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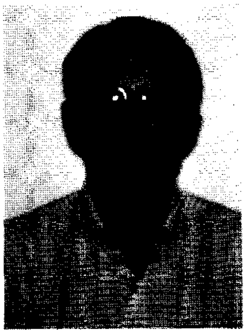

Invited Synthesis Paper:

Influence of abiotic and biotic factors in measuring and modeling soil erosion on rangelands: State of knowledge

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Abstract

The first standardized soil erosion prediction equation used on rangelands was the Universal Soil Loss Equation (USLE). The Revised Universal Soil Loss Equation (RUSLE) was developed to address deficiencies in the USLE by accounting for temporal changes in soil erodibility and plant factors which were not originally considered. Improvements were also made to the rainfall, length, slope, and management practice factors of the original USLE model. The Water Erosion Prediction Project (WEPP) model was developed to estimate soil erosion from single events, long-term soil loss from hillslopes, and sediment yield from small watersheds. Temporal changes in biomass, soil erodibility, and land management practices, and to a limited extent, spatial distribution of soil, vegetation, and land use are addressed in the WEPP model.

To apply new process-based erosion prediction technology, basic research must be conducted to better model the interactions and feedback mechanisms of plant communities and landscape ecology. Thresholds at which accelerated soil erosion results in unstable plant communities must be identified. Research is needed to determine the confidence limits for erosion predictions generated by simulation models so that the probability of meeting specified soil loss values (kg ha^{-1} yr^{-1}) for given management systems can be calculated at specific significance levels. As the technology for modeling soil erosion on rangelands has improved, limitations with the techniques of parameter estimation have been encountered. Improvements in model parameterization techniques and national databases that incorporate vegetation and soil variability are required before existing erosion prediction models can be implemented.

Key Words: Sediment yield, Interrill erosion, Rill erosion, USLE, RUSLE, WEPP

Soil erosion on rangelands was recognized as a serious problem at both local and national scales in the United States in the 1920s; by 1935 soil erosion was considered a national menace on an area covering over one-half of the country (Sampson and Weyle 1918, Bennett and Chapline 1928, Chapline 1929, Weaver and Noll 1935). Soil erosion is an all-inclusive term describing the deflation of the landscape by wind and water. Specific terms like interrill and rill erosion are used to define the detachment of soil particles by raindrop impact and by flowing water, respectively (Table 1). In natural plant communities, the erosion potential of a site is the result of complex interactions between soil, vegetation, topographic position, land use and management, and climate. Soil erosion is a natural process, but the quantity and rate of surface runoff and sediment yield may be altered through land use and management practices.

This paper addresses upland soil erosion and will concentrate on interrill and rill erosion processes. Piping, debris flow, channel scour, side-wall sloughing, head-cutting, and other processes that can significantly affect soil erosion on rangeland watersheds will not be addressed. The influence of abiotic and biotic factors on soil erosion and sediment yield on rangelands is addressed in the first section of this paper. The second section focuses on existing prediction models that were developed for regional or national conditions in the United States. The third section addresses future research needs required to improve our understanding of the erosion process and our ability to model soil erosion.

Abiotic and Biotic Factors that Influence Soil Erosion

There are many abiotic and biotic factors affecting soil erosion and sediment yield on rangelands. Plant and surface cover variables influence runoff and the basic erosion processes of soil detachment by raindrops and concentrated flow, sediment transport, and sediment deposition through the amount and distribution of exposed bare soil, the tortuosity and connectivity of the concentrated flow path, hydraulic roughness, and soil properties of the site (i.e., interrill and rill erodibility). Soil erosion is a function of total standing biomass, biomass by lifeform class (i.e., grass vs. shrub), distance between plants, canopy cover, ground cover or the components of ground cover (rock, litter, plant basal area, cryptogamic crust), bare soil, bulk density, soil texture, soil organic carbon, aggregate stability, the amount of interspace or coppice dune area, number or size of surface depressions, and

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Table 1. Definition of erosion terms.

Term	Definition
Interril erosion	Detachment of soil particles by raindrop impact and their transport by broad sheet flow to concentrated flow areas.
Rill erosion	Detachment and transport of soil particles by concentrated (rill) flow.
Detachment	Dislodging of soil particles from the soil mass by hydrodynamic forces due to raindrop impact and flowing water shear stress.
Transport	Movement of detached soil particles (sediment).
Sediment transport	Ability or power of flowing water to carry sediment.
Deposition	Settlement of detached soil particles.
Sediment yield	Total sediment outflow per unit area measured at a point of reference and for a specific time period (including deposition).
Soil loss	Quantity per unit area and time of soil detached and transport from an area without significant deposition.
Sediment discharge	The rate of movement of a mass of sediment past a point or through a cross section related to velocity of flowing water.

rainfall intensity (Table 2). The complex interaction of these and other abiotic and biotic variables determines how much, when, and where soil erosion will occur.

Vegetation and Management

Vegetation amount, distribution, and lifeform are the primary factors controllable by human activity that influence the spatial and temporal variability of surface runoff and soil erosion on rangelands. Blackburn (1975) found that shrub coppice dunes have significantly different erosion rates than the associated interspace areas. Erosion decreases significantly as plant lifeform changes from short grass to midgrass to tall grass (Thurow et al. 1986, 1988). Grazing management practices impact soil erosion on rangelands through their influence on the type, amount, and distribution of cover (Gifford and Hawkins 1978). By reducing both canopy and ground cover and increasing the number and size of bare soil patches, improperly applied grazing management practices increase the risk that a site will be eroded by both raindrop and concentrated flow path.

In the northern, central, and southern plains grasslands the runoff and erosion potential of a site are closely related to management activity. Prolonged heavy continuous grazing results in significant change in plant community structure in which the more productive tall- and mid-grasses are replaced with less productive short-grasses resulting in increased surface runoff and soil erosion (Rauzi and Fly 1968, Thurow et al. 1988). Other studies have concluded that proper grazing and brush management practices result in infiltration, surface runoff, and soil loss characteristics similar to those of ungrazed landscapes (Blackburn et al. 1982, Blackburn 1983, Weltz and Wood 1986a, 1986b).

Cover

Numerous attempts have been made to establish cover guidelines required for site protection from soil erosion. There are various cover types (i.e., rock, cryptogams, litter, and vegetation), each offering varying degrees of soil protection. The amount and effectiveness of cover necessary for site protection depends upon other factors such as slope, soil type, time of year, and rainfall intensity and duration. Wilcox (1994) found that within the bare interspace areas of pinyon-juniper woodland, most erosion was produced by large convective summer thunderstorms and erosion was slight during the winter, even with high runoff rates from snow melt, due to the absence of raindrop detachment. Generally, the greater the bare soil amount, the greater the erosion rate. Reported levels of cover necessary for site protection range from 20% in Kenya (Moore et al. 1979) to 100% for some Australian conditions (Costin et al. 1960). Most studies indicate that cover of 50 to 75% is probably sufficient (Packer 1951, Orr 1970, Gifford 1984).

