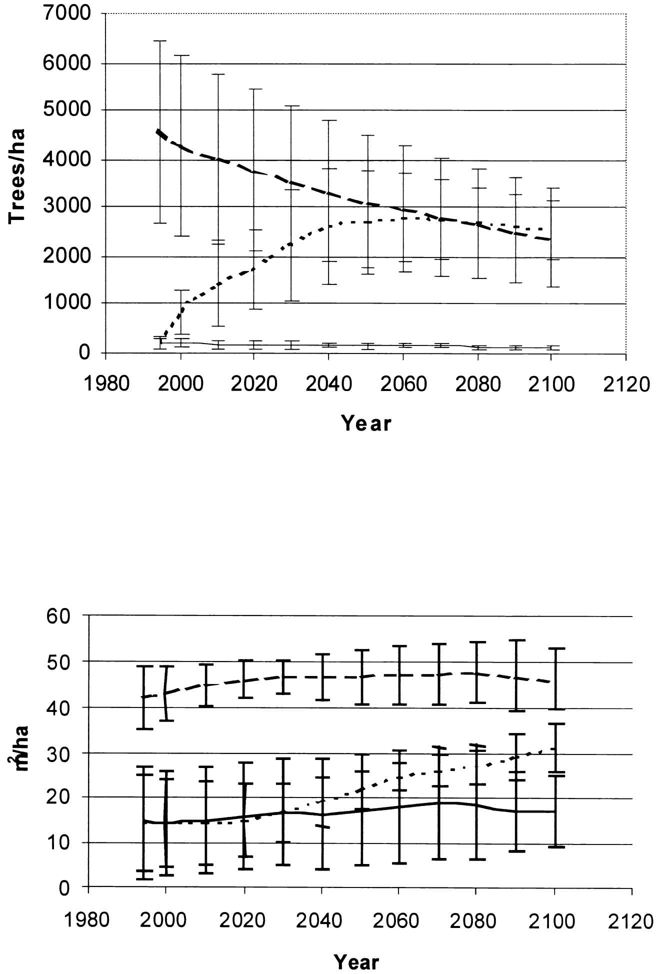

Figure 4. Modeled effects of alternative treatments at the Pearson restoration area, average and standard deviation (bars) of 15 plots per treatment. Tree density is shown in the top graph and basal area in the bottom graph. Upper line in each graph is the control. Lower line is the full restoration treatment (thinning + burning at 4-year intervals). The middle line is the partial restoration treatment (thinning but no burning)

in the restoration thinning to replace dead presettlement trees. Although tree densities in the modeled full restoration area were higher than the reconstructed presettlement densities (about 160 trees/ha vs. 60 trees/ha), approximately 70% of the modeled trees were below 2-cm dbh, representing recurring regeneration that was repeatedly thinned by the fires. Herbaceous production was about 380 kg/ha in 2100, and N allocation patterns were similar to those modeled for the presettlement forest. Forest floor fuel levels stayed low due to the repeated fires (data not shown). These results were relatively insensitive to minor changes in fire regime. Decreasing the fire frequency to a 10-year fire interval, representing the longer extremes of variability in fire intervals in the presettlement fire record (Dieterich 1980), gave nearly identical results in tree density and basal area over the modeled period.

Presettlement-like conditions of forest structure, herbaceous production, N storage, and fuel loading were also observed immediately after treatment in the simulated partial restoration—thinning only—area. In the absence of fire, however, tree density rapidly increased significantly ($p < 0.001$), equaling the control area density within 65 years (Fig. 4). Basal area also increased steadily ($p < 0.001$), increasing 80% above the full restoration treatment basal area as the additional young trees grew. Herbaceous production declined to about 120 kg/ha, and fuel loads increased back to pretreatment levels.

Discussion

Model Performance and Recent Ecological Changes

With appropriate modification of parameters based on southwestern data, FIRESUM accurately modeled changes in overstory and understory structure in an Arizona ponderosa pine forest over the past century. Model validation results were in close agreement with the data from the study area (Table 2) and with the broader changes observed consistently across the Southwest: large increases in tree density and basal area (Cooper 1960; Covington & Moore 1994*a*,*b*; Fulé et al. 1997), reallocation of biomass and nitrogen from herbaceous plants to trees (Arnold 1950; Covington et al. 1997), and accumulation of forest floor biomass (Sackett 1979; Fulé & Covington 1994).

