Slope Stability Effects of Fuel Management Strategies—Inferences From Monte Carlo Simulations[1]

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Abstract: A simple Monte Carlo simulation evaluated the effect of several fire management strategies on soil slip erosion and wildfires. The current condition was compared to (1) a very intensive fuelbreak system without prescribed fires, and (2) prescribed fire at four time intervals with (a) current fuelbreaks and (b) intensive fuelbreaks. The intensive fuelbreak system caused a slight increase in soil slip erosion and a reduction of about 12 percent in average annual wildfire area associated with each of the prescribed fire intervals. All of the prescribed fire strategies greatly reduced wildfires, but resulted in substantial increases in soil slip erosion, with the greatest increase, 282 percent, for the 15-year prescribed fire interval.

Chaparral fires have been an important land management problem in southern California for at least the past 50 years. The fires themselves are a threat to life and property, but postfire erosion is often a greater threat. Fire suppression efforts have been the principal response to these threats. In recent years fire suppression has been supplemented by two vegetation management strategies: fuelbreaks to aid directly in suppression, and prescribed fire to mimic the presumed natural mosaic of age classes and fuel loads, thereby inhibiting fire spread. Neither tactic is expected to eliminate the threat of wildfire; rather, both are aimed at reducing the size of wildfires and making the resultant fire and erosion damage smaller, more predictable, and more manageable.

The probability that fuelbreaks, either alone or with prescribed burning, will accomplish their fire management objectives is relatively high. Whether they will also reduce erosion is uncertain. Both tactics nay increase the risk of soil slip erosion on steep slopes that are partially dependent on (roots for their stability. Soil slips are shallow failures of colluvial soil and ravine fill. Soil slip erosion increases on the areas converted to fuelbreaks because most of the deep-rooted 'vegetation has been eliminated. Prescribed fire, by reducing the mean age of vegetation, and hence root biomass, may also increase soil slip erosion. The severity of soil slip erosion has been linked to storm size and to the state of the chaparral with respect to its burn and regrowth cycle at the time of the storm (Rice 1974).

The outcome of any fire management strategy therefore, depends on the prescribed burning interval, the area maintained in fuelbreaks, the chance occurrence of wildfire, and the timing of landslide-producing storms. Since the chance of fire and its eventual size, as well as the timing and severity of storms, interact stochastically, their joint interaction with fire management strategies is unknown with respect to both erosion and burned area.

In the face of sparse data concerning most of the relevant parameters in the fire-chaparral-storm interactions, this paper reports insights into the soil slip erosion consequences of various fire management strategies, as gained through a Monte Carlo simulation. Our simulation included functions defining the growth and decay of chaparral roots following fires. These phenomena are key links between age and amount of vegetation, storm size, fire occurrence, and soil slip erosion rate. Our model was as eclectic as necessary in order to give a quantitative basis to what is principally a qualitative appraisal of the problem. The path we charted can be widened and straightened by others as better data become available.

MODEL PROCESSES

Vegetation

Plant roots can increase the stability of slopes by anchoring a weak soil mass to fractures in bedrock, by crossing zones of weakness to more stable soil, and by strengthening soil with long fibrous hinders. When vegetation is killed, as by fire or herbicides, the root system decays and root reinforcement of the soil decreases until new roots reoccupy the soil.

In chaparral fires, the aboveground biomass is burned, but the root system remains intact. Some chaparral species such as chamise (Adenostoma fasciculatum H. & A.) have a large root curl which sprouts after. fire. There is adequate carbohydrate reserve in this burl to keep the root system alive for several years. In our model we have assumed the root system will live for 2 years following

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fire before it dies back to be in balance with photosynthate production. The net soil reinforcement by roots is the sum of reinforcement by decaying dead roots and by newly expanding roots of the sprouting brush.