Rock cover on the soil surface has a complex relationship with infiltration and soil erosion processes. If rock fragments are embedded in the soil surface, infiltration rates may be decreased; rock fragments resting on the surface (erosion pavement) can prevent macropores from being sealed, increase infiltration, and reduce soil erosion (Poesen et al. 1990, Poesen and Ingelmo-Sanchez 1992). The rock fragments may provide protection from raindrop impact but do not substantially reduce hydraulic shear stress or rilling in semi-arid shrub dominated landscapes. For large rainfall events, the depth and shear stress of flow in the rills exceeds the resistance offered by the rock fragments and substantial rilling does occur between the shrub dominated coppice dunes (Tiscareno-Lopez et al. 1993).

Cryptogam is a term used to define a collection of nonvascular plants: mosses, algae, lichens, liverworts, and cyanobacteria. The impact of cryptogams on infiltration rates and soil erosion is poorly understood and often contradictory. Cryptogamic crusts can reduce infiltration rates and increase soil erosion by blocking flow through macropores or they may enhance porosity and infiltration rates by increasing water-stable aggregates and surface roughness (Loope and Gifford 1972, West 1990, Eldridge 1993). More research is required before the role that cryptogams play in protecting a site from soil erosion will be fully understood.

Erosion Model Development

The most promising, but difficult (and so far elusive), means for predicting soil loss on rangelands is the development of physically based hydrologic-erosion simulation models that are accurate and simple to use. Three soil erosion models of varying complexity are addressed in this paper: the Universal Soil Loss Equation (USLE), the Revised Universal Soil Loss Equation (RUSLE), and the Water Erosion Prediction Project (WEPP) model. Other erosion prediction models that have been used on rangelands will not be discussed in detail but are referenced in Table 3. The basis of mathematical equations used to estimate soil erosion can be traced to the work of Cook (1936), who identified 3 major variables: (1) the susceptibility of soil to erosion, (2) the potential erosivity of rainfall and runoff, and (3) the pro-

Table 2. Abiotic and biotic variables used in statistical models to estimate soil loss on rangelands.

Variable	Location	Technique	Source
Flow depth	Ariz.	Rainfall simulation	Abrahams et al. (1991)
Bulk density, bare ground, dune interspace, soil surface morphology, carbon weight, sand, silt	Nev.	Rainfall simulation	Blackburn (1975)
Annual runoff, watershed area	Entire U.S.	Reservoirs	Dendy and Bolton (1976)
Ratio of annual precipitation to average temperature, watershed slope, soil particles, greater than 1 mm, soil aggregation index	Western U.S.	Watershed	Flaxman (1972)
Rainfall intensity	Ida.	Rainfall simulation	Goff et al. (1994)
Vegetation cover, slope, percent of watershed with different soil types	S. Dak.	Watershed	Hanson et al. (1973)
Total organic cover, bulk density, soil organic matter, total biomass	Tex.	Rainfall simulation	Hester et al. (1997)
Rainfall, runoff, fine organics, bare ground, forb canopy cover	Ida.	Rainfall simulation	Johnson and Gordon (1988)
Litter, organic matter, aggregate stability, grass/forb standing crop	Kenya	Rainfall simulation	Mbakaya and Blackburn (1988)
Soil moisture, litter, total standing biomass, soil depth, rock cover, bulk density, number of depressions	Tex.	Rainfall simulation	McGinty et al. (1979)
Standing grass, litter, forb cover, vegetation cover	Tex.	Rainfall simlation	McCalla et al. (1973)
Drainage area	Ariz.	Watershed	Renard (1980)
Bare ground, litter, vesicular horizon development	Nev.	Rainfall simulation	Roundy et al. (1978)
Erosion pavement cover	Ariz.	Rainfall simulation	Simanton et al. (1985)
Total bunchgrass cover, total aboveground cover	Tex.	Simanton et al. (1985)	Thurow et al. (1986)
Litter, midgrass cover	Tex.	Simanton et al. (1985)	Thurow et al. (1988)
Standing biomass, basal cover, distance to plants, frequency of plots with no rooted plants	Ariz.	WEPP model	Watters et al. (1996)

tection offered by vegetation. Zingg (1940) evaluated the effects of slope length and steepness on soil erosion and is often cited as the developer of the first erosion prediction equation. He proposed the following equation:

$$X = C\,S^{1.4}\,L^{1.6} \qquad (1)$$

where X is total soil loss from a land slope of unit width, C is a constant of variation, S is slope of the land (degrees), and L is horizontal length of land slope (feet). Smith (1941) included the influences of vegetation (C-factor) and supporting farming practices (i.e., P-factor representing the type, depth, frequency, and direction of mechanical disturbance). He recommended that soil loss be calculated as:

$$A = C\,S^{1.4}\,L^{1.6}\,P \qquad (2)$$

where A is average soil loss per unit area.

Smith and Whitt (1948) proposed the "rational" equation to estimate soil erosion:

$$A = C\,S\,L\,K\,P \qquad (3)$$

where A is the average annual soil loss in tons per acre, C is the average annual rotation soil loss from plots (tons acre^{-1}) for a specific site for a specific crop rotation on a three percent, 1080.71 foot long slope, farmed up and down the slope. The other factors were considered nondimensional multipliers to adjust the plot soil loss for differences in slope (S), length (L), soil group (K), and supporting practices (P).

Universal Soil Loss Equation (USLE)

During the 1960s factors for crop rotation, management, and rainfall for areas of the United States east of the 104th meridian were added to the existing "rational" equation for estimating soil loss from upland areas (Smith 1958, Wischmeier 1959, Wischmeier 1960). This resulted in the USLE soil erosion model (Wischmeier and Smith 1965). A database to address rainfall (R-factors) and cover for rangeland areas west of the 104th meridian was later added to expand the applicability of the USLE (Wischmeier and Smith 1978). More than 10,000 plot years of data, representing a variety of soil, crop, and management practices from multiple research locations across the mid-west were used to statistically derive the USLE model. The USLE groups the physical and land management variables that influence soil erosion into 6 factors. Conversion factors for A, R, and K between U.S. customary units and SI units are given by Foster et al. (1981a). The USLE is defined as:

$$A = R\,K\,L\,S\,C\,P \qquad (4)$$

where:

- A is a computed soil loss per unit area (metric tons • hectare^{-1});
- R is a rainfall and runoff factor based on 22 years of climate records {(megajoule • millimeter)/(hectare • hour • year)$^{-1}$};
- K is a soil erodibility factor based on a slope length of 22.1 m and a uniformly sloping 9% surface in continuously clean-tilled fallow {(metric tons • hectare • hour)/(hectare • megajoule • millimeter)$^{-1}$};
- L is a slope length factor determined as the ratio of soil loss from the field slope (unitless). L is 1 when length is 22.1 m;
- S is a slope steepness factor determined as the ratio of soil loss from the field slope to that from a 9% slope under otherwise identical conditions (unitless);
- C is a cover and management factor determined as the ratio of soil loss from an area with specified cover and management practices to that of continuous fallow (unitless);
- P is a support practice factor determined as the ratio of soil loss with conservation practices to straight-row tillage parallel with the slope (unitless).

