Applying RUSLE 2.0 on burned-forest lands: An appraisal

G. González-Bonorino and W.R. Osterkamp

ABSTRACT: Forestlands disturbed by wildfire commonly constitute major and long-lasting sources of sediment that degrade water quality and cause siltation. Postfire restoration of the resistance to erosion of the forest soil is largely controlled by the rate of regrowth of vegetation and may take several years to return to prefire levels, particularly in areas of high-severity burns in semiarid climate. Time-instantaneous prediction techniques such as the Universal Soil Loss Equation (USLE) fail to describe the long-term effect. The latest version of the Revised Universal Soil Loss Equation (RUSLE version 2.0) includes a time-varying option that can model seasonal or pluri-year variations in biomass and other factors; also, it has revised governing equations and an updated database. RUSLE 2.0 claims to be land-use independent and, thus, it should apply to burned-forest lands with proper input for forest vegetation. This paper discusses this matter and concludes there still exist in RUSLE 2.0 built-in routines and parameters inherited from its agricultural application that hinder its use on burned-forest soils. Moreover, many forest lands are characterized by soil textures and slope gradients that fall near, or outside, the limit of the database used for validating USLE/RUSLE, a condition that may counter RUSLE's overall improvement in precision and accuracy.

Keywords: Erosion, forest fire, RUSLE

Forest fires can greatly accelerate soil loss. They deprive the forest soil of protection from rainfall impact and runoff over large areas, and they change soil properties in ways that may increase local runoff. Soil loss from a burned forest typically decreases rapidly with time postfire. Depending on climate and postburn condition, 3 to 10 years commonly will suffice to restore protection to the soil and lower soil losses to prefire values. The semiarid southwestern United States in particular is characterized by slow regrowth of vegetation and monsoonal rainstorms. There the risk of high soil-loss rates may continue for several years (Figure 1; McNabb and Swanson, 1990).

The Universal Soil Loss Equation (USLE) is commonly employed by the Burned Area Emergency Rehabilitation (BAER) teams to assess the risk of postfire erosion. Being a time-instantaneous prediction technique, the USLE fails to describe the long-term effect, and stresses the high risk of soil loss immediately following a wildfire. RUSLE 2.0, on the other hand, includes a time-varying option that may model seasonal or pluri-year variations in soil loss. RUSLE 2.0 does not explicitly account for a burned-forest scenario, and the official RUSLE database does not include forest vegetation. Nonetheless, given the long-standing success of the USLE/RUSLE erosion-prediction technique on varied land conditions, it seems reasonable to apply RUSLE 2.0 to burned-forest lands.

The main purpose of this paper is to explore whether RUSLE 2.0 is structurally capable of dealing with soil loss from burned forestlands, without need for significant changes. First, an overview is given of the RUSLE 2.0 functionality and mode of operation, emphasizing aspects that relate to the erosion of burned-forest soils. Data on fire-induced changes to the soil and the biomass, and on how they evolve with time postfire, are compiled and used in building a preliminary vegetation database on which to test RUSLE. An equation is proposed describing restoration of the canopy cover, a needed input to the RUSLE database, and test computations are made. In concluding, certain RUSLE limitations with respect to burned-forest lands are identified, and suggestions are made for improvement.

RUSLE 2.0. RUSLE 2.0 was developed by the U.S. Department of Agriculture's (USDA) Agricultural Research Service (ARS). The program, user's guide, and tutorial can be freely downloaded from the RUSLE website supported by the USDA-Natural Resources Conservation Service (NRCS): ftp://fargo.nserl.purdue.edu/pub/RUSLE2/.

RUSLE factors. USLE and RUSLE have the same mathematical structure: $A = R*K*L*S*C*P$, in which A is an estimate of the average annual soil loss from a hillslope, and is computed from the product of R, the average annual erosivity factor; K, the soil-erodibility factor; L, the slope-length factor; S, the slope-steepness factor; C, the cover-management factor; and P, the support-practice factor.

The average annual erosivity factor, R, represents the erosive effects of raindrop impact and overland flow. Its computation is based on depth and intensity of discrete rainfall events summed and averaged over many years. R lends USLE/RUSLE its stochastic nature. Problems arise in lands where snowmelt and freeze-thaw processes are common, and in regions characterized by monsoonal storm regimes; in these areas R is computed differently from the original USLE (cf. Renard et al., 1997; Foster et al., 2003).

