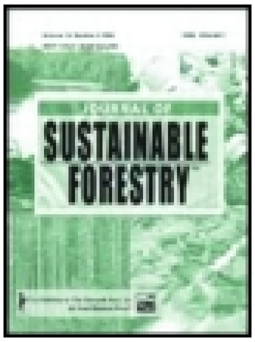

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Chapter 4

Predicting Post-Fire Erosion and Sedimentation Risk on a Landscape Scale: A Case Study from Colorado

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SUMMARY. Historic fire suppression efforts have increased the likelihood of large wildfires in much of the western U.S. Post-fire soil erosion and sedimentation risks are important concerns to resource managers. In this paper we develop and apply procedures to predict post-fire erosion and sedimentation risks on a pixel-, catchment-, and landscape-scale in central and western Colorado.

Our model for predicting post-fire surface erosion risk is conceptually similar to the Revised Universal Soil Loss Equation (RUSLE). One key addition is the incorporation of a hydrophobicity risk index (HYRISK) based on vegetation type, predicted fire severity, and soil texture. Post-fire surface erosion risk was assessed for each 90-m pixel by combining HYRISK, slope, soil erodibility, and a factor representing the likely increase in soil wetness due to removal of the vegetation. Sedimentation risk was a simple function of stream gradient. Composite surface erosion and sedimentation risk indices were calculated and compared across the 72 catchments in the study area.

When evaluated on a catchment scale, two-thirds of the catchments had relatively little post-fire erosion risk. Steeper catchments with higher fuel loadings typically had the highest post-fire surface erosion risk. These were generally located along the major north-south mountain chains and, to a lesser extent, in west-central Colorado. Sedimentation risks were usually highest in the eastern part of the study area where a higher proportion of streams had lower gradients. While data to validate the predicted erosion and sedimentation risks are lacking, the results appear reasonable and are consistent with our limited field observations. The models and analytic procedures can be readily adapted to other locations and should provide useful tools for planning and management at both the catchment and landscape scale.

KEYWORDS. Wildfire, soil erosion, sedimentation, geographic information system, risk assessment

INTRODUCTION

Numerous plot and watershed-scale studies have documented the increase in runoff and erosion following wildfires (Tiedemann et al. 1979). The fire-flood-erosion cycle has been most thoroughly documented in chaparral environments (e.g., Rice 1974; Laird and Harvey 1986; Wells 1987). The risk to life and property are particularly apparent in places such as Southern California (e.g., McPhee 1989; Forrest and Harding 1994) and the San Francisco

Bay Area (Booker et al. 1993), where rapid development has encroached on ecosystems with short fire return intervals.

Large increases in erosion rates have also been observed after wildfire in forested environments (e.g., Helvey 1980; Morris and Moses 1987; Amaranthus 1989; Scott and Van Wyk 1990). The magnitude of the post-fire increase in erosion appears to be highly correlated with fire intensity (Robichaud and Waldrop 1994; Scott 1993), where fire intensity is the relative amount of heat flux (Covington and Moore 1994; Whelan 1995). Post-fire increases in soil erosion rates have also been documented in grasslands (e.g., Cheruiyot et al. 1986; Emmerich and Cox 1994).

The observed increases in runoff and erosion following wildfires or high-intensity prescribed burns have been attributed to several processes. Removal of the vegetative canopy reduces interception and evapotranspiration losses, which then leads to increases in net precipitation, higher antecedent moisture conditions, and increased annual water yields (e.g., Tiedemann et al. 1979; O'Loughlin et al. 1982; Megahan 1983). In high-intensity burns the removal of the protective vegetation and litter increases rainsplash and surface sealing (e.g., DeBano et al. 1979; Beschta 1990; Onda et al. 1996). These changes may be exacerbated by a concomitant reduction in soil organic matter and resulting destruction of soil aggregates (Tiedemann et al. 1979; DeBano 1989; Prosser 1990). In the southwestern US it has been shown that the downslope transport of sediment by dry ravel greatly increases following wildfire (Krammes 1965; Florsheim et al. 1991; Wohlgemuth et al. 1996), and this has also been observed in the Oregon Coast Range (Bennett 1982, cited in McNabb and Swanson 1990).

In some vegetation types fire volatilizes the secondary compounds in the litter and soil organic matter, and the condensation of these organic substances on the underlying cooler soil creates a hydrophobic layer 1-10 cm below the surface (e.g., DeBano et al. 1970; Savage 1974; Wells et al. 1987; Scott and Van Wyk 1990). The strength of this hydrophobic layer increases with fire severity (DeBano and Krammes 1966) and is generally strongest in coarse-textured soils because of their lower surface area (DeBano et al. 1970; Campbell et al. 1977).

The strength and likelihood of a hydrophobic layer also varies with vegetation type. Much of the basic work on hydrophobicity has focused on chaparral communities, but the well-documented association between chaparral and post-fire hydrophobicity is probably partly due to the high intensity of chaparral fires and the propensity of chaparral to grow on coarse-textured soils.

Much less literature is available on the development of hydrophobic layers in other vegetation types, but the literature suggests that more xeric vegetation types have higher concentrations of the secondary compounds that contribute to the development of hydrophobic layers. Post-fire hydrophobic layers have been documented in some pine forests (particularly *Pinus ponderosa* and *Pinus radiata*) (Scott and Van Wyk 1990; Onda et al. 1996) and mixed conifer forests (Dyrness 1976). Hydrophobic layers might also be expected in other vegetation types with substantial surface fuel loadings and high concentrations of secondary compounds, such as oak woodlands, lodgepole pine, pinyon-juniper, and spruce-fir.

Fire-induced hydrophobic layers are of primary concern when they cause infiltration rates to drop below precipitation intensity. The resulting shift in runoff generation from subsurface flow to infiltration-excess overland flow will substantially increase runoff volumes and the size of peak flows from a given rainstorm (Nassari 1989; Scott 1993). More importantly, the combination of rainsplash and surface overland flow can increase erosion rates by one or more orders of magnitude (e.g., Wohlgemuth et al. 1996) and greatly increase the proportion of eroded material delivered from hillslopes to the stream channels (Sampson 1944; Prosser 1990). The resultant increase in sediment supply increases sediment yields and can cause severe aggradation in lower-gradient channels that are not simultaneously subjected to large increases in the size or duration of peak flows (Meyer et al. 1995).

High intensity rainfall events on burned slopes can also generate debris flows. In most cases these are rapidly-moving slurries of water, ash, and sediment triggered by a post-fire reduction in infiltration rates. Such flows can severely scour existing channels and deliver large amounts of material to debris fans or downstream reaches, often with severe consequences to life, property, and aquatic ecosystems (e.g., Klock and Helvey 1976; McPhee 1989; Wohl and Pearthree 1991; Rinne 1996).

While a number of studies have investigated the effects of wildfires on runoff and erosion, we know of no efforts to predict post-fire erosion and sedimentation hazards across a landscape or large catchment. Many ecologists have argued that fire suppression has greatly increased fuel loadings throughout the Western U.S., and consequently the risk of large-scale, high-intensity wildfires (Water Resources Center 1989; Agee 1994; Covington and Moore 1994). The increasing availability of geographic information systems (GIS) and geo-referenced databases make it possible to predict fuel loadings and fire intensity. By combining this information with geo-referenced data on soil texture, slope, precipitation intensity, and stream gradients, we believe that we can predict, at least on a relative basis, the post-fire surface erosion hazard and resultant sedimentation risk.

The approach developed in this paper draws upon the conceptual model that underlies the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1997). RUSLE, which is simply an updated version of the Universal Soil Loss Equation, predicts the combined soil loss from rainsplash, sheetwash, and rill erosion as a product of rainfall erosivity, soil erodibility, slope length, slope steepness, vegetative cover, and management practices. We adapted this widely-used empirical model to post-fire situations by incorporating the additional hazards due to removing the canopy and surface vegetation cover, increasing soil moisture, and developing a hydrophobic layer. The susceptibility of stream segments to sedimentation was assessed from the estimated stream gradients. These broad-scale predictions allow land managers and policy makers to compare fire-induced erosion and sedimentation risks across watersheds, or evaluate relative risk within smaller catchments for planning purposes.