Landslide frequency has been shown to increase after vegetation is removed from metastable slopes (Croft and Adams 1950; Kawaguchi and Namba 1956; Bishop and Stevens 1964; Rice and Foggin 1971; Burgy and Papazifiriou[3]). In situ measurements have shown that soil strength increases as root biomass increases (Endo and Tsuruta 1969; O'Loughlin 1972; Ziemer 1981).

Little is known about the rooting habits of chaparral plants. Hellmers and others (1955) hydraulically excavated the root system of 57 plants at six sites in the San Gabriel Mountains of southern California and reported maximum root length, depth, and radial spread. Information on root biomass in chaparral-covered slopes is scant. Miller and Ng (1977) reported that root biomass of 21-year-old chaparral excavated near Echo Valley in San Diego County ranged from 100 to 1400 g/m^3. Near the same location, Kummerow and others (1977) excavated a 70 m^2 plot and found an average root biomass of 626 g/m^3. In the more humid Mediterranean climate of northwestern California, root biomass of 12- to 20-year-old snowbrush (*Ceanothus velutinus* Dougl.) fields averaged 1050 g/m^3 (Ziemer 1981).

Information on changes in root biomass following fire is lacking. There is better understanding of changes in aboveground biomass following fire (Rothermel and Philpot 1973). Miller and Ng (1977) reported an average root:shoot biomass ratio of 0.58 for chamise; Kummerow and others (1977) reported 0.57. When the root:shoot ratio is applied to the aboveground live brush biomass model of Rothermel and Philpot (1973), an approximation of root biomass changes following fire can be made (fig. 1). This approximation compares favorably with observed root biomass densities in excavated 20-year-old chaparral stands. We have assumed the maximum density of roots in old-growth chaparral is 1270 g/m^2.

In our model, the live and dead root biomass at the time of a fire is based on the age of the chaparral (the number of years since the last fire). The total dead root biomass after a fire is made up of roots that were dead before the fire and live roots that were killed by the fire. These dead roots decay exponentially with time (eq. 1). (Note: all numbered equations referred to are given in the Appendix.) Growth of the remaining live root fraction follows a logistic curve with time (eq. 1). The sum of the live and dead root biomass is the total root biomass (BMASS) in a cell.

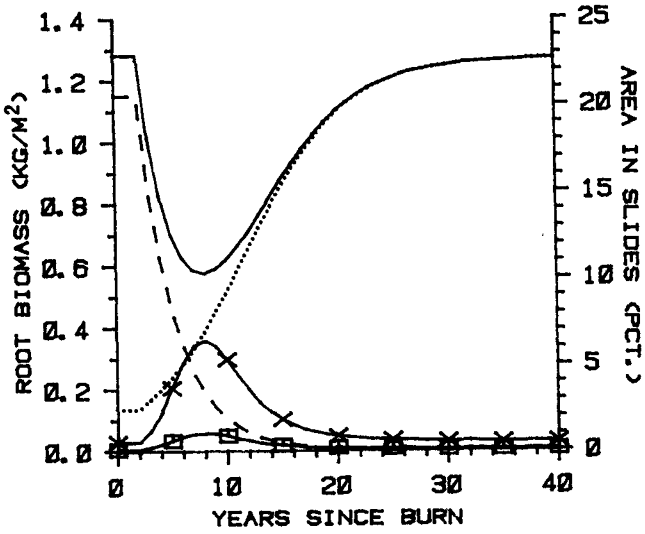

Figure 1--Change in live (······), dead (---), and total (——) root biomass following fire and percent of area in soil slips following fire for a 32-year storm (X——X) and a 9-year storm (⊡——⊡).

Storm Severity

Precipitation data used in our model were obtained from Tanbark Flat on the San Dimas Experimental Forest (Reimann and Hamilton 1959). The record seems to be appropriate since the raingage is located at about the median elevation for chamise and at a middle latitude for the chaparral of southern California. We extended the published data (Reimann and Hamilton 1959) through hydrologic year 1980[4]. From the 47 years of record, we used 176 storms yielding more than 50 mm of precipitation.