The soil-erodibility factor, K, represents the combined effect of susceptibility to detachment and transportability of the mineral particles. Mathematically, K is a coefficient relating rainfall erosivity and soil loss, and has been measured systematically on standard unit plots for many soil types (Wischmeier and Smith, 1978; Foster et al., 2003). For loamy soil types K values may be obtained from the soil-erodibility nomograph (Wischmeier and Smith, 1978; Renard et al., 1997). In addition, values for K for soil map units in the United States are contained in the NRCS state soils database (STATSGO) as the kffact parameter.

Unit plots, on average, measure 22 m (72

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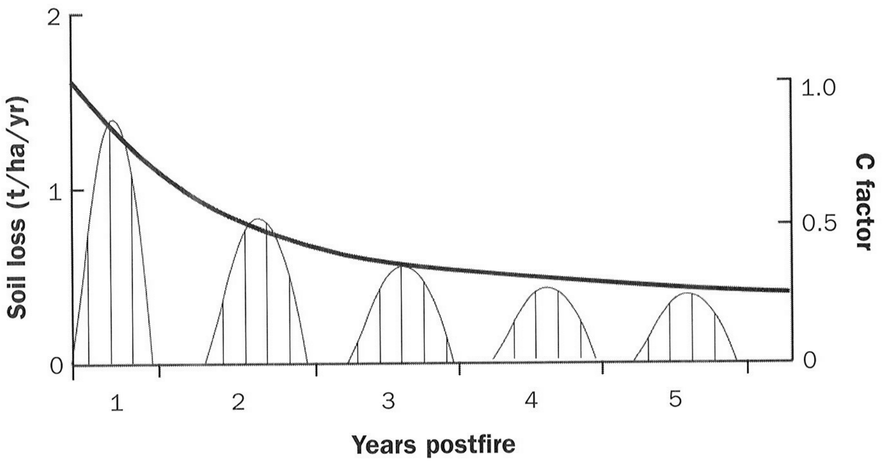
Figure 1
Schematic representation of discrete soil-loss bursts during slow reconstitution of a forest floor, as represented by the diminishing C-factor values; this is typical of rainfall/vegetation regimes in the southwestern United States. Curves redrawn from graphical output from RUSLE 2.0.



ft) in length, lie on hillslopes with 9% gradient, and are maintained in continuous fallow, tilled up-and-down, condition (Foster et al., 2003). RUSLE factors L, S, C, and P are used to scale conditions at the site for which soil loss is to be estimated, to conditions at the unit plot with similar soil and rainfall erosivity. Only the A, R, and K factors have physical dimensions; the remaining factors are dimensionless, ranging from 0, indicating no erosion, to 1 (occasionally larger than 1), indicating that erosive conditions are as bad (or worse) as on the unit plot.

The slope-length factor, L, is indirectly affected by fire. Computation of L includes an effect from the susceptibility of the soil to undergo rill or interrill modes of erosion, and this susceptibility may be altered by consumption of above- and below-ground biomass. RUSLE 2.0 corrects L by considering the effects of soil characteristics, extent of ground cover, and surface gradient (Foster et al., 2003). (This is a case of factor interaction, a common feature of empirical models that poses difficulties in allocating the effects of fire to each factor.) The slope-steepness factor, S, is not affected by burning. Clearly, fire-fighting operations may alter both L and S by opening roads and fire barriers that redirect runoff; this effect, however, can be taken into account in the P factor.

The P factor represents the effects of postfire stabilizing and remediation treatments [see Davis and Holbeck (2001) for a summary of the Department of Interior Burned Area Emergency Stabilization and Rehabilitation (BAER) postfire treatments]. Robichaud et al. (2000) compiled information regarding the effectiveness of postfire treatments. Their study underscores the lack of measured estimates on soil-loss reduction after treatment. Values of the P factor are yet ill-defined for postfire conditions. Following the Cerro Grande Fire in New Mexico (May 2000), the Burned Area Emergency Stabilization and Rehabilitation team proposed values for the P factor resulting from different treatments (BAER, 2000; Miller et al., 2003). The values are apparently based on professional judgement not supported by experiments. Notably, seeding, straw mulching, and contour tree felling were given P values ranging from 0.85 to 0.95, signifying very poor efficiency in reducing soil loss, whereas these three treatments ranked excellent to good in Robichaud et al.'s (2000) evaluation. The conclusion is that further evaluation is required, as is currently being undertaken by other workers. This paper assumes a "No Action" Burned Area Emergency Stabilization and Rehabilitation alternative, that is, no treatment applied. Even without human intervention the burned-forest soil may receive some protection from fallen trees and large branches mimicking contour tree felling. If present, this protection can be accounted for by employing a P-factor value slightly less than 1.