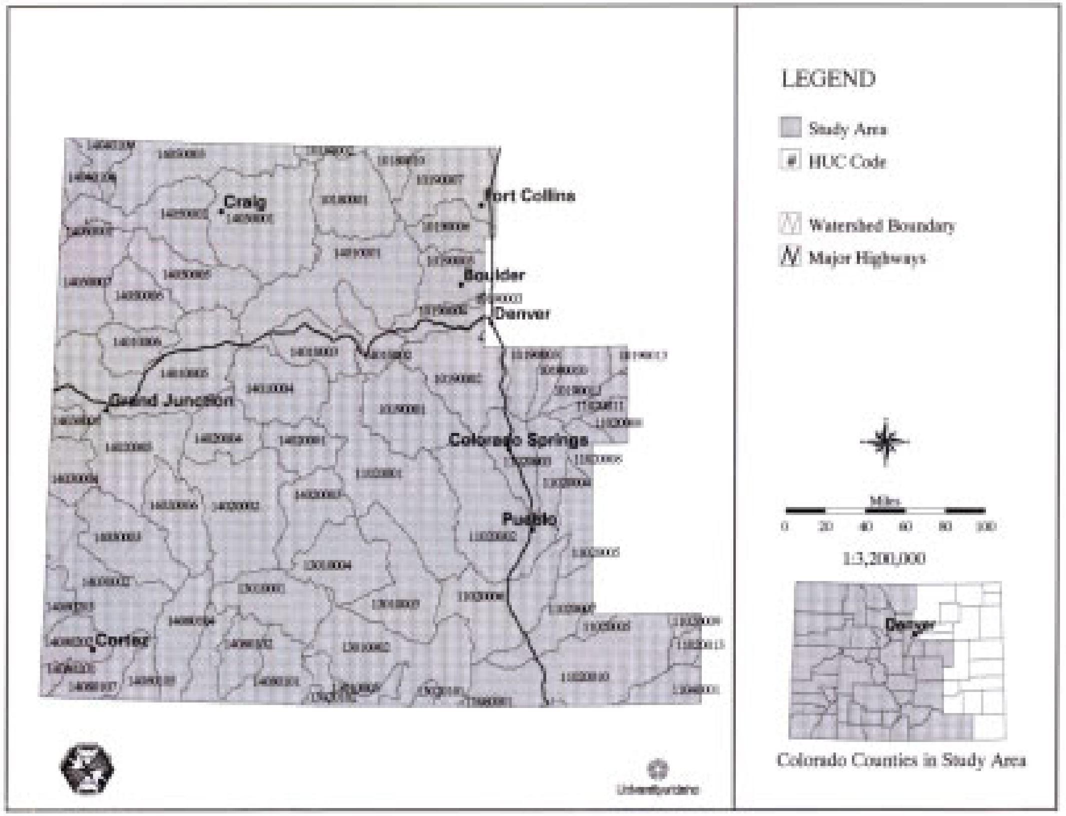
This paper reports on our efforts to: (1) develop a conceptual model for predicting surface erosion risk prior to and following wildfire; (2) apply this model across catchments in central and western Colorado; (3) assess the relative risk of post-fire surface erosion for each catchment; (4) rate the susceptibility of each stream segment to sedimentation; and (5) rank the catchments in western Colorado according to the proportion of stream segments at risk per unit catchment area. Although the results are specific to the study area, we believe that the methodology developed here could easily be adapted to other areas.

STUDY AREA AND DATA SOURCES

As described in Sampson and Neuenschwander (1999), the study area consists of approximately 180,000 km^2 in central and western Colorado. The high plains of eastern Colorado were excluded because these areas are mostly in private agriculture and generally not subject to wildfires. The study area was divided into 72 basins according to the eight-digit hydrologic unit codes (HUC) assigned by the U.S. Geological Survey (Figure 4.1; Table 1). Although many basins were truncated by the state line or the study boundary, the typical size for an entire catchment was 1500-8000 km^2. The average number of ignitions and acres burned for each of the 72 basins was determined from detailed data on wildfires in Colorado from 1986 to 1995 (Neuenschwander et al. 2000).

Geo-referenced data layers were obtained from a variety of sources (Sampson and Neuenschwander 2000). Digital elevation data were on a 90-m grid scale, and these data were used to calculate the average slope between grid squares. The state soils database (STATSGO) provided detailed information on each soil type, but the resolution of the STATSGO database is limited by the 6 km^2 minimum map unit (NRCS 1994). Fourteen vegetation and land cover types were derived from Advanced Very High Resolution Radiometer (AVHRR) coverage, and this was mapped on a one square kilometer grid.

FIGURE 4.1. Hydrologic unit codes (HUC) and catchments in the study area.



Fuel loading and predicted fuel consumption from wildfire were estimated by fuel type for each vegetation and cover type (Neuenschwander et al. 2000). The litter/duff fuel loadings and predicted consumption were used to estimate a qualitative fire severity for each vegetation and land cover type. Similar values were grouped, and this led to four fire severity classes (FI-SEV). We also assigned a propensity to form hydrophobic layers (HYPRO) to each vegetation type according to our best estimate of the amount of secondary compounds in the leaves and litter. Values for the latter index ranged from one to three, and both of these relative rankings were confirmed by foresters and fire scientists familiar with wildfires in Colorado.

For each vegetation type we also assigned a factor to represent the relative increase in soil wetness that would occur if the predominant vegetation was killed by a wildfire. This factor was included because hydrophobic layers are ineffective once they are wetted, so most or all of the winter and spring precipitation will infiltrate into the soil. The reduced evapotranspiration losses during the growing season result in higher soil moisture contents as compared to unburned areas. High amounts of soil moisture will increase the relative likelihood of overland flow due to a saturated soil profile, thus affecting the movement of sediment to the stream channel and the size of peak flows (e.g., O'Loughlin et al. 1982; Beschta 1990). The change in wetness was scaled according to the likely increase in annual water yield. Values for the major vegetation types were derived from paired-watershed experiments (e.g., Bosch and Hewlett 1982; Troendle et al. 1987), predictive models developed by the U.S. Forest Service and EPA (EPA 1980), and observed soil moisture changes in similar vegetation types (Sampson 1944; Klock and Helvey 1976; Helvey 1980). Little or no change in soil wetness was predicted for vegetation types in more arid areas or that would exhibit very rapid hydrologic recovery after a wildfire.

Perennial streams were mapped as vectors from 1:100,000 topographic maps. Pixels with streams were identified, and we assumed the stream length within a pixel to be 90 meters. The total length of streams in a watershed was estimated by multiplying the total number of pixels with streams by 90 m. Stream gradients were calculated as the difference in elevation between adjacent pixels that had been designated as having a perennial stream divided by the stream length. Drainage density was calculated by dividing the estimated kilometers of streams by the area of the catchment in square kilometers. Six catchments were excluded from this portion of the analysis because only the headwaters were located within the study area and the total channel length was too small to be considered representative (i.e., less than 10 km of stream channels).

TABLE 1. List of hydrologic unit codes (HUC), catchment characteristics, composite erosion risk, composite sedimentation risk, percent burned and number of ignitions.

