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EFFECTS OF BASIN AND LAND-USE CHARACTERISTICS ON SUSPENDED-SEDIMENT YIELD IN THE HOUSATONIC RIVER BASIN, WESTERN MASSACHUSETTS

By Gardner C. Bent, Hydrologist, U.S. Geological Survey, 10 Bearfoot Road Northborough, MA 01532

Phone: (508) 490-5041, Fax: (508) 490-5068, E-mail: gbent@usgs.gov

Abstract: Suspended-sediment yield was measured for six subbasins and estimated for two subbasins with different basin and land-use characteristics in the Housatonic River Basin, western Massachusetts from April 1994 through March 1996. Measured yields ranged from 21 to 147 tons per year per square mile, and estimated yields were 82 and 395 tons per year per square mile. Five dams and associated reservoirs, although relatively small in size (each less than 0.20 square miles in surface area), decreased yield in one subbasin by trapping sediment. Suspended-sediment yields were moderately related to the combined percent areas of forested wetlands, non-forested wetlands, and water bodies (-0.50 correlation coefficient) and the percent area of sand and gravel (0.64 correlation coefficient). Yields were highly related to the percent areas of flood-plain alluvium (0.89 correlation coefficient), agricultural and open land (0.86 correlation coefficient), and soils with a high soil-erodibility factor (0.93 correlation coefficient). The five soils in the basin with a high soil-erodibility factor are all silt-loam soils and have a soil erodibility factor of 0.49. The causative effects of the silt-loam soils, flood-plain alluvium, and agricultural and open land on suspended-sediment yields cannot be clearly discerned because they are interrelated, inasmuch as that the silt-loam soils are related to the areas of flood-plain alluvium and of agricultural and open land, and the area of flood-plain alluvium is also related to the area of agricultural and open land.

INTRODUCTION

Studies of suspended sediments transported by a stream are important because of the potential effect sediments have on: recreation on rivers, lakes, and ponds (esthetics); aquatic habitat (for example, burial of fish eggs); water supplies; reservoirs, lakes, and ponds (sedimentation); the design water-treatment plants and reservoirs; stream morphology; and water-quality constituents. Trace metals, pesticides, and polychlorinated biphenyls (PCBs) have a strong affinity for and sorb to soils, sediments, and organic matter. The distribution of these sorbed constituents to sediments in a stream results from suspension, deposition, resuspension, redeposition, and so on, as the sediments move downstream in response to variations in flow.

Suspended sediment has been of particular concern in the Housatonic River because PCBs were detected in the river in the mid 1970s (Gay and Frimpter, 1985). In the late 1970s and early 1980s, several areas of the Housatonic River Basin were investigated for distribution and transport of PCBs. The results of these investigations are presented in Frink and others (1982), Gay and Frimpter (1985), Kulp and Gay (1986), and Kulp (1991). Additional information regarding the distribution and transport of PCBs in the Housatonic River Basin has also been published by other Federal and State agencies and consulting firms since the mid 1980s.

From April 1994 through March 1996, the U.S. Geological Survey (USGS) in cooperation with the Massachusetts Department of Environmental Management (MDEM), Division of Resource Conservation, Office of Water Resources studied the general hydrology of the Housatonic River Basin (Bent, 1999). The study included characterization of suspended-sediment concentrations, discharge, loads, and yields (Bent, 2000). One of the major objectives of this study was to provide a better understanding of suspended-sediment yields and of the relation of the yields to basin and land-use characteristics. This information will assist Federal, State, City, and Town water and land managers in determining the potential transport of PCBs in the river.

The purpose of this paper is to provide information from data collected from April 1994 through March 1996 on suspended-sediment yields at two sites on the Housatonic River, in the area adjacent to the Housatonic River between these two sites, and at five sites on tributaries to the Housatonic River. The relation between suspended-sediment yields and basin and land-use characteristics of the study area is also evaluated.

DESCRIPTION OF THE STUDY AREA

The Housatonic River drains 504 mi^2 (square miles) of western Massachusetts, 217 mi^2 of eastern New York, and 1,232 mi^2 of western Connecticut before discharging into Long Island Sound. The study area for this investigation (fig. 1) is confined to 504 mi^2 in Massachusetts, 26 mi^2 in New York, and 10 mi^2 in Connecticut for a total area of 540-mi^2 (fig.1). The central part of the study area is the lowland area of the Housatonic River Valley and is bordered by the Berkshire Mountains to the east and the Taconic Mountains to the west. Elevations in the study area range from about 635 ft (feet) above sea level at the Massachusetts-Connecticut State border to about 2,600 ft above sea level in the headwaters of the Housatonic River.

Land Use: The study area is mainly rural; it is approximately 67 percent forested, 12 percent agricultural/open, 10 percent urban, 7 percent wetland (forested and non-forested), 2 percent water bodies, and 2 percent barren. Basin and land-use characteristics thought to be most associated with suspended-sediment yield are listed in table 1 for subbasins in the Housatonic River Basin along with suspended-sediment yields determined and estimated for the study.

Geology: The lowlands of the Housatonic River Valley in the study area are underlain primarily by carbonate rocks, the Berkshire Mountains by gneissic rocks with small areas of quartzitic, and the Taconic Mountains by schistose rock (Norvitch and others, 1968, sheet 4). Sand and gravel and flood-plain alluvium deposits overlie till primarily in the upland stream valleys and in the Housatonic River Valley, and overlie about 11 and 5 percent of the study area (fig. 1), respectively.

Soils: Soils that are highly erodible in the study area include the Hadley, Limerick, Linlithgo, Saco, and Winooski silt-loam soils, which have a reported soil-erodibility factor (K-factor) (Wischmeier and Smith, 1978) of 0.49 (U.S. Department of Agriculture, Soil Conservation Service, 1970; 1988; 1989). The K-factor of the other 53 soils in the Massachusetts part of the study area averaged 0.23 and ranged from 0.10 to 0.43 (U.S. Department of Agriculture, Soil Conservation Service, 1988). The Hadley, Limerick, Linlithgo, Saco, and Winooski silt-loam soils tend to be adjacent to stream channels in the study area, especially in the southern part of the basin.

Climate: The mean annual temperature measured at climatological stations in the basin is between 43.4 and 45.6^{o}F (U.S. Department of Commerce, National Atmospheric and Oceanic Administration, 1983; 1994). Mean annual precipitation is between 43.9 and 48.8 in. (inches) and is distributed uniformly throughout the year (U.S. Department of Commerce, National Atmospheric and Oceanic Administration, 1983; 1994). Mean annual snowfall is about 71.4 in. (U.S. Department of Agriculture, Soil Conservation Service, 1988, p. 138). Overall, precipitation during the two-year study was about normal. Precipitation during the first six months of the study was from 1.5 in. below normal to 0.5 in. above normal (U.S. Department of Commerce, National Atmospheric and Oceanic Administration, 1994). During the next year (October 1, 1994 through September 30, 1995), precipitation was from 13.4 to 15.4 in. below normal, whereas precipitation during the next six months was 12.3 to 11.4 in. above normal for the next six months (U.S. Department of Commerce, National Atmospheric and Oceanic Administration, 1994; 1995; and 1996).

Hydrology: From the headwaters of the Housatonic River to the continuous streamflow-gaging station at the Housatonic River near Great Barrington, Mass. (01197500) (fig. 1), the Housatonic River flows 49.7 mi and has a mean channel slope of 16.5 ft/mi (Wandle and Lippert, 1984, p. 19). Five dams (Woods Pond, Columbia Mill, Willow Mill, Glendale, and Rising Pond) (fig. 1) and two smaller dams (not shown in fig. 1) are upstream from station 01197500 (Bickford and Dymon, 1990, p. 35; General Electric Company, 1991). Between station 01197500 and the Massachusetts-Connecticut border (fig. 1), the Housatonic River is about 21 mi long, has no dams, and has a mean channel slope of about 2 ft/mi.

FACTORS AFFECTING SUSPENDED-SEDIMENT YIELD

Hydrologic, physical, and land-use characteristics of the subbasins in the study area were evaluated to explain differences in suspended-sediment yield among the eight subbasins (table 1). Differences in precipitation were not evaluated because precipitation data were not available in all study subbasins; however, total streamflow was evaluated for the six subbasins in which yield was measured (table 1).

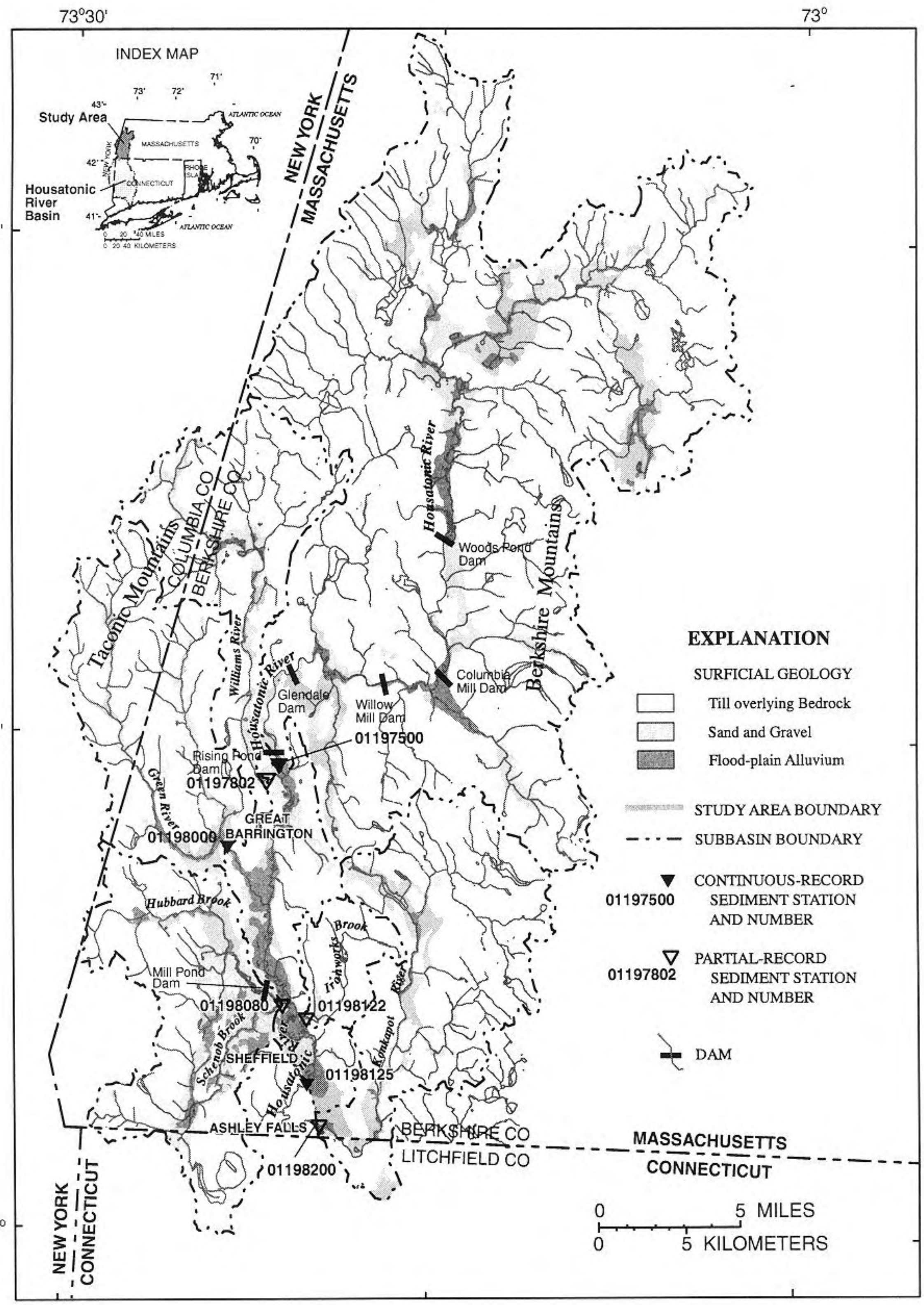

Figure 1. Location of study area; distribution of sand and gravel, flood-plain alluvium, and till; and continuous- and partial-record sediment stations in the Housatonic River basin, western Massachusetts.

Table 1. Suspended-sediment yield, streamflow, and selected basin and land-use characteristics for continuous- and partial-record sediment stations in the Housatonic River Basin, western Massachusetts, April 1994 through March 1996

[Percent values in table rounded to nearest tenth. **USGS station No.:** Locations shown in figure 1. **Total soils with soil erodibility factor of 0.49:** Total soils with soil erodibility factor of 0.49 equals sum of Hadley, Limerick, Linlithgo, Saco, and Winooski silt-loam soils. GRID, surface modeling package in ARCINFO; no., number, USGS, U.S. Geological Survey; $((ft^3/s)/yr)/mi^2$, cubic feet per second per year per square mile; mi^2, square miles; $(ton/yr)/mi^2$, ton per year per square mile,; --, no data.]

USGS station No. and station name	Drainage area (mi^2)	Suspended-sediment yield ($(ton/yr)/mi^2$)	Total of daily mean streamflow ($((ft^3/s)/yr)/mi^2$)	Sand and gravel area (percent)	Flood-plain alluvium area (percent)	Carbonate rocks area (percent)	Gneissic and quartz-itic rocks (percent)	Schistose rocks (percent)
01197500 Housatonic River near Great Barrington, Mass.	282	20.6	694	10.0	3.5	36.8	49.4	13.9
01197802 Williams River near Great Barrington, Mass.	43.2	35.3	621	10.0	2.3	46.6	3.6	49.8
01198000Green River near Great Barrington, Mass.	51.0	77.7	699	8.1	1.9	34.5	1.5	64.1
01198080 Schenob Brook at Sheffield, Mass.	50.0	81.6	--	18.1	8.0	69.4	.0	30.6
01198122 Ironworks Brook at Sheffield, Mass.	11.2	78.4	521	3.4	.1	56.8	37.8	5.6
Area adjacent to the Housatonic River between Great Barrington, Mass., and Ashley Falls, Mass.	27.6	395.3	--	20.2	24.6	55.1	36.3	8.6
01198125 Housatonic River near Ashley Falls, Mass.	465	58.4	711	11.1	4.9	42.5	33.5	24.0
01198200 Konkapot River at Ashley Falls, Mass.	61.1	146.6	655	11.2	1.9	43.4	50.8	5.9

USGS station No. and station name	Mean basin slope (GRID) (percent)	Agricul-tural and open area (percent)	Barren, area (percent)	Agricultural and open area and barren, rocks, mining area (percent)	Water bodies area (percent)	Forested wetlands area (percent)	Non-forested wetlands area (percent)	Total wetlands and water bodies area (percent)
01197500 Housatonic River near Great Barrington, Mass.	7.68	8.4	2.9	11.3	2.2	4.0	3.2	9.4
01197802 Williams River near Great Barrington, Mass.	8.74	14.4	2.2	16.6	.8	4.9	3.7	9.4
01198000 Green River near Great Barrington, Mass.	9.49	17.7	1.0	18.7	.5	1.7	.8	3.0
01198080 Schenob Brook at Sheffield, Mass.	8.06	14.9	2.0	16.9	.7	7.4	4.8	12.9
01198122 Ironworks Brook at Sheffield, Mass.	8.20	12.1	.6	12.7	1.7	4.1	4.0	9.7
Area adjacent to the Housatonic River between 01197500 and 01198125	7.83	26.4	2.8	29.2	.3	.5	3.7	4.5
01198125 Housatonic River near Ashley Falls, Mass.	8.04	11.8	2.5	14.3	1.6	4.0	3.2	8.8
01198200 Konkapot River at Ashley Falls, Mass.	6.78	2.1	1.0	13.1	1.5	1.5	3.7	6.7

Table 1. Suspended-sediment yield, streamflow, and selected basin and land-use characteristics for continuous- and partial-record sediment stations in the Housatonic River Basin, western Massachusetts, April 1994 through March 1996--*Continued*

USGS station No. and station name	Drainage area (mi^2)	Suspended-sediment yield ((ton/yr)/mi^2)	Silt loam area (percent)					Total soils with soil-erodibility factor of 0.49 (percent)
			Hadley	Limerick	Linlithgo	Saco	Winooski	
01197500 Housatonic River near Great Barrington, Mass.	282	20.6	0.2	1.2	0.0	0.6	0.3	2.3
01197802 Williams River near Great Barrington, Mass.	43.2	35.3	.1	.9	.0	.1	.1	1.2
01198000 Green River near Great Barrington, Mass.	51.0	77.7	.1	.6	.0	.0	.8	1.5
01198080 Schenob Brook at Sheffield, Mass.	50.0	81.6	.1	2.2	.0	2.7	.3	5.3
01198122 Ironworks Brook at Sheffield, Mass.	11.2	78.4	.0	.0	.0	.6	.1	.7
Area adjacent to the Housatonic River between Great Barrington, Mass., and Ashley Falls, Mass.	27.6	395.3	5.1	6.8	.0	2.9	5.2	20.0
01198125 Housatonic River near Ashley Falls, Mass.	465	58.4	.4	1.5	.0	.9	.7	3.5
01198200 Konkapot River at Ashley Falls, Mass.	61.1	146.6	.2	.9	.0	.4	.6	2.1

Dams: The suspended-sediment yield at the Housatonic River near Great Barrington (station 01197500), 21 (tons/yr)/mi^2 (tons per year per square mile), was the lowest, , of the eight subbasins. The basin and land-use characteristics of this subbasin are similar to those of other subbasins, but suspended-sediment yield is affected by the five largest dams upstream that trap sediments. Two of the dams, Woods Pond Dam and Rising Pond Dam (only 0.8 mi upstream of the station), have substantial impoundments. Woods Pond has a surface area of about 0.19 mi^2 and depths of about 15 ft (Frink and others, 1982); Rising Pond has a surface area of about 0.07 mi^2 and depths of about 15 ft. Frink and others (1982) report that sediment thicknesses ranged from 0.5 to 6 ft in Woods Pond and from 6 to 8 ft in Rising Pond. The other three dams -- Columbia Mill, Willow Mill, and Glendale – have smaller impoundments and likely trap sediment that otherwise would have been discharged downstream. The sediment transport capacity of a stream can increase immediately downstream of dams that trap sediments, so that the streambed and stream banks are often scoured in these reaches (Collier and others, 1996).

Soils: The area adjacent to the Housatonic River between Great Barrington and Ashley Falls (fig. 1 and table 1) had the largest percentage of soils with a high soil-erodibility factor (20 percent). This subbasin also had the highest percentage of the Hadley, Limerick, Saco, and Winooski silt-loam soils individually. The correlation coefficient (**r**) of percent area of soils with a high soil-erodibility to suspended-sediment yield was 0.93. Suspended-sediment yields are also highly correlated with the individual percent area of Hadley, Limerick, Saco, and Winooski silt-loam soils (**r** = 0.95, 0.89, 0.66, and 0.93, respectively). Although these correlation coefficients are mainly influenced by the area adjacent to the Housatonic River between Great Barrington and Ashley Falls (suspended-sediment yield 395 (tons/yr)/mi^2), they seem reasonable given visual observations of (1) streambank sloughing along the Housatonic River in areas of these silt-loam soils, and (2) the Housatonic River flowing out of its banks and across areas of these silt-loam soils several times during the 2-year study.

The Schenob Brook (station 01198080) subbasin has the second highest percent area of highly erodible soils, but had an estimated suspended-sediment yield of only 82 (tons/yr)/mi^2. This suspended-sediment yield is comparable to that at Green River (station 01198000), 78 (tons/yr)/mi^2 and third lowest percent area of highly erodible soils, and to that at Ironworks Brook (station 01198122), 78 (tons/yr)/mi^2 and lowest percent area of highly erodible soils (table 1). The low suspended-sediment yield from the Schenob Brook subbasin is likely due to (1) the effects of a dam on Hubbard Brook, which drains about 50 percent of the subbasin, and (2) backwater conditions at station 01198080 during moderate to high flows, which caused settling of suspended sediments.

Geology: To assess the possible effects of bedrock and surficial geology on suspended-sediment yield in the study area, the percent area of carbonate rocks, gneissic rocks with small areas of quartzitic rocks, schistose rocks, sand and gravel, and flood-plain alluvium were compared for the eight subbasins. Suspended-sediment yield was poorly correlated to the percent area of carbonate rocks (**r** = 0.28), the percent area of gneissic rocks with small areas of quartzitic rocks (**r** = 0.23), and the percent area of schistose rocks (**r** = -0.38).

Percent area of sand and gravel was moderately related to suspended-sediment yield (**r** = 0.64) and percent area of flood-plain alluvium was highly related to suspended-sediment yield (**r** = 0.89). This high correlation is likely the result of the high percent area of flood-plain alluvium, 24.6 percent, and high suspended-sediment, 395 (tons/yr)/mi^2, for the area adjacent to the Housatonic River between Great Barrington and Ashley Falls. Flood-plain alluvium is highly related (cross correlation) to highly erodible soils, as flood-plain alluvium is mainly mapped (classified) as Hadley, Limerick, Linlithgo, Saco, or Winooski silt-loam soils (U.S. Department of Agriculture, Soil Conservation Service, 1970; 1988; 1989). These soils occur in the flood plain of the Housatonic River and some tributaries. The cross-correlation between the percent area of flood-plain alluvium and the percent area of highly erodible soils for the eight subbasins has a correlation coefficient of slightly less than 1.00.

Land Use: Land uses evaluated for their potential to affect the suspended-sediment yields were the percent area of (1) agricultural and open land, (2) barren land, (3) water bodies, (4) forested wetlands, and (5) non-forested wetlands. Areas of agricultural, open, and barren land likely have a higher suspended-sediment yield (relative to other areas) because these areas generally have a greater percentage of bare soil or greater soil disturbances than other areas. Water bodies, forested wetland, and non-forested wetlands act as sediment traps, and thus tend to decrease suspended-sediment yield.

The percentage of agricultural and open land was highest in the area adjacent to the Housatonic River between Great Barrington and Ashley Falls, where suspended-sediment yield was also highest among the eight subbasins (table 1). The suspended-sediment yield was highly correlated to the percent area of agricultural and open land (**r** = 0.86). Again this is likely a result of the fact that these are the predominant land use types in the area adjacent to the Housatonic River between Great Barrington and Ashley Falls. The agricultural and open land is generally coincident with the highly erodible silt-loam soils, which are mainly found in the area adjacent to the Housatonic River between Great Barrington and Ashley Falls (Bent, 2000). The Housatonic River was out of its banks and flowing across agricultural areas on bends in the river in the area adjacent to the Housatonic River between Great Barrington and Ashley Falls several times during the 2-year study. Four of the five highly erodible silt-loam soils, excluding the Saco silt-loam soil, are reported to be either fairly well or well suited for cultivation for row crops and small grains or for grasses and legumes for hay and pasture (U.S. Department of Agriculture, Soil Conservation Service, 1970; 1988; 1989). Thus, cultivation and exposure of some of the silt-loam soils for agricultural purposes during the dormant (nongrowing) season may provide a source of erodible material.

The percent area of agricultural and open land is highly related (cross correlation) to the percent area of highly erodible silt-loam soils (**r** = 0.85). The percent area of agricultural and open land is also highly related (cross correlation) to the percent area of flood-plain alluvium (**r** = 0.83). These high cross correlations, as well as the one between flood-plain alluvium and highly erodible silt-loam soils, make it difficult to discern causative effects of these three characteristics on suspended-sediment yield. Additionally, the small number of subbasins (eight) makes it difficult to separate these individual basin or land-use characteristics in order explain their causative effects on suspended-sediment yields in the basin.

The percent area of barren land differed slightly among the subbasins (0.6 to 2.9 percent, table 1), but because this type of land use represents such a small percentage of the total area, it was only poorly correlated to suspended-sediment yield (**r** = 0.20). The combined percent area of water bodies, forested wetlands, and nonforested wetlands differed among the subbasins (3.0 to 12.9 percent, table 1) and was moderately correlated to suspended-sediment yield (**r** = -0.50). Water bodies, forested wetlands, and nonforested wetlands likely are efficient in trapping sediment, like the five largest dams on the main stem of the Housatonic River upstream from station 01197500.

Other Factors: Differences in total streamflow (table 1) and in streamflow characteristics (flow durations) were evaluated for their effect on differences in suspended-sediment yield among the eight subbasins. Although total streamflows differed among the six subbasins where suspended-sediment yield was measured (table 1), total streamflow was poorly correlated with suspended-sediment yield (**r** = -0.14). Comparisons of flow-duration curves on a per-square-mile basis for the 2-year study showed slight differences in streamflow characteristics among subbasins, but the differences did not seem to be related to differences in suspended-sediment yield (Bent, 2000). Mean basin slope was evaluated for each of the eight subbasins because steeper slope increases potential soil erosion (Wischmeier and Smith, 1978). Mean basin slopes differed slightly among the subbasins (6.8 to 9.5 percent, table 1) and were poorly correlated with differences in suspended-sediment yield (**r** = -0.27).

SUMMARY AND CONCLUSION

Several basin and land-use characteristics thought to affect suspended-sediment discharge in the subbasins were compared to suspended-sediment yields. The presence of dams decreased the suspended-sediment yield, the percent area of highly erodible soils (Hadley, Limerick, Linlithgo, Saco, and Winooski silt-loam soils) increased yield, the percent area of flood-plain alluvium increased yield, and the percent area of agricultural and open land increased yield. The flood-plain alluvium areas are mainly classified as silt-loam soils, and most of these soils are well suited for agricultural purposes such as row crops, hay, or pasture. Because of this cross correlation and the small number of subbasins, the independent effects of flood-plain alluvium, highly erodible soils, and agricultural and open land on suspended-sediment yield cannot be clearly distinguished.

The relations between suspended-sediment yield and basin and land-use characteristics identified during this study will help water and land managers better understand the potential transport of PCBs and non-point-source loadings in the Housatonic River Basin. These results may be transferable to other basins in the northeastern United States, which have similar basin and land-use characteristics. Because suspended-sediment yield increases with increasing area of soils with a high-soil erodibility factor, water and land managers throughout the United States could use this information to better manage areas of highly erodible soils to control soil erosion to fluvial systems.

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SEDIMENT SUPPLY FROM MOUNT ST. HELENS – 20 YEARS LATER

Jeffrey B. Bradley, Chief Executive Officer, WEST Consultants, Inc., Bellevue, WA; Thomas R. Grindeland, Senior Project Engineer, WEST Consultants, Inc., Bellevue, WA; Hans R. Hadley, Geomorphologist, WEST Consultants, Inc., Bellevue, WA.

12509 Bel-Red Road, Suite 100; Bellevue, WA 98005
Telephone (425) 646-8806; Fax (425) 646-0570
jbradley@westconsultants.com; tgrindeland@westconsultants.com; hhadley@westconsultants.com

Abstract
The May 18, 1980 eruption of Mount St. Helens deposited a debris avalanche of over 3.8 billion cubic yards of silt, sand, gravel, and debris in the upper 17 miles of the North Fork Toutle River valley and another 50 to 60 million cubic yards in the upper portion of the South Fork Toutle River valley. The eruption devastated approximately 150 square miles of evergreen forests, lakes, and wildlife within this area. Mudflows, triggered by the eruption, carried large volumes of sediment from the debris avalanche into the Toutle-Cowlitz-Columbia River system. The resulting sediment deposition caused widespread flooding along the Toutle and Cowlitz Rivers and blockage of the Columbia River Navigation Channel.

Sediment continues to erode from the debris avalanche and is transported down the North Fork Toutle River. The majority of this material now deposits behind the Sediment Retention Structure (SRS), which was completed in 1989. A period of about 20 years has elapsed during which hydrologic recovery of effected watershed areas and watercourses may have partially occurred. A reassessment of the eruption-influenced sediment transport conditions was conducted to quantify the extent of hydrologic recovery and estimate the future supply of sediment to downstream areas. A comparison was made of sediment yield estimates developed for the current study and those developed shortly after the eruption.

INTRODUCTION

In May 1980 Mount St. Helens erupted, removing the upper 1,324 feet of the mountain and depositing approximately 3.7 billion cubic yards of material over an area of 230 square miles (COE, 1999). The resultant debris avalanche buried the upper 17 miles of the North Fork Toutle River to an average depth of 150 feet. Mudflows carried a significant amount of this material downstream into the Toutle, Cowlitz, and Columbia Rivers.

The Mount St. Helens (MSH) Project was formulated to control the projected movement of sediment from the debris avalanche along the North Fork Toutle River and to maintain an optimized level of flood protection downstream along the lower Cowlitz River.
A major element of the MSH Project is the Sediment Retention Structure (SRS) located at RM 13.2 of the Toutle River on the North Fork of the Toutle River. The debris avalanche along the North Fork Toutle River has been evolving since 1980 and differs significantly from when the original SRS design was completed. Sediment deposits upstream of the SRS reached the elevation of the SRS spillway crest between November 1997 and March 1998. The uppermost row of outlet pipes on the SRS was closed in April 1998. An analysis was made to develop improved estimates of the future sediment supply from the debris avalanche.