The C factor is calculated in RUSLE 2.0 as the product of seven subfactors: C = Cc*Gc*Sr*Rh*Sb*Sc*Am, in which Cc is the canopy-cover subfactor, Gc is the surface-cover subfactor, Sr is the surface-roughness subfactor, Rh is the ridge-height subfactor, Sb is the soil-biomass subfactor, Sc is the soil-consolidation subfactor, and Am is the antecedent-moisture subfactor (Foster et al., 2003). Values for these subfactors used in this study are explained below.

The ridge-height subfactor, Rh, relates to mechanically-created furrows that may redirect surface flow and enhance erosion from the steep sides of the furrow. Under the assumption of no human intervention, as assumed in this study, Rh = 1. Soil consolidation, Sc, refers to the increase in the resistance to erosion of a soil with time after a mechanical disturbance. RUSLE suggests 7 to 25 years are required for full consolidation of a tilled land. Forest soils can be assumed to be fully consolidated and Sc is given a fixed value of 0.45.

The antecedent-moisture subfactor, Am, accounts for the observation that when rainfall begins, runoff and erosion are enhanced if the soil is moist. Forest fires are more likely in the dry season and, in addition, forest soil heated by fire tends to dry in the uppermost layer, thus values of Am lower than 1 are to be expected shortly after a forest fire. Nonetheless, Am in RUSLE is strongly related to crop type and management in the Northwestern Wheat and Range Region in the northwestern United States (Renard et al., 1997; Foster et al., 2003), and has not been calibrated for burned-forest soils. In this study Am is taken equal to 1.

The surface-roughness subfactor, Sr, represents the effect of microtopographic irregularities on the soil surface that trap moving sediment, pond water—thereby protecting the soil from raindrop impact—increase infiltration, and reduce runoff. On a forest floor, roughness elements may consist of protruding roots, bunch grass, debris dams, steps (cf. Dissmeyer and Foster, 1980), and buried branches. Roughness of the forest floor may diminish with time, as sediment fills the storage sites. RUSLE 2.0 distinguishes long-term roughness, mainly caused by vegetation, and short-term roughness, originating in mechanical disturbances of the soil by agricultural practices (Foster et al., 2003). Roughness, as defined by RUSLE, is mainly long-term in a burned-forest soil. The Sr subfactor is computed by RUSLE on the basis of a value for random roughness, which in this study is set to 0.01 m (0.4 in).

The remaining three subfactors: canopy cover, Cc, ground cover, Gc, and soil biomass, Sb, describe the protection from rainfall and runoff imparted to the soil by live and dead vegetal matter. Canopy cover relates to the area of above-ground leaves, stems, and branches vertically-projected on the ground, modified by the fall height, that is, the distance from where an intercepted raindrop leaves the canopy to the ground. The lesser this distance, the lesser the particle detachment caused by the impact on the mineral soil. In a forest both over- and understory plants may contribute canopy cover. If their cover areas overlap, the understory prevails in RUSLE for it offers greater protection due to the shorter fall height. Ground cover is provided by live and dead biomass on the mineral soil, such as duff and litter, and protects the soil from particle detachment by both raindrop impact and runoff. Where ground and canopy cover overlap, ground cover prevails for it offers maximum protection from raindrop impact, due to fall height null. Thus, RUSLE acknowledges canopy cover only where it overlies bare ground (Dissmeyer and Foster, 1980).

The soil-biomass subfactor, Sb, represents the effects of buried live and dead organic matter, mainly roots in a forest environment. RUSLE requires root biomass values in the upper 10 cm (0.33 ft) of the soil profile, accounting separately for live and dead root mass. RUSLE computes decomposition of organic matter in the ground cover and the soil biomass pools; the rate of decomposition is mainly governed by a user-defined decay constant, and adjusted according to monthly temperature and precipitation. Arbitrarily, RUSLE relocates into the soil biomass pool 40% of the ground cover biomass lost by decomposition.

A forest fire may alter the soil resistance to erosion by consuming, in part or in total, a) the under- and overstory canopies, b) the duff/litter layer, and c) the root network. An indication of the effect of fire on the forest soil system is represented by burn severity. Burn severity is a measure of the effects of a fire on the soil surface and soil profile, and is rated as high, moderate, or low. (Burn severity is distinct from fire intensity, which is measured as radiant heat flow; a high-intensity crown fire may have little effect on the soil.) Generally, with increasing burn severity, first the canopy, then the duff/litter layer, and finally the root network are consumed (DeBano et al., 1998). A major element in attaining high-severity burn conditions is prolonged duration of the fire, as occurs when a thick duff/litter layer smolders even after the flames are extinguished. In low-severity burn situations, canopy leaves, especially coniferous needles, and scarred branches, may not be entirely consumed by the fire but fall to the ground, augmenting the litter layer at the expense of the canopy. Overall, however, forest fires eliminate varying proportions of above- and below-ground biomass.