HUC	Catchment Area (km^2)	Drainage Density (km/km^2)	Hydrophobicity Composite risk (CHYRISK)	Erosion Composite risk index (CSURFERO)	Percent Streams <2% gradient	Percent Streams >6% gradient	Sedimentation Risk index (CSEDRISK)	Percent Burned 1986-1995	Ignitions per 40 km^2 1986-1995
10180001	3704	0.19	134.5	0.6	39	21	3.36	1.06	0.9
10180002	493	0.11	150.6	0.4	20	42	2.56	0.01	1.1
10180010	1015	0.16	115.5	0.2	26	38	2.76	2.13	0.9
10190001	4165	0.10	106.9	0.6	35	26	3.18	0.03	1.8
10190002	4539	0.11	137.7	2.0	30	31	2.99	0.22	7.3
10190003	1428	0.05	90.1	0.0	47	11	3.72	0.26	2.4
10190004	1431	0.09	120.9	2.5	30	35	2.91	0.01	1.7
10190005	2173	0.13	130.2	0.9	37	26	3.22	1.44	11.2
10190006	1930	0.14	103.6	0.8	35	31	3.09	0.19	7.6
10190007	3819	0.18	59.9	0.1	36	27	3.18	0.56	3.8
10190008*	50	0.19	0.0	0.0	39	9	3.60	0.34	2.4
10190010	820	0.03	55.8	0.0	45	10	3.69	0.00	0.3
10190011	1847	0.07	4.1	0.0	45	13	3.64	0.00	0.1
10190013	298	0.20	0.0	0.0	37	22	3.31	0.00	0.0
10250001*	22	0.03	0.0	0.0	67	0	4.33	0.00	0.0
11020001	7919	0.13	83.9	3.3	26	37	2.79	0.07	1.9
11020002	5956	0.14	45.4	1.7	35	25	3.20	0.63	2.9
11020003	2409	0.14	44.9	0.1	44	18	3.52	0.30	3.4
11020004	1919	0.06	10.8	0.0	57	3	4.08	1.96	3.0
11020005	2385	0.04	13.3	0.0	56	6	3.99	0.55	0.3
11020006	4822	0.11	76.7	1.9	35	23	3.24	0.08	2.0
11020007	2549	0.09	52.3	0.2	42	18	3.48	0.03	0.6
11020008	908	0.06	25.9	0.0	53	4	3.98	0.77	1.7
11020009	380	0.09	0.0	0.0	37	19	3.36	0.00	0.1
11020010	7541	0.09	51.2	0.8	34	24	3.20	0.80	0.9
11020011	864	0.10	2.9	0.0	45	13	3.65	1.04	0.2
11020012	828	0.08	0.3	0.0	49	7	3.84	0.25	0.2
11020013	602	0.06	0.0	0.0	37	15	3.43	0.22	0.3
11040001	1408	0.11	1.6	0.0	29	26	3.06	0.04	0.2
11040004*	38	0.03	0.0	0.0	40	0	3.80	0.00	0.0
11080001*	146	0.08	186.0	7.3	23	39	2.68	0.00	1.1
13010001	3461	0.09	125.1	15.0	28	35	2.88	0.06	0.9
13010002	6513	0.08	93.7	3.4	38	26	3.23	0.06	2.3
13010003	4135	0.05	80.8	2.1	38	25	3.27	1.40	1.7
13010004	3490	0.08	65.6	2.9	34	28	3.12	0.02	1.1
13010005	1390	0.10	125.4	12.4	34	32	3.04	0.13	1.2
13020101	332	0.03	55.0	0.0	70	10	4.21	0.00	0.1

13020102*	214	0.08	206.6	11.2	21	45	2.51	0.01	0.7
14010001	7518	0.13	126.7	0.9	20	45	2.51	0.24	1.5
14010002	1768	0.12	129.4	6.9	22	47	2.49	0.01	2.7
14010003	2523	0.10	139.8	7.0	20	47	2.45	0.44	4.3
14010004	3758	0.07	149.7	19.1	22	44	2.58	0.10	1.3
14010005	8030	0.14	86.3	2.8	27	34	2.86	1.51	4.0
14010006	1821	0.13	113.2	12.6	24	40	2.67	0.13	1.8
14020001	1990	0.09	103.6	1.0	24	42	2.65	0.05	0.8
14020002	6231	0.10	87.4	4.8	24	39	2.69	0.08	1.0
14020003	2858	0.09	94.4	4.2	28	34	2.87	0.10	0.8
14020004	2503	0.11	85.4	0.3	21	43	2.57	0.80	3.3
14020005	4275	0.15	64.7	0.8	25	37	2.77	0.18	2.1
14020006	2945	0.11	81.4	4.2	28	33	2.90	0.07	2.3
14030001	730	0.20	23.4	0.0	25	34	2.83	0.86	2.6
14030002	4930	0.12	62.2	1.4	27	34	2.86	1.16	2.0
14030003	3991	0.12	68.8	1.7	25	37	2.77	0.66	4.5
14030004	1632	0.13	37.1	0.0	20	45	2.51	0.57	2.0
14030005*	5	0.02	40.5	0.0	0	0	3.00	0.00	0.0
14040106	889	0.12	29.3	0.1	23	36	2.74	33.81	8.2
14040109	1146	0.17	23.3	0.0	26	34	2.84	3.28	5.7
14050001	6795	0.14	90.2	1.1	28	35	2.86	0.91	1.3
14050002	4036	0.14	23.7	0.0	31	31	3.01	13.49	5.8
14050003	4441	0.13	64.0	0.4	33	24	3.19	2.12	2.2
14050005	3510	0.15	75.7	1.1	24	40	2.69	2.71	2.6
14050006	2366	0.18	16.3	0.1	25	36	2.77	1.46	8.5
14050007	3811	0.17	26.2	0.2	26	35	2.83	3.52	5.8
14060001	195	0.21	8.9	0.0	19	50	2.38	2.79	5.7
14080101	4205	0.10	102.6	7.1	30	35	2.90	0.18	3.6
14080102	1748	0.07	81.7	5.2	29	34	2.90	1.78	4.2
14080104	2969	0.09	103.1	8.5	28	36	2.83	1.04	4.2
14080105	1111	0.11	74.0	3.0	32	27	3.10	1.32	4.5
14080107	1927	0.15	98.2	3.7	27	32	2.90	21.67	17.2
14080201	397	0.11	66.6	0.4	38	20	3.37	99.65	1.9
14080202	1665	0.14	103.6	0.7	33	25	3.16	0.70	5.7
14080203	954	0.13	53.5	0.0	33	24	3.18	0.02	1.1

* Excluded from sedimentation analysis

Mean	2606	0.110	70.7	2.4	33%	28%	3.09	0.03	2.7
Std. Deviation	2111	0.044	49.0	3.9	11%	13%	0.45	0.13	2.9
Coeff. Variation	81%	40%	69%	165%	35%	46%	14%	426%	108%
Maximum	8030	0.21	206.6	19.1	70%	50%	4.33	99.65%	17.2
Minimum	5	0.02	0.0	0.0	0%	0%	2.38	0.00%	0.0
Skewness	0.92	0.15	0.31	2.38	0.89	□0.57	0.81	6.91	2.43

MODEL DESCRIPTION

The first index we developed assessed the relative risk of generating a fire-induced hydrophobic layer by the function:

$$\text{HYRISK} = \text{FISEV} \times \text{HYPRO} \times \text{TEXTURE} \quad (1)$$

where HYRISK was the hydrophobicity risk index, FISEV was the fire severity by vegetation type, HYPRO was the propensity to form a hydrophobic layer by vegetation type, and TEXTURE was soil texture. Assigned values for FISEV ranged from 0 to 6, while HYPRO ranged from 0 to 3 (Table 2). Given the large number of soils listed in STATSGO, we took the pragmatic approach of rating soil texture as coarse or fine according to the hydrologic soil group (Musgrave and Holtan 1964). High permeability soils (i.e., soils in hydrologic group A or B) were assumed to be coarse-textured and assigned a value of 3. Low permeability soils (i.e., soils in hydrologic groups C or D) were assumed to be fine-textured and assigned a value of 1, as the latter soils would be much less likely to develop a hydrophobic layer following a wildfire. We generally did not assign zero values for FISEV or HYPRO, as this would automatically result in an erosion hazard of zero regardless of slope, soil type, or fire severity. The calculated HYRISK values ranged from 0 to 36 as compared to a theoretical range of 0 to 54. These values were grouped into five classes for calculating risk assessments and three classes for mapping purposes.

TABLE 2. List of vegetation types and their associated fire severity (FISEV), propensity to form hydrophobic layers (HYPRO), and post-fire wetness factor (W).