It is difficult to define exactly what constitutes a soil slip producing storm. Undoubtedly a large number of site conditions and storm sequences could result in soil slips. We have based our model on an extrapolation of the work of Caine (1980), who analyzed 73 descriptions of storms that caused landslides on slopes unmodified by construction, agriculture, or stream erosion at their bases. His data spanned a great variety of climates and vegetative types and included eight observations from southern California chaparral. He found that the landslide threshold was defined by the function

$$I = 14.82\ D^{-0.39}$$

where I is the mean storm rainfall intensity (mm/hr) and D is the storm duration (hr). It seemed reasonable to us that if, in fact, Caine's

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[3] Burgy, Robert H.; Z. G. Papazifiriou. Effects of vegetation management on slope stability, Hopland Experimental Watershed II. Unpublished paper presented at Water Resources Advisory Council Meeting, January 25, 1971, Los Angeles, California, 10 p.

[4] Precipitation data provided by Eldora M. Negley, engineering technician, San Dimas Experimental Forest, Forest Service, U.S. Department of Agriculture, April 20, 1981.

function is the threshold of slope failure, a good index of storm severity might be the distance of a storm from the line defined by Caine's landslide threshold function in the intensity/duration space. We found that with this new function (eq. 2) our 176 storms were lognormally distributed, with the threshold storm approximately one standard deviation above the mean.

Landslide Erosion

The amount of soil slip erosion following fire is related to storm severity and to the degree of vegetation recovery. Following a 9-year return period storm, about 1 percent of an area burned 6 years earlier was in soil slips (Rice and Foggin 1971; Rice 1974), whereas there were no soil slips in a comparable area not burned for 50 years. Three years later the same area was subjected to a much larger storm with an estimated return period of 32 years. About 6 percent of the previously burned area slipped in this storm while 0.7 percent of the 50-year-old chaparral land slipped. There was very little difference between soil slip rate in the old chaparral and that in an area which burned the previous year. In nearby areas which had been converted to grass following the fire, 7 percent of the land slipped in the 9-year storm and 18 percent in the 32-year storm. The amount of slippage observed following the 32-year storm probably underestimates that which would have occurred had there been no 9-year storm 3 years earlier. Some of the most vulnerable areas failed during the previous storm and were not susceptible to re-sliding only 3 years later. However, soil slip scarps often predispose adjacent areas to additional slippage. For purposes of our model we have assumed soil slip susceptibility of previously failed sites (RESID) to follow an exponential recovery rate of the form, e^{-kt}, where $k = 0.23$ and t is the number of years since the area slipped. The amount of soil slips produced by our model (fig. 1) is a function of storm severity (SEVR), root biomass (BMASS), and the residual effect of previous soil slips (RESID) in the cell (eq. 3).

Fire Occurrence

This portion of our model produces a random sequence of fires which conforms to the current frequency and size distribution of chaparral fires in southern California. The number of fire occurrences in each annual cycle of the model was determined by randomly sampling a distribution having the same mean and standard deviation as annual fire occurrences observed on the Angeles, Cleveland, and San Bernardino National Forests during the fire seasons of 1976 through 1980[5].

The point of origin of each fire was located at random. Our model included no variation due to topography or culture, and fire starts on fuel-breaks were not permitted. Consequently, every chaparral location in the model had an equal probability of being the origin of a fire, in contrast to the real-world situation where areas adjacent to roads and development and especially uphill from them, have a higher probability of burning. Given that there was a fire start (eq. 4), the next step in the simulation was to determine the ultimate size of the fire. The probability of burning each cell in our model was based on the size of the approaching fire (eq. 5, 6) and the elapsed time since the previous fire (eq. 7, fig. 2). The elapsed-time function is based on the effect that the age of an average chamise stand has on the rate of spread of a fire driven by a 32 km/hr wind (Rothermel and Philpot 1973). During the first decade following fire, spread of a new fire is assumed to be dependent upon herbaceous vegetation (Rothermel and Philpot[6]). In our model, the rate of spread of a fire in 5-year-old grass or herbaceous vegetation was approximately equal to the rate of spread in 21-year-old chamise. The coefficient of variation of our fire-spread function was assumed to decrease as the mean rate of spread increased (fig. 2). This assumption derives from the expectation that in the early postfire years the amount of fuel present varies considerably and that, as time passes, variation diminishes as the site approaches its ecological potential.