Hydrophobicity and soil crusting. Intense heating of the mineral soil may cause a transient sealing of soil pores by condensed organic compounds, leading to reduced hydraulic conductivity in the soil profile (DeBano et al., 1998; Letey, 2001). The sealing develops at the heat front, a few centimeters below the mineral soil surface, and typically shows a patchy areal distribution. The effect disappears in days or weeks, especially if the soil is moistened (Huffman et al., 2001). This phenomenon is known as hydrophobicity, and is most likely to develop under high-severity burn conditions, particularly beneath smoldering duff/litter. Researchers applying USLE/RUSLE have dealt with hydrophobicity in different fashions. Following the Cerro Grande Fire, the Burned Area Emergency Stabilization and Rehabilitation team observed that hydrophobicity varied according to the type of vegetation, and proposed corrective coefficients ranging in value from 1.1 for mixed conifer forest with grass understory, to 2.5 for ponderosa pine forest with shrub understory (BAER, 2000). USLE erosion estimates were routinely computed and then multiplied by the appropriate coefficient. Miller et al. (2003) included the effect of hydrophobicity in the K factor, and assumed that a hydrophobic soil would have the lowest hydraulic conductivity accepted in the USLE soil nomograph. As explained above, the K factor is precisely defined from unit plots where fire-induced hydrophobicity was absent, and it seems inadvisable to tamper with it by introducing corrective coefficients. Renard et al. (1997) discussed a similar problem regarding the amount of organic matter in the soil, and concluded that long-term effects should be included in the K factor, and short-term effects are dealt with better by the C factor. Following this lead, hydrophobicity, a transient phenomenon, is herein included in the C factor. Modeling hydrophobicity requires data on its spatial distribution, lifetime, and specific effect on hydraulic conductivity of the soil, which are not available at present (Huffman et al., 2001).

Wildfire-induced heat may alter soil aggregation, increasing the susceptibility to erosion; this effect may last a few months (Andreu et al., 2001). Soil crusting, though not reportedly attributed to soil heating, is a chemical and physical phenomenon that may reduce infiltration significantly, thereby increasing runoff (Sumner and Stewart, 1992). A similar reasoning as was given for hydrophobicity applies to soil aggregaton and soil crusting, whereby these effects should be included in the C factor.

RUSLE data files and operations. RUSLE 2.0 stores variable and parameter values needed to compute the factors in a Microsoft Access database comprising fourteen files, or components (Foster et al., 2003). Of interest here are the Climate, Operation, Management, and Vegetation components. The Climate component gathers precipitation and temperature data. The Operation component contains user-defined operations consisting in discrete events, or effects, that change properties of the soil/vegetation system with respect to soil loss. An operation includes one or more effects that occur in sequence on specified dates. The order and number of effects determine the outcome of an operation. The effects in RUSLE 2.0 are: *No effect, Kill vegetation, Live biomass removed, Remove surface cover, Flatten standing residue, Add other cover, Disturb surface,* and *Begin growth.* Note that as defined in RUSLE 2.0, effects are largely land-use independent. The Management component collects all operations and the dates on which they are scheduled to take place.

For the Vegetation component, RUSLE 2.0 classifies biomass into categories and one of the main functions of the effects is to determine how biomass is transferred among, added to, and substracted from, any of these categories. RUSLE 2.0 classifies biomass as: 1) live, above-ground, canopy-forming vegetation; 2) live above-ground vegetation in contact with the soil; 3) live root mass; 4) dead, above-ground standing residue; 5) dead, above-ground flat residue; 6) dead root mass; and 7) dead buried residue. All dead biomass is subject to decomposition in RUSLE, but standing residue does so at a lower rate.

The RUSLE 2.0 effects are summarily described, with emphasis on application to burned-forest soil. The *Kill* vegetation effect (a) terminates use of a particular Vegetation component, (b) sets canopy cover and fall height to null, (c) transfers live-root mass to the dead-root mass pool, and (d) converts live above-ground biomass to standing residue. The *Live biomass removed* effect removes a user-defined percentage of live above-ground biomass, with the option to transfer a given proportion to the above-ground residue pool. The *Remove surface cover* effect eliminates a user-defined proportion of above-ground residue, whether standing or flat. The *Flatten standing residue* converts standing dead biomass to flat residue, which decomposes at a faster rate than does standing residue (Foster et al., 2003). The *Add other cover* effect models the addition of external residue (e.g., mulch) to the soil, specifying its distribution between the above- and below-ground pools. The *Begin growth* effect activates a specified file from the Vegetation component, and sets to zero the day counter. The *Disturb surface* effect loosens the soil and alters its surface roughness (as through plowing); it can be used to bury flat, above-ground residue, and to resurface residue previously buried, restoring it to the above-ground residue pool. RUSLE 2.0 lacks an effect to model the consumption of root mass.