Vegetation Type	Fire Severity	Propensity to Form a Hydrophobic Layer	Post-Fire Wetness
Spruce/fir	6	1	1.5
Mixed conifer	6	2	1.5
Ponderosa pine	3	3	1.3
Lodgepole pine	3	3	1.5
Pinyon/Juniper	1	3	1.0
Aspen/Hardwood	1	1	1.3
Cottonwood/Willow	1	1	1.3
Sage	1	3	1.1
Annual grass	1	1	1.0
Perennial grass	1	1	1.0
Alpine tundra	1	1	1.0
Water	0	0	1.0
Agriculture	1	1	1.0
Grassland/Cropland	1	1	1.0

The inherent erodibility of a site was defined as the soil erodibility index (SOILEROD), and this was calculated for each pixel by:

$$\text{SOILEROD} = \text{K} \times \text{S} \tag{2}$$

where K and S are the soil erodibility and slope factors, respectively, from RUSLE (Renard et al. 1997). K values were obtained directly from the STATSGO database and these generally ranged from 0.10 to 0.45. S was calculated according to the slope angle in degrees (?) by the following equations (Renard et al. 1997):

$$\text{S} = 10.8 \sin? + 0.03 \text{ (for slopes less than 5.14 degrees)} \tag{3}$$

$$\text{S} = 16.8 \sin? + 0.50 \text{ (for slopes greater than 5.14 degrees)} \tag{4}$$

Relative surface erosion risk was predicted by:

$$\text{SURFERO} = \text{HYRISK} \times \text{SOILEROD} \times \text{W} \tag{5}$$

where SURFERO is the surface erosion risk index, HYRISK is the hydrophobicity risk index (equation 1), SOILEROD is the soil erodibility index (equation 2), and W represents the increase in soil wetness by vegetation type (Table 2). The resulting SURFERO values ranged from 0 to 295, and these were grouped into five classes to produce the pixel-scale map of surface erosion risk over the study area (Table 3).

A composite surface erosion risk index (CSURFERO) was determined for each basin by summing the fraction of area in each surface erosion risk class times its respective class rank (equation 6):

$$\text{CSURFERO} = \sum_{i=1}^{5} \left[\frac{\text{number of pixels in class } i}{\text{total number of pixels in catchment}} \times \text{SURFERO class} \right] \tag{6}$$

Similarly, a composite hydrophobicity risk index (CHYRISK) was determined for each watershed by summing the fraction of the watershed in each hydrophobicity class times its hydrophobicity risk class (Table 4):

TABLE 3. Classification of surface erosion risk (SURFERO) values.

Soil Erosion Risk Index	Class and Weighting
0-60	1
60-120	2
120-180	3
180-240	4
240-300	5

TABLE 4. Classification of hydrophobicity risk (HYRISK) values.

Hydrophobicity Risk Index	Class and Weighting
0-7	1
7-14	2
14-21	3
21-28	4
28-36	5

$$\text{CHYRISK} = \sum_{i=1}^{5} \left(\frac{\text{number of pixels in class } i}{\text{total number of pixels in catchment}} \times \text{HYRISK class} \right) \qquad (7)$$

The use of a RUSLE-type methodology ignores the possibility of mass movements and implies that most of the eroded material will be sand-sized or smaller. Since these particles are easily entrained, stream gradient can be used as an index of sedimentation risk (e.g., Benda and Dunne 1987; Montgomery and Buffington 1993). Stream segments with a gradient greater than six percent were assumed to have sufficient energy to transport the eroded sediment to the next downstream segment, and these high-gradient reaches were rated as having little sedimentation risk. Stream segments with a gradient of 2-6% were assumed to have a moderate risk of sedimentation, while segments with a gradient less than 2% were presumed to have a high risk of sedimentation. Sedimentation risk (SEDRISK) values of 5, 3, and 1 were assigned to stream segments with a high, moderate, or low risk of sedimentation, respectively.

These gradient breaks are consistent with studies of downstream sediment transport from debris flows (Benda and Dunne 1987). A two-percent gradient is also a critical break in two of the most commonly-used stream classification systems (Rosgen 1994; Montgomery and Buffington 1997). Montgomery and Buffington explicitly designate streams with a gradient of less than two percent as transport-limited or "response" reaches because of their sensitivity to sediment deposition.

A composite sedimentation risk (CSEDRISK) in each catchment was determined by summing the fraction of stream pixels in a gradient class times the sedimentation risk factor and dividing by the total number of stream pixels within a catchment (equation 8):

$$\text{CSEDRISK} = \sum^{3} \left(\frac{\text{number of pixels in class } i}{} \times \text{SEDRISK} \right) \qquad (8)$$

RESULTS

Landscape-Scale Assessments

The hydrophobicity risk map of the study area (Figure 4.2) indicates that the greatest risk is in the more densely vegetated mid-elevation montane areas. The higher values generally run parallel to the Continental Divide, although the highest elevations have less vegetation and therefore a lower risk.

There are generally more high-risk areas in the western part of the state and on the western side of the Continental Divide than on the eastern slopes. This pattern suggests that the highest risk of post-fire hydrophobicity is generally in the more mesic zones where high fuel loadings can generate high severity fires. Another zone with high risk areas extends to the western border in the central part of the state, and this includes areas similar to the infamous Storm King fire that killed 14 firefighters in July 1994. Patches of high risk areas can also be found in the mid-elevation zones along the Front Range. Lower elevation areas generally have a lower risk of hydrophobicity because of their lower fuel loadings.

The higher spatial resolution of the data used to develop the surface erosion risk map yielded a more detailed delineation of areas with a high erosion risk (Figure 4.3). Areas with the highest risk were in the steeper montane areas that also had a high hydrophobicity index. Areas with particularly high SURFERO values were in the Sawatch Range in the central part of the study area, the San Juan Mountains in the southern part of the state, and the area around the Roan Plateau (just north of Grand Junction in the west-central part of the study area).

Catchment-Scale Analysis

A statewide plot of sedimentation risk did not present any clear patterns, and a more explicit evaluation of the patterns of HYRISK, SURFERO, and SEDRISK was only possible from smaller scale maps. Thus, a complete set of maps was prepared for three catchments (HUC codes 11020002, 14030003, and 14050006) located in different parts of the state with moderate-to-high frequency of ignitions. For illustrative purposes this section will focus only on the middle Arkansas basin in the southeastern part of the study area (HUC code 11020002), as this basin had a moderate CSURFERO (25th of the 72 catchments analyzed).

The mapped hydrophobicity index clearly reflects the coarse scale of the vegetation and soil layers (Figure 4.4). The highest HYRISK values were in the southwestern part of the basin, and comparisons with other maps indicate these areas are generally between 2000 and 3000 meters elevation on lands belonging to the Bureau of Land Management or the San Isabel National Forest.

The areas with the highest surface erosion risk were also in the southwestern part of the basin (Figure 4.5). Although these areas were generally associated with a high hydrophobicity risk index, not all areas with a high hydrophobicity risk have a high surface erosion risk. These results suggest that slope has a relatively strong effect on surface erosion risk, and the comparatively high resolution of the digital elevation model allows a much finer-scale resolution of the relative surface soil erosion risk as compared to the hydrophobicity risk.

Sedimentation risk is also plotted on Figure 4.5. As expected, the smaller, headwater segments generally have the steepest gradients and hence the lowest sedimentation risk. The mainstem of the Arkansas River extends from west to east across this basin, and it consistently has a high sedimentation risk. The lower reaches of the other major streams within this catchment also tend to have a high sedimentation risk, but there often are intervening segments with intermediate sedimentation risks.

Interbasin Comparisons

A third set of analyses examined the interrelationships between different variables and the ranking of each of the 72 basins according to their composite hydrophobicity risk (CHYRISK), composite surface erosion risk (CSURFERO), and composite sedimentation risk (CSEDRISK). As indicated in Table 1, CHYRISK ranges from 0 to 207. The watersheds with the highest CHYRISK are generally in the south-central (San Juan) and north-central mountains. Watersheds in the eastern and northwestern parts of the study area generally had lower CHYRISK values, even though there were often some areas with high HYRISK values within these watersheds.

The distribution of composite surface erosion values was highly skewed, as most basins tended to have relatively low CSURFERO values and only a few basins had relatively high CSURFERO values (Table 1). The skewed distribution of CSURFERO values led us to place each catchment into one of five classes according to the logarithm of CSURFERO (Table 5). Only 23 basins were designated as having a moderate or higher composite post-fire surface erosion risk on a basin-wide scale, while the other 49 basins were classified as having no more than a slight risk for post-fire surface erosion.