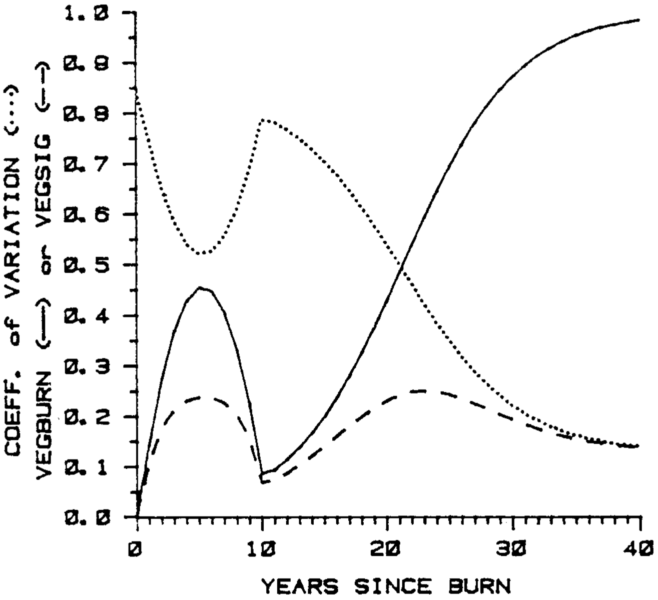

Figure 2--The effect of vegetation age on the mean probability of burning a model cell (VEGBURN) and its standard deviation (VEGSIG) (adapted from Rothermel and Philpot, 1973).

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[5] Fire occurrence data provided by Lynn R. Biddison, director of aviation and fire management, Pacific Southwest Region, Forest Service, U.S. Department of Agriculture, May 1, 1981.

[6] Rothermel, Richard C.; Charles W. Philpot. Mathematical models for predicting chaparral flammability. Unpublished paper, Intermountain Forest and Range Experiment Station, Northern Forest Fire Laboratory, Missoula, Montana, 1972.

Rate of spread was further regulated by a fire-size function (eq. 5), which determined whether the next model cell would be consumed by the fire. Large model elements being approached by small fires have a low probability of burning and, conversely, small elements being approached by large fires have a high probability of burning.

In the computer simulation, once the location of a fire start has been determined, the ignition (eq. 4) and vegetation (eq. 7) functions are consulted to determine if the first cell will be burned. If not, the ignition is ignored and a new random fire start is generated. If the first cell is consumed, the rate-of-spread function (eq. 5), the fire-size function (eq. 6), and the vegetation function (eq. 7) for the adjacent cell are employed to determine the probability that the next cell will also be consumed. This portion of the model is repeated until the fire goes out. If the cell being approached by the fire is a fuelbreak, whether the fire will be contained is determined by reference to an assumed probability that the fire can be stopped at the fuelbreak. That probability (0.8) is an amalgam of the chances of a fuelbreak holding the head, flanks, and rear of a fire; it is based on reported experience of fuelbreak success. Considering the planned simplicity of our model, we do not feel that this treatment of fuelbreaks and other vegetative elements is unwarranted.