Loss of soil protection due to fire can be modeled as a "Burn" operation combining effects *Live biomass removed* and *Remove surface cover*. The first effect removes a percentage of above-ground biomass, with the option of adding scarred branches and leaves to an essentially unburned forest floor, whereas the second effect removes part or all of the duff/litter layer. If virtually all standing vegetation is consumed, but the soil is not severely heated and the root network is not affected, as may occur with a crown fire, the *Kill vegetation* effect may provide a useful description. The problem remains, however, that the live-root mass is added to the dead-root mass pool and becomes subject to decomposition, which may not be true in a forest soil. Consumption of the below-ground biomass, as may occur with a high-severity fire, cannot be modeled in RUSLE because it does not include an effect that removes root mass.

Postfire protective actions, such as mulching, can be modeled by the *Add other cover* effect. This effect may also be used to represent growth of the duff/litter layer. Postfire growth of new forest vegetation could be modeled by the *Begin growth* process, which requires preassembled data for "Live root mass", "Canopy cover", and "Fall height", against time, loaded to the RUSLE 2.0 Vegetation component. The next section proposes an equation for modeling canopy regrowth following a fire.

Postfire restoration of soil resistance to erosion.

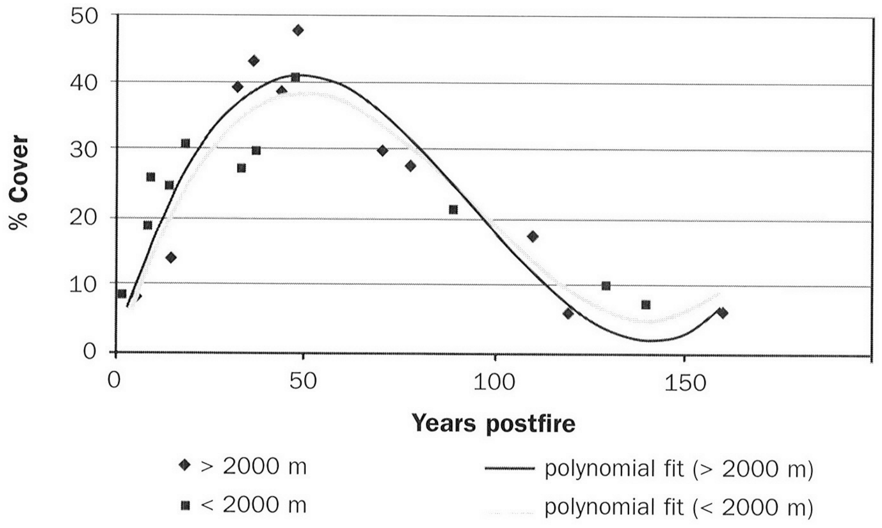
Root network. Values for fine root (the type most effective in binding soil mineral particles) mass in undisturbed forests attain several tons per hectare: 1 to 18 t/ha (0.06 to 0.99 t/ac) were reported for an aspen forest in Alberta, Canada (DesRochers and Lieffers, 2001); average values of 8.2 and 7.8 t/ha (0.48 and 0.43 t/ac), respectively, were estimated for a temperate coniferous forest and a temperate deciduous forest (Jackson et al., 1997). Following most fires, dead root mass declines due to decomposition until vegetation regrowth compensates for the loss and returns it to pre-fire levels. Thus, root mass is characterized as a balance between decomposition and regrowth. In an Amazon forest, root mass after slash burn was 6 to 8 t/ha (0.3 to 0.4 t/ac), declined to 1 t/ha (0.06 t/ac) 6 years later, and then increased to 35 t/ha (1.9 t/ac) 20 years after the fire (Wiesenmuller, 1997). Weltz et al. (1987) published a table of ratios of below-ground to above-ground biomass, and biomass in the upper 0.1 m (0.3 ft) of the soil column to total below-ground biomass, for rangelands. These ratios were used to estimate live- and dead-root mass from total above-ground biomass. This method for estimating total below-ground biomass may not apply well to vegetal systems disturbed by fire because, although the above-ground biomass may be partly eliminated, the below-ground biomass may remain unaffected, and continue to protect the soil integrity.