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The Toutle River Basin primarily drains the northwest and southwest slopes of Mount St. Helens and has a total drainage area of 512 square miles at the confluence with the Cowlitz River. The Toutle River is comprised of three primary tributaries, which include the Green River, North Fork Toutle River, and South Fork Toutle River. They flow roughly parallel to each other from east to west. The Green River flows into the North Fork Toutle River, and the North Fork and South Fork Toutle converge to form the Toutle River approximately 17 miles upstream from the confluence with the Cowlitz River. The North Fork Toutle River is the largest tributary with a drainage area of 172 square miles at the confluence with the Green River. Along the east edge of the Toutle River basin is the Spirit Lake basin. The debris avalanche from the 1980 eruption blocked the natural outlet of Spirit Lake creating a closed basin. In 1985 the Corps of Engineers completed a tunnel from Spirit Lake to the South Fork Coldwater Creek, a tributary to the North Fork Toutle River, in order to regulate the level of the lake.

The May 18, 1980 eruption of Mount St. Helens had the greatest impact on the North Fork Toutle River, as the majority of the debris avalanche deposited in the North Fork Toutle River valley. The Green River and South Fork Toutle River also experienced impacts from the eruption, which consisted primarily of mudflows. The lateral blast from the eruption altered the hydrologic characteristics of all three basins. Some of these altered characteristics included an increase in the magnitude of peak discharge and a reduction in travel time due to the increase in overland flow and reduced resistance in the channel (USACE, 1983).

WATERSHED RECOVERY

Since the 1980 eruption, the Toutle River basin has adjusted itself in various ways. These adjustments include recovery of the watershed, vegetation, and development of the channel system. These ecological and morphological changes alter the hydrologic, hydraulic and sediment transport characteristics of the basin. Available data and information were evaluated to assess the extent and rate of recovery. The objective of this portion of the analysis was to estimate the long-term trend of sediment supply from the debris avalanche to the N.F. Toutle River. Elements of the analysis included evaluation of historic cross section data, channel profile comparisons, historic aerial photography comparison, and computer-based evaluations of digital elevation models of the North Fork Toutle River basin for different time periods.

Historic cross section survey information was evaluated to identify trends in channel cross section development such as channel widening and channel degradation. The rate at which channel cross section development has occurred and whether or not the channel has stabilized was assessed. Surveyed cross sections (USGS, 2000) along the North Fork Toutle, South Fork Toutle, and Toutle Rivers were utilized in the analysis. Cross sections have been repeatedly surveyed at more than 100 locations along these three rivers. Cross section surveys began as early as 1980, and have been resurveyed periodically up through1999. Typical results are shown in Figure 1. Cross sections on the North Fork Toutle, South Fork Toutle, and Toutle Rivers show a general trend of increased cross sectional area since the eruption. A majority of the cross sections have had a significant amount of streambank erosion and channel degradation. As a result, many locations show an increase in channel width and lowering of the thalweg elevation

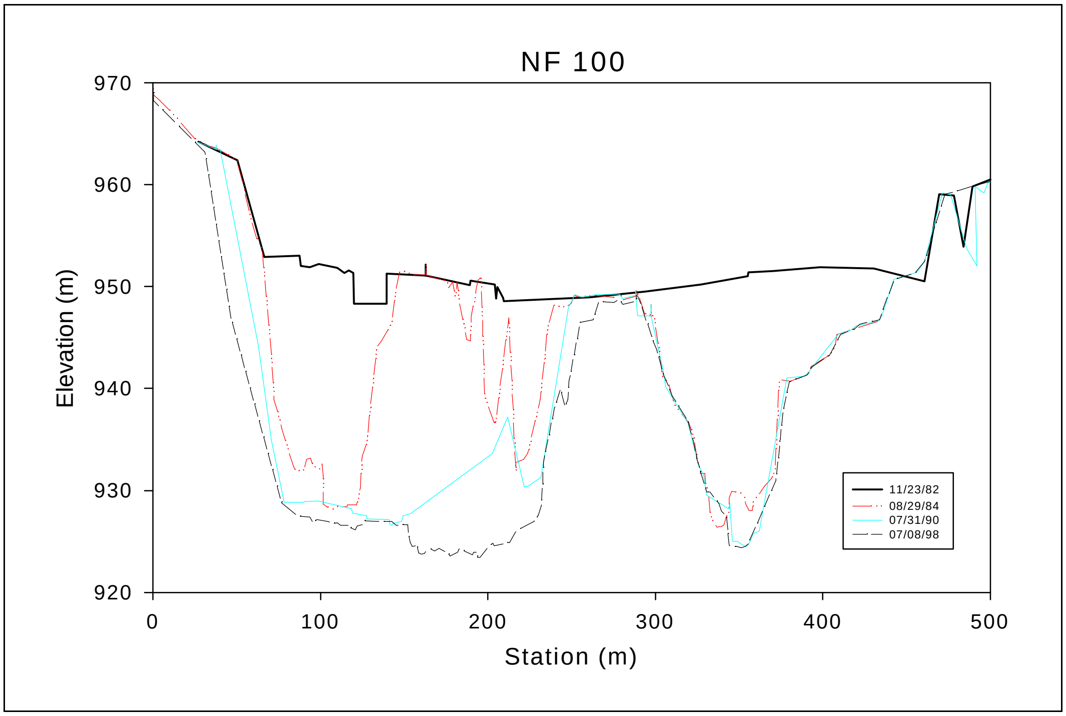

Figure 1 Typical cross section data for the North Fork Toutle River.

A profile analysis was performed for the North Fork Toutle River, Castle Creek, and Coldwater Creek in order to identify changes in the channel slope and thalweg elevation. Digital Elevation Models (DEM' s) (CENWP, 2000) of the North Fork Toutle River above the SRS for the years 1987 and 1999 were used in the analysis. Profiles were extracted from the two DEM' s along the path of the 1999 channel thalweg starting at the upstream end of Loowit Creek down to the SRS along the North Fork Toutle River.

The profile analysis shows that the North Fork Toutle River, Coldwater Creek, and Castle Creek have all degraded between 1987 and 1999, except between the N1 debris dam and the SRS. The most degradation on the North Fork Toutle River occurred near the Coldwater / Castle Creek confluence, and was as much as 40 feet. Up to 40 feet of degradation was also observed on Coldwater Creek, and up to 60 feet of degradation was observed on Castle Creek. More than 100 feet of aggradation was observed upstream of the SRS. Even though the channels degraded significantly, the overall slope of the channels changed very little between 1987 and 1999, except near the SRS.

A plan form analysis was made to observe the condition of sediment erosion and deposition upstream of the SRS. The analysis was made to observe and document geomorphic changes in the river valley over time and evaluate how the occurrence and severity of these channel changes has progressed since the eruption. To perform the analysis, historic aerial photography for the years 1980, 1982, 1983, 1984, 1985, 1987, and 1999 were compared. Analysis of the historical aerial photography indicates that the basin is beginning to recover. The majority of the channels were historically braided since the eruption; however, in many places the density of braided channels has declined, and at a few locations a single thread channel has formed. This would indicate that these channels have become more stable. The emergence of vegetation seen in the 1999 aerial photography adjacent to many of the channels provide additional evidence that hydrologic recovery is beginning to occur. However, the floodplains remain virtually unvegetated, indicating a continued lack of channel stability. The density and aerial extent of vegetation generally increases from upstream to downstream.

The majority of the debris avalanche lacks any significant vegetation while nearer to the SRS there are trees growing on the hillslopes, the floodplain fringe, and even portions of the floodplain. This is likely due to several factors including; lack of sufficient soils and soil moisture to promote vegetative growth on the debris avalanche, reduced impacts from the eruptive blast in the downstream direction, and replanting of private forest land outside of the volcanic monument. Additional evidence of hydrologic recovery can be seen by the stability and extensive vegetation of the delta formation in Coldwater Lake. This would indicate that South Coldwater Creek has started to stabilize

While there are some indications of hydrologic recovery, the aerial photograph analysis also provides clear evidence that recovery is very slow. Hydrologic recovery to pre-eruption conditions has not occurred. The channels continue to shift and widen, and large-scale degradation and bank erosion is still occurring in many areas, as evidenced by the changes in channel plan form and the massive volume of sediment trapped behind the SRS since its completion in 1987.

SEDIMENT SOURCES

Digital Elevation Models (DEM’ s) developed from aerial photography for the years 1987 (pre-SRS) and 1999 in the form of Triangulated Irregular Networks (TIN’ s) were analyzed to estimate the total erosion on the debris avalanche upstream of the SRS as well as the total deposition behind the SRS over the involved time period. Erosion estimates were defined for each of the primary sediment sources (sub-areas) on the debris avalanche. These sub-areas categorized as Elk Rock, Coldwater Creek, Castle Creek and Loowit. Deposition estimates were developed for the North Fork Toutle River between the SRS and N-1 Debris Retention Structure.

The two TIN’ s were converted to overlapping grids with 10 foot by 10 foot cells. The grids were clipped to contain only the data pertinent to the analysis (only the locations of deposition or erosion as seen in the 1999 aerial photography). An elevation difference grid was developed by subtracting the 1987 grid from the 1999 grid showing the location and magnitude of the changes in elevation that occurred between 1987 and 1999. An extensive amount of deposition has occurred between the SRS and N-1 Debris Retaining Structure. In locations nearest the SRS deposition depths exceed 100 feet. The majority of the debris avalanche erosion is associated with the N.F. Toutle River channel upstream of Elk Rock. The most extensive erosion typically occurs along the outside of channel bends where bank erosion has caused elevations changes of up to 180 feet. This suggests that bank erosion has played a major role in the contribution of sediment to the N.F. Toutle River. Site visit observations confirm this conclusion.

The total erosion from the sediment source sub-areas were compared to the sediment deposition volume measured between the SRS and N-1 Debris Retaining Structure and the volume of sediment passing the Kid Valley gage (assumed to be the same as the sediment passing the SRS) to evaluate data consistency. The total amount of erosion was measured to be 88 million cubic yards (MCY). When bulked by 16 percent to account for the reduction in density associated with deposition, the total erosion is estimated to be 102.1 MCY. Total deposition measured between the SRS and N-1 is 90.6 MCY. Suspended sediment passing the Kid Valley gage was estimated to be 11 MCY. It is noted that the Green River enters the North Fork Toutle River above the Kid Valley gage and would account for a small portion of the 11 MCY measured at the gage. Between 1988 and 1998 the Green River was estimated to contribute approximately 0.6 MCY to the North Fork Toutle above Kid Valley. This estimate was based on suspended sediment discharge measurements made from 1988 and 1994 and correlation with the Tower Road suspended sediment record.

The total N.F. Toutle River suspended sediment load that passed the SRS (above the Green River) for water years 1988 through 1998 was estimated to be 10.4 MCY. The deposition behind the SRS plus the estimate of suspended sediment that passed through the SRS totals 101 MCY. This volume is approximately 1 percent less than the total erosion volume estimated for the sediment source sub-areas. The most significant source of sediment has been the Elk Rock and Loowit sub-areas, which have a combined total of approximately 78 percent of the total debris avalanche erosion since 1987. Castle Creek sub-area and Coldwater Creek sub-area make up approximately 12.6 percent and 9.5 percent of the total debris avalanche erosion, respectively.

SEDIMENT YIELD

The average annual sediment yield of the debris avalanche will be influenced by the hydrologic and geomorphic recovery of the watershed and its stream channels. The trend and rate of recovery could be expected to significantly affect the accuracy of the average annual sediment yield estimate. Measured sediment yields at the Toutle River at Tower Road Gage and deposition behind the SRS were used to evaluate existing trends in sediment yield.

As seen in Figure 2, annual sediment yields measured at Tower Road were significantly large during the early 1980' s, but then reduced fairly rapidly throughout the late 1980' s and early 1990' s. This would indicate that recovery in the watershed was causing a reduction in sediment supply to downstream areas. However, this time period was also a period of below average runoff. Total annual runoff was approximately 15 percent below normal for the period 1985 to 1995. A significant increase in sediment yield occurred during the 1996 and 1997 water years as total annual runoff was approximately 45 percent above normal. This would indicate that sediment yield from the watershed is highly dependent upon the hydrology. Variability in the hydrologic cycle would tend to mask trends in the reduction of sediment yield. However, the fact that the sediment yields measured for 1996 and 1997, the largest water years of record, were less than those measured in 1982 and 1983 would indicate that some recovery has taken place. However, the sediment yield in 1996 was nearly the same as occurred in 1984, which further indicate the dependence of sediment yield on the involved hydrology. To account for the dependence between sediment yield and hydrology, the annual sediment yield was divided by the annual runoff to determine the yield of sediment per unit volume of runoff or average sediment concentration. As seen in Figure 3, the yield of sediment in 1996 was approximately 11.7 tons per acre-ft of runoff while the yield in 1984 was approximately 15.3 tons per acre-ft of runoff, a reduction of approximately 24 percent, providing further evidence of watershed recovery.

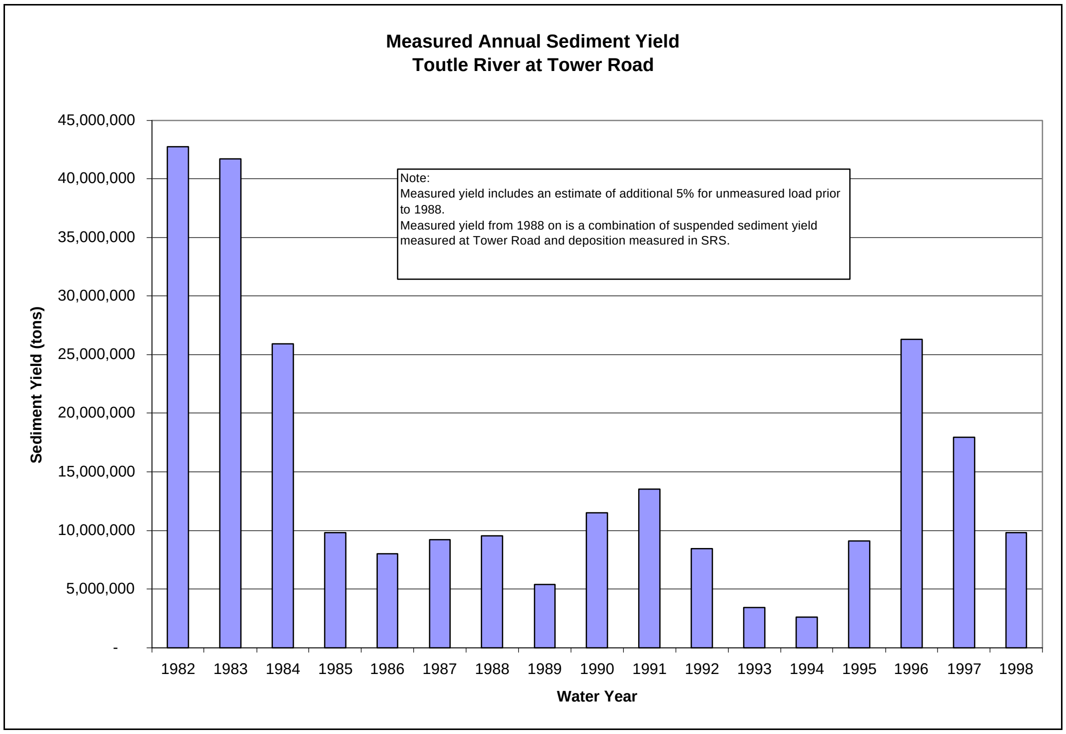

Figure 2. Measured annual sediment yield from Toutle River at Tower Road.

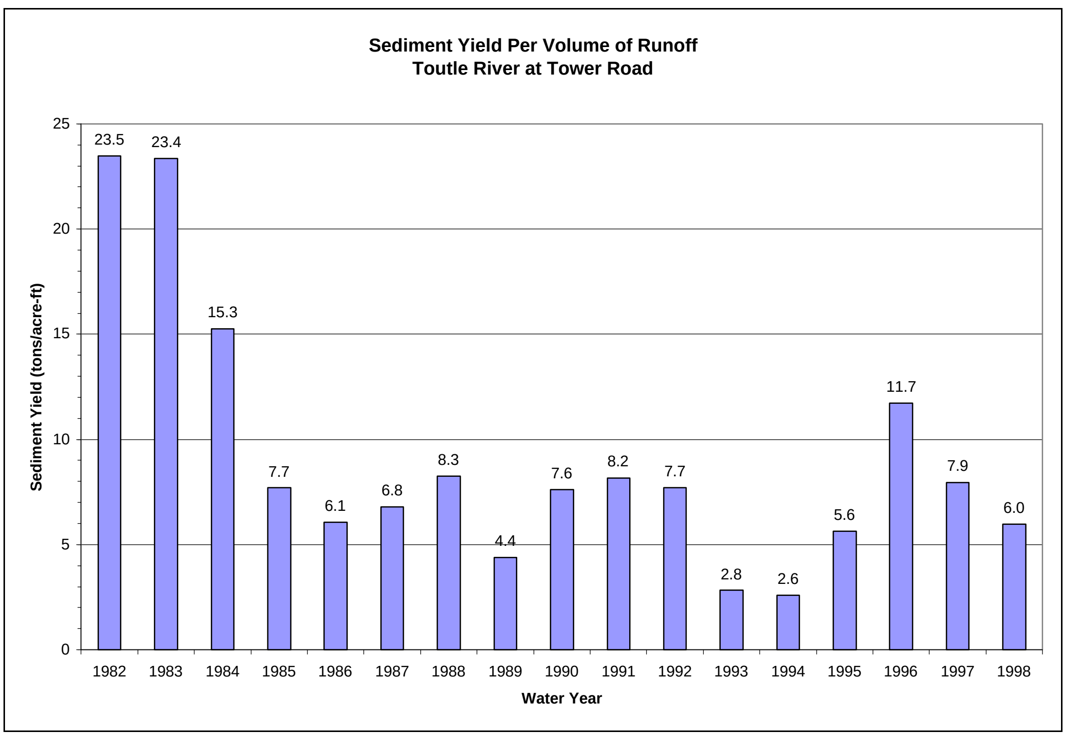

Figure 3. Annual sediment yield per unit volume of runoff.

The average annual sediment concentrations were accumulated on an annual basis to determine if a trend of decreasing average sediment concentration over time is occurring in the system (see

Figure 4). A trend line was fit to the cumulative concentration data to develop a sediment concentration decay curve. This decay curve was then extended to the end of the MSH Project planning period to estimate the reduction in average annual sediment concentration that might be expected to occur by the year 2035. The predicted annual sediment concentrations were multiplied by the average annual runoff volume to estimate the annual yield of sediment from the debris avalanche. Figure 5 shows that over the next 35 years, the annual sediment yield would reduce by 2.3 million cubic yards or approximately 36 percent. It is noted that this estimate is based upon average hydrologic conditions that have occurred between 1931 and 1998. Future hydrologic conditions that differ from those considered in the analysis as well as significant vegetative recovery in the watershed or future volcanic eruptions would significantly alter the estimated rate of reduction in sediment yield.

CONCLUSIONS

Total sediment yield from 1999 to the end of the planning period is estimated to be approximately 185 million cubic yards. When this is added to the approximately 264 million cubic yards that has eroded since 1982, an estimated 449 million cubic yards will have been eroded from the debris avalanche by the year 2035. This is approximately 55 and 31 percent less than estimates of 1 billion and 650 million cubic yards made previously (USACE, 1984).

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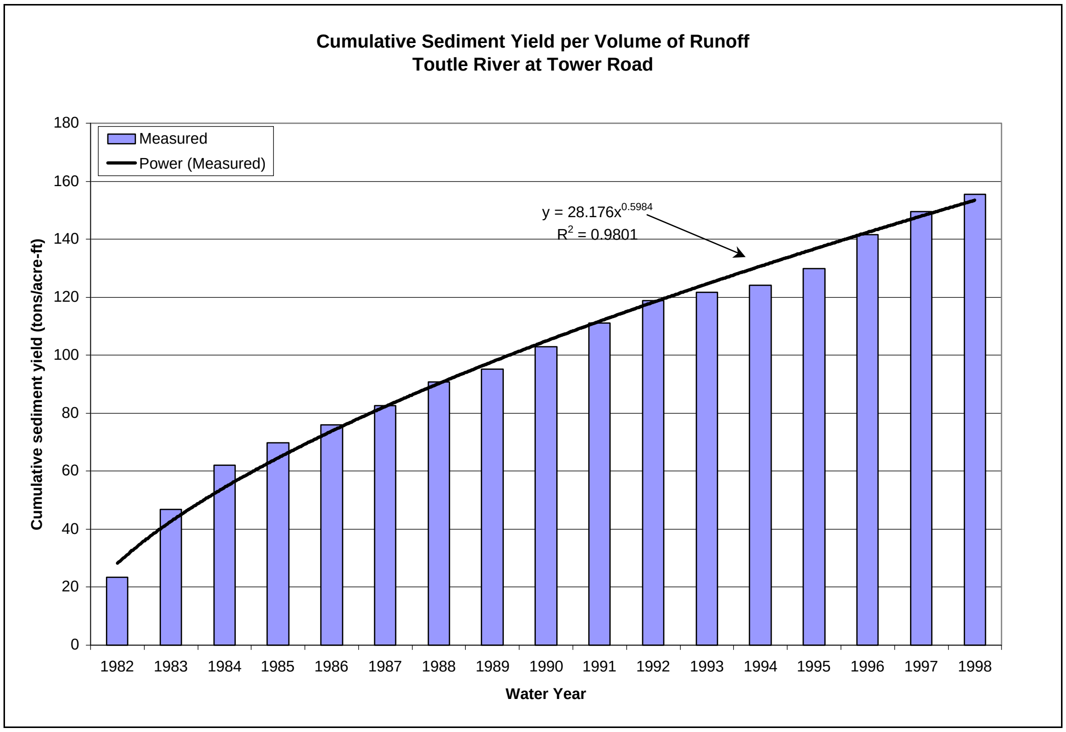

Figure 4. Cumulative sediment yield per unit volume of runoff.

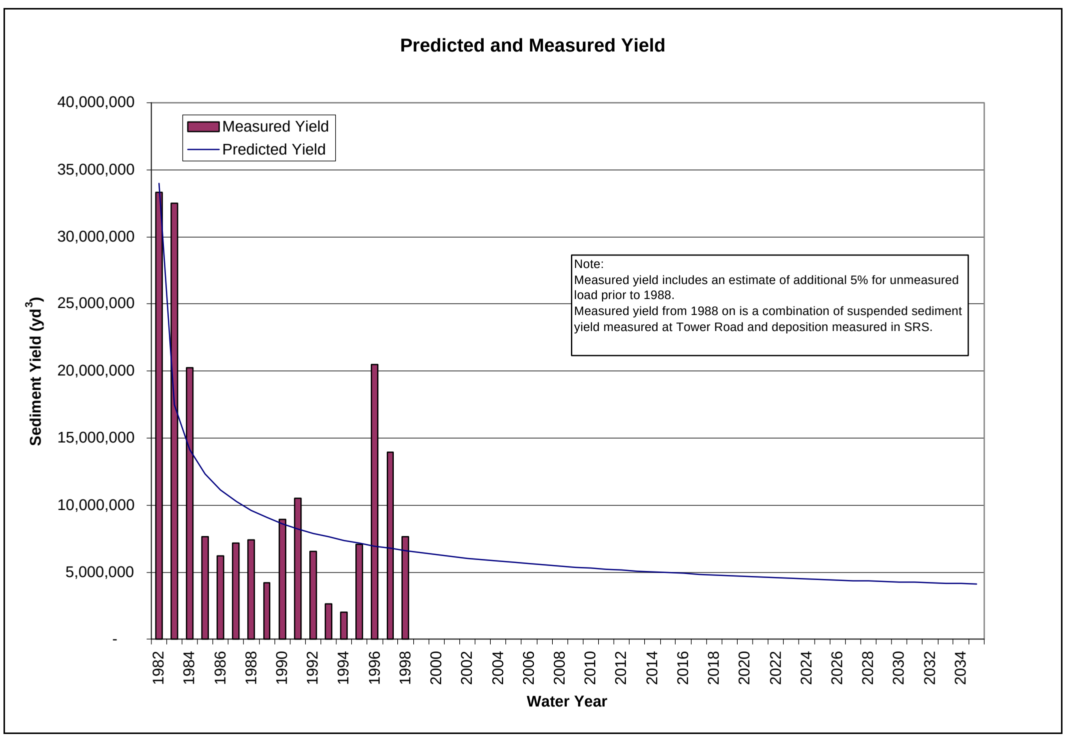

Figure 5. Predicted and measured sediment yield.

SEDIMENT TRANSPORT MODELING FOR THREE GREAT LAKES WATERSHEDS

R.B. Nairn[1], Baird & Associates, Oakville, ON; J.P. Selegean[2], U.S. Army Corps of Engineers, Detroit, MI

1 – rnairn@baird.com; 2 – James.P.Selegean@usace.army.mil

Abstract: The US Army Corps of Engineers has been authorized to develop sediment transport models for Great Lakes tributaries to provide tools that link land use management practices to sediment yield, bed and bank erosion sediment transport and sedimentation (including areas of dredging). This paper describes the development and application of model systems for three watersheds in Minnesota, Wisconsin and Michigan. The three watersheds feature a variety of geologic conditions including glacial till and alluvial outwash. The land use management issues include forestry practices leading to flashier flows and greater bed, bank and valley wall erosion; urbanization also leading to flashier flows and increased bank erosion: and intensive agriculture leading to high silt and sand yields. A variety of model components have been applied to the three watersheds depending on the local requirements and components already in place including: DHI' s UHM (hydrologic component), MIKE11 and MIKE21 (1D and 2D hydrodynamic models); a customized bank and bed erosion model, HSPF (and the GenScn interface), HECRAS/HEC6, SAM, AGNPS and BRANCH1D. ArcView GIS was a key component of all three systems. The paper presents some examples of the application of these systems to understand how different sediment sources contribute to sedimentation in Areas of Concern and Federal Navigation projects.

INTRODUCTION

The US Army Corps of Engineers has been authorized to develop sediment transport models for Great Lakes tributaries under Section 516e of the Water Resources Development Act of 1996. The purpose of the models is to better define and manage the influence of land use practices and sediment loading to Federal Navigation Channels and Areas of Concern. This paper presents an overview of the development, application and testing of sediment transport modeling systems for the Nemadji River watershed that flows in Lake Superior, the Menomonee River watershed that flows into Lake Michigan in Milwaukee and the Saginaw River watershed flowing into Lake Huron.

NEMADJI RIVER WATERSHED

Description of the Watershed: The Nemadji River comprises a 433 square mile watershed. The river flows to Superior Bay at Superior, Wisconsin. The watershed covers three counties (Carleton, Pine and Douglas) in the states of Minnesota and Wisconsin. The region is comprised of roughly 69% forested areas, 18% cropland and pastures, and 11% wetlands and lakes. Roughly one third of the Nemadji River Basin is comprised of glacial till, and glacial lake-laid clay soils commonly known as Red Clay. This Red Clay is considered to be highly erodible, and is prone to extensive mass wasting and bank slumping. Due to high turbidity and sediment loads, an estimated 33,000 tonnes of Nemadji River sediments are dredged annually by US Army Corps of Engineers (USACE) Detroit District. A much higher sediment load flows into and is deposited in Lake Superior. Comprehensive investigations completed for the Nemadji River Basin Project determined that 98% of the sediment yield from the Nemadji Basin is derived from the erosion of the valley walls (NRCS, US FS, 1998). In addition, the sediment delivery ratio (SDR) was found to be almost 98% – indicating that almost all of the sediment that is eroded along the Nemadji Basin tributaries is transported to the mouth of the river. Forestry and timber harvesting practices may have had an impact upon this erosion. The turbidity in the river and dredging in the mouth have an impact on fishing and other recreational uses. A tool to assess the implication of land use planning and the merits of remedial measures is required.

Description of the Model System: The modeling system consists of GIS-database components, a hydrologic model, a river hydrodynamic model and a sediment transport model as shown in Figure 1. ArcView GIS is used to create geographic data, analyze the data, and prepare the data for input to the hydrologic model and the hydrodynamic model. The hydrologic model generates time-series runoff for input to the river hydrodynamic model. The hydrodynamic model provided the hydrodynamic parameters such as water velocity, depth and discharge to the sediment transport model. Lastly, the sediment transport model predicts sediment erosion and deposition in the river. Based on a review of a range of options for the three key model components, the Danish Hydraulic Institute (DHI) MIKE system was chosen for the hydrological model and hydrodynamic model. A customized sediment transport model was developed because the DHI MIKE sediment transport module cannot account for the riverbed and bank erosion in consolidated glacial sediment, the source of almost all the sediment yield.

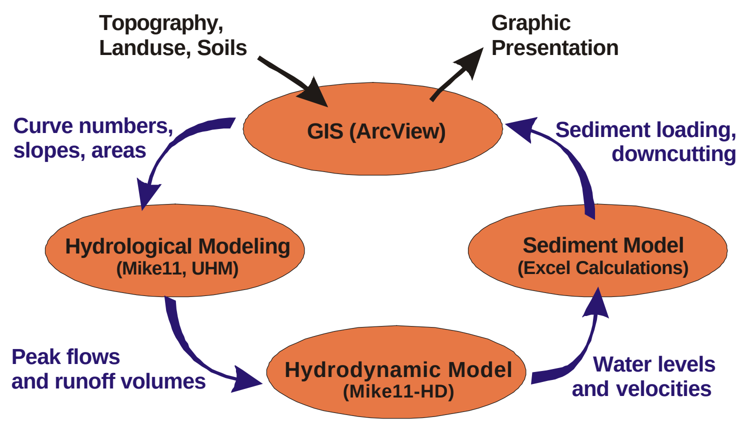

Figure 1. Nemadji Sediment Transport Model System

Calibration and Verification of the Model: Precipitation, discharge, and stage (water level) data are needed for model calibration and verification. The principal input to the model for calibrations and test is hourly precipitation data. USGS stream gage data was used to test the rainfall, runoff and hydrodynamic components of the model.