Limitations of the USLE The USLE is a lumped empirical model that does not separate factors that influence soil erosion, such as plant growth, decomposition, infiltration, runoff, soil detachment, or soil transport. The USLE was designed to estimate sheet and rill erosion from hillslope areas. It was not designed to address soil deposition and channel or gully erosion within watersheds. The applicability, accuracy, and precision of the USLE on rangelands has been debated (Trieste and Gifford 1980, Foster et al. 1981b). In general, the USLE has been found to poorly estimate actual soil erosion on rangelands (Blackburn 1980, Johnson et al. 1980 and 1984, Hart 1984). The potential for improving rangeland estimates of soil erosion with the USLE is limited because of its restrictive structure, reliance on an empirical databases rather than physical processes, and lack of temporal adjustments for factors of soil erodibility (K), cover (C), and management practice (P).

LS-factor The USLE slope length is defined as the distance from the origin of overland flow to the point where runoff reaches a well defined channel or to where slope steepness decreases enough for deposition to occur. Defined concentrated flow paths are not always obvious on rangelands, especially if an area is not eroding. Selection of a typical slope length value involves judgement. The minimum slope length to which the USLE applies is approximately 10 m. The upper limit is even less clearly defined but seldom exceeds 150 m on either forests or rangelands (Dissmeyer and Foster 1980, Foster 1982a).

R-factor Rainfall in the western United States resulting from air-mass thunderstorms is highly spatially variable (Osborn and Renard 1969). For a single thunderstorm on the USDA-ARS Walnut Gulch Experimental Watershed near Tombstone, Ariz., rainfall varied between 25 mm and 50 mm within a distance of 3 km (Renard and Simanton 1975). Because the R-factor is based on a maximum 30-minute rainfall intensity, the variation in the R-factor was magnified from 30 to 100 units over the 3 km distance. Extrapolating the R-factor for more than 1.4 km from a raingauge does lead to serious errors in estimating erosion with the USLE for areas where thunderstorm-derived rainfall controls the erosion process. Renard and Freimund (1994) developed new regression-based methodology to estimate the R-factor for areas in the United States where measured R-factors are unavailable. They utilized precipitation data from 155 weather stations and

Table 3. Comparison of erosion models used on rangelands.

Model	Time step	Erosion	Runoff	Topography	Area	Source
Erosion prediction models						
	Average annual	Sediment yield	NA	NA	Watershed	Renard (1980)
	Average annual	Sediment yield	NA	Uniform	Watershed	Dendy and Bolton (1976)
	Average annual	Sediment yield	NA	NA	Watershed	Flaxman (1972)
USLE	Average annual	Soil loss	NA	Uniform	Hillslope	Wischmeier and Smith (1978)
MUSLE	Event	Sediment yield	NA	Uniform	Watershed	Williams (1975, 1977)
RUSLE	Average annual	Sediment yield	NA	Complex	Hillslope	Renard et al. (1997)
Erosion/Runoff models						
	Event	Soil loss Sediment yield Deposition	Kinematic wave	Complex	Hillslope	Lane et al. (1995)
	Event	Soil loss Sediment yield Deposition	Rainfall excess	Complex	Hillslope	Rose (1994)
AGNPS	Average annual	Sediment yield	Curve number	Complex	Hillslope/Watershed	Young et al. (1987)
CREAMS	Event Average annual	Soil loss Sediment yield Deposition	Curve number	Complex	Hillslope/Watershed	Knisel (1980)
KINEROS	Event	Soil loss Sediment yield Deposition	Kinematic wave Smith-Parlange	Complex	Hillslope/Watershed	Woolhiser et al. (1990)
SPUR	Average annual	Sediment yield	Curve number	Uniform	Hillslope/Watershed	Wight and Skiles (1987)
WEPP	Event Average annual	Soil loss Sediment yield Deposition	Kinematic wave Green and Ampt	Complex	Hillslope/Watershed	Lane and Nearing (1989)

reported that the best predictions ($r^2 = 0.81$) of the R-factor resulted when they separated the database into 2 classes: "winter-type" precipitation distributions and "non-winter-type" distributions. Additional work is still needed to facilitate estimating the R-factor from precipitation data in most areas of the west where the spatial variation in annual and monthly precipitation is greatest.

K-factor Soil erosion is not constant over time. Rainfall simulation studies conducted on 3 semi-arid rangeland soils, cleared of vegetation, indicated that soil erodibility continued to increase throughout a 4-year study (Simanton and Renard 1985). However, studies of mechanical disturbance indicate that soil erodibility decreases with time (Nearing et al. 1988). In 1987, a study on rangeland and cropland was conducted over much of the United States to estimate soil erodibility values for development of new erosion models. Analysis indicated that actual measured interrill and rill soil erodibility values bear little quantitative resemblance to the USLE soil erodibility factor (Laflen et al. 1991b).

C-factor Johnson and Gordon (1988), working on sagebrush dominated rangelands on the USDA-ARS Reynolds Creek Experimental Watershed near Boise, Ida., reported that the combination of K- and C-factors in estimating soil loss from rainfall simulation plots resulted in about 8 times more soil loss from interspace areas than from shrub dominated areas. Actual measured soil losses from interspace areas were 10 times those from sagebrush (*Artemisis* spp.) areas, 7 times more than those from decadent sagebrush areas, and 5 times greater than those from horsebrush (*Tetradymia* spp.) dominated areas. They concluded that there is no mechanism for incorporating information on spatial variability of soil loss into the existing structure of the USLE.

Simanton and Renard (1985), in a similar study on the USDA-ARS Walnut Gulch Experimental Watershed near Tombstone, Ariz., evaluated the interrelationship between the K- and C-factors on 3 rangeland plant communities. The C-factor was calculated for natural plots, assuming that the bare plot C-factor was unity (1.0), and that the calculated K-factor was correct. Because of the method of calculation the C- and K-factors are not independent of each other. An increase in one factor will result in a decrease in the other factor. The C-factor rate of change depended on the type of plant community: the rate of change was similar for a shrub-grass dominated community; the C-factor decreased 2 times faster than the K-factor for a grassland community; and the C-factor changed 6 times faster than the corresponding increase in the K-factor for a shrub-forb dominated community. These results indicate the existing method of calculating the C-factor in the USLE handbook is inappropriate for rangelands and that unique C-factor relationships need to be developed for different rangeland plant communities.

Modifications to the USLE The USLE estimates are based on average long-term annual soil erosion (about 20 year average) and not individual storms. Errors in estimated soil loss from a single rainfall event are large because of the great variation in runoff that can occur when soil moisture and rainfall amount are not considered. To overcome this problem the R-factor was modified to reflect erosion by both raindrop impact and runoff for an individual rainfall event:

$$E = 0.5\, R_s + 3.5\, V_u\, q_p^{0.33} \qquad (5)$$

where E is storm erosivity from rainfall and runoff (MJ mm (ha hour)$^{-1}$), R_s is single storm erosivity (MJ mm (ha hour)$^{-1}$), V_u is storm runoff (mm), and q_p is storm runoff rate (mm hour^{-1}) (Foster et al. 1977a, 1977b).