Duff/litter layer. Litterfall reflects forest biomass production. Lebret et al. (2001) measured litter production in a beech and oak forest in northeastern France, in an oceanic climate of 900 mm mean annual precipitation, and obtained values ranging from 2.1 to 4.7 t/ha/yr (0.11 to 0.26 t/ac/yr), increasing with increasing age of the forest stand. For several months after a fire, biomass production in a burned area decreases, and unburned parts of the duff/litter layer decompose without replacement, leading to reduction of the duff/litter cover. In the Madrean oak woodland studied by Caprio and Zwolinski (1995) in southern Arizona, litter cover decreased from 33% prior to the fire to 16% immediately after the fire, and recovered to 35% within 2 ½ years after the fire.

Decomposition. Dead organic matter on and in the forest floor decomposes at rates that vary mainly with temperature, moisture, and aeration, and secondarily with soil environment and nature of the organic matter. Generally, decomposition is slower in boreal than tropical forests, accounting for the thick duff/litter layers common in boreal forests. A survey of published values for exponential decay constants of undisturbed forests showed that typical values range from about 0.28 yr^{-1} for a boreal forest to 1.5 yr^{-1} for a tropical forest (Gastaldo, 1997). These values for the decay constant are derived from studies in undisturbed forests. A moderating effect of the decay constant value could be expected from changes in moisture and insolation following loss of soil cover due to fire. Nevertheless, in a pine and aspen forest in British Columbia, Prescott and Blevins (2000) noted that litter decay rates in clearcuts and in undisturbed sectors did not differ significantly. The restoration model in this paper assumes that secondary effects are negligible relative to climate control on the decay constant. Wise and Schaefer (1994) observed that leaf litter from the overstory canopy decomposes up to five times more rapidly than does understory leaf litter of the same forest. Thus, using an average decay constant for a forest may overestimate decomposition during inital regrowth, when understory vegetation is dominant.

Regrowth of vegetation. A burned forest may require a century or longer to regenerate a canopy structure similar to that prior to the fire. Long before reaching this state, however, conditions are attained that return the resistance to soil erosion to pre-fire levels. Formal and informal accounts indicate that natural understory regrowth of grasses, forbs, and brush occurs within a few years, whereas trees require many years to develop significant canopy areas (Russell et al., 1998). Moreover, trees compete with understory vegetation and commonly curtail its development (Everett, 1986; Wangler and Minnich, 1996). A general, long-term model taking into account competition among under- and overstory plants, shows the understory vegetation developing at a relatively rapid rate, attaining a maximum level and then declining as trees take over more and more canopy space. This competitive behavior is exemplified by data from Wangler and Minnich (1996) on the postfire evolution of pinyon-

Figure 2
Graph representing long-term postfire evolution of understory vegetation. The decline in understory cover after about 50 years is due to competition from trees. The curves represent pinyon-juniper woodlands in the San Bernardino Mountains, southwestern California. Data sets are for elevations above and below 2000 m (6,600 ft) ASL (data points from Wangler and Minnich, 1996).



juniper woodlands in semiarid southeastern California (Figure 2). During the 30 to 50 years following the burns the land was colonized by sage-scrub, with the addition of chaparral at lower elevations. Pine trees gradually repopulated the burned areas, becoming dense enough to affect shrub growth beneath the shade of their canopies, which accounts for the decline in shrub cover starting at about 50 years postfire (Wangler and Minnich, 1996).

Stickney and Campbell (2000) collected systematic measurements of postfire vegetation regrowth in forests dominated by Douglas-fir, hemlock, and red cedar in the northern Rocky Mountains. The areas have mean precipitation varying from 1,000 to 1,500 mm/yr (40 to 60 in/yr), approximately half of which falls as snow, and lie at elevations ranging from 900 to 1,580 m (2950 to 5180 ft) ASL. Their study included the Sundance (Idaho Panhandle) and Plant Creek (western Montana) areas, which were affected by lightning-ignited wildfires in 1967 and 1972, respectively. In both areas, the above-ground residue and the standing vegetation were consumed by the fires, indicating high-severity burns. Stickney and Campbell (2000) measured percent canopy cover separately for herbs, shrubs, and trees, for postfire time spans of up to 23 years, in six test plots in the Plant Creek area and 13 test plots in the Sundance area. At most sites, trees did not show significant regrowth until 9 years after the burn. Plotted for the first 5 years of regrowth, Stickney and Campbell's (2000) data show a rapid but diminishing rate of growth of the understory canopy cover (Figure 3).