The Rainfall/Runoff model calibrations are carried out to verify the aggregated Curve Number (CN) calculated in the GIS component and the estimated lag time. The CN value is a main parameter influencing the excess rainfall which should be equal to total runoff in a single storm event. The lag time is another key parameter that distributes the excess rainfall in time and influences the duration of the runoff period. Both parameters are adjusted according to calibration results.

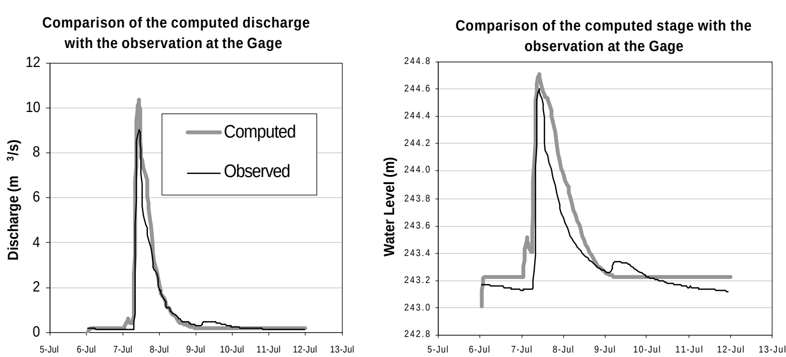

Figure 2. Verification of the Rainfall/Runoff Component

The purpose of hydrodynamic model calibration is to check whether the model parameters such as bottom friction are correct. Bottom friction controls the flow velocity and water level. Since there is no velocity data available, calibration is performed by comparison to measured water levels. An example of a verification test is presented in Figure 2. This figure shows a comparison of modeled water level and discharge with the observed levels at the gage. The calculated results match well with the recorded data. The system has been applied to the Deer Creek and Skunk Creek subwatersheds.

The bed and bank erosion component of the model links bed shear stresses predicted with the hydrodynamic model to erosion through relationships developed from physical model erodibility tests. The critical shear stress and the coefficient relating the rate of erosion to shear stress are further refined through comparison to sediment load data for a period where precipitation and water level gage data are available. Once eroded, entrained clay, silt and sand are then transported using the sediment transport component of the model.

Example Application of the System: Open lands are defined as meadow, pasture or 0-15 year old timber growth. An open land coverage of 65% was identified as a critical value by Verry et al, (1983). The open lands in the Deer Creek subwatershed currently cover about 36% of the total watershed. To investigate the effect of altering this value to a critical level on the hydrodynamics and sediment transport, some of the forest lands of the Deer Creek subwatershed are changed into "open lands" resulting in the 65% open land for this subwatershed.

Curve Numbers are updated using the customized GIS functionality to reflect this change and input into the MIKE11 model and the sediment transport model calculates total sediment load. The scenario of 65% open land area increases the peak discharge by 6% and the total runoff by 9%. However the peak sediment concentration (mg/l) is increased only by about 2.3% while total sediment load (in tonnes) increases by 6.5%. These results are considered preliminary until there is more sediment load data to test the system.

Key Findings: It was a great benefit to rely on an earlier study that defined the key sources of sediment. This provided focus for the selection and implementation of model components, in this case to specifically simulate bed and bank erosion. While almost all rivers have some sediment load measurements, few will have sufficient existing information to both calibrate and verify a model. There are no available and widely used models for predicting bed and bank erosion in heavily consolidated clay sediments such as glacial till and glaciolacustrine clay (a customized approach was developed for this project). Based on feedback from training workshops held with local community representatives, ArcView GIS proved to be an excellent framework for the system due to the familiarity with the interface and functionality.

MENOMONEE RIVER WATERSHED

Description of the Watershed: The Menomonee river watershed in southeastern Wisconsin has a drainage area of about 141 square miles. The watershed lies in four counties, namely Milwaukee, Waukesha, Washington, and Ozaukee. About 70% of the watershed had been urbanized as of 1995. The undeveloped area lies largely in the headwaters in Washington and Ozaukee Counties. The Menomonee River enters the Milwaukee River in the inner harbor of Milwaukee about 1 mile upstream from Lake Michigan. The Menomonee River has been suspected of contributing a significant amount of sediment to the Milwaukee harbor. Because the upper portion of the Menomonee River watershed is predominately agricultural land, undergoing active development, increased sediment loading from bed and bank erosion construction sites and urban areas is of concern.

Description of the Model System: The selected hydrologic model for this system was the USEPA Hydrologic Simulation Program Fortran (HSPF). Since HSPF had already been setup for the Menomonee watershed, it was a logical choice. HSPF allows for the simulation of flows, sediment yield from terrestrial areas and sediment transport in channels. Because the HEC-6 model is being used for detailed channel sediment transport, HSPF was used only for flow data to input into HEC-6 and to obtain information about sediment yield from the land. Another reason HSPF was chosen is that it allows for a continuous long-term simulation.

HEC-6 was chosen to satisfy the hydraulic and sediment transport components of the system for several reasons. First, it is the only comprehensive non-proprietary sediment transport model available. Also, HECRAS river cross-section data were already available, which, through a conversion process, could be used as HEC-6 input. Finally, HECRAS has been adopted by Milwaukee Metropolitan Sewerage District (MMSD) for management of water resources on the Menomonee River. The USACE WRDA96 Section 516e program for the development of sediment transport models for Great Lakes tributaries has an interest in making these models available to local users such as MMSD to better manage sediment yield, delivery and sedimentation issues.

Since the main goal of this project was to produce a system to determine how land use changes (and other anthropogenic factors) affect sediment transport to the Milwaukee harbor, GIS data layers needed to be incorporated into the system. This was done through the use of ArcView, a powerful GIS tool with excellent integration, mapping and visualization capabilities. Since tabular GIS information is directly linked to the map layers in ArcView, once a land use is changed on the map, it is reflected on the table, which is then exported to Excel for further manipulation. A USGS program called GenScn – "A Tool for the Generation and Analysis of Model Simulation Scenarios for

Watersheds" was chosen as an input generation and output presentation tool. GenScn has HSPF incorporated into it and allows the user to create several different HSPF "scenarios" which are saved within one project. Because the overall system requires several models, linkages have been established between some of the models and data layers (GIS, HSPF, HEC-6). These linkages improve the efficiency of applying the system, while at the same time, contributing to a user-friendly interface. Changes in land use directly affect the amount of sediment yield from the land simulated within HSPF, therefore a link was established between the ArcView GIS data layers and the HSPF input files. By changing a land use on the land use map in ArcView, the corresponding tabular data is also changed. This tabular data is then read into Excel where it is broken down into land use/soil groups and sub-basins for HSPF. The next step is to enter this data into the HSPF input file. To properly model the Menomonee River system with HEC-6, local inflows from tributaries and other sources had to be output from HSPF and input into HEC-6. Simulated flow data from HSPF were output in time series format and read into a program called Data Analyzer, developed by Baird & Associates. This program converts the HSPF time series data into the form needed for the HEC-6 input file.

Calibration and Verification of the Model: Calibration and implementation of the hydrologic model component was completed earlier by MMSD. The hydrodynamic model component involves a combination of HSPF and HEC-6. HSPF provides inflows (from tributaries, etc.) that are supplied to the HEC-6 model input. Alone, HSPF is not ideal for detailed hydraulic modeling because of its limitations regarding cross-section spacing. The HEC-6 model is more representative and accurate as it allows for more detailed cross-section data to be supplied along the rivers/tributaries.

The sediment transport model component was the HEC-6 program. Modifications to the source code were made by Baird & Associates in order to accommodate the large Menomonee River watershed network model (~1200 cross-sections). Sediment data from the USGS for 1975-1977 were used for the initial setup of the model. In order to perform satisfactory calibration and verification tests, more recent sediment data will need to be obtained, or the 1975 land use data must be obtained and used in the HSPF model to generate flows and sediment transport for comparison to sediment load data from the 1970's.

Example Application of the System: The HSPF model was run in the continuous mode at 15-min time steps for the period 1940-1997 simulating flows and sediment being washed off the land. The flow outputs from this HSPF model run were input into the HEC-6 model, which was then run for the same period of time. In addition to plotting the HEC-6 output in GenScn, a program developed by Baird & Associates, Spatial Data Analyzer (SDA), was used to animate the HEC-6 time-series sediment load data. Figure 3 shows a snapshot of animated flow velocities along the Menomonee River with GIS data as a backdrop.

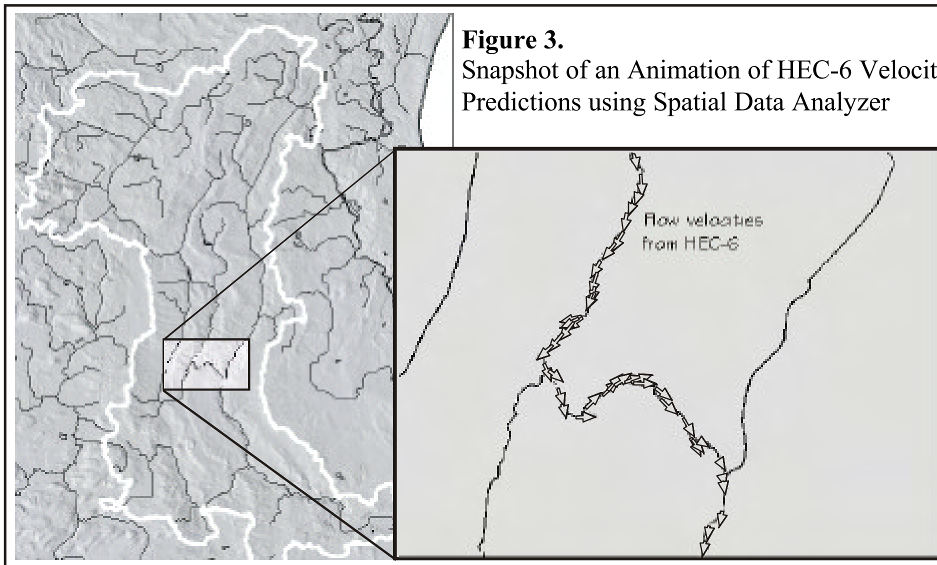

Figure 3.
Snapshot of an Animation of HEC-6 Velocity Predictions using Spatial Data Analyzer

Key Findings: A watershed-based sediment transport modeling system is a valuable tool for the design of river and stream restoration projects, in addition to a tool to managing sediment yield, load and sedimentation. Once again, the available sediment load data was insufficient to both calibrate and verify the model system. Urbanization on this watershed has resulted in flashier flows (higher peak discharges and velocities), increasing bank erosion. This same process must be considered for future impacts to restoration projects that have recently been or will soon be, implemented.

SAGINAW RIVER WATERSHED

Description of the Watershed: The Saginaw Bay Watershed (8,600 square miles) is the largest in Michigan. Twenty-eight major rivers, creeks, or agricultural drains flow directly into Saginaw Bay, but about 75% of the flow to the Bay comes from the Saginaw River. The Saginaw River watershed (6,060 square miles) is generally a large, flat area consisting of agricultural and forested lands with an extensive network of rivers, streams and agricultural drains. The Saginaw River is 22 miles long and most of its flow originates from four major tributaries. The entire watershed is divided into five sub-watersheds by the Saginaw River and its tributaries, including the Cass subwatershed, the Flint subwatershed, the Shiawassee subwatershed, the Tittabawassee subwatershed and the Saginaw River subwatershed (see Figure 4). The developed prediction system has only been applied to the Cass subwatershed. Between 1992 and 1997 the Corps has dredged 275,000 cubic yards on average each year at an average annual cost of $1.1M.

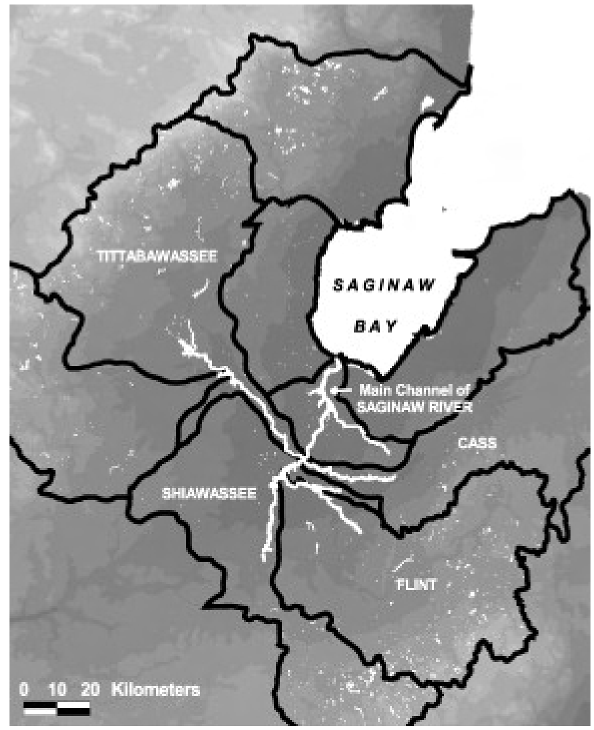

Figure 4. Saginaw River Watershed and its Subwatersheds

Description of the Model System: The overall model system for the Saginaw River is described in Figure 5. The data required to set up the system are geographic data, weather records, and hydrological records. The geographic data includes topographic data defining tributary areas and watershed slopes, soil data presenting soil type and erodibility factor, land use data describing vegetation coverage and fertilization level, stream network, and river bathymetry. These are generally spatial data such that the data can be well presented by GIS. ArcView GIS was implemented to manage and to spatially present pertinent data. This package can eventually form the foundation for an extremely user-friendly platform for interacting with all aspects of the modeling components, while greatly enhancing the value and use of the data. Base maps showing roads, streams and watershed divides provide a spatial context for reviewing the data.

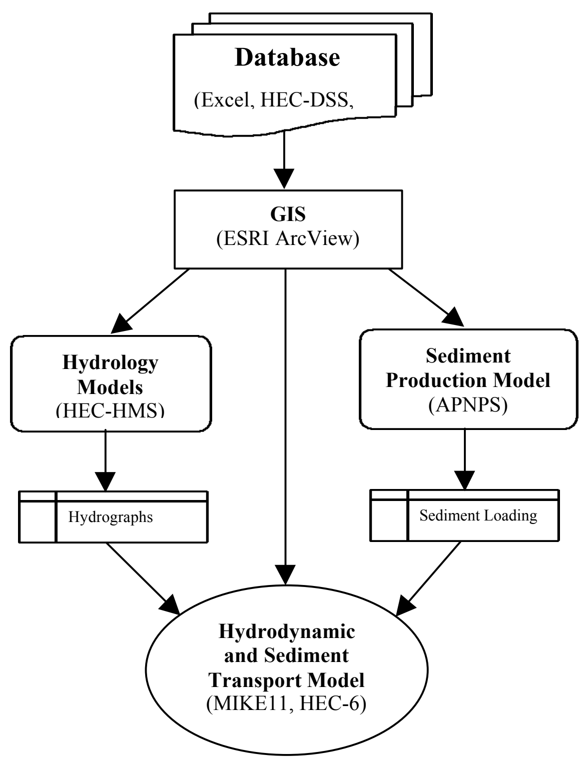

Figure 5. Saginaw River Watershed System

Weather data including precipitation, temperature, and wind data are needed to set up the hydrological model and the watershed sediment production model. Hydrological data including stage, discharge and sediment loading in streams and rivers are used for the calibration of the hydrological and sediment production models and are also used in the hydrodynamic and sediment transport modeling in rivers. These data are generally stored in Excel format or ASCII format.

The US Army Corps of Engineer' s HEC-HMS model was selected for hydrologic modeling. Once the most recent version of AGNPS is implemented, the HEC-HMS system will not be essential.

Sediment yield is predicted with an earlier version of the Agricultural Non-Point-Source Model (AGNPS v5.0), a computer simulation model that predicts runoff volume and peak runoff rate, eroded and delivered sediment, and pollutants for single storm events. This event specific version of AGNPS was previously set up on the Cass River subwatershed by Michigan State University. The watershed may be broken up into cells, so runoff, sedimentation, erosion, or nutrient loading may be evaluated for each cell or at the outlet for the entire watershed. Therefore, areas or cells of erosion or deposition may be identified in the watershed. The input for each cell consists of 22 parameters that identify soil condition, land use, watershed boundaries, water features, and elevation. Estimating the total sediment eroded requires the use of the Universal Soil Loss Equation (USLE), Revised USLE, or Modified USLE.

The earlier version of AGNPS does not estimate the distribution of the sediment load with time, but only the total sediment load produced for each storm event. MIKE11 and HEC-6 require, as input, the sediment load or concentration at each time step. Therefore, a relationship between sediment load and discharge was developed to estimate how sediment load varies during and after the storm event. The latest version, AnnAGNPS (part of the AGNPS98 package), allows for full time series representation.

Sediment eroded from a watershed eventually transports and deposits in streams, rivers and bays. The hydrodynamic and sediment transport models simulate and predict flows and sediment loading in rivers and are key components in the model system. Both hydrodynamic and hydraulic models of the Saginaw River were developed to simulate stage and discharge. The model area extends from Saginaw Bay to the four primary tributaries - the Cass, Flint, Shiawassee and Tittabawassee Rivers (see Figure 4). This investigation included a comparison of a hydraulic model (HEC-6) to a hydrodynamic model (Danish Hydraulic Institute MIKE11). The hydrodynamic MIKE11 was selected in favor of the USGS BRANCH1D and USACE UNET 1D hydrodynamic models because of the more efficient interfaces and higher quality visualization. The initial phase of setting up a sediment transport model for the Saginaw Watershed consisted of developing a sediment transport model of the Saginaw River linking the effects of the Cass River Watershed to the Saginaw River to predict the impact of land use changes in the Cass River watershed on sediment loading.

There are several models available to simulate sediment transport in river systems. Based on a review of the alternatives it was decided to use both MIKE 11 and HEC-6 in predicting both event specific and annual impacts of sediment transport on the Saginaw River. HEC-6 is a 1D variable steady hydraulic model, in which hydraulic calculations are treated as steady state for each flow, while MIKE 11 is a 1D unsteady model. To demonstrate the difference of the sediment transport induced by using the variable steady hydraulic model and the unsteady hydrodynamic model, both MIKE 11 and HEC-6 are used to predict annual impacts of sediment transport on the Saginaw River. Key inputs to the sediment transport model at the upstream boundary condition consisted of sediment load (derived from AGNPS and/or sediment load-discharge rating curves), grain size distribution of sediment being transported as a function of discharge, river bed sediment composition and critical shear stress for erosion and deposition. In all cases, information was not available to the extent required to develop a fully quantitative model. The sediment transport modules of HEC-6 and MIKE 11 are discussed in the next section.

Calibration and Verification of the Model: Extensive stage and discharge data was not found to be available for model calibration and validation. In order to completely satisfy the requirements of a calibration study, a flow/stage data collection program must be implemented.

The dredging records in the Saginaw River are the only available information located or found for calibration of the sedimentation simulations. The total dredging averaged over 11 years from 1982 to 1993 is assumed to be equal to the annual deposition rate. This assumption implies that over the full length of the river there is no net erosion or deposition in the long-term. It is difficult to assess the validity of this assumption without additional data. The assumption also implies that most of the deposition occurs only in the dredged channel and not adjacent parts of the river bed. The calibration period is the calendar year of 1991 because the hydrological data at the Saginaw and

Essexville gages are relatively complete in that year and the flow conditions were representative of average for the period 1982-1993. Under the same boundary conditions, sediment deposition in the Saginaw River is carried out using both the HEC-6 and MIKE 11 models.

The "calibration" of the HEC-6 model consisted of selecting the sediment transport prediction that provided the best estimate of sedimentation derived from the dredging data. In the HEC-6 model, there are two methods for the calculation of clay and silt deposition rates: Method 1 - Krone (1962); and Method 2 - Computes deposition rate using Krone (1962) and erosion rate using Ariathurai (1976). The annual sediment deposition rates computed by these two methods was compared to the annual average dredging rate (using Meyer-Peter & Muller' s equation for sand transport). The accumulated sediment deposition from the Mouth to River Mile 18, for which the dredging records are available to be compared, are calculated and listed in Table 1 below. It is seen that using Method 2 for clay and silt transport predicts less sediment deposition than Method 1. There are eleven (11) methods available to estimate sand transport in the HEC-6 model. We have selected several empirical equations as listed in the table to determine which method best predicts the sediment transport in the Saginaw River. From the table, it can be seen that the combination method of Toffaleti and Meyer-Peter & Muller (hereafter referred to as the "Combination method") produces relatively accurate results compared with the annual dredging volume, but still results in a 33% underprediction. Regardless of the equation, the HEC-6 model underpredicts the deposition at both the mouth of the Saginaw River and in the middle reach from the river. In contrast, the MIKE 11 predictions (which rely on the approach of van Rijn, 1984) over predict the deposition by a factor of two.

The annual sediment load carried into the Saginaw Bay (see Table 1) varies depending on method used for estimating the sediment transport rate. The amount deposited in the river is relatively small compared to the total load passing through the river into the bay. In the model estimates, the predicted deposition rate will be very sensitive to the assumed grain size composition for the sediment load at the upstream boundary condition. Improved estimates of incoming grain size composition at the upstream boundary of the model may result in better estimates of total deposition for HEC-6. Clearly in order to select the most appropriate model it is also essential to quantify the outgoing transport into Saginaw Bay. A study is just being completed with a 2D hydrodynamic and advection-dispersion model in addition to an evaluation of deposition and dredging rates to assess the amount of sediment transported into the bay.

Table 1. Results of Sediment Transport Modeling for the Navigable Section of the Saginaw River

Models and Empirical Methods			Deposition (cu-yds)	Relative Error (%)	Incoming (cu-yds)	Outgoing* (cu-yds)
	Method for Clay and Silt	Method used for Sand Transport				
HEC-6	Method 1 (Krone, '1962)	Toffaleti (1966)	57,464	-48.8	247,455	189,991
		Yang (1973)	67,138	-40.2	247,455	180,317
		Meyer-Peter & Muller (1948)	53,033	-52.7	247,455	194,442
		Combination	74,338	-33.7	247,455	173,117
		Madden (1985)	52,191	-53.5	247,455	195,264
		Copeland (1990)	55,643	-50.4	247,455	191,812
	Method 2 Krone (1962) and Ariathurai (1976)	Madden (1985)	24,736	-78.0	247,455	222,719
		Meyer-Peter & Muller (1948)	27,073	-75.9	247,455	220,382
MIKE 11			214,552	91.2	247,455	32,903
Annual Dredging Volume Averaged Over 11 years			112,195	---	---	---

*Note: Transported into Saginaw Bay, Lake Huron

Example Application of the System:
To demonstrate the difference between the unsteady model, MIKE 11, and the variable steady model, HEC-6, the sediment deposition rates along the river computed by both models are compared with the average annual dredging rate and shown in Figure 6. River Mile 0 coincides with the river mouth at the downstream end of the model domain. The HEC-6 result shown in Figure 10 is computed using Method 1 for clay and silt transport and the Combination equation for sand transport. The HEC-6 model underpredicts the sediment deposition in the downstream section of the river near the mouth. This may be induced by the steady hydraulic calculating scheme, which may not account for the flow speed reduction at the mouth resulting in increased deposition. In contrast, the hydrodynamic model, MIKE 11, reproduces the physical process of sediment deposition near river mouth reasonably well.

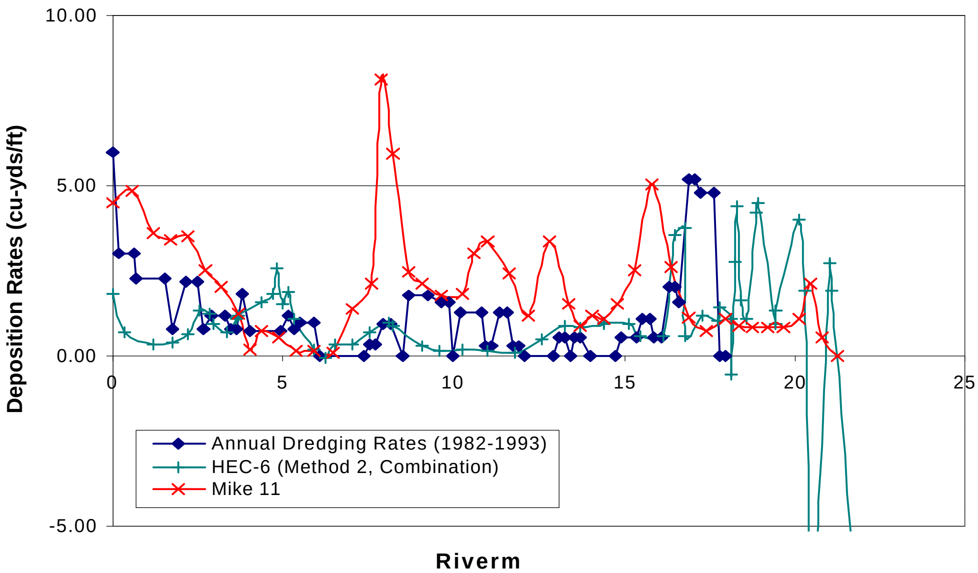

Figure 6. Comparison of deposition rates computed by HEC-6 and MIKE 11 to the dredging rate

Key Findings: Using the same upstream sediment load boundary condition estimated with a sediment rating relationship, the predicted annual sedimentation in the navigable section of this river was in the range of 50,000 to 200,000 cubic yards per year compared to the estimate based on annualized dredging quantities of 100,000 cy/year. The only difference in total deposition was the amount of sediment transported beyond the river mouth into the bay. There were significant differences between the hydraulic or steady state model (HEC-6) and the hydrodynamic model (MIKE11) in where the sedimentation occurred in the river with the hydrodynamic model providing a somewhat better comparison to the sedimentation pattern near the river mouth inferred from the dredging quantities. AGNPS appears to be the best available model to estimate sediment load from agricultural land. However, the AGNPS98 package with AnnAGNPS providing continuous simulation of hydrologic and sediment transport processes is much more versatile and useful. The AGNPS v5.0 model used in this study must be updated to AnnAGNPS.

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A GIS ADAPTATION OF *PSIAC* FOR PREDICTING SEDIMENT-YIELD RATES

Kathleen Woida, Geologist, USDA-NRCS, Des Moines, IA; Kathryne B. Clark, Dept. of Geography, University of New Mexico, Albuquerque, NM

INTRODUCTION

Since the 1970's, the Natural Resources Conservation Service (NRCS) in New Mexico, Utah, Arizona, and Nevada has used the Pacific Southwest Inter-Agency Committee (PSIAC) method for predicting watershed sediment-yield rates. These rates are important in the design of flood-retarding and erosion-control dams as well as conservation management practices that reduce sediment loading to streams. The PSIAC (1968) method considers nine factors in determining the sediment-yield rate for a drainage area. These are surface geology, soils, climate, runoff, topography, ground cover, land use, upland erosion, and channel erosion and sediment transport. Although originally recommended for areas of 10 mi^2 miles or greater, Shown (1970) indicated good results when applied to basins between 0.02 and 7.5 mi^2. Early studies (Shown, 1970; Renard, 1980; Johnson and Gebhardt, 1982) showed the average difference between PSIAC predicted rates and measured rates—from reservoir sedimentation surveys—to range between 15% and 45% in the southwest U.S. In New Mexico, the average difference was 27% for ten flood-retarding reservoirs surveyed in the 1990's (NRCS, unpublished data).

The accuracy of the sediment-yield rates predicted by PSIAC depends in large part on the sampling intensity, that is, the number of ratings conducted for a given watershed and how well they represent the watershed's variability. The more comprehensive and systematic the sampling technique, presumably, the more accurate the results. This premise suggested the usefulness of a geographic information system (GIS), which over the past decade has proven to be an effective tool for the analysis of large numbers of spatially diverse watershed parameters. In this study, we developed a GIS application to analyze the nine PSIAC watershed factors in two small controlled watersheds having moderate and high spatial variability. We then compared the results to PSIAC rates obtained in the conventional manner, and to measured sedimentation rates acquired from reservoir surveys and adjusted for trap efficiency. The results tentatively suggest improved accuracy and provide a spatially refined visual map of watershed sensitivity.

STUDY AREAS—LOCATION AND DESCRIPTION

Santa Cruz Site 3A: This 2.2 mi^2 watershed, on BLM land, is situated about a mile north of the village of Chimayo in mountainous north-central New Mexico. When flowing, its two major arroyos discharge into the Santa Cruz River, a tributary to the Rio Grande. The watershed is extremely variable, consisting of sloping mesa in its upper section, dramatic badlands in its middle section, and gently sloping valley floor in the bottom. Average watershed gradient is 8%. Ground cover varies from bare mudstone and volcanic tuff, to sparse range vegetation, to locally thick grass cover. The NRCS flood-control dam at this site has been in operation since 1972.

Hatch Valley Site 3: This is a 3.3 mi^2 watershed on BLM land that drains into the Rio Grande near the town of Hatch, in south-central New Mexico. The watershed heads in the foothills of the Caballo Mountains. It has an average gradient of 3.5% and shows less variability in ground cover, upland erosion, and land use than the Santa Cruz site. Most of the watershed is underlain by Tertiary alluvial fan deposits varying from weakly cemented sandstone and conglomerate to mudstone. The NRCS dam at the site has been in operation since 1958.