The slope length factor (L) also varies for different types of storms and should be adjusted if the USLE is to be used to estimate soil loss from a single rainfall event:

$$L = (y/y_u)^n \qquad (6)$$

where y is slope length (m), y_u is length of unit plot (22.1 m), and n is a slope length exponent (usually 0.5) (Foster 1982b). The slope length exponent (n) varies with the potential amount of rill erosion and should be increased by 0.1 when rill erosion is higher than normal and decreased by 0.1 when rill erosion is minimal (Renard and Foster 1983). The normal density of rills per unit area is not clearly defined. Areas with rill density greater than 1 per 1 m width across the slope should be considered higher than normal and areas with rill density of less than 1 per 5 m width across the slope should be considered below normal.

Renard et al. (1974) provided a modification for the USLE to address soil loss from channel erosion. A further modification to predict individual storm/sediment yield and route sediment from small watersheds through large watersheds resulted in the Modified USLE (MUSLE) model (Williams 1975). To accomplish this the rainfall/runoff (R) factor was replaced with a term that combines storm-runoff volume (Q in m^3) and peak-runoff rate (q_p in m^3 sec^{-1}). The MUSLE model is defined as:

$$A = 9.05\, (Q\, q_p)^{0.56}\, K\, L\, S\, C\, P \qquad (7)$$

where sediment yield is given in megagrams for the watershed area rather than kilograms per square meter and K has units of Mg h/ha N. The other terms are as defined in the USLE. Replacing the R-factor increased accuracy in estimating sediment yield from single storm events on watersheds (Williams 1977). Channel erosion, gully erosion and deposition in impoundments must be accounted for separately and either added to or deleted from the estimated sediment yield (Williams 1978).

Revised Universal Soil Loss Equation (RUSLE)

Advancements in erosion science (i.e., techniques to address slopes over 20%, compound slopes, and time varying adjustments for soil erodibilities and cover for cropland) since the release of the USLE in 1978 were incorporated into the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1997). The RUSLE model maintains the simple linear form of the USLE as a product of 6 factors, but subfactors that reflect current knowledge of erosion science are used to calculate each factor (Tables 4 and 5).

A new methodology for estimating R-factors for RUSLE based on more than 1,000 National Weather Station rain gauges has resulted in as much as a 7-fold increase in R-factor estimates. The R-factor has been adjusted to account for soil erosion on partially frozen soils and on soils with ponded water where the erosivity of raindrop impact is reduced.

The K-factor now accounts for seasonally varying erodibilities. Erodibilities are highest in the spring and lowest in mid-autumn following rain compaction. Rock fragments in and on the soil surface are not accounted for in RUSLE. Rock fragments on the soil surface are treated as surface cover in the C-factor, while rock in the soil profile of coarse-texture soils is assumed to reduce permeability and is reflected in the K value (Renard and Ferreira 1993).

The S- and L-factors have been modified for slopes greater than 20% and are a function of the soil's susceptibility to rill ero-

Table 4. Comparison of model inputs for climate, soils, and topography components of the USLE, RUSLE, and WEPP rangelands erosion models.

Model components	USLE	RUSLE	WEPP
Climate	Rainfall Energy	Rainfall Energy	Rainfall Volume
	Rainfall Intensity	Rainfall Intensity	Rainfall Duration
			Ratio of peak rainfall intensity to average rainfall intensity
			Ratio of time to peak rainfall intensity to rainfall duration
			Relative humidity
		Frost free period	Frost free period
		Monthly temperature	Maximum daily air temperature
			Minimum daily air temperature
			Wind velocity
			Wind direction
			Daily solar radiation
Soils	Organic matter	Organic matter	Organic matter
	Sand	Sand	Sand
	Silt	Silt	
	Soil structure	Soil structure	
	Permeability	Permeability	
		Rock fragments	Rock fragments
			Clay
			Cation exchange capacity
			Soil water content
			Saturated hydraulic conductivity
			Interrill erodibility
			Rill erodibility
			Critical sheer
			Depth of horizon
Topography	Length of slope	Length of slope	Length of slope
	Angle of slope	Angle of slope	Angle of slope
			Width of slope
Slope	Uniform	Uniform	Uniform
		Complex	Complex

sion relative to interrill erosion. Soil loss is much more sensitive to slope steepness than to changes in slope length (Renard and Ferreira 1993). The modified S- and L-factors in RUSLE result in generated soil loss values considerably lower than the USLE values, although these new algorithms have yet to be verified with experimental data.

The P-factor has been the least defined of all factors for rangelands. The P-factors for several mechanical renovation techniques have been incorporated into RUSLE and require the user to estimate the random roughness, surface cover, and reduction in runoff as a result of the treatment.

The C-factor is used both within the USLE and the RUSLE models to reflect the effect of management practices (i.e., grazing or burning) on cover conditions and erosion rates. The C-factor can vary from near zero (for a dense grass area with no exposed bare soil) to 1.5 for a freshly disturbed soil surface. The C- and K-factors for rangelands can be simulated as time variant or average annual over the simulation period. When the time variant option is utilized, the RUSLE model computes the C-factor by 15 day increments. Monthly temperature, average frost free period, and a litter decay coefficient are needed when the time variant C-factor is utilized. The time variant C-factor is not recommended for use on rangelands because the added complexity of defining the litter decay coefficient and below-ground biomass over time does not increase the accuracy of the estimated erosion rate. A brief description of the subfactors contributing to the calculation of the C-factor is presented. The C-factor for rangelands is estimated as:

$$C = PLU\ CC\ SC\ SR\ SM \tag{8}$$

where:

- PLU Prior land-use subfactor;
- CC Canopy cover subfactor;
- SC Soil cover subfactor;
- SR Surface roughness subfactor.
- SM Soil moisture subfactor.

Each of these subfactors in turn is expressed by an equation so that a value can be computed for most rangeland situations.

The prior land-use subfactor is based on the time since last disturbance, root biomass and buried organic material in the upper 100 mm of the soil. The canopy cover subfactor is related to the fractional cover of the soil surface provided by above-ground plant biomass and the height that raindrops fall after leaving the plant and impacting the soil surface. The soil surface cover subfactor is related to the fractional cover of the soil surface that is covered by non-eroding material (basal area of plants, rocks and organic litter). The surface roughness factor is based on the random roughness of the soil surface and the root biomass in the upper 100 mm of the soil. The soil moisture factor was included to address unique erosion problems of croplands in the Northwestern Wheat and Range Region of eastern Oregon, Washington, and Idaho and should not be used on rangelands

(SM set to 1.0 for rangelands). A complete description of the subfactor equations is provided by Renard et al. (1997) and Weltz et al. (1987).

The RUSLE model was compared to the USLE for 3 different soil-vegetation assemblages using a large rotating boom rainfall simulator on the USDA-ARS Walnut Gulch Experimental Watershed near Tombstone, Ariz. (Weltz et al. 1987). Three surface conditions were evaluated: natural vegetation; clipped plots where all standing vegetation was removed; and bare plots where all above ground biomass and surface cover was removed. Both dry and wet soil moisture conditions were evaluated twice a year (spring and fall) over a 4-year period. The regression coefficients of predicted versus observed erosion for the different model comparisons were used to evaluate the different models (Table 6) and indicate that the models were similar in predicting soil loss. In each instance, the slope of the line is less than unity, demonstrating that the predicted values of soil loss were substantially less than the measured values.