Basins with the highest CSURFERO values tended to be clustered in the southern and central mountains, and, to a lesser extent, in the west-central part of the state (Figure 4.6). This pattern suggests that one reason for the relatively low CSURFERO values in many basins is the large amounts of range and agricultural lands. Low post-fire erosion rates on the large ex-

FIGURE 4.2. Predicted hydrophobicity risk (HYRISK) by pixel in the study area.

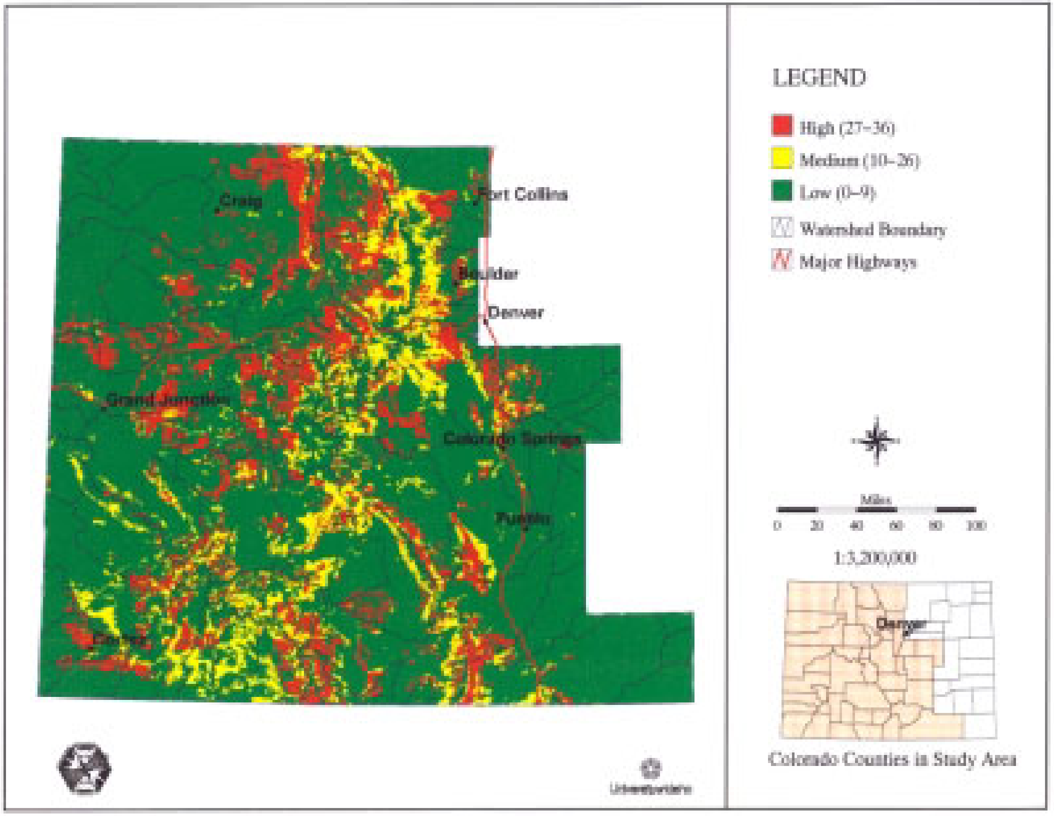


FIGURE 4.3. Predicted surface erosion risk (SURFERO) by pixel in the study area.

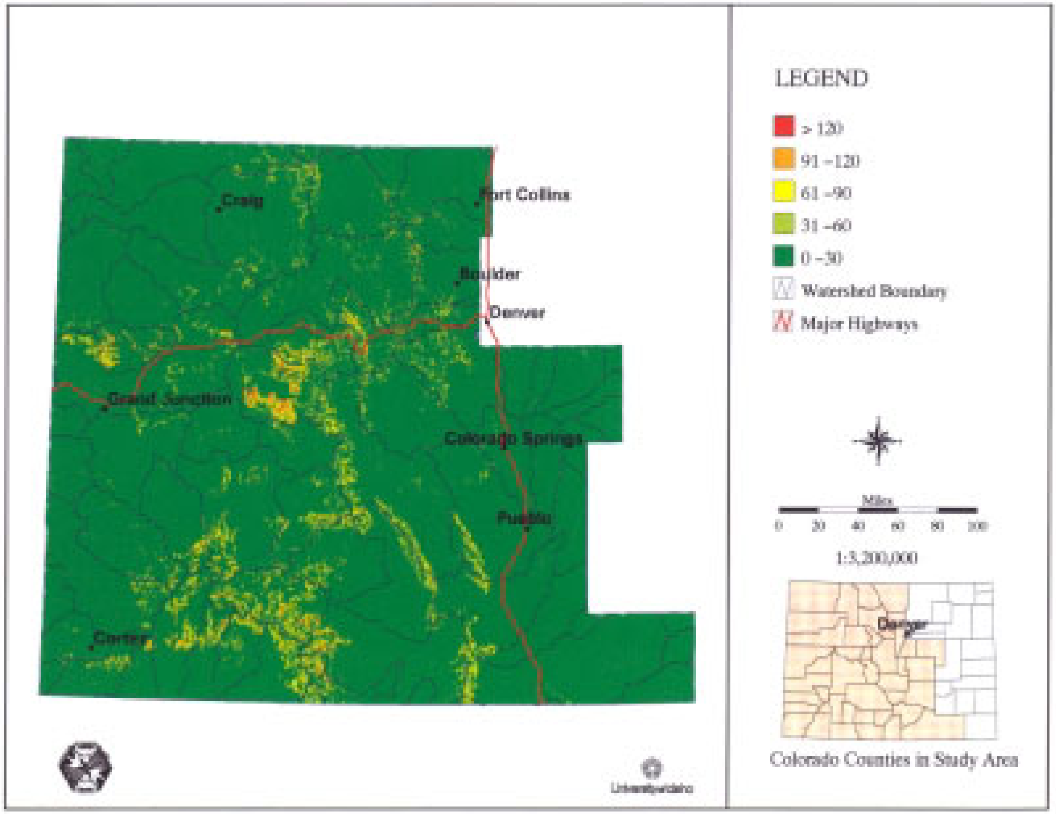


FIGURE 4.4. Predicted post-wildfire hydrophobicity in the middle Arkansas basin.