METHODS

Conventional PSIAC Procedure: Using the conventional procedure followed by NRCS in New Mexico, we subdivided each watershed into general areas characterized by obvious differences in erosion and sediment-yield potential. The Santa Cruz watershed was subdivided into three major land areas based on topography and soil type. The Hatch watershed was subdivided into three sections based on land use and management. In the field, we rated the nine factors of the PSIAC (1968) formula at four or five representative locations in each watershed section. We then calculated average point totals for the watershed sections and a weighted average point total for each watershed. The latter were converted to average sediment-yield rates.

Data for the GIS Model: We constructed a GIS map layer for each of the first two PSIAC factors—geology and soils—from digital data and digitized paper maps of the two watersheds. PSIAC point values for each geologic or soil map unit were assigned on the basis of field observations. Because of the small basin size, we assigned each watershed a single value for the climate factor, the same value that was assigned in the conventional rating procedure described above. For the topography layer, we used 1:24,000 scale USGS 30-meter resolution digital elevation models (DEMs), which then provided a base for data analysis of all layers at a 30-meter resolution. Point values for slope categories were assigned directly from the conventional PSIAC field form, Chart E. For the factors ground cover, land use, upland erosion, and channel erosion and sediment transport, we began by delineating map areas from 1:6,000 scale 1993 color-infrared aerial photographs. PSIAC point values for the map units were then assigned on the basis of field observations and NRCS field office information. To ensure consistency of data, all field delineations and point-value determinations were made by the same person that had done the conventional rating. The Santa Cruz site required one day and the Hatch site one and a half days of field work.

For the runoff layer, we constructed a specific algorhythm that generated a single average runoff value for each 30-meter pixel on the basis of NRCS soils hydrologic-group data, the ground cover layer, and the topography (slope category) layer.

The GIS Model: The GIS coverages were created using ESRI Arc/Info and its modules Arc/Grid and ArcTables. Models were developed to correct aberrant DEM values, mosaic and subset DEMs, and classify imagery with Erdas Imagine 3.1. Vector coverages were first digitized and then translated into raster files based on values assigned within the vector coverages. Both study areas were run through a percent slope model to create topography layers. These layers were then classified into the traditional PSIAC topographic ranges. The layer for runoff values was compiled using a sub-model incorporating soil permeability, ground cover, and slope values. The climate factor was added in arithmetically since this value was the same for the entire watershed. The values for all the PSIAC factors were totaled for each 30-meter pixel unit, which created a separate rating for each 30-meter square area of the watershed. The ratings were then added together, and divided by the total number of pixels. This average point value was converted to a sediment-yield rate using the conventional PSIAC (1968) chart.

RESULTS AND DISCUSSION

Santa Cruz Site 3A: The conventional rating method produced an average sediment-yield rate of 1.82 acre-feet per square mile per year (ac-ft/mi^2/yr) for the Santa Cruz watershed. This is a weighted average of separate rates for the upper, middle, and lower portions of the watershed of 0.86, 2.91, and 0.54 ac-ft/mi^2/yr respectively.

The GIS adaptation yielded a considerably lower average sediment-yield rate of 1.18 ac-ft/mi^2/yr for the watershed. A look at the average value of each PSIAC factor for the two methods (Table 1) suggests which factors were most affected by the application of a GIS—namely topography and channel erosion/sediment transport, each of which showed a drop of three points in the average point value. A comparison of the range of values for each factor (Table 1) demonstrates the ability of the GIS method to better capture the spatial variability of the watershed. This is best illustrated here by the ground cover factor (Figure 1), which goes from a 6-point range (3 to 8) to an 18-point (-7 to 10) range.

Table 1. Values of PSIAC Factors for Conventional and GIS Methods (higher values indicate greater potential for sediment yield).

	Santa Cruz Site 3A				Hatch Valley Site 3			
	Average		*Range*		*Average*		*Range*	
PSIAC Factor	Conv.	GIS	Conv.	GIS	Conv.	GIS	Conv.	GIS
Geology	5	5	3 to 5	3 to 5	4	4	4 to 5	2 to 5
Soils	6	6	4 to 7	4 to 7	5	5	4 to 8	4 to 8
Climate	6	6	6	6	6	6	6	6
Runoff	6	6	3 to 8	3 to 9	4	4	3 to 5	1 to 8
Topography	13	10	2 to 20	0 to 20	9	5	1 to 15	0 to 20
Ground Cover	6	6	3 to 8	-7 to 10	-1	-3	-5 to 10	-9 to 10
Land Use	3	2	-1 to 8	-3 to 10	2	2	2 to 5	-2 to 5
Upland Erosion	16	17	8 to 22	6 to 25	12	10	9 to 19	0 to 19
Channel Erosion	18	15	12 to 24	7 to 25	11	9	9 to 11	0 to 13

Figure 2 shows sediment-yield rates across the watershed grouped into five standard categories: Low (< 0.2 ac-ft/mi^2/yr), moderate (0.2-0.5), moderately high (0.5-1.0), high (1.0-3.0), and very high (>3.0). A comparison of Figure 2 to Figure 3, which maps the conventionally derived rates, illustrates the obvious spatial refinement of the GIS adaptation of PSIAC.

Hatch Valley Site 3: The conventional rating method yielded an average sediment-yield rate of 0.57 ac-ft/mi^2/yr for the Hatch Valley watershed. This is a weighted average of separate rates for the upper, middle and lower portions of the watershed of 0.52, 0.54 and 0.82 ac-ft/mi^2/yr respectively.

The GIS model calculated a lower average sediment-yield rate of 0.39. The topography factor was impacted the most by application of a GIS, with a drop in the average point value from 9 to 5. The ranges of values for runoff (Figure 4), channel erosion/sediment transport (Figure 5), and upland erosion showed the greatest relative increase in variability (Table 1). Figure 6 shows the standard sediment-yield categories for the Hatch Valley watershed.

Comparison with Reservoir Survey Data: NRCS engineering personnel conducted 1-foot contour surveys of reservoirs at the two sites in 1998-99. Total storage capacity loss at Santa Cruz Site 3A over the period of operation was determined to be 80 acre-feet. After adjustment for trap efficiency of the structure, this translated to an average watershed sediment-yield rate of 1.37 ac-ft/mi^2/yr. At Hatch Valley Site 3, storage loss over the period of operation was 57.6 acre-feet, for an average sediment-yield rate of 0.44 ac-ft/mi^2/yr (NRCS, 1999a and 1999b).

In both watersheds, the GIS model results were closer to the sediment survey results than were those of the conventional method (Table 2). In both cases, the GIS model underestimated and the conventional method overestimated the rates. Conventional results in previous studies have not shown a consistent trend toward overestimation or underestimation. At least in the case of the Santa Cruz site, the high numbers of the conventional method may be partially explained by the generalization of the topography factor in the middle section of the watershed. Here the badland topography, with its many near-vertical slopes, was given an average point value of 20. This is the highest value possible for this factor and represents average slopes of 30% or steeper. On the other hand, the topography layer in the GIS adaptation, with its 30-meter resolution, was able to account for the many small areas in the badlands having low slopes, such as bottoms of channels and tops of buttes.

The lower-than-measured rates calculated by the GIS model may be a product of temporal variations in sediment-yield rates throughout the period of record. In the case of the Santa Cruz site, current rates appear to be much lower than at certain intervals during the past. Earlier surveys at three other Santa Cruz sites within a two-mile radius of Site 3A indicated much higher rates during the period 1966 to 1971, ranging from 4.1 to an astonishing 12.1 ac-ft/mi^2/yr. The PSIAC procedure is designed to predict the average long-term sediment-yield potential of a watershed, whereas the survey measured a relatively short period of time (27 years) that included a critical interval with much higher than average rates.

Table 2. Comparison of GIS and Conventional PSIAC Results to Survey Results.

Method	Santa Cruz Site 3A		Hatch Valley Site 3	
	Sediment Yield (ac-ft/mi^2/yr)	Difference (%)	Sediment Yield (ac-ft/mi^2/yr)	Difference (%)
Reservoir Survey	1.37		0.44	
Conventional PSIAC	1.82	+37	0.57	+30
GIS Adaptation	1.18	-14	0.39	-11

CONCLUSIONS

The study has shown that it is possible to build a geographic information system that mimics the PSIAC procedure at a much more detailed level. For these two small New Mexico watersheds, the GIS adaptation yielded more accurate sediment yield-rates than did the conventional approach, with differences of 11% and 14% respectively, compared to 30% and 37% for the conventional method. However, more sites would need to be compared to test this hypothesis of improved accuracy.

The time required to produce the GIS version—roughly 1-2 days of field work and two weeks of office/computer time—is considerably greater than that required by the conventional method. However, in addition to potentially improving accuracy, the GIS model offers much more detailed information about a watershed's susceptibility to erosion and sediment yield. Although a more detailed picture could be obtained conventionally simply by performing more ratings in the watershed, only about 150 ratings per watershed could be completed in the same amount of time as the entire GIS procedure required. This compares to roughly 6,000 and 11,000 rating calculations provided by the GIS model for the Santa Cruz and Hatch sites respectively. The GIS model provides a detailed view of each factor, as well as sediment-yield potential, over the entire watershed. The runoff and the topography factors in particular are greatly enhanced through the spatial resolution of the DEMs and the great computational ability of the GIS. This spatial refinement would make it possible to rapidly identify causes, type, and magnitude of land sensitivity at specific locations in a watershed slated for change or development, such as an urban fringe watershed. Values for the ground cover and land use factors can also be easily changed in the GIS model to project the impact of planned development or land treatment practices, or to reflect actual watershed improvements as they evolve.

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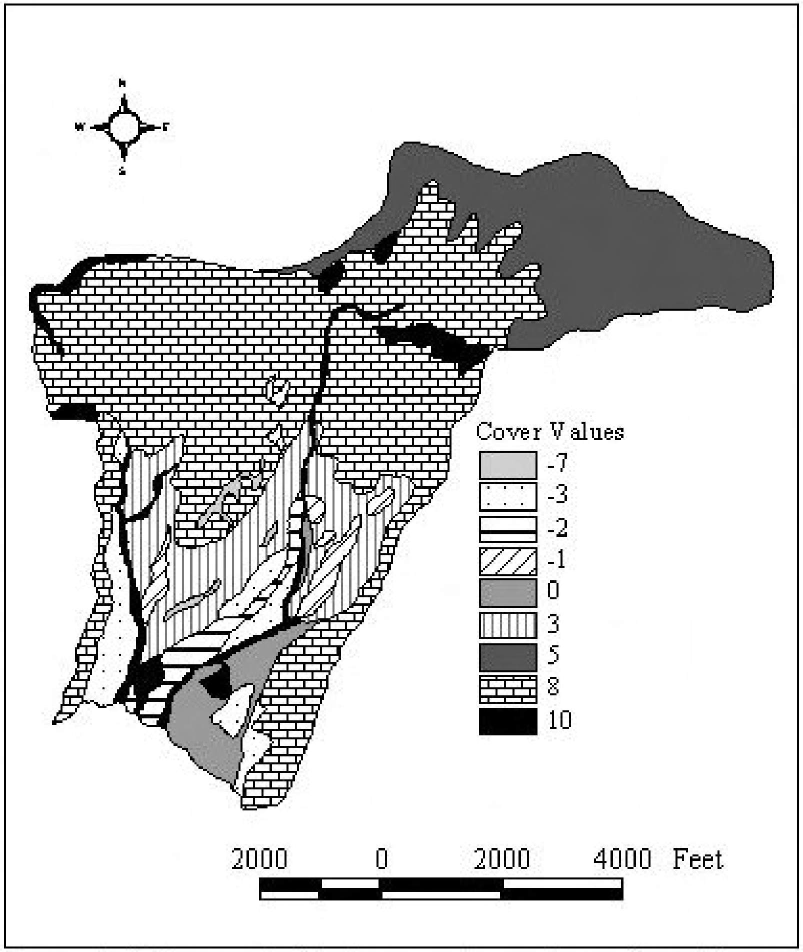


Figure 1. GIS map units for the ground cover factor, Santa Cruz watershed.

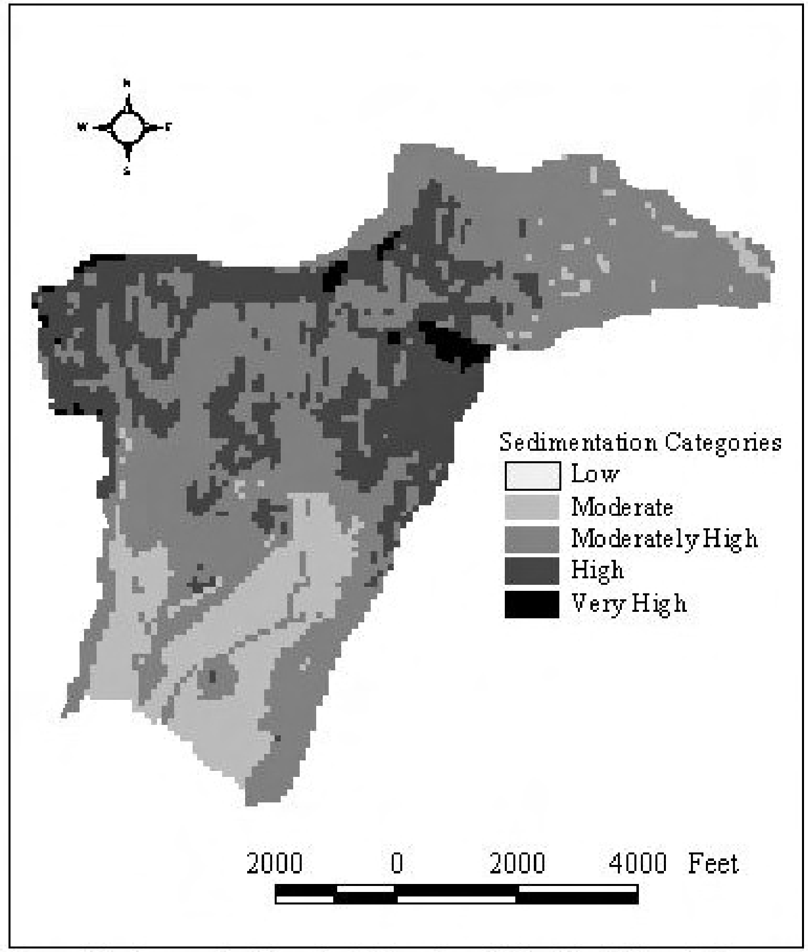


Figure 2. General sediment yield rates obtained with the GIS model, Santa Cruz watershed.

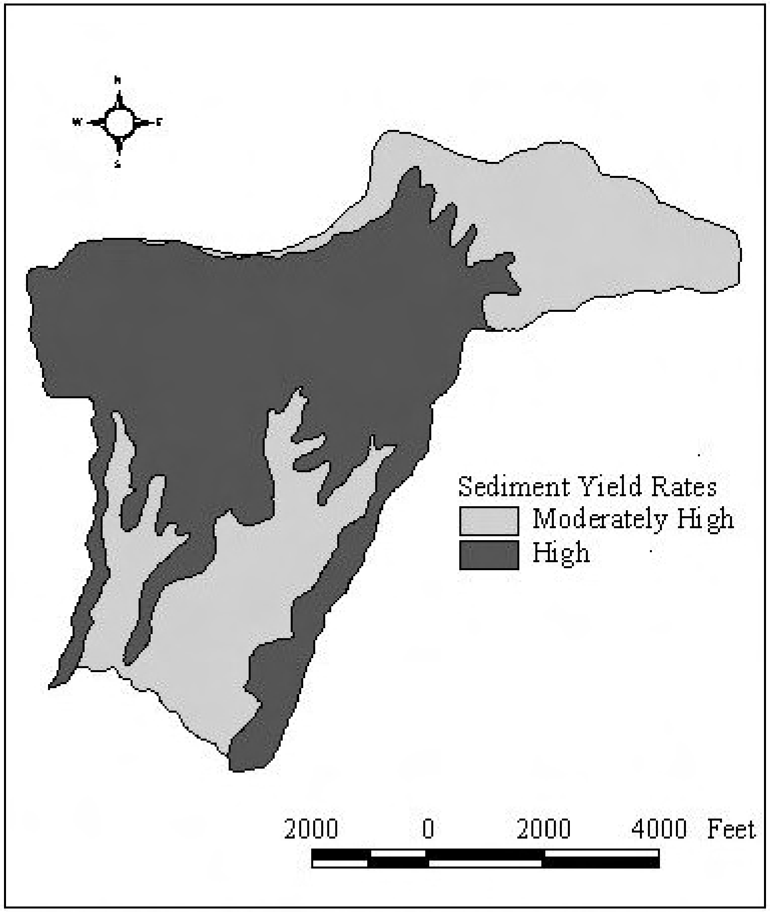


Figure 3. General sediment yield rates obtained in the conventional manner, Santa Cruz watershed.

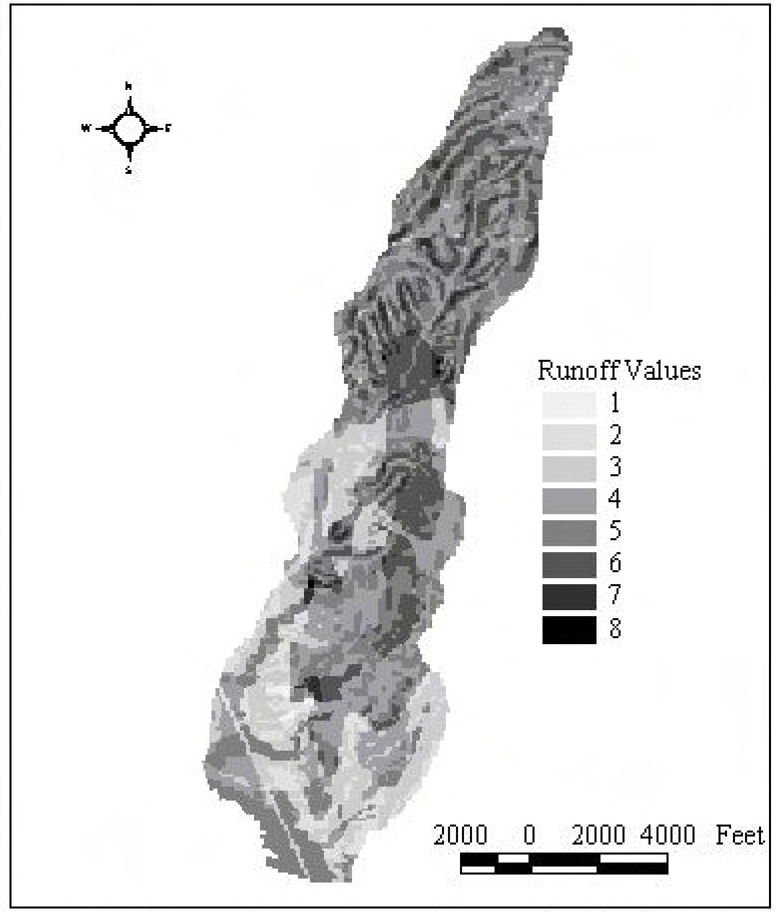


Figure 4. GIS map units for the runoff factor, Hatch Valley watershed.

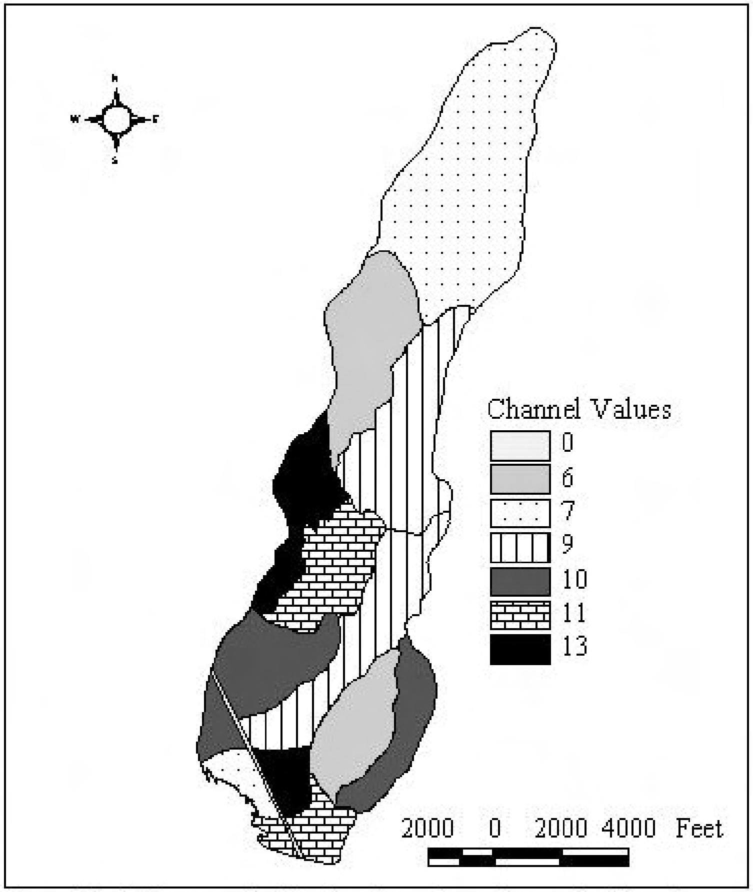


Figure 5. GIS map units for the channel erosion and sediment transport factor, Hatch Valley watershed.

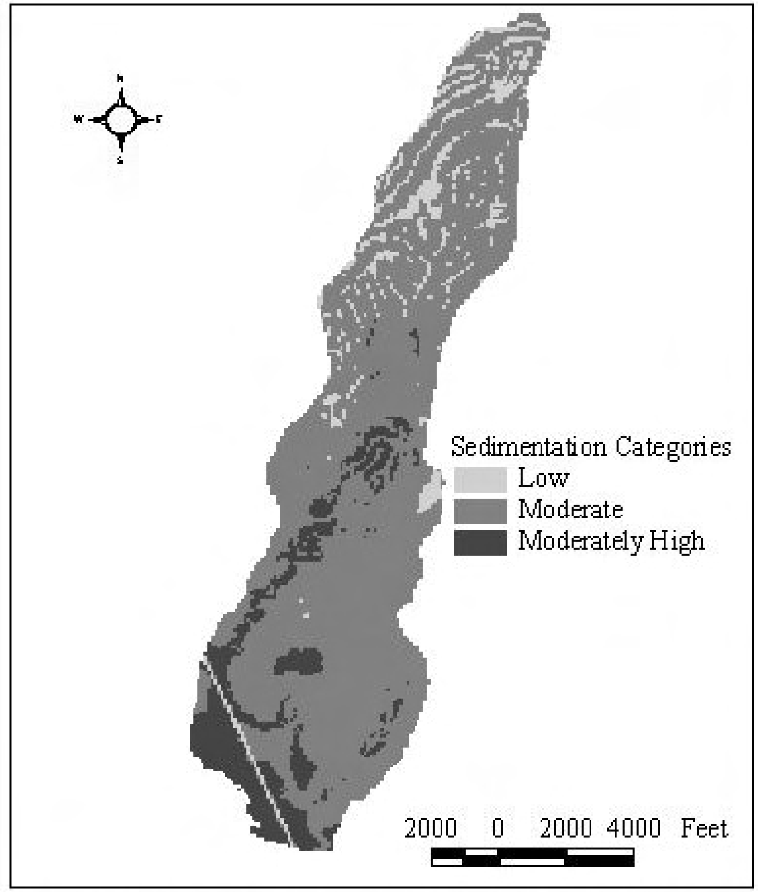


Figure 6. General sediment yield rates obtained with the GIS model, Hatch Valley watershed.

FIRE EFFECTS ON SEDIMENT AND RUNOFF IN STEEP RANGELAND WATERSHEDS

Frederick B. Pierson, Jr., Research Hydrologist, USDA-ARS; Kenneth E. Spaeth, Rangeland Hydrologist, USDA-NRCS; Deirdre H. Carlson, Support Scientist, University of Idaho, Boise, ID.

USDA-ARS Northwest Watershed Research Center, Boise, ID 83712-7716
208/422-0700, fpierson@nwrc.ars.usda.gov

INTRODUCTION

Fire is a natural component of the Intermountain sagebrush-steppe ecosystem (Wright and Bailey 1982) with a return period of 25 to 100 years, depending on community type and natural fuel load and distribution. However, fuel and land management activities in the past century have placed wildland values such as soil and water quality at greater risk from wildfire. Increased soil erosion over natural levels following wildfire can lead to loss of soil productivity. Additionally, higher runoff rates from severely burned landscapes can lead to flooding, sedimentation, and increased risk to human life and property. This increased risk of runoff and erosion following wildfire continues to generate concern at the expanding urban-wildland interface throughout the western United States.

While the hydrological consequences of fire have been widely examined in forest ecosystems, few studies have examined wildfire impact on rangeland hydrology and erosion. Most of these studies have shown an increase in runoff and erosion rates the first year following fire, with recovery to pre-fire rates generally within five years (Wright and Bailey 1982). Timing and extent of recovery is highly dependent on slope and vegetation type (Branson et al. 1981, Knight et al. 1983, Wright et al. 1982, Wilcox et al. 1988). Many rangeland plant communities have naturally occurring hydrophobicity (DeBano and Rice 1973), but litter and vegetative cover protect the soil and enhance infiltration. Fire removes this protective covering, exposing the soil to erosion by raindrop impact and overland flow. Fire can also vaporize some of the organic compounds on the soil surface and distill the rest downwards, creating concentrated hydrophobic layers within the upper soil. Degree and longevity of hydrophobicity is dependent on compounds present and intensity and duration of fire (Wright and Bailey 1982). Fire can also reduce the organic matter content in the upper layers, thus reducing infiltration.

Major unknowns associated with rangeland wildfire are effects on vegetation and soil conditions affecting hydrologic processes, including infiltration, surface runoff, erosion, sediment production and transport, flooding, and the effectiveness of mitigation practices. The USDA-ARS Northwest Watershed Research Center (NWRC) has been investigating the impact of fire on rangeland hydrology and erosion in the mountains above Boise, Idaho (Boise Front) and in the Pine Forest Range near Denio, Nevada. The objective of the NWRC investigations are to quantify fire impacts on infiltration capacity, runoff, and erosion following fire, gain insight into the processes involved and determine how long the fire effects persist.

STUDY AREA AND METHODS

The Idaho study site is located on the Boise Front immediately above the city of Boise, Idaho (Eighth Street Fire) and the Nevada study site is located approximately 50 km south of the town of Denio, Nevada (Denio Fire). Both sites have vegetation consisting of bitterbrush (*Purshia tridentata*) /big sagebrush (*Artemisia tridentata* spp. *Wyomingensis*) /bluebunch wheatgrass (*Pseudoroegneria spicata*) - Thurber' s needlegrass (*Stipa thuberiana*) communities on south aspects, and big sagebrush/Idaho fescue (*Festuca idahoensis*) communities on north slopes (Interagency Fire Rehabilitation Team 1996). Some areas are characterized by an increase in three-awn (*Aristida spp*), Sandberg' s bluegrass (*Poa sandbergii*), and rabbitbrush (*Chrysothamnus* spp). Soils on both sites were derived from granite and consisted of fine gravelly coarse sandy loams, shallow (south slopes) to very deep (north slopes), well drained, on slopes of 35 to 60%.

Treatments on the Boise Front included combinations of slope aspect (north and south) and fire intensity (moderate, high). Treatments in Denio consisted only of intensely burned, north facing aspects. In both studies, burned sites were compared to unburned sites with the same soil type and vegetation as that found before fire. Sampling on both burned and unburned sites was stratified based on coppice areas (areas strongly influenced by the existence of a shrub) and interspace areas (areas between shrubs primarily dominated by grasses and forbs) (Pierson et al. 1994).

A portable oscillating-arm rainfall simulator with specifications as described by Meyer and Harmon (1979) was used to achieve intermittent rainfall similar to naturally occurring rainfall. Simulations were run on undisturbed plots, 0.5 m^2 in size, without pre-wetting. Soil moisture for all plots was extremely low (generally <10%). Rainfall was applied at a rate 67 mm hr^{-1} on the Boise Front and 85 mm hr^{-1} on the Denio sites. Runoff samples were collected at two-minute intervals throughout the 60-minute simulation and analyzed for runoff volume and sediment concentration. Infiltration capacity for each two-minute interval was calculated as the difference between applied rainfall and measured runoff. Suspended sediment samples were weighed, dried at 105^o, and re-weighed to determine soil loss. Plot vegetative cover for each plant functional group was ocularly estimated, and vegetation within plots was harvested, dried and weighed following each simulator run to determine vegetation biomass by functional group for each plot. Microtopography was estimated using a point-frame to measure average deviation of the soil surface compared to a flat surface.