In a similar comparison Renard and Simanton (1990) evaluated the USLE and RUSLE models at 17 sites in 7 western states using the procedures described above. The differences in the comparisons between the 2 models involve the K-, LS-, and C-factors. They concluded that RUSLE did a better job of estimating soil loss than USLE for naturally vegetated and clipped plots although both models were poorly correlated with actual soil loss. Both RUSLE and USLE gave improved soil loss estimates when the bare soil treatments were included in the analysis with the vegetated and clipped treatments. However, as in the previous study, the slope of the line was less than unity for the RUSLE model, demonstrating that the predicted values of soil loss were substantially less than the measured values.

Benkobi et al. (1993a), working with rainfall simulation from 1 m^2 plots on a sagebrush-grassland area in Idaho, reported that RUSLE significantly underestimated soil erosion and the slope of the line was near zero indicating a very poor relationship between measured and predicted soil loss. They replaced the surface cover subfactor (SC) with a multiple regression equation based on litter and rock cover in an attempt to improve prediction of soil loss. This new equation did not substantially improve the estimate of soil erosion, and both versions of RUSLE significantly underesti-

Table 5. Comparison of model inputs for plant community and management practice components of the USLE, RUSLE, and WEPP rangeland erosion models.

Model components	USLE	RUSLE	WEPP
Plant community[1]	Canopy cover	Canopy cover	Canopy cover coefficient
	Ground cover[2]		
	Plant height	Fall height	Plant height
	Type of roots		
		Bare soil[3]	
		Standing biomass	Standing biomass
		Root biomass	Root biomass
		Random roughness	Random roughness
			Distance between plants
			Canopy diameter
			Litter biomass
		Rock cover	Rock cover
			Cryptogam cover
			Leaf area index coefficient
			Drought tolerance coefficient
			Carbon nitrogen ratio of litter
			Root turnover coefficient
			Minimum temperature for growth
			Maximum temperature for growth
			Potential plant productivity
			Day of peak standing crop
		Litter decay coeffecient	Litter decay coeffecient
Management		Mechanical	Grazing
		Rock fragments	Animal weight
		Hydrologic group	Animal number
		Surface roughness	Change in accessibility
		Surface cover	Digestibility
		Strip cropping	Grazing period
		width of field	Supplemental feed
		slope along ridge	Burning
		Terraces	Biomass removed
		spacing	Accessibility
		bottom slope	Change in net primary productivity
			Herbicide
			Percent kill
			Change in net primary productivity
			Change in accessibility

[1]With the USLE and RUSLE models, only one plant community can be evaluated. With the WEPP model, up to 10 plant communities can be evaluated on each hillslope.
[2]USLE gound cover is usually considered to be litter cover only.
[3]RUSLE ground cover includes litter, rocks, gravel, cryptogams, and basal plant area.

Table 6. Comparison of RUSLE and USLE soil loss predictions to observed soil loss on rangelands.

Surface condition			RUSLE		USLE		
Surface cover[1]	Soil Moisture[2]	No. of plots	Slope	r^2	Slope	r^2	Data source
Bare	Dry and Wet	94	0.77	0.52	0.64	0.53	Weltz et al. (1987)
Bare, clipped and vegetated	Dry and Wet	190	0.77	0.77	0.66	0.70	Weltz et al. (1987)
Clipped and vegetated	Dry and Wet	181	1.06	0.36	0.39	0.08	Renard and Simanton (1990)
Bare, clipped and vegetated	Dry and Wet	181	0.69	0.66	0.91	0.62	Renard and Simanton (1990)
Vegetated	Dry	11	0.11	0.67			Benkobi et al. (1993a)
Vegetated[3]	Dry	11	0.06	0.81			Benkobi et al. (1993a)
Vegetated	Wet	11	0.03	0.14			Benkobi et al. (1993a)
Vegetated[3]	Wet	11	0.03	0.50			Benkobi et al. (1993a)

[1]The bare treatment had all vegetation and ground cover removed, the clipped treatment had all standing vegetation removed, and vegetated is the the natural condition of the site.
[2]Soil moisture refers to the antecedent moisture condition of the soil surface: dry refers to the soil surface at about wilting point conditions, and wet is 24 hrs after first rainfall simulation with the soil surface at about field capacity.
[3]Modified surface cover subfactor for RUSLE model.

mated soil erosion (Table 6). Soil loss was most sensitive to changes in values of the slope steepness and slope length factors (Benkobi et al. 1993b).

Water Erosion Prediction Project (WEPP)

The Water Erosion Prediction Project (WEPP) model is a process-based erosion simulation model that operates on a daily time step (Nearing et al. 1989, Flanagan and Nearing 1995, Flanagan and Livingston 1995). This allows the incorporation of temporal changes in soil erodibility, management practices, above and below-ground biomass, litter biomass, plant height, and canopy and ground cover in the prediction of soil erosion. Linear and nonlinear slope segments, multiple soil series, and multiple plant communities within a hillslope can be represented with the model. The WEPP model can be applied under many rangeland conditions where water erosion occurs, including concentrated flow channels ranging in size from 1–2 meters in width by 1 meter in depth. Stream-bank sidewall sloughing and head cutting in gullies are not addressed in the WEPP model.

Model component 3 The rangeland version of the hillslope model can be divided into 7 conceptual components: climate, topography, soils, hydrology, erosion, management, and plant growth and decomposition (see Tables 4 and 5 for a list of model inputs). The hydrology component utilizes the Green and Ampt equation to calculate infiltration. A semi-analytical solution of the kinematic wave equation is used to route rainfall excess (runoff).

The erosion component of the model uses a steady-state sediment continuity equation to predict the movement of suspended sediment in a rill (Nearing et al. 1990):

$$\frac{\partial G}{\partial x} = D_f + D_i \tag{9}$$

where x represents distance downslope (m), G is sediment load (kg sec^{-1} m^{-1}), D_f is rill erosion rate (kg sec^{-1} m^{-2}), and D_i is interrill erosion rate (kg sec^{-1} m^{-2}). Rill erosion is positive for detachment and negative for deposition. Net soil detachment in rills is calculated for the case when hydraulic shear stress exceeds the critical shear stress of the soil and when sediment load is less than sediment transport capacity. Rill detachment is calculated as:

$$D_f = D_c \left(1 - \frac{G}{T_c}\right) \tag{10}$$

where D_c is detachment capacity by concentrated flow (kg sec^{-1} m^{-2}), and T_c is sediment transport capacity in the rill (kg sec^{-1} m^{-2}). When hydraulic shear stress exceeds critical shear stress for the soil, detachment capacity (D_c), is calculated as:

$$D_c = K_r (T_f - T_c) \tag{11}$$

where K_r (sec m^{-1}) is a rill erodibility parameter, T_f is the flow shear stress acting on the soil particles (Pa), and T_c is the rill detachment threshold parameter (or critical shear stress) of the soil (Pa). Net deposition in the rill is computed when sediment load (G) is greater than sediment transport capacity (T_c). Interrill erosion is a function of baseline interrill erodibility (K_i), rainfall intensity (I), average unit discharge of runoff from interrill areas over time of excess rainfall (q_i) (m^2 sec^{-1}), weighted interrill sediment delivery ratio based on roughness (R_i)(unitless), canopy cover (C), ground cover (G), rill spacing (R_s), rill width (R_w), and is calculated as:

$$D_i = K_i\, I\, q_i\, R_i C\, G \left(\frac{R_s}{R_w}\right) \tag{12}$$

The relationships developed to calculate K_r and K_i and the effect of canopy and ground cover on rangeland soil erosion are discussed in detail by Lane and Nearing (1989), Laflen et al. (1991a, 1991b), Simanton et al. (1991), and Weltz et al. (1997).