Implications of these types of changes in forest structure have been widely discussed for crownfire hazard (Covington & Moore 1994*b*), tree growth, mortality, and regeneration (White 1985; Biondi 1996), and understory production (Arnold 1950). The modeling results also show that the substantial increases and reallocations in biomass from herbs to trees have strongly impacted nitrogen (N) storage over the past century. Changes in the sources of N have implications for herbivores. Nutrient and biomass resources increased for those feeding directly on pine trees, such as *Dendroctonus* spp. and *Ips* spp. (bark beetles), *Sciuris abertii* (tassel-eared squirrels), and *Coloradia pandora* (pandora moth). However, the herbaceous understory represents 99% of the plant species diversity at the study area (1 tree species, 96 grass, forb, and shrub species). These species, which constitute the habitat resources for the nonarboreal invertebrate and vertebrate herbivores, declined nearly 90% in biomass and N. A drop of this magnitude undoubtedly had impacts across trophic levels and food webs of the forest.

Long-term simulations of presettlement conditions provide an opportunity for assessing the range of natural variability in forest structure. The concept of the range of natural variability (RNV) provides a point of reference against which to evaluate changes in ecosystems and a criterion for measuring the success of ecological restoration treatments and other ecosystem management experiments (Kaufmann et al. 1994; Morgan et al. 1994; Moore et al. 1999). In simplest terms, the range of natural variability refers to the "composition, structure, and dynamics of ecosystems before the influence of European settlers" (Swanson et al. 1994). As a conservative estimate, the plants and animals that evolved together with characteristic climate and disturbance regimes for many thousands of years should be well-adapted to these conditions and presumably less well-adapted to rapid anthropogenic alteration of ecosystems (Kaufmann et al. 1994; Swanson et al. 1994). Without modeling, our understanding of the temporal variability in forest structure is limited. Detailed quantitative evidence, such as dendroecological data, photographs, and records are all most available and most precise relatively close to the present, such as the circa 1870–1880 time of fire exclusion (Mast et al. 1999; Moore et al. 1999), whereas progressively much less detail can be reconstructed for 1780, 1680, and before. To some extent, however, reconstruction of the RNV in forest structure in space can also serve as a proxy for reconstruction in time. The sampling of large landscapes includes a variety of differences in aspect, elevation, soils, microhabitats, lightning intensities, and human occupancy. These factors are likely to represent the majority of ecosystem influences over past times, although past proportions varied depending on climatic factors and human population and cultural changes.

The fluctuations in forest structure under long-term simulation (500 years) had coefficients of variation ranging from 4 (basal area) to 9% (tree density), calculated from the averages for all 90 plots every 50 years. The simulated forest structural changes over 500 years are probably well within the RNV of presettlement forest structure on this 4.7-ha study area. Over larger and more diverse landscapes, more variable conditions would contribute to greater structural diversity. For example, reconstruction of ponderosa pine and gambel oak forest structure at the onset of fire exclusion in 1883 at Camp Navajo, about 16 km west of Flagstaff, had coefficients of variation of 76% in tree density and 85% in basal area over a 558-ha area (Fulé et al. 1997). In this local example, temporal variability at the GPNA site is much less than presettlement spatial variability at Camp Navajo, a logical outcome for a small plot compared to a landscape. As FIRESUM, FIRE-BGC, and other models are refined in the Southwest, a more extensive examination of temporal and spatial variability will be possible.

Although highly resolved mechanistic models like FIRE-BGC hold tremendous potential for the future, FIRESUM and other JABOWA-type models will probably remain useful because of their relatively low data requirements and computational simplicity. For FIRESUM to be applied in broader southwestern settings, needed refinements will include local parameterization of three conifer species included in the original FIRESUM model, *Pseudotsuga menziesii* (Douglas fir), *Abies lasiocarp* (subalpine fir *a*), and *Picea pungens* (Engelmann spruce), and addition of several important new species: *Quercus gambeli* (gambel oak *i*), *Abies concolor* (white fir), and *Populus tremuloides* (aspen). *Pinus edulis* (pinyon pine) and *Juniperus osteosperma*, *J. monosperma*, and *J. deppeana* (junipers) are also major species in southwestern forests at xeric woodland ecotones. Modeling of herbaceous production and forest floor dynamics will also have to be modified for the variety of southwestern environments. Application of the model over spatially explicit landscapes will support investigation of large-scale phenomena such as landscape connectivity and spread of contagious processes. We also continue to develop related software tools to support analysis of model outputs.

Comparing Ecological Restoration Treatments

Ecological restoration is one of many ways in which the negative effects of recent ecological changes could be addressed. For example, a variety of tree thinning regimes could address forest density, herbaceous production goals, or fuelbreak objectives, while various fire prescriptions or mechanical fuel treatments could be designed to reduce forest floor fuels. The distinguishing feature of an ecological restoration approach is close emulation of a reference condition based on the historic, indigenous ecosystem (Society for Ecological Restoration 1993). The design of the GPNA restoration treatments included close attention to the presettlement forest structure and spatial pattern, fire regime, reference fuel and burning characteristics, native plant communities, and historic data from the site (Covington & Moore 1994*a*; Edminster & Olsen 1996; Covington et al. 1997; Fulé et al. 1997; Mast et al. 1999).