RESULTS AND DISCUSSION

Boise Front, Idaho: Fire had little effect on the initiation of overland flow. Runoff on all plots began within 4 minutes after start of rainfall (67 mm hr^{-1}, 5 mm cumulative rainfall) for both burned and unburned sites on north and south facing slopes (Figure 1). All sites also reached their peak runoff rates from 8 to 12 minutes following the start of rainfall (Figure 1). Very dry soils, waxy substances on the soil surface, or fire can create a temporary hydrophobic soil condition (Robichaud 2000, De Bano et al. 1967), such that during the first minutes (or longer) of rainfall, water beads on the soil surface and quickly runs off the plot. The water repellency can deteriorate as simulated rainfall continues, resulting in a gradual infiltration rate recovery over the duration of the run. This phenomenon was observed for the unburned plots on both the north and south slopes, and on the moderate and high intensity burns for the north slope, but disappeared within 30 minutes after start of simulated rainfall (Figure 1). For the remainder of the rainfall simulation, there was no difference in infiltration rates demonstrated between unburned sites and north-facing slopes with either moderate or high intensity fires.

Fire intensity had the greatest impact on runoff on the south slopes (Figure 1). Intensively burned south-facing slopes had the lowest cumulative infiltration (34.2 mm) followed by the moderately burned south slopes (46.2 mm), compared to the unburned south slopes (58.0 mm). This represents nearly a two-fold increase in runoff produced by the fire. Fire intensity also produced significantly greater erosion from south aspects as well (Figure 2). Cumulative erosion was up to 34 times higher on intensively burned south slopes compared to unburned conditions or even burned north aspects. This was due to the devastating removal of nearly all the vegetative ground cover that protects the soil and much of the soil organic matter that helps to bind soil particles together.

Closer examination of the spatial variability of infiltration and erosion processes following fire was accomplished by studying the differing effects of fire on coppices (areas directly under shrubs) and interspace areas (areas between shrubs). While both the coppice and interspace sites on south slopes were affected by fire, the fire had the greatest impact on infiltration of coppice areas where terminal infiltration rates were reduced 62 % compared to the unburned treatment. Infiltration on north slopes was little affected by burning and thus, no significant differences were found between coppice and interspace areas. The greatest difference in sediment yield came from intensely burned south-facing interspace sites, where severely burned interspace sediment yield increased immensely over that of even the moderate burn. Aspect differences in fire impacts on erosion have also been noted on forests, where south-facing slopes yielded six times the sediment as north-facing slopes (Marques and Mora 1992). This difference was attributed to the denser vegetative cover and more developed soils found on north-facing slopes. These findings are also consistent with observations made following an intense thunderstorm that occurred over the study area where the south-facing slopes had the highest concentration of rills and suffered significant soil losses.

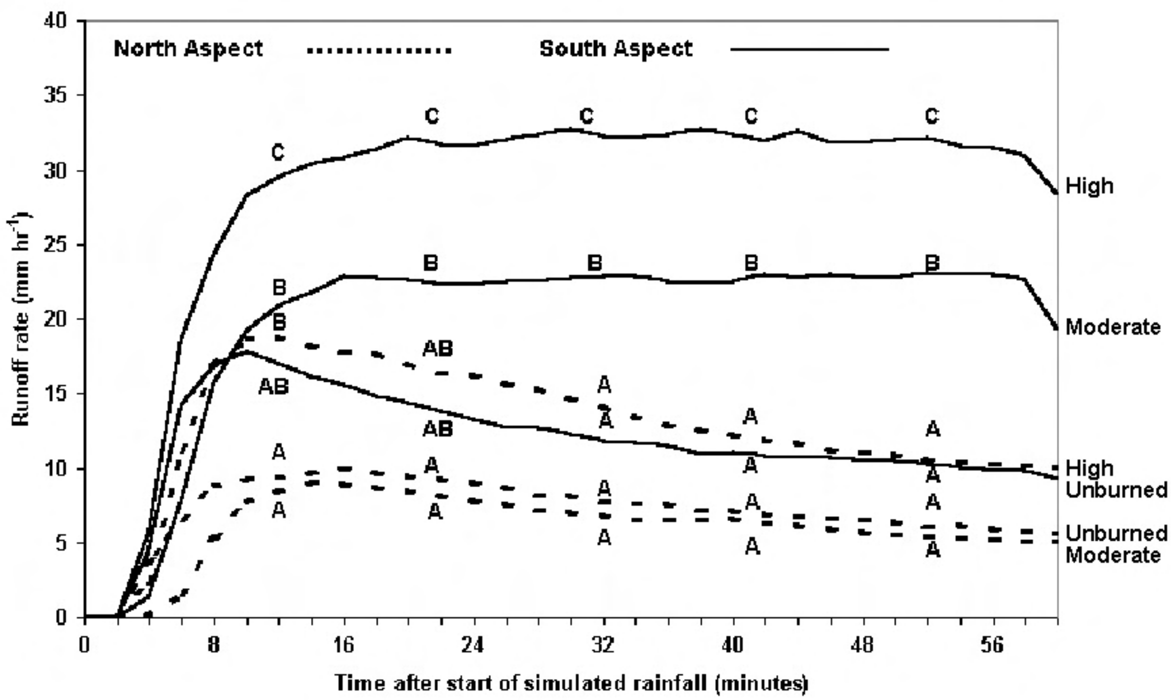


Figure 1. Runoff rate over time for burned (high and moderate fire intensity) and unburned, north and south slopes, Eighth Street Fire, Boise Front, Idaho. Means for each time period with a different letter are significantly different ($P \leq 0.05$).

Microtopography (R=0.51), clay content (R= -0.41), litter biomass (R=0.38), shrub canopy cover (R=0.35), and litter basal cover (R=0.35) were the variables most correlated with infiltration rate over the entire data set. On north slopes, microtopography tended to be greater on interspace (20.1 mm) vs. coppice (18.1 mm) sites and was unaffected by fire. Greater interspace microtopographic relief was probably associated with the dominance of perennial grasses (bunchgrasses). On south slopes, microtopography was reduced by fire, with the greatest reduction on the high intensity interspace site (9.8 mm) where essentially all vegetation and litter was removed. Ground cover (primarily litter) on unburned north slopes approached 100% for both coppice and interspace, but litter biomass was 4 times greater on coppice as compared to interspace sites. South slope unburned coppices had greater basal cover and litter biomass than the interspace sites. Basal cover was much reduced on all burned slopes (1.8 to 5.0 %), as was litter biomass. Virtually none of the litter remained after the fire, and litter biomass had not significantly increased one year after burning.

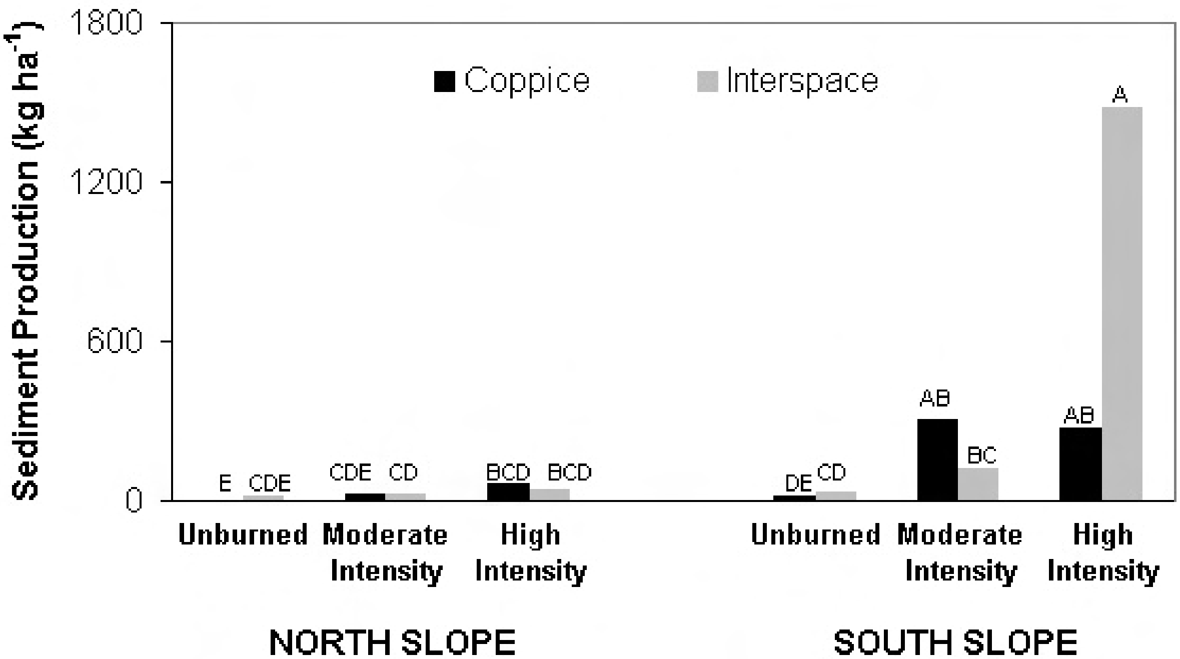


Figure 2. Average total sediment yield (kg ha^{-1}) for burned (high and moderate fire intensity) and unburned coppice and interspace areas on north and south slopes, Eighth Street Fire, Boise Front, Idaho. Means across aspect and fire intensity with a different letter are significantly different ($P < 0.05$).

Microtopographic relief and vegetation can provide crucial surface storage during the initial non-wettable period. Surface storage reduces the amount of runoff by holding water on the plot so that it can eventually infiltrate. Rainfall intensity rather than duration is the critical factor in determining hydrologic response on steeper slopes where initial storage capacity is limited. The study showed that north slopes experienced less runoff even on high intensity burn areas because the microtopography was not significantly reduced despite reduction in litter and cover. This was likely due to the predominance of robust perennial bunchgrasses on the site. The south slopes, however, lacked the microtopographic relief to provide initial storage, so most of the precipitation quickly ran off.

Denio, Nevada: All results presented for this study are preliminary and do not represent rigorous statistical analysis of the data. The study has not yet concluded and final conclusions are not possible.

All burned and unburned sites showed a rapid runoff response with overland flow beginning within 0-4 minutes after initiation of rainfall (85 mm hr^{-1}) (Figure 3). This result is consistent with the rapid hydrologic response measured on the Boise Front. Runoff from all unburned sites showed greater initial runoff from interspace areas compared to coppices, consistent with findings from other studies of the effects of coppice areas on infiltration processes (Pierson et al. 1994). In contrast, runoff from burned sites all had greater initial runoff from coppices that gradually decreased throughout the rainfall period

(Figure 3). In addition, fire had the greatest impact on erosion from burned coppices as well (Figure 4). However, the fire impacts on infiltration and erosion were relatively small, but do indicate that fire can produce a hydrophobic condition on the densely vegetated coppices whereby infiltration is decreased and soil erodibility is increased.

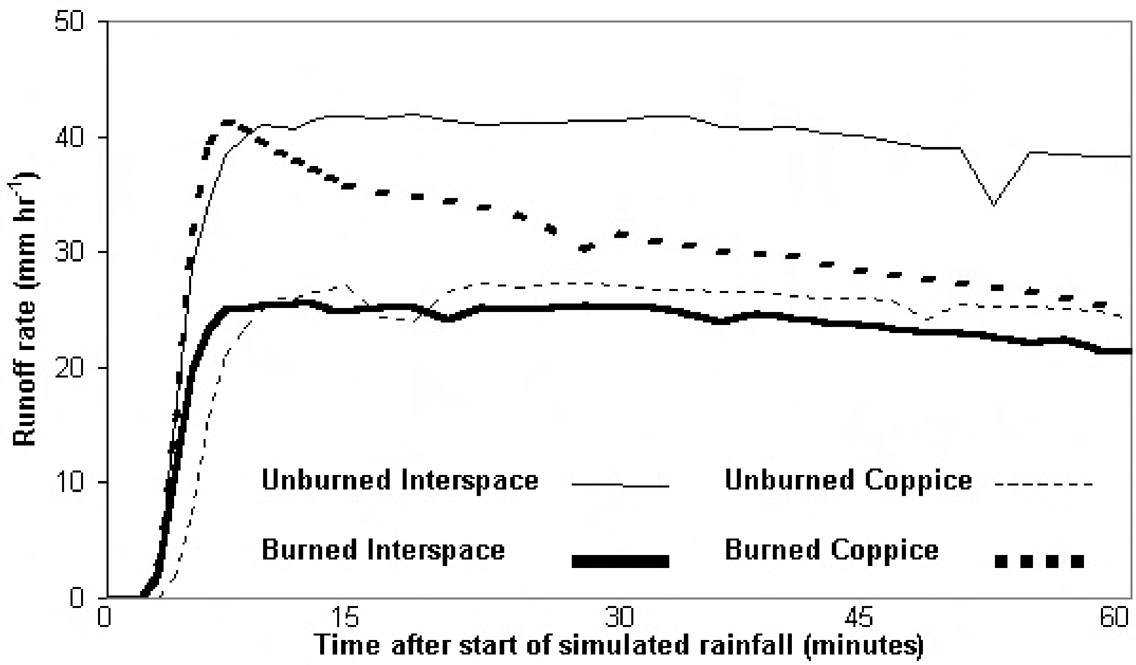
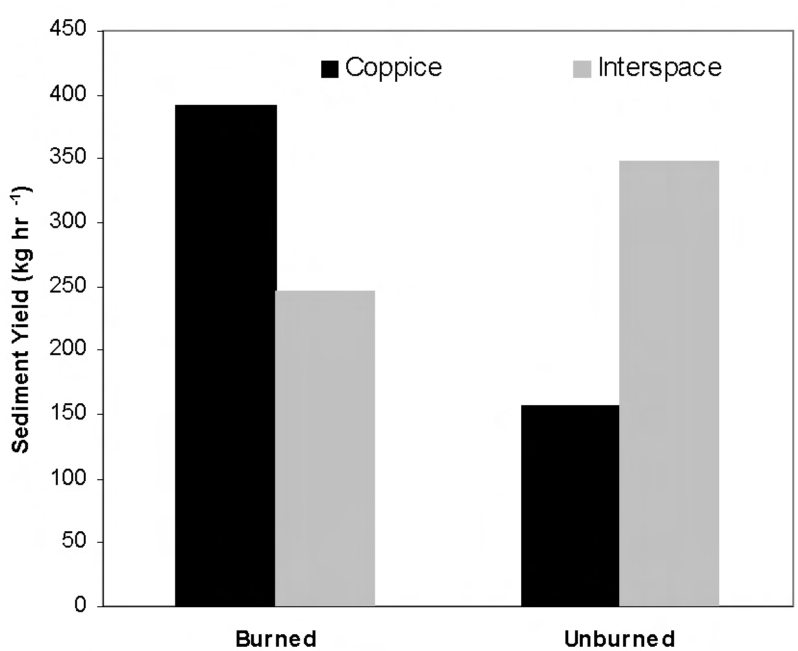


Figure 3. Runoff rate over time for burned and unburned coppice and interspace areas, Denio Fire, Pine Forest Range, Nevada.

Figure 4. Average sediment yield for burned and unburned coppice and interspace areas, Denio Fire, Pine Forest Range, Nevada.



Significant rilling was observed on the burned Denio study sites following periods of rapid snowmelt during the first winter following fire (Figure 5). No rills were observed on the unburned sites. Measurements of rill erosion were then initiated, but results have not yet been summarized. However, very strong differences were observed in overland flow characteristics and rill formation between burned and unburned areas (Figure 6). While fire may not significantly affect infiltration and interrill erosion in these sagebrush/grass ecosystems, it may have a profound effect on concentrated flow paths and rill erosion processes.

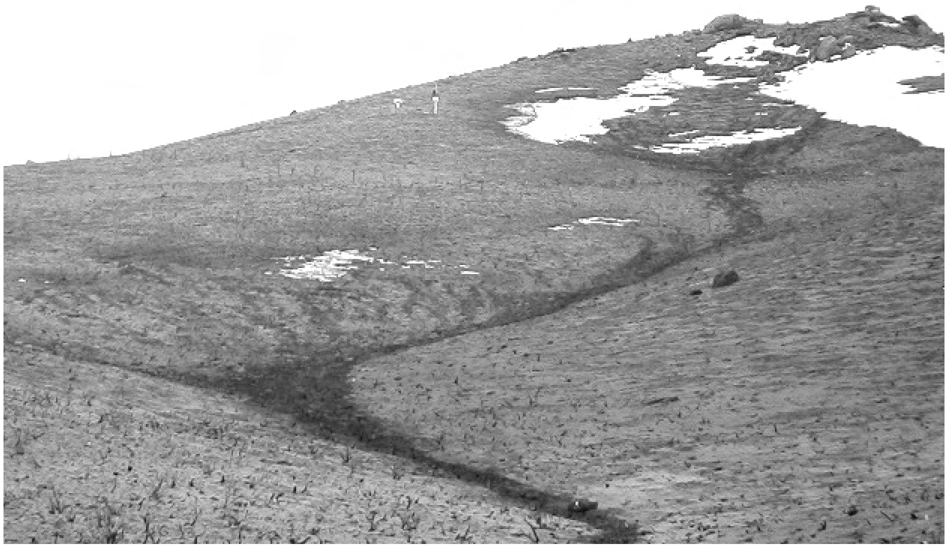


Figure 5. Photo of rill erosion during period of rapid snowmelt in late March following the Denio Fire, Pine Forest Range, Nevada.

Figure 6. Contrasting photos of concentrated flow and subsequent rill formation between intensely burned areas on the Denio Fire and adjacent unburned areas, Pine Forest Range, Nevada.

IMPLICATIONS

The results of these studies provide a relative index of the increased risk of runoff and erosion following fire. The results imply that south-facing slopes are at greater risk for increased runoff and erosion, and this risk increases as the fire intensity increases. Increased risk comes primarily from the interspace areas that have become devoid of protective vegetative and basal cover. Follow-up treatments for rehabilitation may be most cost-effective when applied to these south-facing slopes. While initiation of runoff (0-4 minutes) was similar for all of these sites, the amount of runoff was primarily determined by both initial storage capacity (microtopography) and soil surface characteristics (soil properties and vegetation).

Burning of litter can create hydrophobic soil conditions in forest systems, and is common where fire burns vegetation with waxy coatings or secondary compounds, resulting in the wax reforming a coating on soil particles (DeBano 1981). This type of fire effect did not seem to dominate on sagebrush grassland. Natural hydrophobicity possibly due to soil surface dryness appeared to be a natural part of the system, at least during dry summer periods. Therefore, fire-induced soil hydrophobicity was not an apparent factor in increasing runoff and erosion from steep burned slopes in this study. Microtopographic relief following fire seemed to provide the best indicator of infiltration rate and sediment yield following fire on steep slopes.

Unanswered Questions: While the Boise Front and Denio studies provide insight into hydrologic processes on steep rangeland slopes and the impacts of wildfire, there are still many unanswered questions. Although hydrophobic soils seem to be a natural part of sagebrush plant communities in dry summer periods, how does the response differ with season of the year? Is this type of hydrophobicity consistent with similar plant communities on different soils? Does wildfire exacerbate or initiate hydrophobic soils on other rangeland communities? If so, what is the spatial and temporal distribution of hydrophobic soils? Is microtopography an important factor on all steep rangelands, or is the factor specific to certain plant community/soil associations? What is the immediate response of sagebrush communities to wildfire, and how does the response change over time? How does recovery differ between sagebrush communities? Are wildfire effects highly site specific, and how can we begin to provide predictive capabilities about the impact of wildfire?

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SEDIMENT TRANSPORT REGIMES AFTER A WILDFIRE IN STEEP MOUNTAINOUS TERRAIN

John A. Moody, Hydrologist, U. S. Geological Survey, Denver, Colorado;
U.S. Geological Survey, MS 413, Denver Federal Center, Lakewood, CO, 80225; phone: 303-236-0606; fax: 303-236-5034; jamoody@usgs.gov;

Abstract: Intense rains after a wildfire created catastrophic erosion and deposition in a mountainous watershed. The result was a slug of sediment filling the main channel and parts of many tributaries up to 4 m thick. This changed a supply-limited system to a transport-limited system. During the four years after the wildfire, three different transport regimes were identified. These regimes were not persistent enough to transport sediment as a wave but changed frequently, thus altering the morphology of the sediment slug.

INTRODUCTION

Wildfires are a natural disturbance that lowers the intrinsic threshold of erosion in a watershed; when followed by rain in steep mountainous-terrain, a dramatic erosional response can occur. Steep mountainous chan nels, in general, are supply-limited systems, but the erosion-response after a wildfire can provide so much sediment that these channels become transport-limited (Martin and Moody, unpublished data). Conditions for sediment transport in these mountainous channels and in similar ephemeral channels differ from the widely studied low-gradient, transport-limited sand channels (Leopold and Emmett, 1977; Brownlie, 1981) and from the steeper and rougher, supply-limited gravel channels (Milhous, 1973; Parker et al., 1982). Channel slopes often exceed 2%, the relative roughness (particle diameter/water depth) is nearly 1, and the beds are sometimes composed of an unsorted coarse mixture of sand and gravel rather than well-sorted sand. Other natural disturbances such as hurricanes and volcanic eruptions can also create transport-limited systems as do some anthropogenic activities such as mining and clear cutting. The eruption of Mount St. Helens in 1980 created a debris avalanche and debris flows that deposited slugs of sediment in the Toutle, Cowlitz, and Columbia River channels (Voight et al., 1981; Simon, 1999). Mining activities often deposit slugs of sediment into channel networks (Gilbert, 1917; Knighton, 1989). These inputs of sediment resulting from disequilibrium of the fluvial system have been defined by Nicholas et al. (1995) to be sediment slugs if they persist over time scales greater than the time of a flood event. They are classified by the spatial scale and the impacts they impart on the fluvial system. Macroslugs are controlled by in-channel processes, scale as the channel width, and result in minor channel changes . Larger megaslugs are controlled by local sediment supply and valley-floor configuration, and cause major channel changes whereas still larger superslugs are controlled by watershed-scale sediment supply and cause major changes in valley-floor morphology.

The movement of sediment slugs in flumes and perennial streams and rivers has been modeled as translational and dispersive sediment waves (Gilbert, 1917; Pickup et al., 1983; Knighton, 1989), as stationary and dispersive sediment waves (Lisle et al., 1997), as sediment bores (Needham and Hey, 1992), or as a stochastic process characterized by random particle movement (Pickup and Higgins, 1979; Griffiths, 1994). Wave-like transport of natural bedload has been described by Meade (1985) for the East Fork River in Wyoming over a spatial scale equal to about 100 channel widths. He related transport and storage to the changes in slope over pools and riffles caused by the increase and decrease in water level in response to the unsteady flow from snowmelt. Wave-like pulses of sediment have also been observed at fixed points during the duration of a flood event (Lekach and Schick, 1983) in ephemeral channels.

This paper describes the transient sediment-transport regimes of a superslug that was created by initial basin-scale erosion after a wildfire. The time scale (about 4 years) was greater than the flood-event time scale but shorter than the time scale for complete recovery of the channel to pre-fire conditions. This system, unlike many described and modeled in the literature, is characterized by unsteady and intermittent flow with a non-uniform channel; it represents typical systems found in steep mountainous terrain burned by wildfires.

BACKGROUND

The wildfire burned 4690 hectares in two watersheds (Buffalo and Spring Creeks) located in the Colorado Front Range southwest of Denver, Colorado in May 1996. This area of the Colorado Front Range is characterized by short duration, intense rainfall events in the summer (Henz, 1974) that produce flashfloods and longer duration runoff from snowmelt in the winter and early spring. The main channels are steep (the slope of Buffalo Creek is about 1% and the slope of Spring Creek is about 4%) and baseflow in these channels may be elevated because the tree canopy and ground vegetation was destroyed which decreased evapotranspiration (Figure 1). The primary erosional event was a thunderstorm on 12 July 1996 that was much greater than the estimated 100-yr, 1-hr rainstorm (Miller et al., 1973; based on empirical equations developed from 6 and 24-hr precipitation data). The initial erosion in Buffalo Creek was relatively minimal, and subsequent sediment deposition was in the form of alluvial fans from each tributary with sediment thickness decreasing in the main channel downstream from the fan. Although the Buffalo Creek flood plain was buried near the mouth of each tributary, it was, in general, preserved throughout the length of the valley. Erosion and deposition in the main, east-west channel of Spring Creek was much different. Initial erosion (probably during the rising limb of the hydrograph) occurred across the entire valley and removed any pre-existing flood plain; alluvial fans were deposited at the mouths of tributaries (probably on the falling limb of the hydrograph) and were connected with in-channel sediment deposits up to 4 m thick, producing a sediment superslug in Spring Creek occupying about 5000 m along the main channel and filling the entire valley.

METHODS

Morphology The morphological changes of this superslug were monitored for four years (June 1996 to June 2000) by a series of closely spaced channel cross-sections in a study reach near the mouth of Spring Creek. This reach, which was 1490 m long and extended from the mouth upstream to a stream gage, was selected to encompass more than one wavelength of degradation and aggradation of a possible sediment wave. The width of the valley in Spring Creek was about 30 m; cross sections were spaced 10 m apart to provide detailed measurements of changes in morphology and in the total volume of stored sediment. Changes in volume at several neighboring cross sections were very similar during 1997, so in 1998, 1999, and 2000 the interval between cross sections was increased to approximately 30 m. Initially, the absolute location and elevations of each cross section were measured by using a Nikon 720 DTM, but after the absolute location had been established, a surveying level, metric tape, and surveying rod were used to remeasure the elevations at each cross section. Initial post-fire but pre-flood bed morphology was determined from 1:12,000 stereo photographs taken in June 1996, and the morphology of the superslug was determined as the difference between the pre-flood and additional post-flood 1:12,000 stereo photographs taken in August 1996.

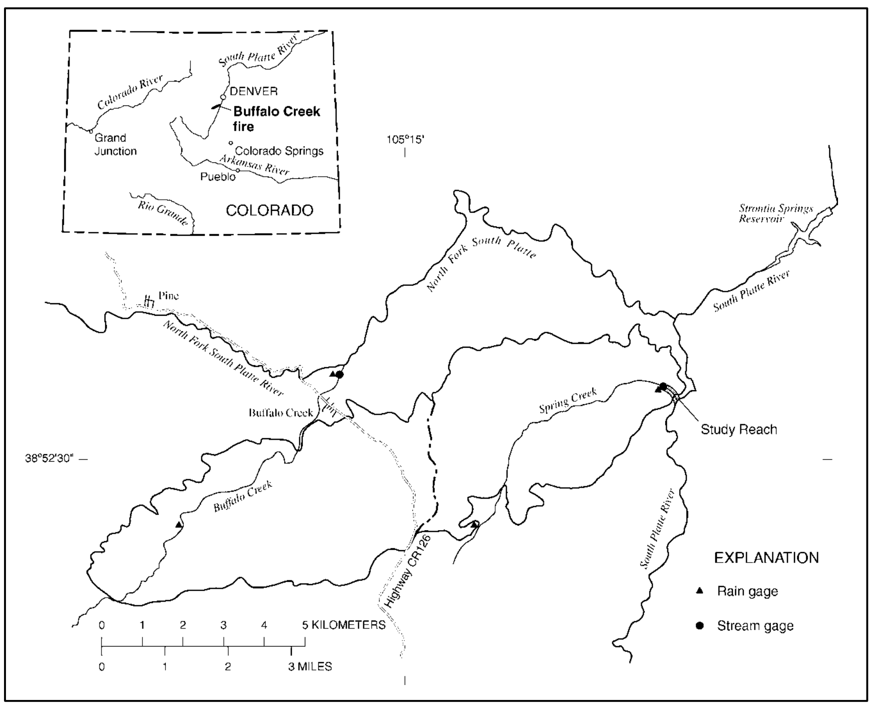

Figure 1. Location of the areas burned in two watersheds (shown by different crosshatching patterns) at elevations between 1880 and 2360 meters above sea level. Transport regimes were monitored in the study reach near the mouth of Spring Creek.

Particle-size Distributions Particle-size characteristics of the superslug were determined by collecting: 1) sediment cores from unburned hillslopes to characterize the source of sediment, 2) surficial-sediment samples to assess longitudinal variations , 3) vertical stratigraphic samples to differentiate between bedload and flash-flood deposits, and 4) a large-volume sediment sample coupled with surface mapping of boulder deposits to characterize all the bed material available for transport. The particle-size distribution for each sample was measured by using standard sieves at 1-phi intervals after 15-20 minutes on a RoTap machine. The distributions were computed as the slope of a third-order polynomial spline fit to the cumulative data (R.F. Stallard, per. comm., 1997), and are characterized by the median diameter and the dispersion or phi deviation (Inman, 1952) which equals 1.0 in a very well sorted sample with only one phi size class.

RESULTS AND DISCUSSION

Initial Morphology Cross-valley and longitudinal surface topography of the superslug in the main channel of Spring Creek appeared relatively smooth and was confined by steep and sometimes nearly vertical hillslopes with colluvial material as well as bedrock outcrops of the easily eroded Pikes Peak granite (Moore, 1992). The longitudinal surface topography was broken by several steep bedrock outcrops which had been eroded and had essentially no sediment deposits. Within the 1490-m study reach, the valley width was narrowest at the upstream end (10 m), widest at the mouth (45 m), and the mean valley width was 27 m (Figure 2A). Pre-flood bed slopes were slightly more variable (standard deviation = 38%) than the widths (standard deviation = 20%) and the mean bed slope was 0.040. After the flood, the superslug had a mean bed slope of 0.041 and was less variable (standard deviation =27%) than the pre-flood channel (Figure 2A). At each channel cross-section, the sediment deposition was measured as the change in area or change in sediment volume per unit channel length by using photogrammetry methods. This area was converted to an equivalent thickness for a uniform channel with a width of 27 m. The mean sediment deposition was 0.6 m thick (Figure 2B). It was thickest at the mouth (2.6 m), which represents a truncated deposit that had dammed the South Platte River, and thinnest at narrow cross sections (0.20-0.23 m) where bedrock outcrops were exposed in the channel.