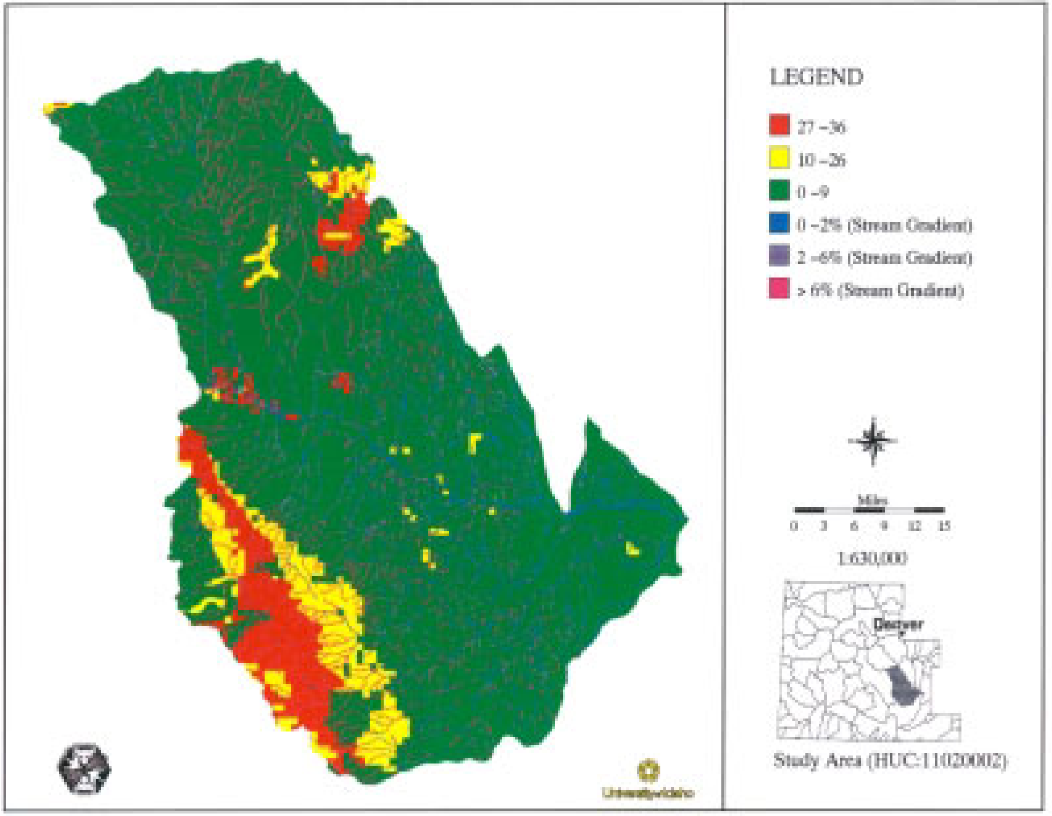


FIGURE 4.5. Predicted post-wildfire soil erosion and sedimentation risk in the middle Arkansas basin.

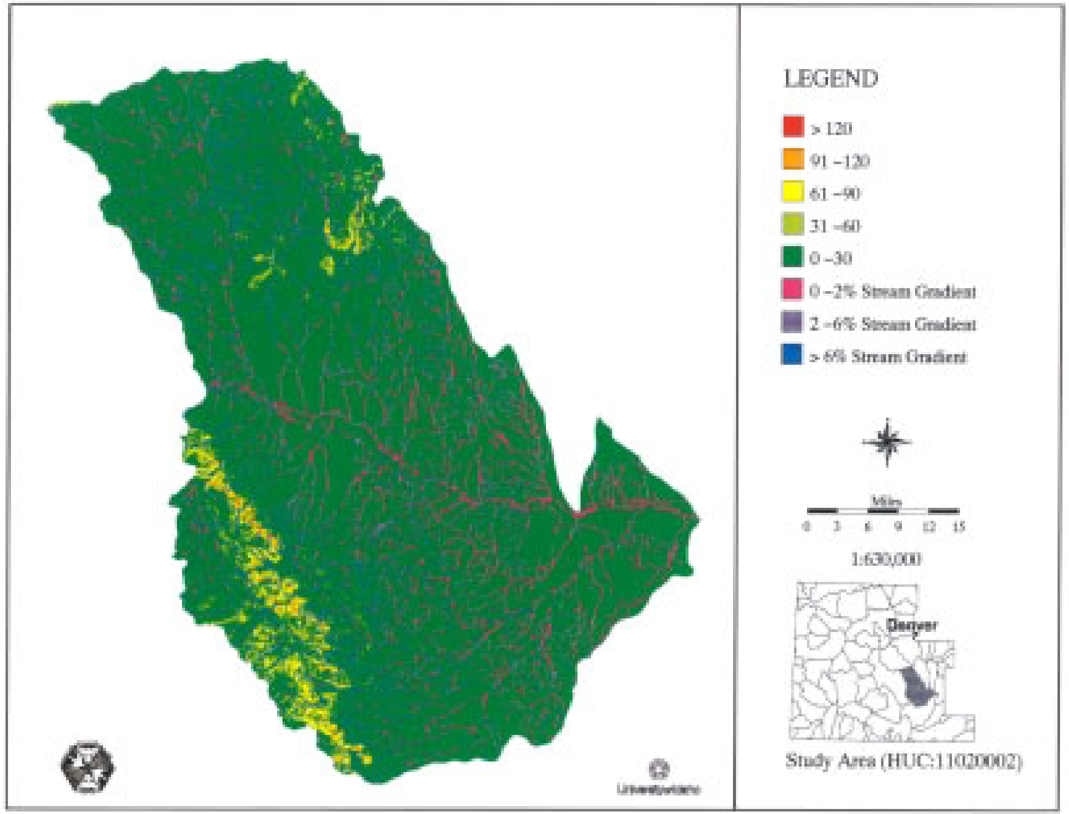
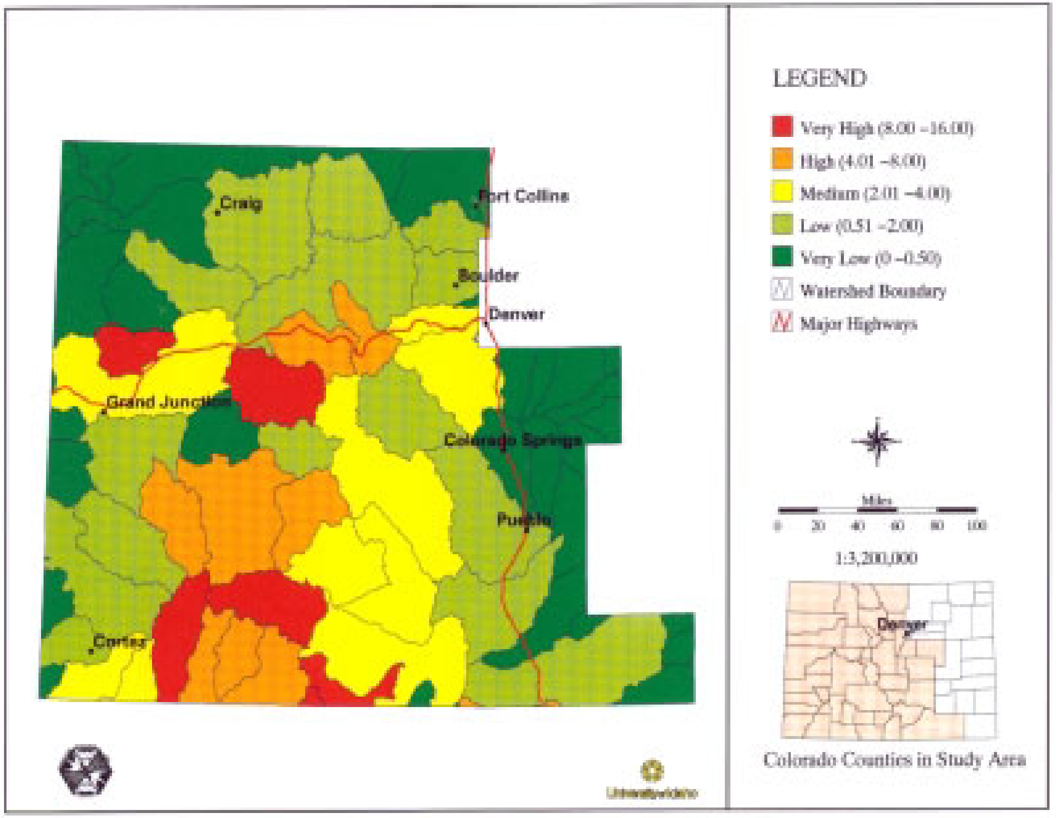


FIGURE 4.6. Map of composite soil erosion risk (CSURFERO) by catchment.



panses of these vegetation types would compensate for smaller areas with higher SURFERO values (e.g., Figure 4.5), thereby yielding a lower composite value.

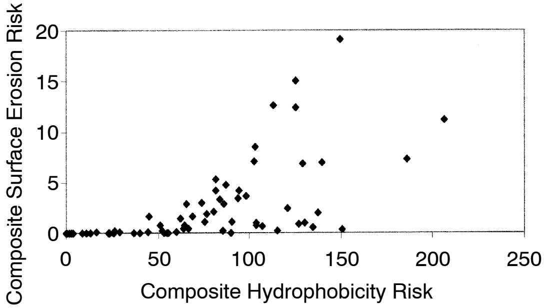
This "dilution" of high SURFERO values raises the issue of whether CHYRISK and CSURFERO depend in part on basin size. Higher CSURFERO values, for example, might be more likely in smaller basins because there would be less area in lower elevation zones with lower SURFERO values. Many basins in the southern mountains were truncated by the state line (Figure 4.1), and this also might lead to disproportionately high CSURFERO values. However, the correlation coefficient between watershed area and CSURFERO was only 0.10, and only 0.23 between watershed area and CHYRISK. Scatter plots showed no consistent trend between CSURFERO or CHYRISK and catchment area.