Apart from the distinct differences between the treatments observed in the simulation results, other ecological factors beyond the scope of the model may affect the forest. Maintenance of extremely high basal area in the control is probably not possible for the next century; in reality the control plots are most likely to experience high tree mortality either from fire or insect or disease pathogens (Wilson & Tkacz 1996). The highly stressed presettlement trees in these crowded stands are already showing evidence of significant growth declines (Biondi et al. 1994; Biondi 1996) and high mortality rates (Sackett et al. 1996; Mast et al. 1999). Essentially two alternatives are possible: either the control can be successfully protected from allogenic disturbance, as implied in the FIRESUM analysis, or a stand-replacing disturbance event will occur. Neither outcome is desirable from a viewpoint that seeks to restore natural structure and process. Fires or pathogens that kill trees over a small area at one time may not be entirely outside the range of frequent-fire regime conditions (e.g., see Fulé & Covington 1997 for an example in northern Mexico). Even at the small scale of the control area (approximately 1.5 ha), however, the native forest was highly uneven-aged (Mast et al. 1999). Because the experimental site is surrounded by dense forest, wildfire in the control area today would not be contained in a small area. Contemporary landscape-scale wildfires, such as the 6,500 ha Hochderffer fire in 1996, ignited about 14 km north of GPNA, have a far greater impact in terms of soil heating, erosion, plant mortality, and loss of seed sources than presettlement fires.

Simulation of the two restoration treatments showed early positive results, consistent with experimental observations at the study area of increased herbaceous production (Covington et al. 1997), increased tree photosynthesis rates and resin flow, decreased tree moisture stress (Feeney et al. 1998), and increased N transformation and soil respiration rates (Kaye and Hart 1998*a*,*b*). Rapid gains in tree growth, crown development, and herbaceous production were observed in the first two decades following thinning of dense young trees at a nearby Fort Valley study area (Ronco et al. 1985). However, the partial restoration treatment (thinning only) rapidly diverged from the full restoration treatment (thinning and burning) as trees became reestablished. Although tree regeneration can be sporadic in the Southwest (Schubert 1974), tree establishment in the absence of fire has occurred repeatedly since settlement. Even in the absence of regional-scale events such as the 1919 seed year (Savage et al. 1996), open forests with seed trees eventually become denser. For example Fulé et al. (1997) published a repeated photographic scene from Camp Navajo, Arizona, showing seedlings in 1942,that grew into dense sapling and pole stands by 1995. A thin-only treatment could be applied repeatedly to control regeneration, and specific basal area targets could even be maintained through commercial thinning (Edminster & Olsen 1996). From a restoration perspective, such a management regime would be less desirable than full restoration because of the loss of the variety of ecological functions played by fire, as well as the soil and habitat impacts of repeated mechanical entries.

Alternative views toward restoring fire-excluded forests have been characterized as a debate between "process restorationists"—who argue that restoration of key ecological processes, especially fire, will eventually restore natural ecological conditions—and "structural restorationists"—who argue that forest structure and fuels must be restored before reintroducing fire (Stephenson 1996). A restoration approach based on fire alone has been popular in areas with management restrictions on mechanical tree thinning, such as national parks (Parsons et al. 1986). In the Sierra Nevada, Stephenson (1996) suggests that fire-only treatments in sequoia/mixed-conifer forests can restore forest structure nearly as well as thinning and burning. Modeling studies in the Sierra Nevada have reached similar conclusions (van Wagtendonk 1996; Miller & Urban 2000).

In contrast a prescribed fire-only treatment was not tested at GPNA because more than two decades of extensive local experimentation have shown that the ponderosa pine forests in our study region are unlikely to regain natural forest structure from burning alone. Interval burning studies were initiated on the Fort Valley and Long Valley Experimental Forests in 1976 and 1977 to test the ecological effects of reintroducing low-intensity fire at intervals ranging from 1 to 10 years (Harrington & Sackett 1990). Prescribed fires reduced accumulated fuel and mobilized forest floor nutrients (Covington & Sackett 1992) and moderately stimulated tree growth (Peterson et al. 1994) but thinned only a limited number of small trees. Even trees as small as 10-cm dbh were highly resistant to prescribed fire, so the study areas have retained thousands of trees/ha, well above presettlement reference levels. Techniques that increased fire intensity, such as ring firing, led to higher mortality of small trees but were difficult to apply due to the presence of continuous vertical fuels, the need for dry conditions, and the desire to protect old-growth trees from excessive scorch (Harrington & Sackett 1990). The long-term interval burning studies remain in place, but more comprehensive restoration experiments including both structural and process restoration are the focus of current research (Sackett et al. 1996; Fiedler et al. 1996; Covington et al. 1997).