Redevelopment of the vegetation may be slow in semiarid regions. For the San Bernardino Mountains in southern California, Wangler and Minnich (1996) reported 8% cover by shrubs one year after fire (compare regrowth curves in Figures 2 and 3). In more humid environments recovery is faster. An aspen forest in Wyoming, at 2400 m ASL, showed a significant decline in production of grasses and forbs from 1.8 to 1.2 t/ha (0.099 to 0.066 t/ac) during the first year after prescribed fire, but by the second year production had risen to 2.8 t/ha (0.15 t/ac), twice that in the unburned areas (Bartos and Mueggler, 1981). Caprio and Zwolinski (1995) measured herbaceous and litter biomass in unburned and burned parts of a Madrean oak woodland in the Santa Catalina Mountains, southern Arizona. Ten months after the fire biomass was in the order of 3.7 t/ha (0.20 t/ac) in unburned areas, and 1.3 t/ha (0.071 t/ac) in burned areas, and recovered to 2.9 t/ha (0.16 t/ac) 30 months after the fire. The La Mesa (6,700 ha (16,600 ac) burned in 1977), Dome (7,100 ha (17,500 ac) burned in 1996), and Oso (2,700 ha (6,700 ac) burned in 1998) fires burned ponderosa-pine-dominated forests in the Jemez Mountains of northern New Mexico. Veenis (2000) reports on the status of vegetation regrowth in these fire areas by year 2000. The La Mesa fire area showed 5-m (16-ft) high stands of ponderosa pine amidst open grassland; the Dome fire site, aided by reseeding, was covered by grassland and was being colonized by brushy oaks and aspen groves; and the Oso fire area was covered by grasses and small shrubs (Veenis, 2000).

The two well-documented cases of postfire forest regrowth published by Wangler and Minnich (1996) and Stickney and Campbell (2000), as well as the other summarized reports, indicate that grasses, forbs, and shrubs

Figure 3
Graph with polynomial describing postfire regrowth of understory vegetation. Data points are averages for Plant Creek (n=6) and Sundance (n=13) fire areas (Stickney and Campbell, 2000).

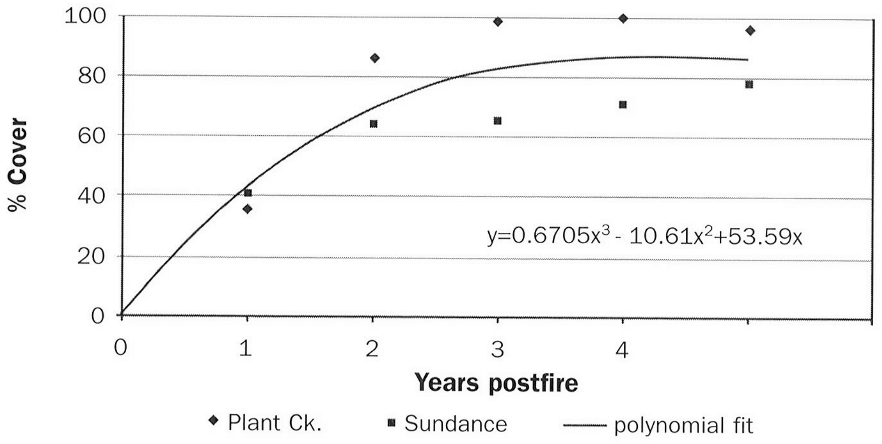


Table 1. Input values for computation of the C factor for different burn conditions. Values in italics relate to the method of Dissmeyer and Foster (1980).

line		Low burn severity	High burn severity
1	Ground cover	70	30
2	Canopy cover	60	10
3	Effective canopy cover	10	0
4	Root netork extent	100	60
5	Fall height (m)	0.5	0.5
6	Random roughness (m)	0.01	0.01
7	*Bare soil*	*20*	*60*
8	*Bare soil w/fine roots*	*100*	*50*
9	*USLE*	*0.01*	*0.10*
10	RUSLE 2.0 (range for 5 yr)	0.029-0.026	0.08-0.079

are the vegetation types first to recolonize the burned land (cf. Everett, 1986), and that, in temperate climate, their canopy may cover 90% of the ground by the time tree canopy becomes significant. Consequently, in certain situations, restricting the time of erosion prediction to a few years postfire allows a significant simplification of the vegetation regrowth model by not having to account for the overstory.

Results and Discussion

Two approaches are possible when using RUSLE to estimate soil loss from burned-forest lands. One is to mount a "Pristine forest" operation to model the canopy, the duff/litter cover, and the extent of the root network previous to the fire. A "Burn" operation, describing the removal of live biomass and surface residue, is then implemented followed by a "Restoration" operation to account for the reconstitution of the soil resistance to erosion. This approach can be implemented in RUSLE 2.0 if no root mass is consumed during the burn. To circumvent this limitation, this paper follows a second approach whereby time-varying, postfire, values are entered for above- and below-ground biomass, ignoring the forest/soil condition prior to the burn.