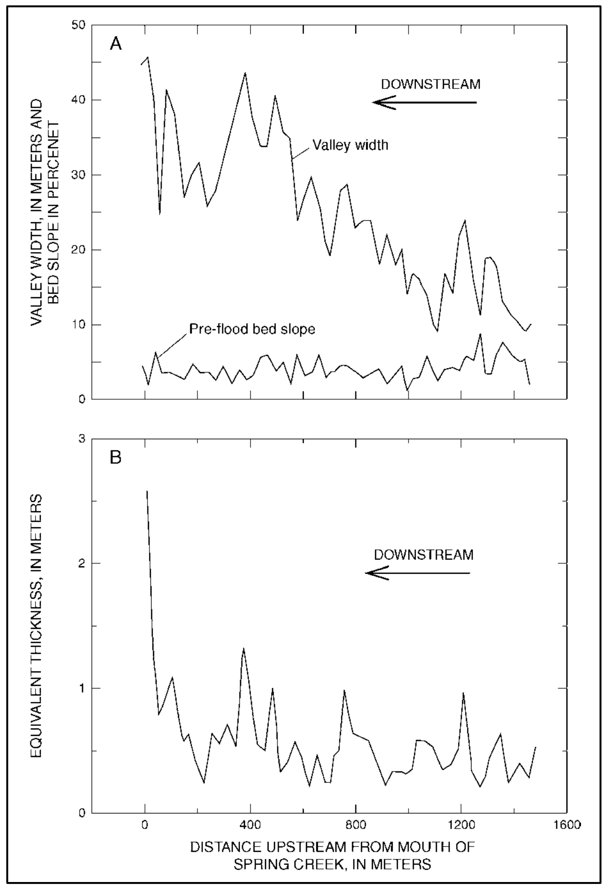

Figure 2. Initial morphology of the superslug created by severe erosion after a wildfire. A. Valley widths were measured in 1997 and the bed slopes (before the flash flood on 12 July 1996) were measured approximately every 30 m from 1:12000 aerial photographs. B. Equivalent thickness of sediment is the thickness in a uniform channel 27 m wide. Sediment deposited after the flood was measured as the difference in elevations between pre- and post-flood aerial photographs.

Particle-Size Bed material composing the superslug had a bimodal distribution with components from the hillslopes and from the pre-flood channel banks. Particle-size distribution of the hillslope sources had a median diameter of 2.8 mm, a dispersion of 4.6, and was similar to the particle-size distribution of the surficial sediment of the superslug (Figure 3). Characteristics of surficial sediment varied from upstream to downstream and indicated that some sorting had occurred. At about 5000 m upstream from the mouth of Spring Creek, the median diameter of the surficial sediment was about 2.5 mm and was consequently better sorted (dispersion = 3.4) than the hillslope sources. The median diameter increased downstream and was about 4.4 mm (dispersion = 2.9) at the mouth.

The second component of the bimodal distribution came from channel banks and had a median diameter of 110 mm and a dispersion of 3.2. This material was eroded from the outcrops of Pikes Peak granite which is often decomposed at the soil interface and easily broken into gravel-sized material (grus) or plucked as boulder-sized material by the flow. Boulders often formed bars on the surface after the major floods, and field observations suggest they may have been deposited along the edge of an eddy in expanding reaches during the flood.

Horizontal layers of sediment were observed in the eroded banks of the superslug. However, detailed sampling indicated that apparent layers of coarse sand and gravel interbedded with fine sand were not continuous over distances much greater than 1-2 m. Rather than layers, these flash-flood deposits were actually lenses of coarse sand and gravel similar to bedload deposits in reaches of braided channel. Here, bedload was observed to be deposited by very shallow flow (0.01-0.05 m deep) as coarse sand and gravel mid-channel bars or islands (about 0.2-0.5 m wide and 0.5-1 m long) that randomly diverted the flow in these braided reaches. These small bars or islands were later covered by finer bedload material transported by a diverted channel from upstream and formed lenses of coarse sand and gravel. The median diameter of the coarse lenses in the flash-flood deposits (4.7 mm) was larger than those of the bedload deposits (3.6 mm) but neither was well sorted (dispersion >3.5).

<u>Sediment Transport Regimes</u> Three sediment transport regimes were identified. These are combinations of flow conditions and bed conditions and are referred to in this paper as uniform, discontinuous, and unsteady regimes.

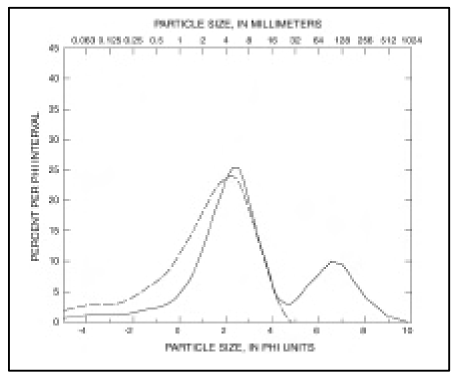

Figure 3. Particle-size distribution of the hillslope source of sediment (dashed line) and the channel bed material (solid line) composing the superslug. The median diameters of the hillslope source, the surficial sediment in the bed material, and the bank material in the bed material are 2.8 ±0.4, 4.6 ±2.7, and 110 ±3.2 mm respectively.

The uniform regime had steady and spatially continuous flow and non-cohesive and cohesive bed conditions depending upon the time of year. Relatively uniform water discharge (0.074- 0.21 m^3s^{-1}) occurred in the early spring from snowmelt and during the summer as prolonged and possibly elevated baseflow from summer rains percolating into the highly fractured granite. This flow eroded the non-cohesive surficial sediment of the superslug in some reaches creating an incised channel about 1-2 m wide with characteristically 0.5 high banks and a bed armored by the coarse bed material (median diameter of 110 mm) derived from the banks (Figure 3). Sediment was transported downstream (0.32-1.2 kg $s^{-1}m^{-1}$) and deposited in other reaches creating braided channels about 20-40 m wide which filled the entire valley. This alternating pattern of incised and braided reaches was observed along the entire 5000 m length of the superslug and was measured in detail in the study reach. For example, between October 1997 and May 1998 (Figure 4) when there were about 60 days of active transport during snowmelt runoff in the spring, the volume of sediment (4700 m^3) eroded in the reach between 1390 and 745 m was a little greater than the volume of sediment (3500 m^3) deposited downstream in the reach between 745 and 110 m. The difference in volume represents an average transport rate of 0.4 kg s^{-1} (using a bulk density of 1700 kg m^{-3}) out of the study reach. During the winter, relatively uniform discharge (about 0.02 m^3s^{-1}) also occurred from snowmelt on south-facing hillslopes; however, the surficial bed material was now frozen (and therefore cohesive). As a result, the incised reaches were narrower and had higher banks (~1 m). The sediment transport downstream from these narrow incised reaches often exceeded the carrying capacity of the narrow channels, resulting in overbank flows that created levees and irregular mounds of sediment as the sediment refroze during the night. Repeatedly during the winter seasons, sediment accumulated as an in-channel fan in the reach between 700 and 600 m, which was also a storage reach during the other seasons.

The discontinuous regime had very low discharge (<0.01 m^3s^{-1}) flowing over the non-cohesive surficial sediment. This regime frequently occurred during the summer. Surface water was discontinuous, disappearing into the bed material, depositing the bedload, and creating an in-channel fan. The fan moved very slowly upstream by deposition on the upstream edge and erosion on the downstream edge as the subsurface water emerged, creating a slip face on the fan and a narrow (~0.5 m) incised channel.

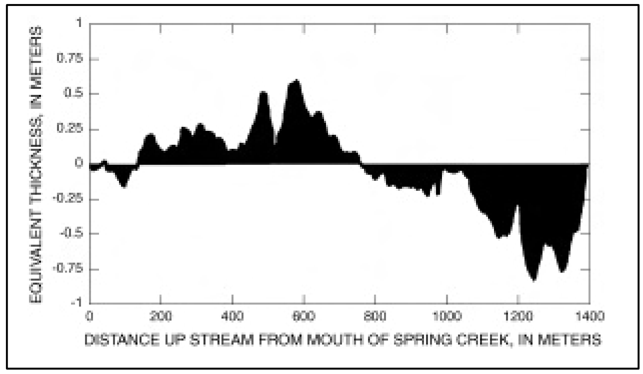

Figure 4. Equivalent thickness of eroded (negative) and deposited (positive) sediment in the main channel of Spring Creek between October 1997 and May 1998 (see figure 6, time interval 6).

The unsteady regime represents relatively short times when the discharge changed from 0.02-0.20 m^3s^{-1} to 20-180 m^3s^{-1} during flash floods caused by heavy rainfall from summer thunderstorms. Rainfall in excess of about 10 mm h^{-1} seemed to produce general overland flow and flash floods. Time to peak discharge on the rising limb was typically 0.1-0.5 hours, and the falling limb usually lasted longer than 3 hours. The number and magnitude of flash floods was greatest in 1997 (1 year after the fire) and has since decreased. The initial flash flood on 12 July 1996 was the largest, producing a watershed average unit discharge of 24 $m^3s^{-1}km^{-2}$. The next largest event was on 31 August 1997 (unit discharge of 8.6 $m^3s^{-1}km^{-2}$), and third largest event occurred on 31 July 1998 (2.8 $m^3s^{-1}km^{-2}$). Much of the sediment which had moved as bedload (1-32 mm) during other regimes was probably transported as suspended load during some of these flash floods. During the 31 August 1997 event, the estimated minimal total transport was 4000 kg s^{-1} (based on volume of sediment deposited in the study reach which acted like a bedload trap because of its expanding width); about 80% was suspended load, and 20% was bedload.

These flash floods were, in general, net depositional processes resulting in aggradation. The average equivalent thickness of deposited sediment after the12 July 1996 flood was 0.6 m, after the 31 August 1997 flood it was 1.0 m, after 31 July 1998 flood it was 1.03 m, and by May 2000 the sediment in the study reach was 1.1 m thicker than in June 1996. The deposition, however, within the study reach is spatially variable (Figure 2) and seems to reflect the spatial variability in bed slope and valley width.

Deposition Model: Deposition from major flash floods that filled the entire valley were predicted by a simple analytical model involving bed slope and valley width. This was an extension of the observations by Meade (1985) that deposition depended upon variations in slope to include deposition that depended upon variations in width. The analytical model predicts net deposition or net erosion for a single event but does not predict the changes during an event. Deposition from smaller flash floods could not be predicted with this analytical model because the water surface width was less than the valley width, confined within very erodible banks, and therefore the width was unknown. The sediment transport, G, was assumed proportional to the total shear stress to 3/2 power, the shear stress was estimated as the depth-slope product, and the velocity approximated by Manning's equation. For constant water discharge the sediment transport is then:

$$G = C\frac{S^{21/20}}{w^{9/10}}$$

where C is a proportionality constant, S is the bed slope, and w is the valley width. Deposition and erosion within a reach are then proportional to the difference between the transport at the upstream and downstream boundaries. These boundaries were the channel cross sections which were approximately 30 m apart for the initial flood on 12 July 1996 (when the deposition was determined photogrammetrically) and were approximately 10 m apart for the flood on 31 August 1997 (when the deposition was determined by ground surveys before and after the flood). The spatial variability predicted by this simple model is very similar to the actual deposition (Figure 5A) after the 12 July 1996 flood. When differences between sediment transport were computed over 3 reaches (~90 m), the difference between predicted deposition and measured deposition was less. The difference between the predicted and measured deposition was relatively greater between 1000 and 1400 m where the valley is narrow than between 20 and 1000 m where the valley is much wider. A possible explanation is that the bed roughness may have been greater in the wider reaches where willows and cottonwood trees were more numerous than in the narrower reaches. This analytical model assumes a constant roughness, and if the roughness in the narrow reach between 1000 and 1400 m was reduced there would be a decrease in the predicted thickness and an improvement in the prediction by the model. Differences between the measured and predicted deposition is also large near the mouth of Spring Creek at the junction with the South Platte River where the channel conditions change abruptly. Actual deposition was probably controlled by the water level in the South Platte River which created an effective slope that was possibly much smaller than the slope in the Spring Creek channel upstream from the mouth used in the model. This simple model was also applied to the deposition from the 31 August 1997 flash flood (Figure 5B) and once again reproduced the general spatial pattern of deposition which for this flood was measured at about 10 m intervals. The best predictions of deposition, however, were achieved when the bed slope was spatially averaged over 3 reaches (about 30 m) producing "regional" slopes similar to those used for the initial flood in 1996. In addition, the differences in the measured and predicted thickness were less when differences between sediment transport were computed over 3 reaches (~30 m) or approximately the valley width. Deposition seems to be controlled by a "regional" slope averaged over the distance of several reaches upstream. This distance was larger (~ 90 m) for the 12 July 1996 flash flood (estimated peak discharge of 510 m^3s^{-1}) than the distance (~30 m) for the 31 August 1997 flood (estimated peak discharge of 180 m^3s^{-1}).

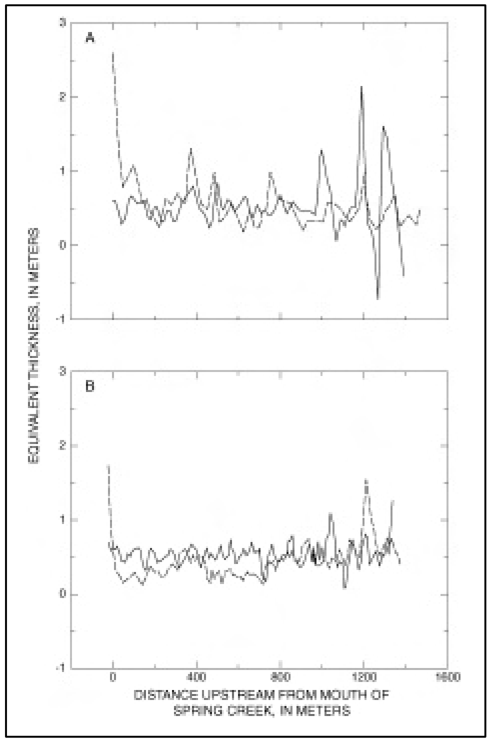

Figure 5. Analytical model for net erosion and deposition as a result of a flash flood. A. The measured equivalent thickness (dashed line) for the 12 July 1996 flash flood (see figure 2 and figure 6, time interval 1) is compared to the predicted thickness (solid line). The proportionality constant C was set arbitrarily to 10 to produce similar amplitudes and an offset of 0.5 m was applied. B. The measured equivalent thickness (dashed line) for the 31 August 1997 flash flood (see also Figure 6, time interval 4) is compared to the predicted thickness (solid line) where the proportionality constant C was set arbitrarily to 10 to produce similar amplitudes and an offset of 0.5 m was applied.

CONCLUSIONS

The transport of sediment for this superslug can not be described by a sediment wave during the transient period consisting of the first four years after its creation. Three different transport regimes existed at different times and for varying lengths of time and no single transport regime existed long enough in space or time to consistently move sediment. The unsteady regime is represented by time intervals 1, 4, 9, 10, and part of interval 14 (Figure 6), the discontinuous regime is represented by intervals 2, 3, 8, and 11, the uniform regime with non-cohesive bed conditions by intervals 5, 13, and part of 14, and 15, and finally the uniform regime with cohesive bed conditions by intervals 6, 7, 12, and 16. No translational sediment wave was observed to propagate downstream, which in Figure 6, would appear as a slug moving from right to left (along the spatial axis) and from top to bottom (along the time axis), and no diffusing stationary wave is evident.

Deposition during flash floods probably occurs on the falling limb of the hydrograph when the sediment is moving as bed load. Flash-flood deposits were coarser than bedload deposits that had formed at lower discharge. Neither deposits had vertical stratification that would indicate they were formed by the settling of suspended sediment. The spatial variability of the deposits depends mostly on the spatial variations of bed slope and valley width but also on the bed roughness which is difficult to determine during a flash flood.

The hillslopes sediment fluxes have decreased to pre-fire in 3-5 years (Moody and Martin, unpublished data), and thus the runoff response to rainfall has decreased and the unsteady nature, characteristic of the transient phase of this transport-limited system, is decreasing. The implication is that with steadier flow a stable and armored channel will probably form within the entire length of the superslug. The bulk of sediment in the superslug will go into long-term storage and will probably move only when the next disturbance such as wildfire changes the erosion threshold of the watershed.

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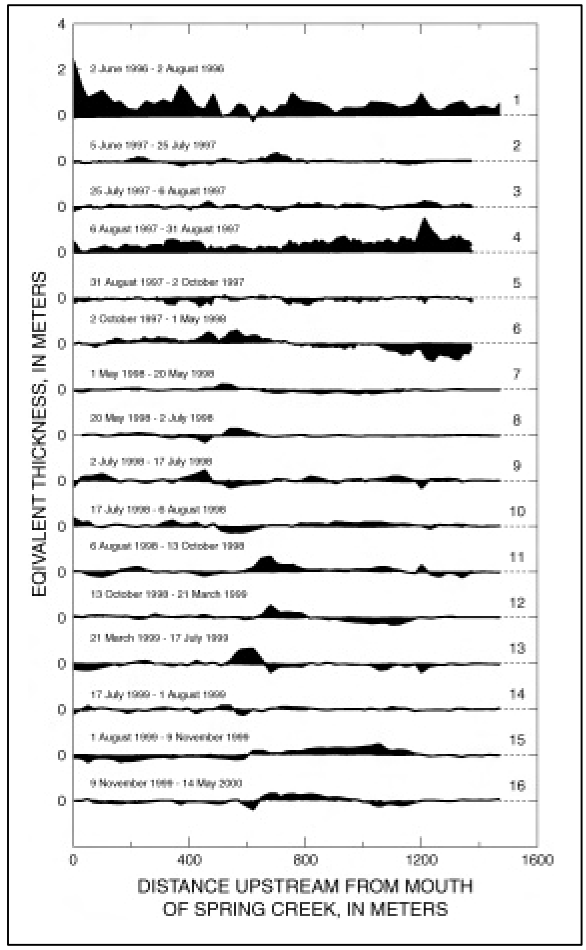

Figure 6. Changes in the morphology of the superslug within the study reach. The changes in equivalent thickness between successive surveys of the study reach are shown with time increasing downward. The time intervals are not equal but represent 16 measurements of change over the four years following the wildfire and initial erosional and depositional event.

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PRESCRIBED FIRE AS A SEDIMENT MANAGEMENT TOOL IN SOUTHERN CAIFORNIA CHAPARRAL WATERSHEDS

Peter M. Wohlgemuth, Hydrologist, USDA Forest Service, Pacific Southwest Research Station, Riverside, California

Mailing Address: Riverside Forest Fire Laboratory, 4955 Canyon Crest Drive, Riverside, California 92507. Phone: (909) 680-1538. Fax: (909) 680-1501. Email: pwohlgemuth@fs.fed.us

Abstract: Land managers in southern California have speculated that prescribed burning could reduce the soil erosion generated by catastrophic wildfires. A unique opportunity to test this notion arose when a wildfire swept over a field experiment measuring hillslope erosion from a prior prescribed burn. Results indicate that 1) fire severity may directly affect erosion response, 2) postfire hillslope erosion levels may return to normal within three years, and 3) prescribed fire may reduce the erosion produced by subsequent wildfires. These results suggest that prescribed fire may be an effective sediment management tool in southern California chaparral brushfields.

INTRODUCTION

In southern California, where human development has encroached on fire-prone chaparral brushfields, the threat of wildfire to life, property, and infrastructure is well documented (Conard and Weise 1998, Riggan and others 1994). In addition, fires render the landscape susceptible to enhanced erosion, flooding, and sedimentation with the onset of heavy winter rainstorms (Rice 1974, Wells 1981). This has prompted expensive emergency watershed rehabilitation measures on the part of land managers to protect downstream values at risk (USDA Forest Service 1992).

Prescribed burning is one possible way to protect communities from catastrophic wildfires. Selectively burning strategic corridors and buffer zones can reduce hazardous fuel conditions adjacent to urban areas (Green 1981). Prescribed burning also creates a mosaic of vegetation age classes on the landscape that may reduce the fire severity in the event of a destructive wildfire and aid suppression efforts (Conard and Weise 1998, Riggan and others 1994). Thus, using prescribed fire as a vegetation management tool can reduce both watershed damage and suppression costs.

Prescribed fire might also be used as a sediment management tool. Burning smaller areas at lower fire intensities should generate less site disturbance and reduce the loss of soil and nutrients compared to a large wildfire (Green 1981). Given the potential for damage and cleanup costs of accelerated postfire erosion, any reduction in postfire sediment yields could be highly beneficial.

Unfortunately, the effectiveness of prescribed fire as a sediment management tool in chaparral has not been adequately demonstrated in the field. Pase and Lindenmuth (1971) reported that a prescribed fire generated only 10 percent of the sediment produced by a wildfire in a comparable area in central Arizona. In southern California, watersheds burned in a moderate-severity prescribed fire yielded only 35 percent as much water and sediment as watersheds burned under a high-severity prescription that was intended to simulate a wildfire (Riggan and others 1994). While both these studies suggest that the erosion response after a prescribed fire is lower than after a natural burn, neither was able to address the impact of prescribed burning on subsequent wildfires.

The purpose of this paper is to report the findings of a field experiment that may illuminate the use of prescribed burning as a sediment management tool. The results document the postfire hillslope erosion response from two adjacent and presumably nearly identical sites for three distinct fire cases: a moderate severity prescribed burn, a moderate to high severity short-interval reburn, and a very high severity wildfire.

STUDY AREA AND METHODS

The Belmar study area is located in the Peña Canyon watershed on the southern slopes of the Santa Monica Mountains above the coastal community of Malibu, California (Figure 1). Although the Belmar area constitutes an unreplicated case study, the geology, soils, climate, and vegetation are typical of much of the east-west trending mountain ranges of southern California (Table 1).

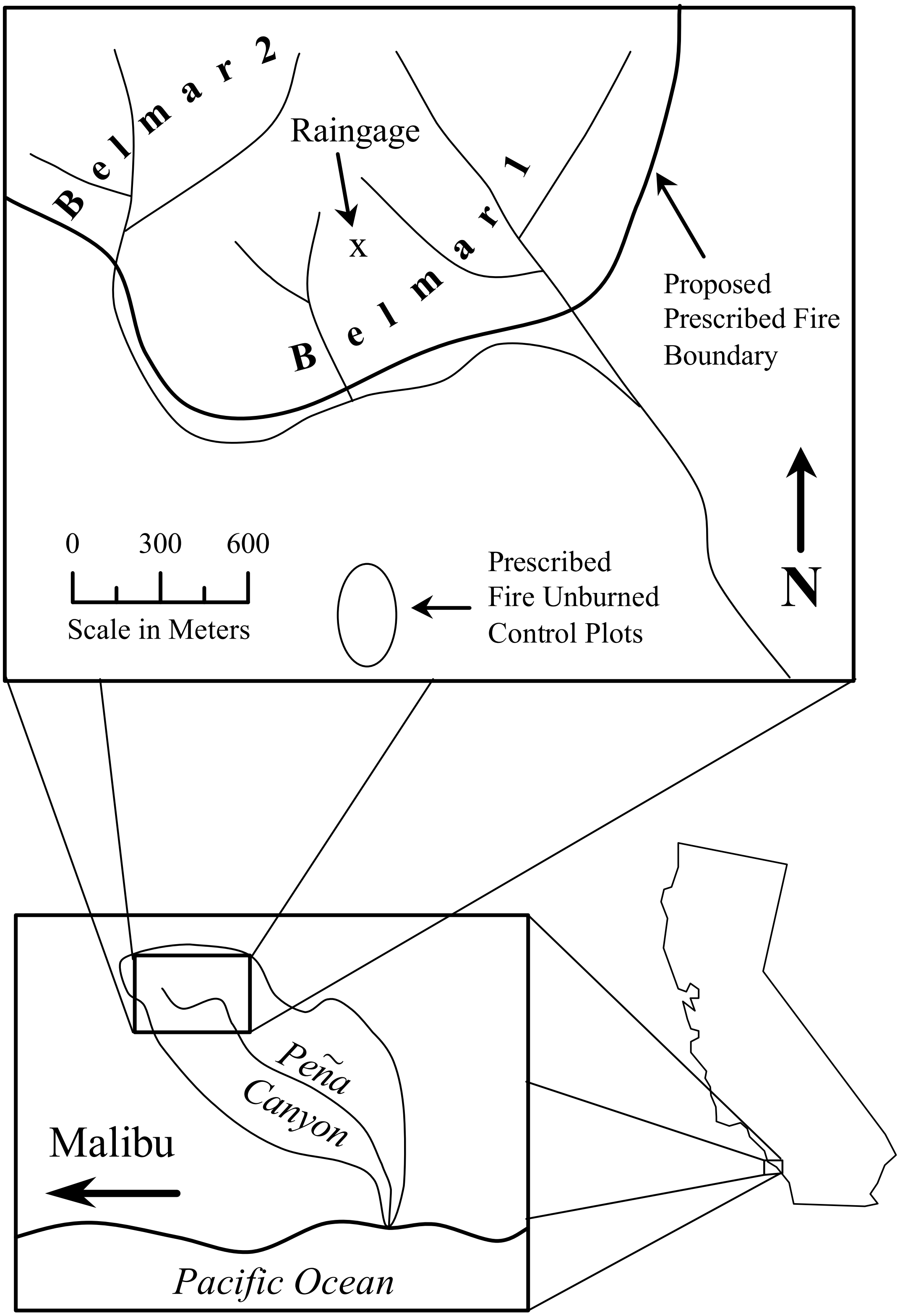

Figure 1. Location of the study area.

Table 1. Characteristics of the Belmar study area.

Attribute	Value
Distance from the coast	2 km
Mean elevation	450 m
Mean aspect	south
Mean slope angle	28 degrees
Mean slope length	37 m
Parent rock type	sedimentary, primarily sandstone
Soil texture	sandy loam
Mean annual precipitation	467 mm
Vegetation type	mixed chaparral
Vegetation cover	40 percent

The Belmar site was established in January 1988 by installing 70 erosion plots over a 60 ha (150 ac) area in 30-year old mixed chaparral that was planned to be burned in a prescribed fire. Each erosion plot consisted of a segmented sheet metal sediment collector trap with a 1.5 m aperture parallel to the slope contour (Wells and Wohlgemuth 1987). The unbordered erosion plots were situated at midslope positions along several interflueves. The potential plot contributing area extended to the hillslope crest, and varied in length from 9 to 115 m. Ten of the plots were established outside the proposed prescribed fire boundary to serve as unburned controls (Figure 1).

The area was burned in June 1988, but, because of a sharp rise in relative humidity during the prescribed fire operation, the fire consumed the vegetation on only about half the site. Based on the depth of soil char, the diameter of the remaining plant stems, and the consumption of ground litter and foliage, the prescribed fire burned with moderate fire severity (Wohlgemuth and others 1998). The 22 completely burned original erosion plots plus 17 plots newly-established after the prescribed fire (n=39) are designated the Belmar1 site (Figure 1).

The entire Peña Canyon watershed subsequently burned in the wind-driven Old Topanga Fire of November 1993. The wildfire burned under more severe weather conditions than the prescribed fire. Consequently, based on the above criteria, the prescribed burn section of the Belmar site reburned with moderate to high fire severity, while the previously unburned vegetation was completely consumed with a very high fire severity (Wohlgemuth and others 1998). The 24 original erosion plots burned in the wildfire but not in the prescribed fire (including the original control plots) plus 12 plots newly-established after the wildfire (n=36) are designated the Belmar2 site (Figure 1).

Sediment was gathered periodically from the collector traps and taken to the laboratory, dried, and weighed. The preburn erosion record for both sites consisted of a single collection of sediment that accumulated between January and June 1988 (Figure 2). Postfire erosion was measured with decreasing frequency for five years after each fire.

The erosion data were aggregated into wet season (roughly November to March) and dry season (April to October) collection periods, based on the precipitation records from an onsite raingage (Wohlgemuth and others 1998; Figure 1).

The original erosion plots that became the Belmar2 site were abandoned after failing to burn in the 1988 prescribed fire. Measurements following the prescribed burn at the Belmar1 site were initially terminated after five years of monitoring in the spring of 1993. Thus, gaps exist in the erosion record of both sites before they were reactivated following the fall 1993 wildfire. Although the magnitudes of the missing data points were probably small, in order to compare the erosion response for both Belmar sites over the entire 10-year life of the project, estimation procedures were needed to fill in these gaps. The missing erosion values for the unburned Belmar2 site were estimated from the relationship between the Belmar2 prefire erosion measurements and those from the unburned controls. The missing values from both sites for the 1993 dry season were estimated by multiplying the median 1992 dry season erosion rate (amount divided by duration) by the duration of the 1993 dry season.

Because of unequal sample sizes, non-normally distributed data, and a desire to avoid numerical transformations in analyzing the data, central tendency and dispersion were characterized as the medians and semi-interquartile ranges of the distributions. Although the same plots were used at the Belmar1 site after both the prescribed fire and the reburn, for consistency in testing against the Belmar2 site, subsequent statistical analysis employed the unpaired non-parametric Wilcoxon rank sum statistic (Dixon and Massey 1969) to compare the postfire erosion responses of the various fire scenarios.