Additionally, a burn-only treatment was not included in the FIRESUM analysis because fire modeling capabilities are limited to surface fires, so simulation results can be misleading if fuels are actually available to support crownfire. The fires we simulated burned in presettlement or restored fuels, usually corresponding to fire behavior fuel model 2 (grass and timber litter, Anderson 1982). Forest structures were open and dominated by large, fire-resistant trees with very few ladder fuels such as shrubs or small trees. Under these circumstances, torching is rare and active crownfire essentially impossible even under the relatively dry fire conditions characteristic of the natural spring and summer fire regime (Table 1). The surface fire module incorporated in FIRESUM, based on the fire behavior model of Rothermel (1972), provided realistic estimates of fire intensity for input to the seedling mortality and fuel reduction modules. In contrast, passive and active crownfire behavior are likely under the same weather conditions in the control fuels, which have high vertical fuel continuity, a low crown base height, and high canopy density. Crownfires result from modeling stand structures like those of the control area with the fire simulator FARSITE (Finney 1998), which incorporates crownfire-modeling capabilities. Modeling a fire-only treatment with FIRESUM in the control fuels would result in an underestimate of fire intensity and an overestimate of the thinning effect of fire, because fire behavior could be manipulated in the model by reducing fuel moisture and increasing wind speed to achieve any desired level of tree thinning. In reality, however, the control fuels can easily support crownfires even under average fire season burning conditions.

A modeling study by Miller and Urban (2000) suggested that relatively severe prescribed fires in the Sierra Nevada could approximate the effect of mechanical thinning over a time span of a few centuries, because intense burns will thin trees, repeated fire will regulate new tree regeneration, and the excess established, fire-resistant, overstory trees will eventually die, reestablishing reference forest conditions. Even a regime of less-intense "natural" fires is predicted to eventually achieve the same goal (Miller & Urban 2000). For many southwestern ponderosa pine forests, however, such an outcome is unlikely for two reasons. First, real fires severe enough to thin 25–35-cm dbh pines are likely to crown in unthinned stands and would be difficult to manage under any circumstances. This outcome cannot be accounted for in modeling, because the fire model in FIRESUM as well as that used by Miller and Urban (2000) have only surface fire capabilities, as discussed above. Second, the long-term restoration approach implies that dense stands will be able to persist for centuries without destruction by wildfire. Recent fire trends in the Southwest (Swetnam & Betancourt 1998), coupled with increasingly strong El Niño–Southern Oscillation (ENSO) events such as the 1996 La Niña episode, during which the largest and most costly crownfires in southwestern history occurred, suggest that long-term maintenance of dense forests is unlikely.

Repeated underburning would help reduce crownfire danger by lowering fuels in dense stands (van Wagtendonk 1996), but fine fuels reaccumulate quickly in ponderosa stands (Davis et al. 1968; Harrington & Sackett 1990). Restoration treatments, which include biomass removal, are more likely to provide long-term crownfire protection. Repeated burning is necessary in any event, but as long as the burning units are appropriately situated with respect to protection of surrounding areas, burning in open forest conditions after thinning will permit much wider prescription windows because the risk of intense fire behavior is low.

In practice southwestern forest restoration will include a mix of practices as treatment areas expand from small experimental sites. On many public and private lands, mechanical thinning offers the combined benefits of rapid restoration, crownfire protection, and the opportunity to offset treatment costs to some extent with the value of small-diameter trees. Pure ecological restoration goals (restoration *sensu stricto*, Aronson et al. 1993) are unlikely to be adopted uniformly; instead, mixtures of several thinning prescriptions and set-asides will lead to diverse landscapes on large-scale projects. On lands such as parks and wilderness areas, where management objectives seek to minimize mechanical activities, thinning may be limited to the minimal level needed to safely reintroduce fire or protect boundaries or developments. Prescribed fire will be central in every restoration approach—underscoring the fact that the term "structural restoration" is a misnomer. As shown by the partial-restoration treatment simulation, a successful restoration approach depends on reinstating and maintaining the full spectrum of natural processes, including demographic and successional changes, and disturbance. Throughout the Southwest, another kind of restoration is increasingly needed: reestablishment of indigenous ecosystems on severely burned landscapes. It is important to test a variety of restoration treatments while we can still intervene to limit the damage.

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