MODELS TESTED

Ten models were tested. The one representing current conditions was most important, as only this could be tested against reality. The modeled area represented approximately 80,000 ha divided into "watersheds" of 400 ha or more. (The capabilities of our computer dictated the 400-ha minimum watershed size.) We adjusted our fire functions to increase the probability of burning a 400-ha watershed in order to compensate for the model's inability to explicitly consider smaller fires (eq. 6). This adjustment yielded an annual wildfire area distribution (Table 1A) in agreement with historical records (Philpot 1974). Initially we assumed that the vegetation had an average age of 27.5 years and a standard deviation of 15 years. After several trial runs we found that, <u>in our model</u>, the current fire regime yielded a mean vegetation age of 32 years, and a standard deviation of 27.5 years. For all subsequent runs the model was initialized based on these parameters. The model had a random distribution of cell clusters ranging in size from 1200 to 8000 ha. Ages were randomly assigned which reproduced the appropriate distribution of age classes.

Watersheds were assumed to be rectangular and twice as long as wide. Fuelbreak area was taken to be 100 meters in width times half the perimeter of the watershed (assuming that adjacent watersheds supply the other half-perimeter of fuelbreaks). In our model of current conditions, fuelbreaks surrounded randomly located watersheds having areas typical of fuelbreaked watersheds on the Angeles, Cleveland, and San Bernardino National Forests. The fire- and landslide-producing capabilities of the current condition were contrasted with those of nine other conditions: regularly-spaced fuelbreaks enclosing watersheds of 2000 ha without prescribed fires, and prescribed fire intervals of 15, 20, 25, and 30 years with the current fuelbreak system and the 2000-ha fuelbreak system.

DISCUSSION

The ability of a model to predict actual conditions depends on the assumptions used to construct the model. In a complex natural ecosystem, such as the fire subclimax chaparral, model assumptions are imperfect, at best. As knowledge of the parameters influencing the ecosystem improves, the model can evolve to convey actual conditions more accurately. As a first approximation, however, even an imperfect, simple model, such as ours, may identify the potential result of planned management strategies, given the set of assumptions which land managers now use and the current state of knowledge of ecosystem interactions.

From our 10 model runs, several trends emerged regarding the influence of prescribed fires and fuelbreaks on wildfires and soil slip erosion.

<u>Prescribed Fires</u>

In our model all prescribed fires were assumed to be uniformly successful in reducing fuel volume and none escaped. Prescribed fires dramatically modified the amount of area burned by wildfires each year. Under the current fuelbreak system with no prescribed fires, the long-term average area burned annually by wildfires was estimated to be about 2.2 percent (fig. 3). The average annual wildfire area dropped as prescribed fire intervals

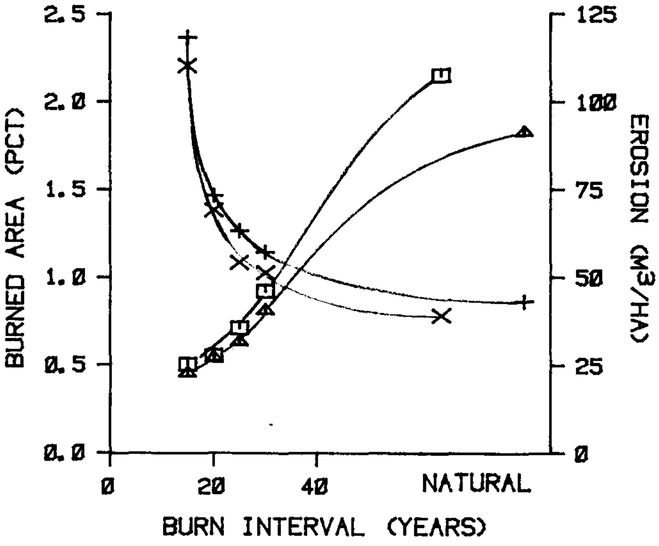

Figure 3--Prescribed fire interval effects on wildfire area with current (❑) and 2000-ha (Δ) fuelbreak densities and annual soil slip erosion rates with current (x) and 2000-ha (+) fuelbreaks.

Table 1--Interval between prescribed burns related to expected frequency of area burned annually by wildfire.