RESULTS

The magnitude of the initial postfire erosion response of the prescribed burn and the reburn appear similar (Table 2), despite the differences in fire severity. Results of the statistical analyses (Table 3) confirm that the two cases were comparable in the first two years postfire, but show that the prescribed burn produced significantly more ($p=0.03$) erosion than the reburn over a five-year study period. In contrast, the erosion response of the wildfire was ten to twenty times greater than the other two cases during the first two postfire wet seasons (Table 2), but tested similar to the reburn and less than the prescribed burn for years 3-5 (Table 3). Overall, sediment production after the wildfire was significantly greater ($p<0.01$) than the prescribed fire or the reburn over a five-year study period (Table 3).

In addition to the moderate fire severity, the modest erosion response after the prescribed burn was probably due to the prevailing drought conditions that produced both low rainfall totals and low intensities (Figure 2). Conversely, the reburn/wildfire experienced a low rainfall year followed by a high rainfall year, then two years of nearly average precipitation (Figure 2).

Despite the relatively rapid recovery period, soil erosion from the wildfire was greater than from the other two fire cases combined. Over a 10-year period for nearly identical site characteristics and rainfall patterns, the prescribed fire and short-interval reburn on the Belmar1 site together produced only 17 percent of the soil erosion generated from the wildfire and estimated preburn record at the Belmar2 site (Figure 2).

DISCUSSION

Factors governing postfire erosion response in southern California--apart from inherent site characteristics--are precipitation, vegetation regrowth, and perhaps the depletion of hillslope sediment supply. Generally, postfire soil erosion is more pronounced in wet years than during sub-normal rainfall years (Rice 1974). A greater cover of herbaceous vegetation regrowth has a better ability to retard soil movement (Beyers and others 1998). There is also speculation (Wohlgemuth and others 1998) that much of the loose, easily erodible soil may be removed in an initial postfire flush, exposing more structured soil material at the surface during subsequent years.

The short-interval reburn produced only moderate erosion initially and very minor erosion during the subsequent years. This probably reflects the patterns of vegetation regrowth, as the herbaceous cover was 2-3 times greater after the reburn than after either the prescribed fire or the wildfire (Beyers and others 1998). Presumably, the seed bank from the herbaceous plant community following the prescribed fire sprouted vigorously after the reburn, and the additional cover afforded greater site protection against postfire erosion, even in a very wet year. This would explain the results of the foregoing statistical analyses, and would account for the lower levels of erosion after the

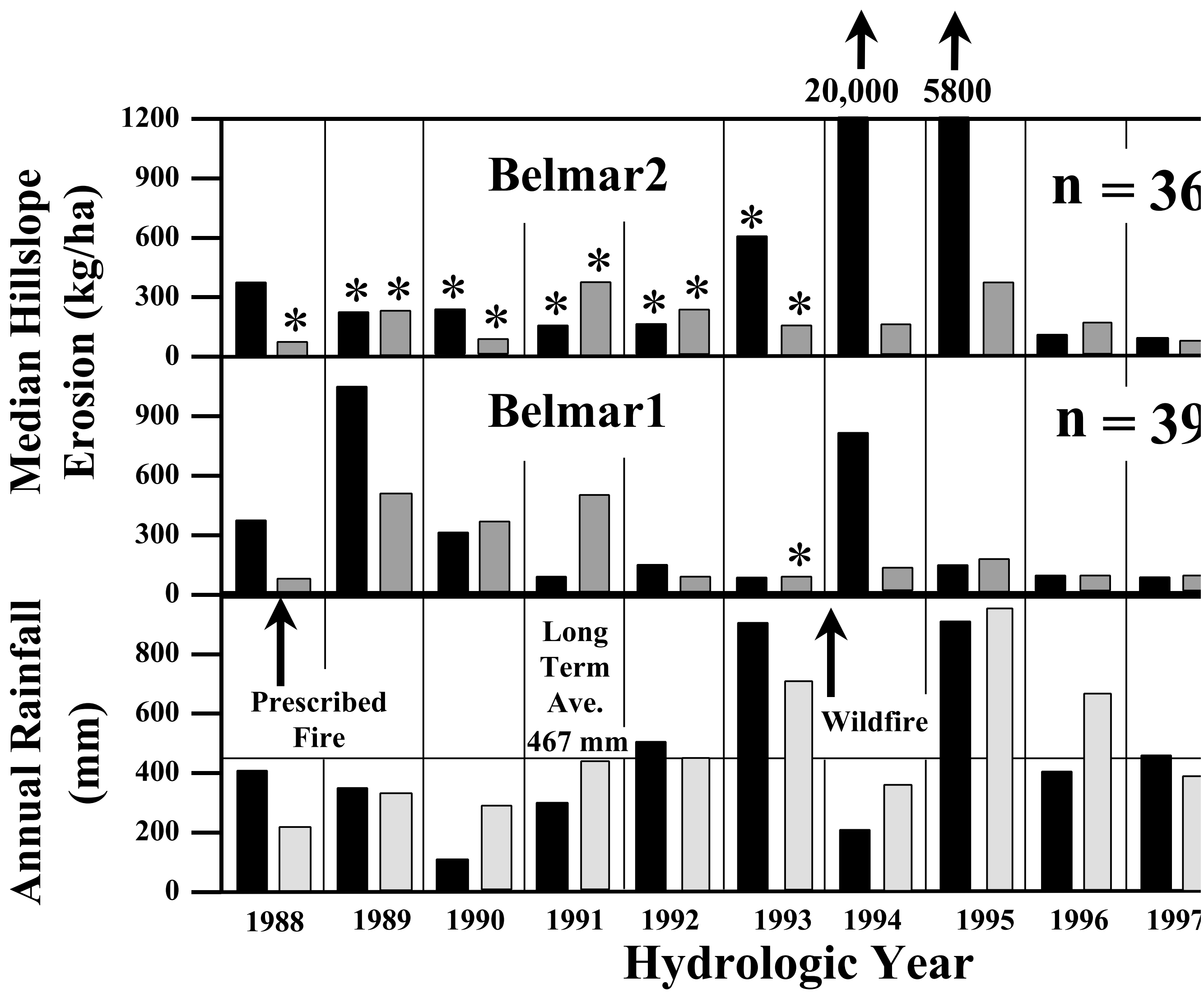

Figure 2. Hillslope erosion and rainfall at the Belmar sites by hydrologic year (October 1 to September 30). The up season erosion; the patterned histograms are dry season erosion. Histogram widths are standardized and do not repr * are estimated values (see text). Sample size for the winter/spring 1988 preburn collection was 22 and 24 for the Be respectively. Sample size for the estimated Belmar2 preburn period was 24. The lower black histograms are annua histograms are rainfall intensity.

Table 2. Postfire erosion by year and season.

		Belmar1		Belmar2
Year postfire		Prescribed burn	Reburn	Wildfire
		-----------------------Kilograms of sediment per hectare		----------------------
Year 1				
	Dry season	55 ± 64[1]	[2]	[2]
	Wet season	1083 ± 430	774 ± 1182	20183 ± 11633
Year 2				
	Dry season	504 ± 432	98 ± 130	126 ± 31
	Wet season	255 ± 334	106 ± 90	5825 ± 5991
Year 3				
	Dry season	337 ± 790	144 ± 152	353 ± 334
	Wet season	85 ± 110	69 ± 104	81 ± 39
Year 4				
	Dry season	502 ± 659	74 ± 116	110 ± 136
	Wet season	121 ± 172	38 ± 20	73 ± 45
Year 5				
	Dry season	73 ± 73	22 ± 27	49 ± 40
	Wet season	59 ± 76	38 ± 16	63 ± 69

[1]Median and semi-interquartile range (the nonparametric equivalent of mean and standard deviation).

[2]As the rainy season commenced shortly after the wildfire, there was no appreciable Year 1 postfire dry season.

Table 3. Results of the Wilcoxon rank sum tests.

Comparison	Score	p-value
Rx Burn vs Reburn, 5 yr	2.229	0.03
Rx Burn vs Wildfire, 5 yr	-9.154	<0.01
Reburn vs Wildfire, 5 yr	-10.926	<0.01
Rx Burn vs Reburn, 2 yr	1.558	0.13
Rx Burn vs Wildfire, 2 yr	-10.206	<0.01
Reburn vs Wildfire, 2 yr	-11.017	<0.01
Rx Burn vs Reburn, year 3-5	3.480	0.01
Rx Burn vs Wildfire, year 3-5	2.345	0.02
Reburn vs Wildfire, year 3-5	-1.552	0.12

reburn than after the prescribed burn. Alternatively, the lower level of erosion in the higher rainfall years following the short-interval reburn may reflect a depletion of hillslope sediment supply.

The wildfire generated substantial sediment movement for both the low and high rainfall years, then abruptly ceased (Figure 2). The amount of herbaceous cover following the wildfire was nearly identical to the amount after the prescribed burn in a similar low rainfall year (Beyers and others 1998). This strongly suggests fire severity is a major factor in governing the magnitude of postfire erosion, and is consistent with previous observations (Pase and Lindenmuth 1971). This reflects the greater degree of site alteration (foliage and litter consumption, surface soil structure disruption, degree of soil non-wettability) associated with higher severity fires. In view of the initial postfire erosion spike following the wildfire, the abrupt return to baseline erosion levels may seem remarkable. However, research has shown that burned hillslopes typically recover to prefire erosion rates in 2-4 years (Wohlgemuth and others 1998).

Just as fire in southern California chaparral is inevitable, so is accelerated postfire erosion. While it has been shown that prescribed fire may reduce the short-term erosion response of a subsequent wildfire, it is unlikely that the long-term (millennial) patterns of sediment production will be affected. Moreover, the optimal prescribed fire recurrence interval for sediment management has yet to be demonstrated. However, it appears that the judicious use of prescribed fire could buffer the erosion spikes produced by catastrophic wildfires, and should be considered by land managers as a legitimate sediment management tool.

CONCLUSIONS

This example from southern California suggests that prescribed fire may be an effective sediment management tool. A wildfire generated twenty times as much sediment as a prescribed burn in the firstpostfire winter with similar amounts of rainfall and vegetation regrowth. Postfire erosion recovered to normal levels in as little as two years, although it is unclear to what degree these measured responses reflect vegetation regrowth or the depletion of loose surface soil. Prescribed fire was associated with lower hillslope erosion from a subsequent wildfire in the Belmar area, with the prescribed fire plus the reburn together producing only 17 percent of the sediment generated by the wildfire.

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CHANNEL CHARACTERISTICS AND LARGE ORGANIC DEBRIS IN ADJACENT BURNED AND UNBURNED WATERSHEDS A DECADE AFTER WILDFIRE

By Ronald B. Zelt, Hydrologist, U.S. Geological Survey, Cheyenne, Wyoming

INTRODUCTION

Wildfire can affect not only vegetation communities but also hydrologic and geomorphic processes. Its geomorphic importance relates to the sensitivity of landscapes to fire-caused changes in vegetation and soils, and consequent changes in hydrology. Increased overland flow and soil erosion may result when vegetation is removed. Reduced transpiration increases soil moisture levels, and increased exposure to sun and wind can alter the magnitude and timing of snowmelt runoff. Changes in burned watersheds impact stream channels through altered hydrology, sediment inputs, or riparian disruption that produce effects on channel erosion, sediment storage and transport, and aquatic life. Debris torrents may dramatically alter stream channels and sediment loads.

Many fire-caused changes have immediate and short-term effects. Delayed or long-term effects are due to such processes as root decay of fire-killed trees, loading of stream channels with large organic debris (LOD), or fluvial adjustments of channels to increased runoff. Current understanding of fire effects on channel characteristics and LOD is limited by the large variability in effects due to the number of affected processes and controls. Due to this variability, the magnitude and duration of fire effects cannot be predicted reliably.

The 1988 wildfire season climaxed in early September as the Clover-Mist Fire burned beyond the eastern boundary of Yellowstone National Park (YNP) and consumed most of the vegetation in Jones Creek watershed, a headwater drainage of the North Fork Shoshone River in Shoshone National Forest (Greater Yellowstone Coord. Comm., 1989). However, less than 8 percent of Crow Creek watershed was burned, despite being adjacent to Jones Creek (fig. 1). The wilderness watersheds of Crow and Jones Creeks provide a rare opportunity to compare fluvial processes in a severely burned watershed with those in an undisturbed adjacent watershed having very similar physical characteristics, without the confounding influences of activities such as timber harvest, roads, or mining.

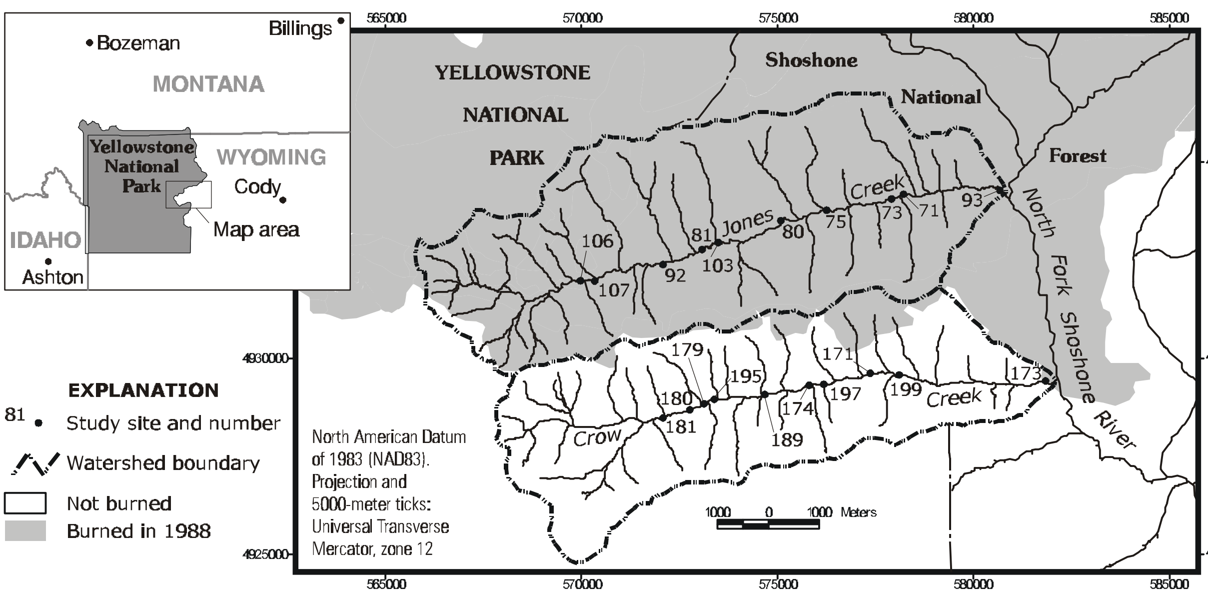

Figure 1 – *Location of study area and study reaches in northwestern Wyoming.*

This paper reports preliminary results from a case study of channel conditions during the postfire recovery of Jones Creek, in contrast with Crow Creek. The study focuses on channel characteristics (including width, substrate, and pool characteristics) and differences in LOD loading and characteristics that may be attributed to severe wildfire disturbance. However, observed differences between the streams cannot establish fire as the causal agent because no pre-fire data exist (see Troendle and Bevenger, 1996). Specific research hypotheses include: (1) Jones Creek is wider and has more fine sediment stored within its channel than does Crow Creek; and (2) following a decade of presumably accelerated input of burned LOD, debris loading is larger in Jones Creek than in Crow Creek.

STUDY AREA

Crow and Jones Creeks drain steep terrain in the northern part of Wyoming's Absaroka Range adjacent to YNP (fig. 1). Elevation ranges from 2030 to 3240 m (above mean sea level) in Crow Creek watershed (49.5-km^2), and from 2082 to 3280 m in Jones Creek' s slightly larger drainage (66.8-km^2). Slope steepness of 70 percent of each watershed ranges between 16 and 60 percent (Troendle and Bevenger, 1996), averaging 42 percent for Crow Creek, and 41 percent for Jones Creek. The distributions of slope aspect and elevation zones also are similar for both watersheds (see Troendle and Bevenger, 1996).

Mean annual temperature is about 0°C, with January mean daily minimum of -17°C and July mean daily maximum of about 22°C (Martner, 1986). Analysis of precipitation and elevation data (G.S. Bevenger, U.S. Forest Service, written commun., 1992) suggests that mean annual precipitation in the study area is about 1350 mm on the ridge tops and decreases to about 450 mm at the watershed outlets.

The northern Absaroka volcanic field consists of deeply eroded andesitic and basaltic lava flows and epiclastic deposits derived from them, some air-fall tuffs, and a variety of related intrusive rocks (Nelson and Pierce, 1968; Smedes and Prostka, 1972). Upper slopes are bedrock outcrops or talus slopes, whereas glacial till and rock outcrops dominate lower slopes. Valley floor deposits include sand and gravel stream deposits, bouldery debris-flow fans, and avalanche debris. The distribution of geologic units is quite similar for both watersheds (see Troendle and Bevenger, 1996).

The distribution of vegetation communities in these watersheds is strongly affected by elevation. Sagebrush dominates the upland communities at elevations below about 2130 m, but vegetation and hydrology change distinctly near that elevation threshold (Marston and Anderson, 1991). Engelmann spruce, subalpine fir, Douglas fir, and whitebark pine compose the predominant climax community in the subalpine zone that extends to upper timberline near 3100 m.

Snowmelt runoff dominates the hydrology of the study streams. Because of the near permanence of the snowpack at the higher elevations, streamflow is sustained between periods of precipitation (Lowry and Smalley, 1993). Postfire streamflow data circumstantially indicate that for Jones Creek a 34 percent increase in water yield occurred during 1989-1992, with the difference appearing during the spring rise of the snowmelt hydrograph (Troendle and Bevenger, 1996). Those daily streamflow data show that the frequency of the largest flows (those exceeded about 1 percent of the time) was similar for both streams, but a flow of 80 L/s/km^2 (exceeded about 5 percent of the time in Crow Creek) occurred nearly twice as frequently in Jones Creek.

Available data for 1989-93 indicate that most of the total annual suspended-sediment yield was associated with high flows generated during snowmelt (Troendle and Bevenger, 1996). However, peak sediment yields for the two streams were associated with summer storms. Annual sediment yield was more than four times larger in Jones Creek than in Crow Creek for 1989-1992 (Troendle and Bevenger, 1996), and for 1993 (Druse et al., 1994). Channel bed and banks appear to have been the primary sources of increased sediment load (Troendle and Bevenger, 1996) due to the devastation of the riparian vegetation (Young, 1994).

METHODS

Stream order and channel gradient were computed for each stream segment using geographic information system (GIS) techniques. A sample of 10 third-order segments was selected from each creek from among segments deemed most likely to have riffle-pool morphology (average gradient less than 0.03 m/m). At each selected stream segment, a study reach with length of 20 average bankfull channel widths was defined onsite. A study reach that appeared to have a gradient gentler than adjacent reaches was selected, thus targeting reaches assumed to be most sensitive to changes in sediment inputs (Lisle and Hilton, 1992). A few study reaches have well-developed riffle-pool sequences (ideal for sensitivity to changes of interest), but most reaches have plane-bed channels with small pools only. Nearly all data for study reaches were collected during late July through mid-September 1999, during low flow conditions; four reaches also were visited in August 1998, when substrate size distributions were sampled.

Bankfull channel width was measured at 4 to 8 riffle locations per reach and at 4 to 6 pools. Reach-average bankfull width was computed as a length-weighted average of riffle width and pool width. Residual water volume and fine-sediment volume were measured in 6 pools in each study reach, using methods of Lisle and Hilton (1992), except that only 5 pools were measured at 2 reaches. Reach length and downstream position of each pool were measured approximately along stream centerline. Eleven to 18 soundings were made along 3 or 4 transects across each pool, using a graduated steel rod 0.95 cm in diameter. Abrupt changes in resistance to penetration of the rod as it passed from sand or fine gravel to imbricated coarse gravel and cobbles indicated the base of the fine sediment deposit. Composite samples of the fine sediment in 3 pools per study reach were collected using a pipe dredge and submitted for laboratory sieve analysis to determine grain-size distribution (GSD). Results by size fraction were averaged within a reach to compute a reach-average GSD for pool fines. Reach-average channel gradient was measured using a graduated rod, a stabilized hand level with 5x magnification, and a tape-measured distance along the channel.

Streambed-surface particle size was sampled from two riffles in each study reach using the grid-sampling variant of the Wolman (1954) pebble count. In 1998, intermediate diameters were measured using a ruler, whereas a US SAH-97 particle-size analyzer (template) was used in 1999. For each reach, the percentages by size fraction for each riffle were averaged to compute a reach-average GSD on riffles.

LOD was defined as fallen pieces of wood with diameter greater than 10 cm and at least 2.0 m in length. Total length and length within or suspended above the active channel were measured for each piece. Diameter was measured at the mid-length point. Volume was estimated as equivalent to a cylinder having the same length and diameter. LOD loading, as volume per unit length of channel, was summed per reach. In some reaches, LOD jams were too large and dense to access or reliably inventory every piece; in those cases, all principal pieces seen, together with most other accessible pieces, were measured. The degree to which each piece of LOD was anchored was recorded as the number of ends attached or buried in banks, or number of sides buried in channel sediments (Young, 1994). Notation was made of each piece that was in contact with another piece of debris.

Statistical testing procedures used S-Plus 2000 analysis software (Mathsoft, Inc., Seattle, Wash., rel. 2, 1999). Tests of differences between streams used the Wilcoxon rank-sum test, except for a few variables that had sample distributions satisfying the assumptions for the t test. Correlations were tested using the large-sample approximation of significance for Spearman' s rank correlation coefficient (R_s). Multiple linear regression models were fitted using least-squares regression or least-trimmed-squares regression. Standard diagnostic plots were used to check for gross violations of the assumptions underlying normal linear regression. Significance of linear regression models was tested using the F ratio of the explained variation to unexplained variation divided by their respective degrees of freedom (Ott, 1988). In this study, test results with probabilites (p-values) under the null hypothesis of less than 0.05 were considered to be significant and results with p-values from 0.05 to 0.10 to be marginally significant.

CHANNEL CHARACTERISTICS

Channel Width. Average channel widths are chiefly a function of drainage area. However, Crow Creek is much narrower than expected near its mouth at site 173, just downstream from where part of the flow is diverted into a ditch. Data for that reach were excluded from further analyses of channel width. Reach-average bankfull width (W) was greater in Jones Creek, on average, than in Crow Creek (table 1), as was reach-average bankfull width of pools (Wp). However, the rank-sum test results given in table 1 ignore the general relation of width to drainage area.

In a simple linear regression model of W, drainage area (DA) explains 74.6 percent of the variance. Recognizing that differences in channel width also might be related to postfire increases in runoff (i.e., the difference between streams) and LOD loading (see table 2), expanded linear models were examined using stepwise selection. The selected model includes the number of large LOD jams (N.LGJAMS) and the dummy variable for stream identity (WS) as independent variables. The least-squares regression-fitted model,

$$W = 5.97 + 0.0010*DA + 0.36*N.LGJAMS + 0.59*WS \quad (1)$$

explains 86.7 percent of the variability in average bankfull width with a residual standard error (RSE) of 0.90 m, and is highly significant ($F = 32.57$, $p < 0.0001$). The explanatory variables are each significant: DA ($t = 5.463$, $p = 0.0001$), N.LGJAMS ($t = 2.788$, $p = 0.0138$), and WS ($t = 2.653$, $p = 0.0181$). Bankfull width of Jones Creek is 0.59 m (with 0.22 m standard error) greater (or 6.6 percent wider) than that of Crow Creek, after accounting for relations with drainage area and large LOD jams.

Bankfull widths of pools and riffles also were analyzed with multiple regression models including DA. For Wp, DA alone explains 50.4 percent of the variance, but pool width also is significantly correlated with reach gradient (G, $R_s = -0.775$), pieces of LOD per accumulation (PC.ACC, $R_s = 0.547$), number of LOD jams (N.JAMS; $R_s = 0.518$), pieces of LOD in jams (R_s = 0.553), and percentage of LOD occurring in jams (PCT.JAM, R_s = 0.599). Stepwise selection indicated DA and reach gradient as the best pair of independent variables for explaining pool width, with 77.4 percent of variance explained with a RSE of 1.30 m. However, there is no significant difference between streams in bankfull width of pools (p = 0.1970) after accounting for differences in drainage area and reach gradient.

Table 1 – *Summary of reach-average channel characteristics in low-gradient reaches.*

[Sample size is 10 reaches for each stream, except that for channel width variables sample size is 9 for Crow Creek and 10 for Jones Creek; COV, coefficient of variation (i.e., standard deviation divided by mean).]

Channel characteristic	Variable name	Crow Creek			Jones Creek			p-value for rank-sum test of different medians
		Mean	Median	COV (%)	Mean	Median	COV (%)	
Bankfull width (m)	W	8.9	8.6	16.3	11.2	11.2	21.3	0.0350
Bankfull pool width (m)	Wp	9.8	9.8	14.5	12.4	12.6	22.7	.0350
Bankfull riffle width (m)	Wr	8.7	8.2	14.4	10.6	9.7	20.3	.0947
Gradient (m/m)	G	.022	.023	25.1	.016	.015	51.2	.1051
Grain size on riffles, percent finer than 8 mm	GS_r8	12.3	12.1	29.5	8.4	8.1	28.5	.0115
Grain size on riffles, percent from 128 to 256 mm	GS_r256	7.6	5.4	86.5	11.7	11.6	70.4	.2176
Grain size on riffles, percent coarser than 256 mm	GS_r256+	2.1	0.7	136	6.2	4.0	123	.2475
Median grain size on riffles (mm)	$R.d_{50}$	42	38	41.2	58	50	45.8	.1037
Pool spacing, in channel widths	SPAC.w	3.9	3.7	31.6	3.8	3.8	38.0	.8534
Residual pool mean depth (cm)	D_r	16	16	28.2	18	18	29.2	.4813
Residual volume per pool (m^3)	V_r	4.2	3.4	82.7	6.4	5.8	82.3	.3527
Ratio of pool fines volume to residual volume	V^*_w	.20	.17	58.6	.23	.23	33.2	.1575
Grain size of pool fines, percent finer than 0.6 mm	$GS_p0.6$	24.2	18.3	48.9	27.4	29.6	52.4	.7394
Grain size of pool fines, percent from 0.6 to 1.18 mm	$GS_p1.18$	13.7	11.3	39.8	13.9	13.8	35.3	.4813
Grain size of pool fines, percent from 1.18 to 2.36 mm	$GS_p2.36$	11.3	9.8	45.0	12.0	11.5	14.5	.3445
Grain size of pool fines, percent coarser than 2.36 mm	$GS_p2.36+$	50.8	54.4	36.3	46.7	45.8	33.9	.2475
Median grain size of pool fines (mm)	$P.d_{50}$	3.5	3.0	59.9	2.2	1.8	69.3	.0716

For riffle width (Wr), DA alone explains 76.2 percent of the variance, but Wr also is significantly correlated with PC.ACC (R_s = 0.585), N.JAMS (R_s = 0.656), JAM.100m (R_s = 0.552), pieces of LOD in large jams (R_s = 0.688), and PCT.JAM (R_s = 0.624). A multiple linear regression model using DA, N.JAMS, PC.ACC, and WS as predictors explains 87.5 percent of the variance in riffle width with RSE of 0.90 m. The coefficient for WS (0.48 m, with 0.23 m standard error) indicates the average width by which Jones Creek reaches exceed Crow Creek reaches' average riffle width, after accounting for differences related to drainage area and LOD accumulations.

Channel Gradient. The reach-average gradient (G) data summary (table 1) shows the mean for Crow Creek reaches is 0.022 m/m, versus 0.016 for Jones Creek, a marginally significant difference (t = -1.876, p = 0.0769). The third-order length of Crow Creek is 1.3 km shorter than that of Jones Creek, its mouth is 52 m lower, and Crow Creek enters its third-order valley at an elevation 60 m higher than does Jones Creek; thus, the elevation change of third-order Crow Creek is 112 m greater than Jones Creek. GIS data for all third-order segments of each stream indicate average gradient of 0.038 m/m for Crow Creek versus 0.027 m/m for Jones Creek.

Bed Substrate on Riffles. For mean GSD on riffles, 90-percent confidence intervals (fig. 2) were computed pointwise for each mean percentage-finer break point individually, not as a simultaneous confidence interval. Nevertheless, the 90-percent confidence intervals clearly overlap for all particle sizes greater than 10 mm, indicating no difference between streams. Reach-average GSD in riffles as summarized for seven size fractions (phi-unit intervals from 8 to 256 mm; results for 3 fractions listed in table 1) showed no significant differences between streams, except for the fraction finer than 8 mm (GS_r8), with Jones Creek riffles being fines-poor compared with

Crow Creek. The largest differences in mean percentages by size fraction were for the fractions less than 8 mm, from 128 to 256 mm, and larger than 256 mm in diameter, but variability among the reaches within in each stream was much greater for the coarser fractions, as indicated by large coefficients of variation (table 1).