Plant growth is simulated as a function of temperature and soil water content. Historical climate data or data stochastically generated by CLIGEN (Nicks and Lane 1989), a weather generator that has been parameterized to yield a weather sequence for nearly 1,000 stations in the United States, can be utilized. The soil-water balance is updated as a function of daily evapotranspiration, precipitation, runoff, and drainage. The growth rate of above-ground biomass for rangeland plant communities is simulated by using a potential-growth curve, which is defined with either a unimodal or a bimodal distribution of plant growth (Alberts et al. 1989, Weltz and Arslan 1990). The potential-growth curve represents the aggregate total production for the plant community. The flexibility of the potential-growth curve permits description of either a warm or cool-season plant community or a combination of the 2 communities. Plant parameters calculated by daily simulation include canopy height and cover, above-ground standing biomass, plant density, leaf area index, litter mass and cover, basal plant cover, rock and cryptogam cover, total ground cover, root biomass, and root distribution with depth.

The model provides 4 management options within the rangeland component: grazing, fire, herbicide application, and complete protection. The user can define the type, severity, and timing of the management activity to be simulated. A hillslope within the model can be subdivided to represent 10 overland flow planes. Each overland flow plane can represent a different soil type, vegetation community, or management activity. Multiple hillslopes can be defined to comprise a watershed. This versatility allows the user to represent a wide range of management practices.

The grazing option allows for as many as 10 rotations of livestock within a year on each overland flow plane and livestock can be rotated from one hillslope to another or within a hillslope. The user can control the weight and number of animals to represent either domestic livestock or wildlife. The effect of grazing is represented by removal of standing biomass with a corresponding reduction in canopy and basal plant cover. Grazing increases transfer of standing dead biomass to litter. Trampling by livestock alters the hydraulic roughness of the soil surface through the interaction of the amount and type of ground cover. It is the interactions of vegetation and surface cover with runoff that determine soil erosion and deposition across the landscape.

The watershed option of the WEPP model will estimate soil loss and deposition from one or more hillslopes within a watershed. With the watershed option, unique climate and rainfall distributions can be assigned to each hillslope to represent spatially and temporally varying rainfall. The model computes sediment delivery from small watersheds and computes sediment transport, deposition and detachment in small channels and impoundments within the watershed. The watershed model can be used to identify zones of soil loss and soil deposition on the hillslope, within channels and gullies, and estimate sedimentation of livestock ponds.

The WEPP watershed model is limited to "field size" areas. For rangelands, this area is estimated to be about 800 ha. There are no explicit limits on size of watershed to which the model can be applied; rather, the user must exercise judgement based on the dominant erosion process. The model does not simulate either baseflow or overbank flooding. The model will have limited use in riparian areas where shallow ground water tables influence runoff, plant growth, and plant community dynamics. The model does not address soil erosion effects from springs or seepage areas.

Model evaluation Soil erodibility for the WEPP model is conceptually different from soil erodibility as defined for the USLE and RUSLE. Soil erodibility within the USLE combines infiltration, runoff, and soil detachment processes of rainfall and flowing water, and is averaged over space and time. Within WEPP soil erodibility is separated to represent soil erosion by rainfall detachment (interrill erodibility) and detachment by flowing water (rill erodibility and critical shear stress). The basic erodibility design used in the WEPP field studies included a bare treatment whereby the soil surface was scalped to a depth of 5 mm and all rock and biomass was removed. Slope steepness ranged from 5 to 15% and slope length was 10.7 m. In addition, 4 small (about 1 m^2) interrill plots were evaluated (Simanton et al. 1987). Interrill erodibilities were determined by measuring erosion rates and dividing these by the square of rainfall intensity. Interrill erodibility (kg sec^{-1} m^{-4}) is highly variable on rangelands and varied by a factor of 174 (Laflen et al. 1991b).

With the experimental design used by Laflen et al. (1991b) and by Simanton and Renard (1985) to develop WEPP erodibilities, only total sediment yield at the end of the plot was measured. There was no direct measurement of the contribution of soil loss from either rill or interrill erosion processes on the natural plots. Only 1 site had noticeable rills before or after the rainfall simulation treatments (Simanton et al. 1991). To determine rill erodibility and critical shear stress, an iterative optimization scheme was used (Page et al. 1989). Rill erodibility and critical shear stress varied by factors of 75 and 190, respectively (Laflen et al. 1991b). To determine rill erosion on rangeland required that soils be tested in standard condition; hence the bare treatment. It was recognized that the bare treatment utilized in these experiments was not equivalent to naturally occurring bare soil because of the disturbance of surface crusts and prior interactions with plants.

The WEPP model has been evaluated for numerous rangeland situations in the western United States. The model has been shown to give good results in predicting runoff volume and peak discharge in the southwest. Evaluation of the hydrologic component of the WEPP model for semi-arid desert shrub and grassland unit source watersheds on the USDA-ARS Walnut Gulch Experimental Watershed showed that the model does a good job in fitting observed and predicted runoff volume and peak discharge (Stone et al. 1992, Tiscareno-Lopez 1994). Data from the USDA WEPP rangeland field experiments (Simanton et al. 1991) were used to test the model's ability to predict sediment yield at 16 locations in the western United States (Kidwell 1994). The WEPP model predicted runoff volume and peak discharge within 2% of the observed data and sediment yield within 16% of observed sediment yield.

Mokhothu (1996) evaluated the WEPP watershed option on the 1.9 ha Kendall sub-basin of the USDA-ARS Walnut Gulch Experimental Watershed. The study assessed the effects of scale on distributed water erosion parameters such as interrill and rill erodibility and predicted sediment yield. To accomplish this, the watershed was split into 1, 2, 3, 6, 8, and 10 contributing hillslopes using geostatistical analysis on data collected on a 20 m grid over the entire watershed. Block kriging was used to split the watershed into cascading planes composed of hillslopes and overland flow elements based on measured vegetation characteristics.

Distribution of vegetation parameters by multiple hillslopes to represent the measured variability did not improve the prediction of runoff and sediment yield at the watershed outlet. However, averaging vegetation estimates for a single plane watershed configuration gave poorer results for predicted runoff and sediment yield than did higher hillslope configurations. The WEPP model produced plausible results for runoff volume, peak discharge, and sediment yield when the number of hillslopes was increased from the 1 to the 8 hillslope watershed configuration. No further significant improvements were realized under the 10 hillslope configuration. The erratic nature of predicting sediment yield was attributed to the fact that the WEPP model does not address the temporal variability of rill and interrill erodibility parameters during continuous simulation as well as the model's limitations in representing the spatial variability that occurs on rangelands.