Annual Wildfire Area(pct)	Prescribed Fire Interval (years)				
	(†)	30	25	20	15
	--Expected Frequency (years/century)--				
	A. Current Fuelbreak System				
0.0	16.5	30.7	37.6	43.5	45.7
0.0-1.5	30.9	42.8	43.4	44.0	43.5
1.5-3.0	25.7	19.7	15.9	10.7	9.5
3.0-4.5	15.1	5.5	2.5	1.5	1.0
4.5-6.0	4.8	0.7	0.2	0.2	0.0
6.0-7.5	3.3	0.3	0.1	0.1	0.1
7.5-9.0	1.4	0.0	0.0	0.0	0.0
>9.0	2.3	0.1	0.2	0.1	0.1
	B. 2000-ha Fuelbreak System				
0.0		32.2	36.8	42.6	48.3
0.0-1.5	41.5	52.5	54.4	50.6	47.1
1.5-3.0	23.4	12.4	7.5	6.1	4.0
3.0-4.5	12.6	2.3	1.1	0.5	0.4
4.5-6.0	3.8	0.4	0.1	0.1	0.1
6.0-7.5	2.0	0.2	0.1	0.1	0.1
7.5-9.0	1.1	0.0	0.0	0.0	0.0
>9.0	0.3	0.0	0.0	0.0	0.0

(†) Prescribed burning not done.

were more frequent and the average age of the chaparral decreased. A prescribed fire interval of 30 years produced an expected average annual wildfire burn of 0.9 percent of the area, or about 43 percent of the average annual area burned by wildfires under the current system. With a prescribed fire interval of 15 years, the average annual wildfire area averaged only 0.5 percent of the area, or about 23 percent of the current average annual wildfire area.

Not only was average annual wildfire area reduced by prescribed fire, but the distribution of fire sizes shifted to smaller fires (table 1A). For example, with the current system, there were no wildfires larger than 400 ha in 16.5 years per century, whereas with a 15-year prescribed fire interval there were no wildfires larger than 400 ha in 45.7 years per century. With the current system, in 11.8 years per century, wildfires burned more than 4.5 percent of the area and in 2.3 years per century, more than 9 percent of the area. Using a 15-year prescribed fire interval, wildfires burned more than 4.5 percent of the area only twice in a millennium.

Prescribed fire had an erosional penalty. Under the current fuelbreak system, the long-term soil slip erosion rate was about 39 m^3/ha/year (fig. 3). This rate increased to 110 m^3/ha/year when a 15-year prescribed burn interval was employed, which is a 282 percent increase in average annual soil slip erosion.

Table 2--Interval between prescribed burns related to expected frequency of annual slip soil erosion rate.

Annual Erosion Rate (m^3/ha)	Prescribed Fire Interval (years)				
	(†)	30	25	20	15
	--Expected Frequency (years/century)--				
	A. Current Fuelbreak System				
0	54.0	53.1	54.0	53.4	54.4
0-50	22.8	19.5	18.6	17.1	14.3
50-100	9.8	9.5	7.9	6.6	5.7
100-200	8.3	10.2	10.4	9.3	6.1
200-400	4.0	6.1	7.4	9.6	9.5
400-600	0.5	1.1	1.3	3.2	4.7
600-800	0.3	0.2	0.2	0.6	2.9
800-1000	0.0	0.2	0.1	0.2	1.1
>1000	0.1	0.0	0.0	0.1	1.2
	B. 2000-ha Fuelbreak System				
0	55.0	52.5	53.6	54.2	54.2
0-50	21.9	20.0	18.2	16.9	15.0
50-100	7.4	7.8	7.3	6.3	4.5
100-200	9.6	9.8	9.3	8.4	6.3
200-400	4.9	7.9	8.7	9.4	8.7
400-600	0.8	1.7	2.4	3.6	5.7
600-800	0.2	0.2	0.2	0.6	2.8
800-1000	0.0	0.1	0.2	0.2	1.8
>1000	0.1	0.0	0.0	0.1	1.2

(†) Prescribed burning not done.