The effect of other variables besides the predicted hydrophobicity on surface erosion risk was assessed on a watershed scale by plotting CSURFERO against CHYRISK (Figure 4.7). Overall, the composite hydrophobicity index explained only 40% of the variation in the composite erosion index. This correlation and a comparison of Figures 4.2-4.5 confirm that a moderate

TABLE 5. Classification of composite surface erosion (CSURFERO) values and the number of catchments in each class.

Composite Surface Erosion Risk	Class	Number of Catchments
0-0.5	Very little risk	34
0.5-2.0	Slight risk	15
2.0-4.0	Moderate risk	9
4.0-8.0	High risk	8
8.0-16.0	Very high risk	6

FIGURE 4.7. Plot of composite hydrophobicity risk (CHYRISK) versus composite soil erosion risk (CSURFERO) for each catchment in the study area.



CHYRISK is necessary for a high CSURFERO, but a high CHYRISK does not necessarily result in a high CSURFERO.

Drainage Density and Composite Sedimentation Index

Calculated drainage densities ranged from 0.03 to 0.21 km/km^2 (Table 1). These relatively low values are due in part to the use of 1:100,000 maps to delineate channels, as well as the implicit assumption that each stream pixel has only 90 meters of channel. There was considerable variation between basins in the number of stream segments in the different gradient categories. For example, the proportion of streams with gradients less than two percent ranged from 19 to 57%, and the proportion of streams with gradients greater than six percent ranged from 4 to 47% (Table 1).

The calculated composite sedimentation risk (CSEDRISK) ranged from 2.4 to 4.2 (Table 1). Catchments with the highest sedimentation risk were generally in the eastern part of the study area where drainage densities were low and most streams had gradients of less than two percent (Table 1; Figure 4.8). Steeper catchments in the central and west-central parts of the study area had lower composite sedimentation risks.

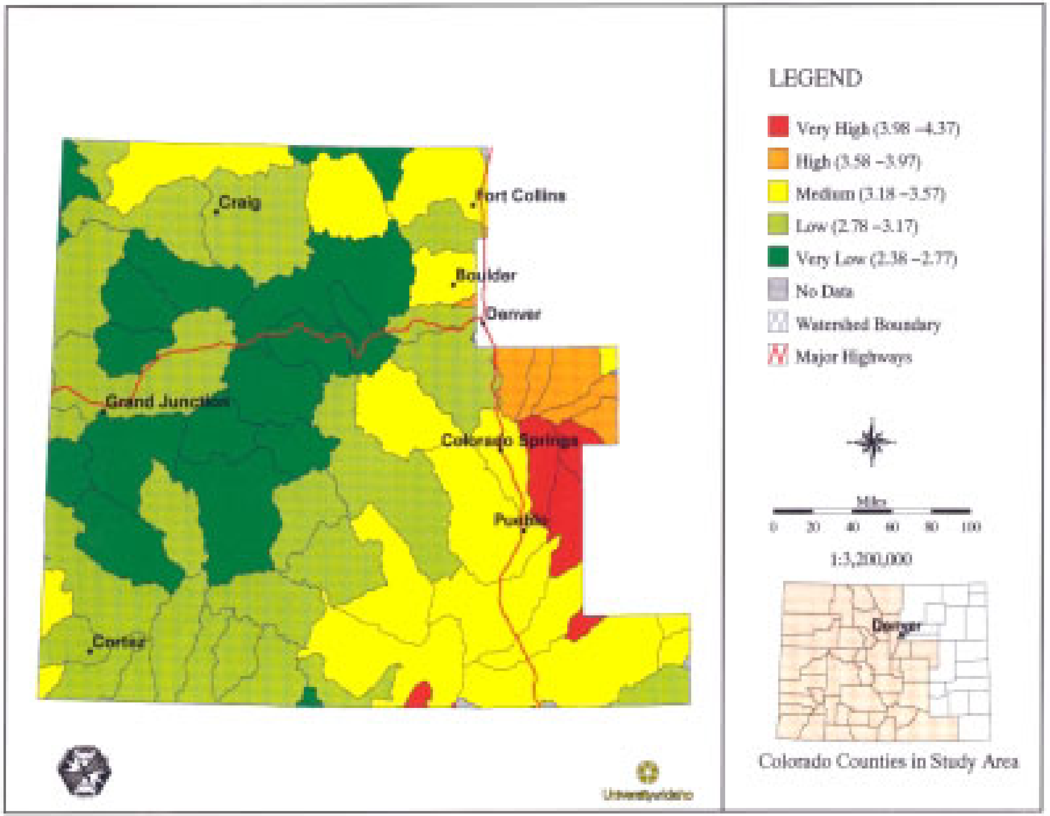
DISCUSSION

The procedures developed in this paper for predicting erosion and sedimentation risks are consistent with the limited literature on post-fire erosion and existing conceptual models. Both the patterns of risk and the correlation analyses suggest that the results are credible as a first estimate. However, before the results can be used to assist in setting policy or guiding future management, several other issues must be recognized. These include the temporal and spatial scale of the analyses, model limitations and validation, and the use of relative versus absolute predictions.

Temporal and Spatial Scale of the Analyses

An important limitation of this work is that the estimated erosion and sedimentation risks do not consider the frequency or varying intensity of wildfires within a vegetation type, or the rate of recovery. To properly quantify risk, the predicted erosion and sedimentation rates need to be distributed over long-term average cycles of burning and recovery. Differences in the relative frequency of fire might cause some vegetation types, such as sagebrush or ponderosa pine, to have much higher long-term average erosion rates than vegetation types with higher post-fire erosion rates, but less frequent fires.

FIGURE 4.8. Map of composite sedimentation risk (CSEDRISK) by catchment.



Different vegetation types will also have different rates of regrowth and hence different patterns of erosion rates over time. Resprouting species, such as aspen, should recover more quickly than spruce-fir. Recovery rates will also vary by erosion process. Surface erosion, for example, generally declines rapidly a fire, while mass movements triggered by a decrease in root strength will tend to be most frequent 5 to 10 years after a fire (USFS 1981). Ideally site- and process-specific recovery rates could be predicted from combining spatial data on key factors, such as aspect, vegetation type, elevation, and annual precipitation. The incorporation of fire frequency and recovery rates could alter the present catchment-scale rankings of erosion and sedimentation risks.

For this reason, we compared the mean number of ignitions and mean area burned by catchment to our catchment-scale estimates of composite soil erosion and sedimentation risks. While none of these plots showed any significant correlation, the highest CSURFERO values were associated with a very low probability of being burned and a relatively low probability of an ignition event. Although data on ignitions and area burned are only available for a 10-year period (1986-1995), these results indicate that our predictions of erosion and sedimentation risk need to consider the likelihood of burning.

In general, the catchments with the largest burned area were rangeland catchments. Relatively frequent, low-intensity fires in these basins may not result in much surface erosion or sedimentation risk for several reasons. First, these areas have limited amounts of precipitation and runoff. Second, these basins lack the fuel loading to generate an intense fire that could burn off much of the surface organic matter and possibly create a hydrophobic layer. Thus, basins with smaller, more severely burned areas could ultimately have much higher erosion and sedimentation rates than these large rangeland catchments that burn more frequently.

These arguments for lengthening the temporal scale are reversed with regard to the spatial scale. Basin-scale comparisons may be useful to identify higher-risk zones, but these will not have the necessary spatial resolution to guide where one should initiate a more intensive fuels treatment program.

Working at a spatial scale appropriate to vegetative manipulation or other types of intervention will also facilitate a more explicit spatial linkage between predicted erosion and sedimentation risk. At present sedimentation risk is assessed independently of the likely sediment production upstream of that location. In reality the upstream sediment production needs to be routed through the stream network to the segments of concern (Bunte and MacDonald in press). Not all streams will have the same resource value, and the incorporation of site-specific values (e.g., habitat for an endangered fish or a domestic water supply reservoir) can only be done on a smaller-scale with more specific data.