Initial, postfire values for duff/litter and canopy covers, and for root-network extent, were postulated for hypothetical low- and high-severity burn conditions (Table 1). Then the evolution of these variables over 5 years postfire were calculated on a spreadsheet and pasted into the "Growth chart" in the Vegetation component. Time-varying canopy cover was calculated by means of the equation in Figure 3, and added to initial canopy cover.

The postulated extent of the duff/litter cover immediately after the burn, for each burn-severity condition, was described by adding residue mass to the table for "Add other cover" in the Management data file until the observed percentage cover was obtained in the "Surface after planting" box. This addition was set on the same day of the "Begin growth" effect. Residue mass was added. (RUSLE takes input in mass units and converts to percent cover applying a user-defined coefficient.) Further addition of residue as vegetation regrows was modeled by periodically adding external residue. The decomposition of residue was set k = 1.0 yr^{-1}, for a temperate climate. RUSLE 2.0 computes decomposition of the above- and below-ground residue according to the exponential decay constant and the temperature and precipitation values in the Climate component. The postfire evolution of the root network was approximated as the balance between loss through decomposition and gain from canopy regrowth times the initial root mass value, assuming root mass is linearly related to canopy growth. RUSLE requires a value for the yield at harvest of the vegetation associated to the "Begin growth" effect. For this case, an estimated annual yield was entered for understory vegetation consisting of grasses and shrubs.

RUSLE 2.0 was run and C-factor values were obtained for each hypothetical burn condition (Table1, line 10). Using the same set of conditions, C-factor values also were obtained by the method of Dissmeyer and Foster (1980, their Tables 3 and 5), which requires consideration of the effective canopy cover, that is canopy onlapping bare soil, and the portion of bare soil maintaining a dense population of fine roots (Table 1, line 9).

Given appropriate input, RUSLE 2.0 computes reasonable C-factor values for low- and high-severity burn conditions. The number of biomass-related values that must be input is large and not easily obtainable. Compared to USLE, RUSLE 2.0 is much more data-intensive and more complex to operate. Having access to a forest vegetation database comparable to that available for crops could greatly encourage the routine application of RUSLE 2.0 to burned-forest soils. The Prognosis Model (Wykoff et al., 1982), developed by the U.S. Forest Service, may serve as a starting point for this task. A few changes, such as incorporating an effect permitting removal of dead and live root mass, would help to model high-severity burns, although it has been shown here that this deficiency can be overcome.

Considering the greater effort involved in applying RUSLE 2.0 instead of USLE, it must be evident to the user that there is an advantage in greater accuracy in estimating soil loss from burned-forest lands. In this respect, two characteristics of RUSLE 2.0 need discussion, namely its calibration on the basis of standard tilled soils, and its reliance on yield for essential computations. The validity of extrapolating standard agricultural plot conditions to land that may never have been subject to tillage needs evaluation. Assuming complete consolidation of the soil may not be sufficient to compensate for fundamental differences between the reference and the actual soil. This point has been made by Renard (1986) and Weltz et al. (1998) regarding rangelands, and it applies as well to forested lands. Also, many forested lands occupy slopes significantly steeper and coarser-grained than the standard plots used for validating USLE/RUSLE.

The yield parameter in RUSLE 2.0 strongly influences soil-loss estimates. Its effect is partly explained in the RUSLE 1.06 and 2.0 users' guides but it does not appear in the published equations. Thus, its effect on computations is unclear. In the case of a natural, non-harvested, forest vegetation, annual biomass production might be substituted for yield in RUSLE 2.0, as has been done in the present study. Doubt remains, however, whether this substitution is appropriate. Perhaps the final version of RUSLE 2.0 user's manual should include a definition and complete description of the yield parameter's role in soil-loss computations.

Summary and Conclusions

RUSLE 2.0 is a powerful erosion-prediction technique and its routine application to burned-forest lands is of interest to land-conservation managers. This paper shows that RUSLE 2.0 may need modification before it is applied directly to such situations. A forest vegetation database is needed as part of the RUSLE distribution to facilitate its use. Erosion prediction over time spans of a few years may benefit by restricting attention to the understory vegetation. For time spans of more than 10 years, however, the overstory must be considered. In addition, it is suggested that RUSLE developers explain at greater length the use of the yield parameter and how it affects computations, as a means of showing that RUSLE 2.0 is land-use independent.

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