The effect of prescribed fire on the distribution of soil slip erosion rates is more complex (table 2). The distribution of storm severities fixes the frequency of soil-slip-producing storms. Thus, the frequency of years with no soil slips is independent of fire management strategy. The important change in the distribution of soil slips is the marked increase in the frequency of larger events as the prescribed fire interval decreases--especially in the strategy utilizing a prescribed fire interval of 15 years. That strategy increased the number of years with soil slip erosion greater than 200 m^3/ha by almost fourfold to almost 1 year in 5. This prescribed fire interval concentrates the age distribution of the chaparral around the trough of minimum root strength (fig. 1), resulting in extreme sensitivity to storms having soil-slip-producing potential.

Fuelbreaks

The addition of a very intensive fuelbreak system, isolating each 2000-ha watershed, had much less effect on the distribution of average annual wildfire area than the various fire management strategies (table 1), but it shifted the fire-size distribution toward the smallest fires. The intensive fuelbreak strategies are also associated with an average reduction of about 12 percent in the long-term average annual wildfire area (fig. 3).

The intensive fuelbreak system had little effect on the distribution of erosion rates associated with the various prescribed fire intervals (table 2). Neither did it seem to have an appreciable effect on the average annual soil slip erosion rate associated with various prescribed burn intervals (fig. 3).

CONCLUSIONS

In common with most land management decisions, the choice of a fire management strategy requires careful evaluation of trade-offs. Generally, those strategies achieving the greatest reduction in area burned by wildfires also carry the greatest soil slip erosion penalties. This is because reduced root strength and slope instability accompany managed reductions in fuel volumes.

A hypothetical 2000-ha fuelbreak system appears to add only modest fire management benefits to the various prescribed fire strategies, and has negligible effect on soil slip erosion. Consequently, it appears that increases in fuelbreak density beyond the present state will have to be justified mainly on bases other than reduction of burned area or erosion.

It should be kept in mind, however, that the increased erosion rate we have estimated applies to areas having a distribution of slopes matching the average for chaparral in southern California. If the various strategies we have discussed were applied exclusively to the 30 percent of the chaparral area having slopes steeper than 50 percent, the erosion rates would be approximately three times greater. On the other hand, if the manager can restrict his activities to the 70 percent of the chaparral growing on slopes less than 50 percent, the erosional consequences that we have estimated for the various fuel management strategies will almost vanish. Planning a fuel management strategy which capitalizes upon this dichotomy, is the challenge facing chaparral land managers.

LITERATURE CITED

Bishop, Daniel M.; Stevens, Mervin E. Landslides on logged areas in southeast Alaska. Juneau, Alaska: Northern Forest Exp. Sta., Forest Serv., U.S. Dep. Agric.: 1964; Res. Paper NOR-1. 18 p.

Caine, Nel. The rainfall intensity-duration control of shallow landslides and debris flows. Geogr. Ann. 62A(1-2):23-27; 1980.

Croft, A. R.; Adams, John A. Landslides and sedimentation in the North Fork of Ogden River, May 1949. Ogden, Utah: Intermountain Forest and Range Exp. Sta., Forest Serv., U.S. Dep. Agric.: 1950; Res. Paper 21. 4 p.

Endo, Taizo; Tsuruta, Takeo. The effect of the trees' roots upon the shear strength of soil. 1968 Annual Report of the Hokkaido Branch, Forest Exp. Sta.; Sapporo, Japan: 1969; 157-182.

Hellmers, H.; Horton, J. S.; Juhren, G.; O'Keefe, J. Root systems of some chaparral plants in southern California. Ecology 36(4):668-678; 1955.

Kawaguchi, Takeo; Namba, Senshi. Landslides and erosion control. Rin-Go-Shiken-Hokoku 84:43-66; 1956.

Kummerow, Jochen; Krause, David; Jow, William. Root systems of chaparral shrubs. 0ecologia (Berl.) 29:163-177; 1977.