Weltz et al. (1997) evaluated sediment yield estimates from the WEPP model with data collected from rainfall simulation and soil erosion experiments conducted on 20 rangeland sites from a wide range of soil and vegetation types across the western United

States. One hundred and twenty rainfall events were used to test the WEPP model under 2 scenarios: i) the rangeland option and ii) using adjustments to the interrill erodibility from the cropland option. Total sediment yield values for each event were compared with the WEPP model predicted sediment yield. The results indicate that the current WEPP rangeland option underestimated sediment yield while the cropland option significantly overestimated sediment yield on rangelands.

A limitation with the WEPP model is that the model does not have feedback mechanisms between the simulated climate or the management option and the plant growth model. The plant growth model operates under steady state assumptions. The model does simulate reduced plant growth as a function of drought stress within a year but there are no carry-over effects to future years to simulate death rates or alterations in species abundance as a function of natural or anthropogenic stress. The same potential growth rate is maintained regardless of the previous stress applied to the plant community. This limitation needs to be recognized or unrealistic results may be attained when using the grazing option of the model under continuous simulation, whereby it is very possible to configure a grazing scenario that will result in different potential growth rates or even different plant communities using heavy continuous stocking rates.

The lack of feedback mechanisms between soil loss and plant growth in WEPP and almost all other simulation models that are used to estimate soil loss on rangelands is a further problem. Using the continuous simulation option of the WEPP model, a management scenario could easily be constructed that would result in sustained plant growth even though estimated soil loss was significantly greater than the estimated Natural Resource Conservation Service (NRCS) published soil loss tolerance (T) value for the site. We could find no published work that directly measured soil loss and its effect on plant productivity or the sustainability of rangeland ecosystems that would validate the NRCS concept of T for rangelands. Significant new research needs to be initiated that relates soil and associated nutrient loss to site sustainability before these types of interactions can be incorporated into continuous simulation models like WEPP.

Next Generation Research Needs

Data Collection

Predicting erosion processes has progressed rapidly since the development of computers and introduction of a wide range of soil erosion models have been developed. To utilize these models and to develop new alternative models that better reflect the feedback mechanism between soil erosion and sustainable land use require that new data and new methods for data collection, storage, and retrieval by users be developed that are cost effective and efficient to implement. This requires that several new baseline abiotic and biotic variables be collected: plant height; distance between plants; canopy diameter, canopy cover, and aboveground standing biomass by functional plant group (i.e., annual, sod- or bunchgrass, half-shrub) that are based on relationships to erosion and not forage characteristics; litter biomass and the distribution of litter (under plant canopy or in the interspaces); rock cover; cryptogamic cover by functional group (i.e., lichen or moss); size and connectivity of bare soil patches; percentage of bare soil that is exposed to direct raindrop impact versus bare soil under plant canopy; random roughness; and the abundance and size distribution of roots by class in the surface 10 cm of the soil.

On rangelands most runoff and soil erosion are generated from bare soil interspaces rather than from vegetated coppice dunes or vegetation patches. Hillslopes with identical average exposed bare soil will have significantly different erosion rates depending on the spatial distribution of the bare soil. Bare soil beneath canopy cover is protected from raindrop impact and has a very low probability of being detached and contributing to sediment yield from the hillslope. The distribution and connectivity of the bare soil interspaces and vegetation patches are more important than the absolute amount of bare soil in determining potential runoff and soil erosion rates.

Rill erosion are initiated in the bare soil interspaces when the runoff velocity (hydraulic shear force) exceeds the resistance of the soil. For the rill erosion process to continue downslope, a cascading series of bare interspaces must exist and not be intercepted by vegetated patches for the entrained sediment to contribute to total sediment yield measured at the base of the hillslope. Runoff intercepted by vegetated patches can decrease runoff volume through either direct reduction in runoff volume, higher infiltration rates and capacity of the vegetated patch, or by providing detention storage areas for runoff generated from the bare interspaces. Each of these processes provides a negative feedback to the erosion process by reducing the velocity of the runoff. The reduced velocity results in deposition of entrained sediment because the transport capacity of the runoff has been exceeded. Erosion is further constrained because the reduced velocity inhibits the runoff water's ability to detach additional soil particles downslope.

Traditionally vegetation properties have been estimated using located line-intercept methods, belt-transect, or point-intercept methods, or by sampling quadrats. These methods involve measuring vegetation properties along randomly determined strips, lines, belts, or quadrats across the landscape. Soil erosion is a 3-dimensional process and therefore spatially distributed data collection techniques, at a minimum in 2 dimensions (across the hillslope and down the hillslope), need to be developed if we expect to make significant improvement in estimating soil erosion at the hillslope, watershed, or landscape scale.

Rainfall Simulators as a Tool to Measure Soil Erosion

Rainfall simulators are probably the most common tool used to evaluate the interaction between management practices and abiotic and biotic factors to measure soil erosion on rangelands. However, current rainfall simulators have several limitations and disadvantages: the expense involved in their construction and operation; cost and logistics of supplying water to remote locations; most simulators do not produce drop-size distributions that are representative of natural storms; most simulators can not replicate the temporal variability of rainfall intensity within a storm; steep slopes (> 15%) may not be able to be sampled by trailer or truck mounted simulators; ecosystems with plants greater than 3 m typically can not be sampled due to limitations in the height at which the simulators can be safely operated; and, areas treated are small (1 m^2 to 40 m^2) and may not be representative of the spatial gradient of soil and vegetation associations down a hillslope or represent all soil erosion processes.

Small plot (≤ 1 m^2) rainfall simulators only address interrill erosion processes and do not address soil detachment by concentrated flow, sediment transport, or deposition processes. Large plot (> 30 m^2) rainfall simulators have been used to address both rill and interrill erosion with limited success (Simanton et al. 1985, Simanton et al. 1991, Simanton and Emmerich 1994, Goff et al. 1992, Hart et al. 1985, Abrahams et al. 1991). Large plots integrate the coppice dune and interspace areas found on rangeland and provide a mean erosion response for the hillslope. However, the applied rainfall energy is less than the expected energy from convective thunderstorms which results in less rilling than expected from natural rainfall events. Furthermore, with current technology and experimental designs, there is no way to identify or validate the rate of soil loss from different contributing areas (shrub coppice dune vs. interspace) or to determine which erosion process (rill vs interrill) generates the soil loss.

Despite the limitations of existing rainfall simulators for reproducing natural rainfall events, their advantages for performing artificial, but controlled and replicated experiments in a cost-effective manner over a short time period necessitates their use in obtaining many hydrologic and soil erosion parameter values. When rainfall simulators are used in conjunction with long-term monitored plots and watersheds using natural rainfall events, the resulting information is helpful in understanding the interaction between abiotic and biotic relations and soil loss on rangelands. New modular, programable, variable intensity rainfall simulators that can reproduce the natural variability in rainfall energy and intensity (25 to 200 mm/hr), function on slopes with gradients > 40%, and slope lengths > 20 m are required if we expect to fully understand and predict hydrologic and erosion processes at hillslope and watershed scales on rangelands.

Databases

To be widely applied, erosion prediction technology must be usable by technicians at the field level. To meet this objective, the technology must encompass an integrated system of tools on 3 levels: database generation, user interface, and simulation models. National relational databases that contain climate, soils, topography, land-use, management-practice, and vegetation data are required to implement the new generation of erosion-simulation models. These natural resource databases will allow uniform application of erosion technology by all user groups at the local, county, state, and national levels. Development of a national rangeland database will avoid duplication of effort and time in collecting and maintaining separate databases.