At the hillslope scale the location of a fire relative to the stream channel is an important control on the amount of sediment that is actually delivered to the stream channel (EPA 1980; Scott 1993). Higher resolution vegetation, soil, and topographic data are necessary to evaluate the delivery of sediment down a hillslope to a channel. If the analysis is limited to smaller areas, these spatial issues could be explicitly considered, and this would lead to more realistic assessments of potential environmental effects and better management guidelines.

Model Validation, Refinements, and Limitations

The predicted erosion risks are based on a relatively simple conceptual model of surface erosion from rainsplash, sheetwash, and rilling. In other areas wildfires have been shown to increase the number of landslides (Tiedemann et al. 1979) and the rate of dry ravel (Krammes 1965; Florsheim et al. 1991; Wohlgemuth et al. 1996). Consultations with fire and watershed scientists suggest that these other erosion processes are of lesser importance after wildfires in Colorado. Field observations of recently burned areas in the Colorado Front Range (Buffalo Creek and Pingree Park) indicate that dry ravel can be an important process, but surface runoff is usually the primary mechanism for delivering eroded sediment to the stream channels (Morris and Moses 1987).

Debris flows and channel erosion may also be important sediment sources, but these are much more difficult to predict. After the July 1994 South Canyon fire in western Colorado large amounts of sediment were delivered to the channel network by wind erosion, dry ravel, and rill and gully erosion. Heavy rains in early September then eroded more material from the burned hillslopes and scoured an estimated 70,000 m^3 of material from the hillsides and the channels as hyperconcentrated flows and debris flows (Cannon et al., 1998). Meyer et al. (1995) found a similar topographic sequence of rill erosion, debris torrents, and main channel scour after the 1988 Yellowstone fires.

In other areas it may be necessary to predict the likelihood of mass movements following wildfires. The development of such a model would need to incorporate other factors, such as root cohesion and pore water pressures, but the formulation could largely follow the methodology used by Dietrich et al. (1994) to predict susceptibility to landslides. Any effort to predict mass movements will require higher resolution data, as the occurrence of mass movements is highly dependent on topographic convergence and slope. Accurate mapping of these characteristics requires as small a pixel size as is feasible (Quinn et al. 1995). Higher resolution data are also needed because most slides will be smaller than the 90 × 90-meter pixels used here.

Several studies have noted that the post-fire surface erosion risk is highly dependent on the occurrence of a high intensity storm (e.g., Krammes 1965; Renard et al. 1996; Prosser and Williams 1998). This means that our erosion prediction model could be improved by adding either a deterministic or a stochastic precipitation component. Jarrett (1990) has already noted that the occurrence of large runoff events in Colorado varies with elevation, and it would be relatively easy to incorporate a precipitation factor, such as the 2-year 30-minute precipitation event, into the surface erosion risk index (equation 5).

Further refinements might include stochastic components to account for the seasonal timing of both fires and future precipitation events. Wildfires in the early summer, for example, are more likely to be followed by high intensity rainfall events as was observed at Storm King Mountain in 1994 and Buffalo Creek near Denver in 1996. Negative impacts from hydrophobic soil layers and soil erosion are much less likely if a late fall fire is first subject to snowfall rather than rainfall events. The likely timing of a wildfire will vary with vegetation type and location, just as the likelihood of a given precipitation event will vary over the course of a year and with location. Hence a more accurate assessment of post-fire erosion and sedimentation risks will require a combination of deterministic functions, based on location and vegetation type, with stochastic components to represent the relative likelihood of different fire and rainfall events.

A major limitation in the development and application of the model used in this study is the uncertainty over the strength and persistence of a fire-induced hydrophobic layer. Observations at Buffalo Creek and Pingree Park suggest that most of the erosion was due to the complete elimination of the surface vegetative cover and the pulverization of the soil by burning off the organic matter and breaking down the soil aggregates. The resulting fine-textured, cohesionless surface layer was highly susceptible to rainsplash, soil sealing, and, on the steeper slopes, dry ravel. The observed high density of rills may have been due to a combination of surface sealing and the development of a hydrophobic layer. Although surface sealing and a hydrophobic layer reduce infiltration by different processes, the effect of each process on runoff is similar. A high fire intensity is critical to both processes and the two processes may be synergistic.

The persistence of fire-induced hydrophobic layers has not been rigorously evaluated. A number of studies have documented a rapid decline in post-fire erosion rates, and this is taken as *de facto* evidence for a breakdown of the hydrophobic layer (e.g., Morris and Moses 1987; Prosser and Williams 1998). Root growth, animal burrowing, and a variety of other physical, biological, and chemical processes all will act to break up a hydrophobic layer, and we are not aware of any study that has shown accelerated erosion for more than four years after a fire. In the absence of detailed studies, the more persistent increases in erosion could be ascribed to a reduction in cover rather than a persistent, fire-induced hydrophobic layer.

A serious limitation to the use of our surface erosion prediction models is the relative absence of plot-scale data from the study area on post-fire hydrophobicity, runoff, and surface erosion rates. The limited data from Morris and Moses (1987) are not sufficient to calibrate, much less validate, the models developed in this paper. Observations from Buffalo Creek and Pingree Park suggest that the models presented here may overemphasize the development of a hydrophobic layer, but the relative results may well be accurate because the factors controlling post-fire erosion and runoff-generation (i.e., loss of surface cover, surface sealing and hydrophobicity) are similar regardless of which process is reducing infiltration rates. Buffalo Creek also may not be typical, as a high-intensity fire in early summer was followed by a 1-hour rainfall event that may have a recurrence interval of 100 years or more (R. Jarrett, U.S. Geological Survey, pers. comm., 1996). The high intensity of this rainfall event may have masked the effect of a hydrophobic layer relative to less extreme rainfall events.

The other sources of post-fire erosion that are not considered in our model are the effects of fire suppression and rehabilitation. Efforts to control a wildfire typically involve the construction of fire lines in rugged terrain, and these are usually constructed with little regard to streamside management zones or post-fire erosion rates. The relative importance of erosion from suppression efforts will depend on the particular fire, landscape, and type of suppression activities, but the implications of fire suppression efforts must also be considered if one is predicting post-fire erosion and sedimentation risks. Similarly, one should also include the reduction in erosion associated with post-fire rehabilitation efforts (MacDonald 1989). Limited research suggests that fire rehabilitation efforts have had mixed success in reducing post-fire sediment production and delivery (Taskey et al. 1989; Booker et al. 1998; Wohlgemuth et al. 1996).

Relative vs. Absolute Predictions

A final issue is the context of our predictions. Should the predicted erosion and sedimentation risks be evaluated relative to pre-disturbance erosion and sedimentation rates in the area of interest, or on a more absolute scale, as in this paper? Data and time limitations forced us to utilize an absolute scale in this paper, but land managers may wish to focus their efforts on areas and stream segments predicted to have the greatest change in erosion and sedimentation relative to pre-fire conditions. On the other hand, a large percentage increase in erosion may be relatively meaningless in areas where the pre-fire erosion rate is low. Thus an approach that evaluates both the absolute increase and the increase relative to background is preferable to accurately assess risk and evaluate management options.

CONCLUSIONS

The increasing availability of geo-referenced databases makes it possible to develop and apply conceptual soil erosion and sedimentation models across a range of spatial scales. In central and western Colorado the predicted post-fire surface erosion rates were highest in steep areas with vegetation types that could support high-intensity fires. Predicted surface erosion risks were low in most of the study area.

Sedimentation risks were based on stream gradient. Areas with a higher proportion of low-gradient streams had the greatest sedimentation risk.

The methods and models developed here can be adopted for use elsewhere, but there is an urgent need to assess these predictions against field data. Several possible improvements in the models and the approach were identified, and these include the addition of both deterministic and stochastic components. The predictions also could be improved by using higher-resolution topographic, soils, and vegetation data. In the absence of specific field data, we believe that the models developed here can provide useful comparisons of surface erosion and sedimentation risks across a range of spatial scales. Such information is a necessary first step to guide future analysis and management activities.

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