Miller, Philip C.; Ng, Edward. Root:shoot biomass ratios in shrubs in southern California and central Chile. Madrono 24:215-223; 1977.

O'Loughlin, Colin Lockhart. The stability of steepland forest soils in the Coast Mountains, southwest British Columbia. Vancouver, British Columbia: Univ. of British Columbia; 1972; Ph.D. thesis. 147 p.

Philpot, Charles W. The changing role of fire on chaparral lands. Proceedings of the symposium on living with the chaparral; 1973 March 30-31; Riverside, California: Sierra Club; California Div. Forestry; U.S. Forest Serv.; 1974; 131-150.

Reimann, Lyle F.; Hamilton, Everett L. Four hundred sixty storms--data from the San Dimas Experimental Forest. Pacific Southwest Forest and Range Exp. Sta., Forest Serv., U.S. Dep. Agric.: 1959; Misc. Paper 37. 101 p.

Rice, R. M. The hydrology of chaparral watersheds. Proceedings of the symposium on living with the chaparral; 1973 March 30-31; Riverside, California: Sierra Club; California Div. Forestry; U.S. Forest Serv.; 1974; 27-34.

Rice, R. M.; Foggin, G. T., III. Effect of high intensity storms on soil slippage on mountainous watersheds in southern California. Water Resources Research 7(6):1485-1496; 1971.

Rothermel, Richard C.; Philpot, Charles W. Predicting changes in chaparral flammability. J. For. 71(10):640-643; 1973.

Ziemer, R. R. Roots and the stability of forested slopes. Proceedings of the erosion and sediment transport in Pacific rim steeplands symposium; 1981 January 25-31; Christchurch, New Zealand: Int. Assoc. Hydrol. Sci. Pub. 132; 1981; 343-361.

APPENDIX

Eq. 1. Root biomass as a function of time since last burn and live and dead root biomass at last burn.

For chaparral:

$$LIVE(t) = 1270/(1 + C * e^{-0.22T})$$

$$DEAD(t) = 0.905 * LIVE(0) * e^{-0.3T} + DEAD(0) * e^{-0.5T}$$

where: $C = (1270 - L_O)/L_O$,
$L_O = 0.095 * LIVE(0) + 12$,
t = years since last burn,
$T = t-2$, for $t \geq 2$, and
$= 0$, for $t < 2$.

For fuelbreaks:

$$LIVE(t) = DEAD(t) = 0$$

Eq. 2 Storm severity as a function of storm duration and rainfall amount.

$$SEVR = \log(0.081 * I^{0.93} * D^{0.36})$$

where: I is the mean rainfall intensity in mm/hr,
D is the storm duration in hours.

Eq. 3 Percent of area in landslides given storm severity, SEVR, residual slide effects, RESID, and summed live and dead root biomass, BMASS.

$$P = \frac{(2.2 * 10^9) * SEVR^{1.13}}{BMASS^{3.6} + (2.4 * 10^9)} - RESID$$

and

$$RESID(new) = RESID(old) + P$$

Eq. 4 Probability of the first cell igniting if cell is of average age.

$$SPREAD = 5.74 * SIZE(CELL)^{-0.577}$$

where: SIZE(CELL) is the area of the cell where the possible ignition occurs.

Eq. 5 Probability of fire spreading to the next cell if it is of average age.

$$SPREAD = \left[\frac{FIRAREA + SIZE(CELL}{SIZE(CELL)}\right]$$

where: FIRAREA is the current size of the fire.

Eq. 6 Fire-spread function adjusted to compensate for area burned in fires smaller than minimum cell size (applies to second cell of a fire only).

$$SPREAD = 0.556 * SPREAD$$

Eq. 7 Fire-spread function adjusted for age of vegetation.

$$SPREAD = 1.284 * SPREAD * N(VEGBURN, VEGSIG)$$

where: N(VEGBURN, VEGSIG) is a normal random variable having the mean and the standard deviation corresponding to the age of the vegetation in figure 3.