One approach which should be investigated is the use of expert or knowledge-based systems to generate the required model parameters. Plant-growth and litter-decay coefficients are only available for a few plant communities. Knowledge based systems that can communicate with the user and translate their knowledge into model parameters are required before complex erosion models can be implemented uniformly across the United States or in other countries. The process of building national rangeland plant-growth, soils, and climate databases must include research objectives that incorporate spatial and temporal variability and mechanisms to address scaling parameters from plots to hillslopes to entire watersheds. In addition, funding and resources need to be assigned to implementing training and technology transfer to successfully deliver this new generation of simulation models.

Landscape Surface Description

Environmental changes in the West are exemplified by vegetation changes from grasslands to shrublands (Branson 1985). This conversion has resulted in substantially increased erosion rates and major impacts on landform stability and geomorphic processes (Parsons et al. 1996). Most of the current methods of estimating soil loss and surface runoff assume uniform distributions of vegetation and surface cover across the landscape. Techniques to describe the distributions of vegetation and the rates of change in both spatial and temporal scales of plant species, plant canopy, and surface cover are required before significant improvements can be developed and validated in the modeling of ecosystem dynamics at either the field or watershed scale to predict surface runoff and soil erosion.

The role of other properties of surface soil crusts (chemical or physical) needs to be better defined in all erosion models. Methods are needed for predicting which soils crust and the degree to which the sealing affects infiltration rates and interrill and rill erodibilities for different soils. Temporal changes in crusts and their effect on infiltration and soil erosion after drying and cracking, freezing and thawing cycles, and emergence and establishment of seedlings must also be addressed for future soil erosion modeling efforts.

Soil and Plant Parameters

Technologies for modeling runoff and soil loss have greatly improved, but improvements in model accuracy are often lost in the techniques used to estimate model parameters (e.g., infiltration, interrill and rill detachment parameters, and their temporal and spatial variations). Improvements in model parameter estimation techniques and our understanding of the interactions between vegetation, soil, and grazing practice induced temporal and spatial variability are required before the full potential of our hydrologic and erosion modeling capabilities are achieved. For instance, no model currently addresses the enrichment of surface rock cover (formation of erosion pavements) as a function of soil erosion processes. Fundamental research is needed to develop field techniques to describe and predict the effect that rock fragments have on rangeland infiltration rates, rate of soil loss, and rate of erosion pavement formation.

None of the existing soil erosion models represent contributions of individual species to canopy or litter cover or separate the influence of species or functional plant groups on infiltration, runoff, and erosion rates within a plant community. Most of these models can be configured to represent the differences between plant communities, but not the contribution of individual plant species within a community. Research needs to be initiated to incorporate species composition, species replacement, and feedback mechanisms that result in changes in soil and hydrologic properties: soil texture, organic matter, root distribution, macroporosity, bulk density, aggregate stability, and interrill and rill erodibility. If future hydrology and erosion models are going to predict the effect of land management practices on erosion, they need to address the physical processes and mechanisms that drive the soil erosion processes.

Statistical Analysis

Natural processes are inherently variable. The deterministic models reviewed here do not provide information on the reliability of predicted output. Information is needed to determine the confidence limits for erosion predictions generated by continuous simulation erosion models. Research needs to be done whereby the change in the selected input parameter could be related to a change in the predicted output variable. New research should be undertaken concerning the construction of confidence intervals on predicted sediment yield for all types of erosion models. This would allow the probability of meeting specified soil-loss tolerance levels for a given management system to be calculated at a specific significance level.

Soil Erodibility

The ability to conceptualize and develop erosion models has exceeded the ability to design and quantify the component processes of interrill and rill erosion with traditional rainfall simulation field experiments. With 2 unknowns (soil detachment from interrill and rill erosion processes) and only 1 known value (total sediment yield), there is no direct way to validate the erosion process and determine if erosion models are correctly proportioning the sediment yield measured on rangelands from natural plots. Field experiments need to be designed to directly allow for internal validation of soil detachment from interrill and rill erosion process simultaneously to fully validate process based erosion models. Limitations with current data collection methods prevent full evaluation of any erosion model which addresses both rill and interrill processes to determine if the under-prediction or over-prediction of sediment yield on rangelands is the result of representing the erosion process with inappropriate functional equations or if the limitation is in having an adequate sample size to address the variability in soil erodibility of native rangelands. The current form of the interrill erodibility equations does not capture the inherent differences in soil erodibilities that result from chemical interactions (e.g., dispersability of the soil as a function of sodium content). New equations and/or adjustment factors need to be explored to account for chemical as well as physical factors that affect erodibility of rangeland soils. Fundamental research is needed to determine under what rainfall intensities, storm duration, slope length, and slope steepness conditions rilling of rangeland soils will occur.

The concept of the unit fallow bare plot from repeated plowing as used in cropland to define baseline soil erodibility does not apply to rangelands. Interrill soil erodibilities on a single phase of a Pierre soil series near Cottonwood, S. Dak. and a Woodward series near Woodward, Okla. under different historic land uses (cropland and grazed rangelands) were compared (Weltz et al. 1997). The cropland baseline soil erodibility was calculated from fallow plots in the soil's most erosive state (i.e., immediately following plowing) (Laflen et al. 1991b). The severity of this treatment removed any residual influence of previous soil consolidation, land use, and vegetation. The baring of the soil surface under different rangeland treatments resulted in variable disturbance for similar phases of a soil series due to the variation in vegetation (both type and amount) and rock content of the soil. This treatment causes non-reproducible experimental results for a given phase of a soil series and does not necessarily produce the most erosive state of the soil series. The residual root biomass and organic matter left in the soil after baring rangeland plots greatly influences the baseline soil interrill erodibility. However, there is currently no way to separate the historic and current vegetation influence, land use, and management effects from the inherent soil interrill erodibility.

Soil erodibilities measured during rainfall simulation experiments conducted at various rangeland sites varied yearly and depended on vegetation and soil type (Simanton and Emmerich 1994). Time related changes in erosion rates on rangelands need to be evaluated over a multi-year period using multi-plot studies. Biotic factors, both flora and fauna, significantly influence the variability of soil interrill erodibility and need to be considered before the interactions between soil interrill erodibility and soil erosion on rangelands can adequately be defined. Until techniques are developed to define the inherent soil interrill erodibility independent of vegetation and land use influences, the ability to significantly improve soil erosion estimates on rangelands will not be achieved.

Summary

Development of improved erosion technology will require the development of new methods to represent the spatial and temporal variability of landscape surfaces. Furthermore, the development of expert systems is required to provide default plant-growth and soil erodibility coefficients to effectively use and implement continuous-simulation models like WEPP. New research techniques to quantify rill initiation and propagation are required before significant improvements in estimating soil erosion on rangelands can be achieved and incorporated into existing and future erosion models. To apply new process-based erosion technology, basic research is needed for modeling the interactions and feedback mechanisms of plant communities and landscape ecology to identify when accelerated soil erosion will result in unstable plant community dynamics. With the new generation of erosion simulation models, the statistical probability that a specific land-use practice will exceed a specific soil-loss tolerance value can start to be addressed.

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