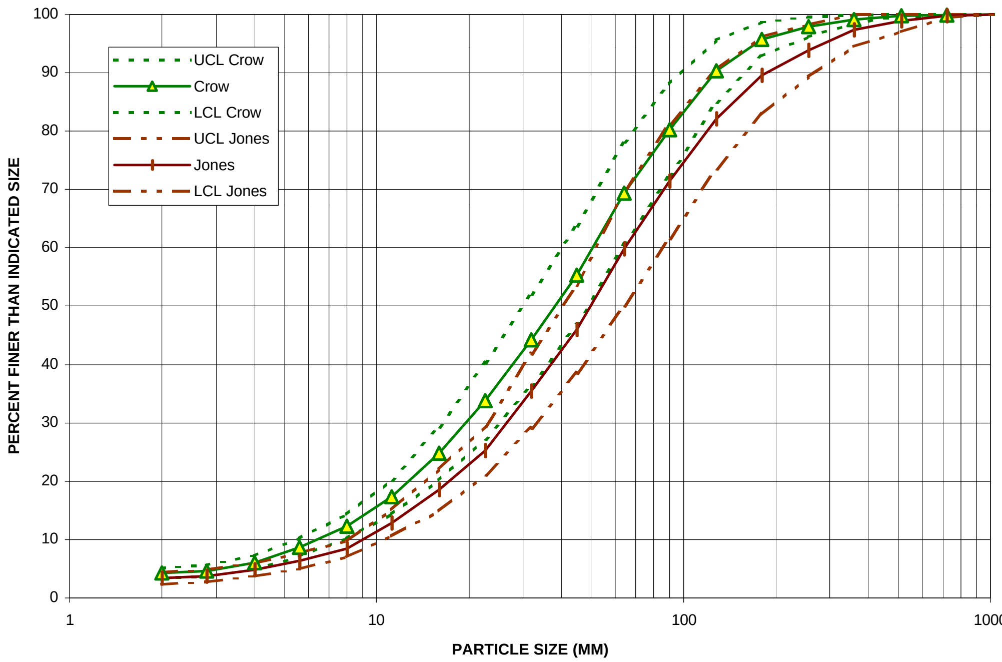

Figure 2 -- *Mean grain-size distributions on riffles of 20 low-gradient reaches (UCL and LCL are upper and lower limit, respectively, of 90-percent confidence interval).*

Median diameter of substrate on riffles (R.d_{50}) averaged 58 mm in Jones Creek reaches and 42 mm in Crow Creek. A multiple linear regression model explains 68.6 percent of the variability in R.d_{50} using reach gradient (G) and WS as independent variables, with RSE of 14 mm. WS is a significant explanatory variable ($t = 4.64$, $p = 0.0002$), meaning that R.d_{50} for Jones Creek was 16 mm coarser (with 3.4 mm standard error) than that for Crow Creek, after accounting for differences in reach gradient.

<u>Pool Characteristics.</u> For both streams, average pool spacing (SPAC.w) is about 3.8 times bankfull channel width (table 1). Most reaches have average pool spacing between 2.5 and 5 times bankfull width. Mean depth of residual pools (D_r) is significantly correlated with reach gradient (G, $R_s = -0.802$), percentage of channel length in pools (PctPool, $R_s = 0.779$), and W ($R_s = 0.523$), in addition to a marginally significant correlation with SPAC.w ($R_s = -0.444$, $p = 0.0528$).

Residual volume of pools (V_r) is significantly correlated with reach gradient (G, $R_s = -0.811$, $p = 0.0004$). To obtain valid linear regression models, the logarithm of V_r was used as the dependent variable. The simple linear model using reach gradient as the independent variable explains 64.2 percent of the variance, with a RSE of 1.7 m^3.

Median reach-average ratio of residual-pool fine sediment volume to residual volume (V^*_w) is 0.17 for Crow Creek and 0.23 for Jones Creek (fig. 3). Rank correlations with V^*_w are significant for percentage of channel length in pools (PctPool, $R_s = 0.675$), pool spacing (SPAC.w, $R_s = -0.636$), D_r ($R_s = 0.565$), and V_r ($R_s = 0.459$). Multiple linear regression of V^*_w with pool spacing and WS indicates a significant inverse relation with pool spacing ($t = -3.145$, $p = 0.0056$), with no significant difference between streams ($p = 0.7066$). For both streams, where pools are closely spaced, they are more filled in by fine sediment at low flows. Multiple linear regression of V^*_w with D_r and WS indicates a significant relation with D_r ($t = 3.0449$, $p = 0.0070$), with no significant difference between streams ($p = 0.7829$). For both streams, deeper pools are associated with greater filling by fine sediment. Stepwise model selection resulted in the following preferred multiple linear model, which explains 57.2 percent of the variance with a RSE of 7.0 percent of V_r.

$$V^*_w = 0.013 + 0.0215^*W + 0.0050^*PctPool - 0.000039^*DA \quad (2)$$

The average size distributions of pool fines were summarized for four size fractions (table 1), but none of the rank-sum tests for differences between streams produced significant results. Median diameter of pool fines ($P.d_{50}$) averaged 2.2 mm in Jones Creek reaches and 3.5 mm in Crow Creek. Rank correlations with $P.d_{50}$ are significant for D_r (R_s = -0.869, p = 0.0001), G (R_s = 0.840, p = 0.0003), V_r (R_s = -0.836, p = 0.0003), and V^*_w (R_s = -0.617, p = 0.0071). Results of stepwise selection indicated that a simple linear model using reach gradient provided the best fit. The fitted model explains 64.2 percent of the variance in $P.d_{50}$ with RSE of 1.2 mm. Addition of WS to the model does not improve its explanatory ability (p = 0.8257). Results for the GSD of pool fines indicate no difference between streams exists after accounting for the relation with reach gradient.

LARGE ORGANIC DEBRIS

Piece frequency and size. Overall, 970 pieces of LOD were inventoried in Crow Creek, versus 1,042 pieces in Jones Creek. The mean number per 100 m of channel (PC.100m) was 63.2 for Crow Creek and 61.5 for Jones Creek (table 2). Reach-average LOD diameter (DIA) for Crow Creek was 2.2 cm larger than that for Jones Creek, a significant difference (p = 0.0433). However, neither reach-average total piece length nor active-channel piece length for Crow Creek were significantly greater than that for Jones Creek (table 2). Based on medians, Crow Creek reaches average significantly greater total volume per piece (TVOL, p = 0.0262) and active-channel volume per piece (AVOL, p = 0.0376) than Jones Creek.

Table 2 – *Summary of large organic debris (LOD) characteristics in low-gradient reaches.*

[Sample size is 10 reaches for each stream; COV, coefficient of variation (i.e., standard deviation divided by mean).]

LOD characteristic	Variable name	Crow Creek			Jones Creek			p-value for rank-sum test of different medians
		Mean	Median	COV (%)	Mean	Median	COV (%)	
Pieces per 100 m of channel	PC.100m	63.2	61.1	27.1	61.5	50.4	46.7	0.4961
Mean piece diameter (cm)	DIA	21.9	21.4	10.3	19.7	18.9	10.1	.0433
Mean piece length (m)	LEN.Tot	7.09	7.10	11.8	6.69	6.50	14.5	.2799
Mean active-channel piece length (m)	LEN.Act	5.06	5.11	11.4	5.02	4.75	22.5	.4813
Total volume per piece (m^3)	TVOL	0.36	0.33	21.7	0.28	0.26	36.9	.0262
Active-channel volume per piece (m^3)	AVOL	.250	.234	22.8	.206	.195	49.6	.0376
Total volume per 100-m channel length (m^3)	TVOL.100m	22.4	21.6	25.2	18.9	13.8	77.6	.1655
Active-channel volume per 100-m channel length (m^3)	AVOL.100m	15.5	14.6	29.0	14.2	8.9	92.3	.2176
Number of anchors per piece	N.ANC	.76	.77	30.0	.59	.57	31.4	.0262
Percentage anchored	Pct.ANC	54.5	54.5	24.7	45.7	47.5	21.8	.0615
Percentage with > 1 anchor	Wel.ANC	16.2	14.9	50.3	10.5	8.6	67.9	.0615
Accumulations per 100 m	ACC.100m	8.3	8.9	31.3	5.7	4.9	34.6	.0170
Pieces per accumulation	PC.ACC	6.5	6.5	34.0	8.0	7.4	32.6	.1061
Percentage of pieces occurring in accumulations	PCT.ACC	80.2	82.2	11.7	74.8	78.3	15.6	.3073
Large jams per 100 m	JAM.100m	1.5	1.5	79.3	1.6	1.5	60.9	.8797
Pieces per large jam	PC.JAM	19.7	17.7	34.2	21.4	21.3	32.1	.0992
Percentage of pieces occurring in large jams	PCT.JAM	40.8	45.3	58.0	52.5	57.2	29.9	.1537

LOD Loading. A multiple linear regression model for active-channel LOD loading (AVOL.100m) was fitted using W and WS as independent variables. The model fitted using all 20 reaches explained 45.1 percent of variance with a RSE of 7.5 m^3 per 100-m length of channel, but site 93 was a large positive residual outlier. Excluding site 93, the trimmed model explains 35.9 percent of the variance with a RSE of 5.2 m^3, and is significant (F = 4.48, p = 0.0284). The difference between streams is significant (t = -2.881, p = 0.0109), indicating that the LOD loading in Crow Creek is 4.0 m^3 greater (with 1.4 m^3 standard error) than that in Jones Creek, after accounting for channel width differences. Variability in LOD loading among the reaches of Jones Creek was much greater than that for Crow Creek (table 2). This reflects both a greater tendency for Jones Creek reaches to have small loadings if they had few large jams, and the influence of site 093 at the mouth of Jones Creek, which had 7 large LOD jams and the largest LOD loading of any study reach.

Piece Anchoring. Frequency of occurrence decreased with increasing number of anchors, from 1,008 pieces without anchoring to 89 pieces with 3 or more anchors. As indicated by mean number of anchors per piece (N.ANC, table 2), LOD in Crow Creek is better anchored, on average, than that in Jones Creek ($p = 0.0262$). Percentage of pieces being anchored (Pct.ANC) or well-anchored (more than one anchor, Wel.ANC) is greater for Crow Creek, but with only marginal significance ($p = 0.0615$).

LOD Accumulations and Large Jams. More than 75 percent of LOD pieces inventoried were in contact with another piece. The number of debris accumulations per 100 m length of channel is greater in Crow Creek than in Jones Creek (table 2; $p = 0.0170$), though the Crow Creek accumulations tend to be smaller (mean of 6.5 pieces versus 8.0 pieces; table 2).

The number of large jams (accumulations of 10 or more pieces) ranged from 0 to 3.3 per 100 m for Crow Creek and from 0.5 to 3.6 per 100 m for Jones Creek. Piece counts of debris per large jam were marginally greater in Jones Creek than in Crow Creek (median of 21.3 versus 17.7; $p = 0.0992$).

DISCUSSION

Increases in channel width and riffle substrate coarseness confirm theoretically predicted channel responses to increased runoff following burning of the riparian and hillslope vegetation. The increase in streamflow was most pronounced on the rising limb of the snowmelt hydrograph (Troendle and Bevenger, 1996), which may correspond to increases in soil moisture (due to postfire transpiration reductions) that are likely greatest from late summer until spring snowmelt is underway. Earlier timing of peak snowmelt due to loss of forest cover also contributes to the differences in spring runoff.

A large slope failure in 1996 affected an area of tens of hectares and deposited debris across the flood plain and directly into Jones Creek. Also, postfire debris flows were evident in several tributaries to Jones Creek (M.E. Smith, U.S. Geological Survey, Cheyenne, Wyo., unpub. data, 1989; personal observ.). The sediment inputs from these events may have reduced the channel cross-sectional area and contributed to bank erosion and channel widening (e.g., Ryan and Grant, 1991). Smaller LOD jams in Crow Creek may correspond to less flow deflection, greater bank stability, and narrower bankfull width. At several very large LOD jams in Jones Creek, recent bank erosion and channel widening were apparent. Thus, the wider channel of Jones Creek appears to be the result of multiple postfire conditions: increased runoff, increased sediment inputs, flow deflection by enlarged LOD jams, and decay of stabilizing forest-root mat.

Although the rank-sum test result for difference between streams in V^*_w (table 1) was not significant, the data suggest that two outliers in the Crow Creek data (sites 171 and 179) cause the insignificant result. Site 171 has the gentlest average gradient among the Crow Creek study reaches and is part of an aggradational segment just upstream of a tributary fan that extended across the valley floor about 25 years ago (based on aerial photographs, and from the size of the even-aged stand of lodgepole pines covering the fan). The reach gradient of site 179 is at the 25th percentile for studied reaches of Crow Creek, and the reach is immediately downstream of the mouth of a tributary that is actively headcutting its incised channel into a fan deposit. If the conditions at these two reaches are excluded, being responses to site-specific anomalies, then the rank-sum difference becomes significant ($p = 0.0171$). One might conclude regarding the general condition of the study streams that pool filling with fine sediment is greater in Jones Creek, likely as a result of increased fine-sediment inputs following wildfire. However, these anomalous reaches suggest that local disturbances of small magnitude with occurrence intervals of a few decades can have measurable impacts at the reach or segment scale in Absaroka streams, even in the absence of recent wildfire.

Smaller average piece size of LOD in Jones Creek reflects increased loading from younger, fire-killed trees. Many burned trees are still standing, but low root strength is common. Young (1994) reported a larger mean diameter for Jones Creek than for Crow Creek in 1990 (in the downstream segment of each stream). However, his results also show that Jones Creek LOD tended not to be in younger apparent-age categories and was more commonly in a rotted condition than Crow Creek debris, suggesting that much of the Jones Creek LOD he measured was older, pre-fire debris. To further compare the 1999 data with Young' s results, data were summarized for the downstream study reach of each stream for LOD having diameter of at least 15 cm. Mean piece diameters of 24 and 27 cm for Crow and Jones Creek, respectively, reported by Young (1994) are similar to the 27 and 25 cm mean diameters, respectively, measured in 1999. Young's 1990 piece counts of 32 and 15 pieces per 100 m for Crow and Jones Creeks, respectively, contrast with 31 and 76.5 pieces per 100 m in 1999. However, Young included only LOD in or above the wetted channel, whereas the 1999 data included nonwetted active channel areas, undoubtedly accounting for much of the difference in piece frequency for Jones Creek's widened downstream reach. But increased LOD inputs since 1990 also appear likely, given that the minimum frequency of 15-cm or larger LOD among Jones Creek reaches was 21 per 100 m in 1999. Field evidence was noted of snow avalanches having delivered large numbers of burned trees locally into Jones Creek, in addition to those from windthrow, bank erosion, debris torrents, and large slope failures.

LOD was better anchored and more frequently occurred in small accumulations in Crow Creek. The most common type of LOD anchoring observed in these streams was attachment by roots. Root decay of fire-killed trees probably contributes to less frequent anchoring of LOD in Jones Creek, as does the larger percentage of pieces occurring in large jams, compared with Crow Creek. Although LOD mobility was not measured in 1999, the differences in measured characteristics and loadings suggests strongly that LOD mobility continues to be greater in Jones Creek, as was reported for 1990 by Young (1994).

The analysis phase of this study continues; other conclusions may be premature at this time. This and other studies of the effects of wildfire on channel characteristics and LOD will increase understanding of the magnitude and duration of longer-term fire effects, post-disturbance recovery rates, and the role of wildfire in landscape evolution.

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EFFECTIVENESS OF AGRICULTURAL BMP' S IN REDUCING STORMFLOW SEDIMENT (TSS) IN TELLICO CREEK, MACON COUNTY, NORTH CAROLINA

David A. Braatz
Duke Power Company, Mail Code MGO3A3
13339 Hagers Ferry Road
Huntersville, North Carolina 28078 USA
Dabraatz@duke-energy.com
704-875-5430

INTRODUCTION

Tellico Creek is a tributary of the Little Tennessee River, in the southern Appalachian Mountains in western North Carolina. Headwater reaches, starting at the Appalachian Trail, consist of stream types A and A+; lower reaches, including our study area, are B3a channels. The region is largely forested, with limited agriculture. Water quality is generally very good: specific conductance is low (7 to 20 umho/cm); total alkalinity is low (2.0 to 4.6 mg-CaCO3/L); and nutrient concentrations are low. Total suspended solids (TSS) have been monitored at up to eight sites in the 1591-ha (6.05 sq mi) Tellico Creek study area since 1990, as part of Duke Power Company' s erosion control assessment program for transmission lines. Construction activity such as clearing tower sites and building or upgrading access roads in this area of high rainfall and rugged terrain poses significant potential for erosion and for sedimentation impacts to surface waters. North Carolina' s Division of Water Quality (DWQ) has identified sedimentation as the biggest threat to mountain streams.

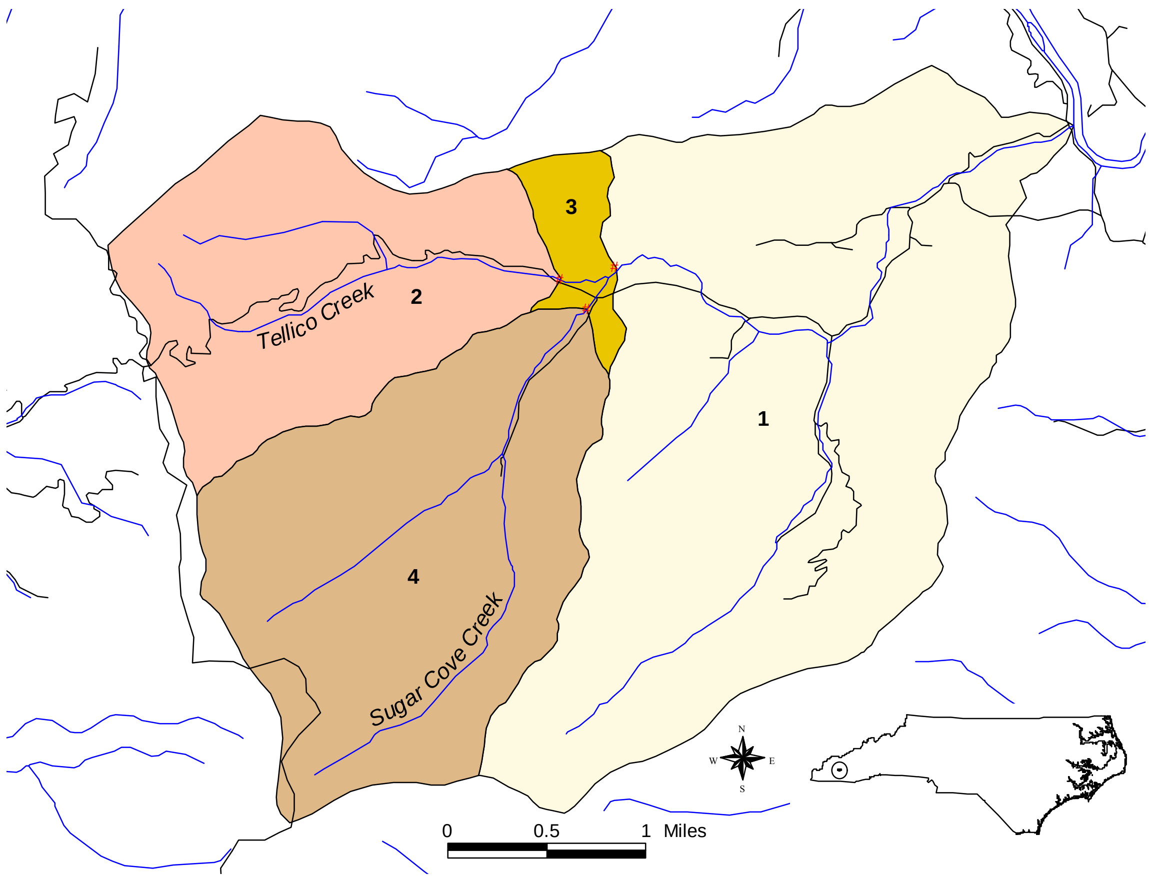
An earlier study documented the successful use of construction BMP' s, and the subsequent reduction of stormflow sediment concentrations. BMP' s to improve and stabilize roads (such as installing culverts and broad-based dips, maintaining silt fences to reduce off-site sediment, promptly seeding disturbed areas, and respecting vegetated buffer zones along the creek), all contributed to successful completion of the transmission line project (Siler, et al., 1995). However, lower reaches of the creek, including a major aquaculture operation, remained subject to sediment concentrations exceeding 10,000 mg/L after the initial program. A small family farm was identified as the source of 90% of the stormflow TSS. The current study focuses on that farm, and on three sampling sites (Figure 1): Upper Tellico and Sugar Cove Creek (the main stream and a major tributary, both just above the farm), and Lower Tellico (just below the farm, and also below the confluence of Tellico and Sugar Cove Creeks). A cooperative program between stakeholders was initiated in 1994 to address the agricultural impact to the stream. The study objective was to eliminate the 90% of stormflow TSS estimated to originate on the agricultural property, so that the quantity of sediment leaving the farm would be no greater than the quantity of sediment entering the farm from upstream.

METHODOLOGY

Best management practices (BMP' s) were applied to the farm that was identified as a significant source area for suspended sediment in Tellico Creek. The BMP' s included fencing cattle from the stream; installing a steel bridge for pasture access; installing a gravity-fill watering tank on a gravel pad; creation of a settling basin between a garden and the creek; and some minor landscaping to improve runoff patterns on the farm. Streambank erosion was not judged to be a problem, and riparian areas inside the fencing were allowed to vegetate naturally.

Three monitoring sites are addressed in this study, relative to the agricultural focal point: The Upper Tellico (UT) site is located above the farm, 492 m above the Lower Tellico (LT) site, and 385 m above the confluence of Tellico and Sugar Cove creeks. The Sugar Cove Creek (SC) site is also above the farm, 407 m above LT, and 300 m above the confluence of Tellico and Sugar Cove Creeks. The Lower Tellico (LT) site is 107 m below the confluence of Sugar Cove and Tellico Creeks, and 33 m above the uppermost raceways of a major trout-rearing operation (Figure 1). The confluence of Tellico and Sugar Cove Creeks is located at approximately 35 16' 34.5" N. Lat., 83 31' 56.3" W. Long.

Figure 1. Tellico Creek watershed: study area includes sub-basins 2 (Upper Tellico, 632 ha); 3 (farm area above Lower Tellico, 83 ha); and 4 (Sugar Cove, 876 ha).



From 1990 to 1996, baseflow samples were collected with a depth-integrating sampler (US DH-75P), in duplicate, at approximately twice-monthly intervals, using the equal-transit-rate method; they were composited from at least ten verticals across the stream by the equal-width-increment (EWI) method (Guy and Norman 1970; Vanoni 1977). From 1996 to 1999, single baseflow grab samples were collected biweekly, from the relatively shallow (6 to 10 inches), well-mixed riffles;

this change was instituted for time and cost savings, and in consideration of the consistently low baseflow TSS concentrations. A vertical series of single-stage suspended sediment samplers (US U-59B) was employed to automatically collect representative TSS samples on the rising stage of storm hydrographs, when erosion and worst-case sediment impacts were most likely to occur. These stormflow samplers were designed for automatic collection of samples in remote areas, where adequate sampling by other means is impracticable (Interagency Committee 1961). For purposes of this report, "baseflow samples" refers to the depth-integrated or grab samples collected manually at the given stream stage on the date of site visits; "stormflow samples" are the single-stage samples collected automatically on a rise in stage from baseflow.

Stream stage and the stages at which each of the single-stage samplers filled were measured to the nearest 1/4 inch, against a fixed stake to which the samplers were attached. Depending on the spacing of single-stage samplers on the stake, and hence the desired degree of accuracy, peak flood stages could usually be estimated within 2 to 3 inches (e.g., the midpoint between the highest filled and the lowest unfilled samplers). When all samplers were filled (or destroyed), peak flood stage was estimated if possible by observation of scouring levels or debris deposits.

Sediment concentrations were analyzed according to EPA-recommended standard methods. The analytical detection limit was considered to be 4 mg/L TSS, and any sample concentrations <4 mg/L were recorded as 4 mg/L. Subsampling has proven to be inaccurate for samples with extremely high TSS concentrations; therefore, for such samples the *entire* sample was analyzed to determine the sediment concentration. Sampling procedures included Duke Power Company QA/QC guidelines, and chain of custody records, for state-certified laboratory analyses.

Suspended sediment concentrations at SC and UT were weighted by discharge to estimate delivery to the farm. Relative to discharge at LT, discharge at SC contributed an average 63.7% (S.D.=3.4) and discharge at UT contributed an average 33.9% (S.D.=4.7); the remaining 2.4% originating on the farm was not monitored. Theoretical suspended sediment delivery to the farm (0.652 SC + 0.347 UT) was compared to estimated sediment leaving the farm (at LT); the difference was considered to indicate the role of the farm as a sediment source or deposition area. Relative to the entire study area (above LT), the sub-basin area above SC was 55.1% and the area above UT was 39.7%. Therefore, relative to Site LT, the farm property where agricultural BMP' s were applied contributed just 5.2% (83ha) of the total study area (1591ha), and an average 2.4% of discharge.

RESULTS

A total of 856 baseflow samples (280 to 292 per site), and 524 stormflow samples (152 to 189 per site) was collected. From the baseline period (Nov 1990 to Mar 1994) through the period of post-BMP monitoring (May 1995 to Mar 1999), mean baseflow sediment concentrations at UT, SC, and LT decreased from 16.5 to 7, 6.5 to 5.4, and 8.5 to 6 mg/L, respectively. From the baseline to post-BMP period, mean stormflow sediment concentrations in the control watershed (SC) increased over five-fold, from 108 to 531 mg/L. At UT, mean stormflow sediment increased over eight-fold, from 282 to 2400 mg/L; and below the BMP work, at LT, it increased only 30%, from 1340 to 1740 mg/L (Table 1).

Table 1. Average stormflow suspended sediment concentrations (TSS, mg/L), before and after application of best management practices (BMP' s).

	SC			UT			LT		
	Mean	S.D.	N=	Mean	S.D.	N=	Mean	S.D.	N=
Pre-	108	137	92	282	461	64	1339	3001	101
interim	224	196	19	2174	3466	15	2361	2890	22
Post-	531	1302	72	2400	5909	73	1736	6243	66

A sediment transport assessment indicated that 87% of stormflow sediment originated on the farm before BMP' s; this dropped to 62% originating on the farm during the BMP application period, and dropped further to 32% for the post-BMP period, based on mean sediment concentrations (Table 2). Of three storms analyzed individually, a pre-BMP storm in 1991 indicated that 99% of stormflow sediment originated on the farm; and post-BMP storms in 1995 and 1999 indicated that between 25 and 38%, respectively, of stormflow sediment originated on the farm (Table 2). The change in magnitude of component TSS values indicates that the percentage attributed to the farm may not be the best measure. While 37.6% of the TSS at LT during the 1999 storm event was attributed to the farm as a source area, that excess from the farm was a mere 19 mg/L in 1999, as opposed to 3583 mg/L from the farm during the 1991 storm event. Comparing TSS concentrations during these two similar-magnitude storm events (Table 2), there was a 99% reduction in TSS (mg/L) attributable to the farm. Two major floods were also compared: the flood of April 1994, in which the farm reach aggraded significantly, and a 1998 flood in which the farm served as the source of an estimated 61% of the downstream stormflow sediment. Although the average TSS concentration increased 30% during the post-BMP period at LT, this was primarily due to the 400% increase at SC and 750% increase at UT. While much of the sediment leaving the farm is now attributed to new upstream sources, it is also suspected that some sediment still originating on the farm (post-BMP' s) may in fact be due to unsampled road runoff, rather than farm runoff, that comprised the unmonitored 2.4% of LT discharge delivered to the stream just above LT.

Table 2. Discharge-weighted contributions of average stormflow TSS (mg/L) from upstream sites, compared to actual TSS at Lower Tellico, to estimate the percentage of TSS originating on the farm.

Date(s):	Baseline 1990-94	Interim 1994-95	Post-BMP 1995-99	One storm 3-5-91	One storm 10-16-95	One storm 2-3-99
From UT	99	761	840	22	2050	14
From SC	70	146	345	21	301	18
UT + SC	169	907	1185	43	2351	31
Actual LT	1339	2361	1736	3626	3115	50
From Farm	1170	1455	551	3583	764	19
Farm as % of LT TSS	87.4	61.6	31.7	98.8	24.5	37.6

Analysis of the frequency distributions of stormflow sediment concentrations reinforced the conclusion that while the agriculture BMP' s were effective, the UT and SC basins both deteriorated during the study. From the baseline to post-BMP period, the median stormflow sediment concentration, C50 (50% of samples being greater than the given concentration), at SC increased from 60 to 124 mg/L; and at UT it increased from 103 to 125 mg/L. Water quality below the BMP work, at LT, improved as the median concentration decreased from 159 to 94 mg/L (Table 3). While the C80 at SC increased from 24 to 38 mg/L, at UT it was similar (40 and 36 mg/L), and at LT the C80 decreased from 44 to 29 mg/L. Similarly, on the high end of concentrations, the C20 at SC increased from 182 to 365 mg/L; at UT it increased from 325 to 2900 mg/L; and at LT the C20 decreased from 1960 to 1050 mg/L. Only the highest 5% of samples increased significantly at LT following BMP's. Before BMP's, the ranked stormflow TSS concentrations at the three study sites demonstrate the relatively degraded water quality at LT attributed to the farm area. After BMP's, the distribution of stormflow sediment concentrations indicates that LT is very similar to UT and SC; this is due to a combination of a reduction in sediment sources on the agricultural property, as well as degradation of the waters coming onto the farm from UT and SC.

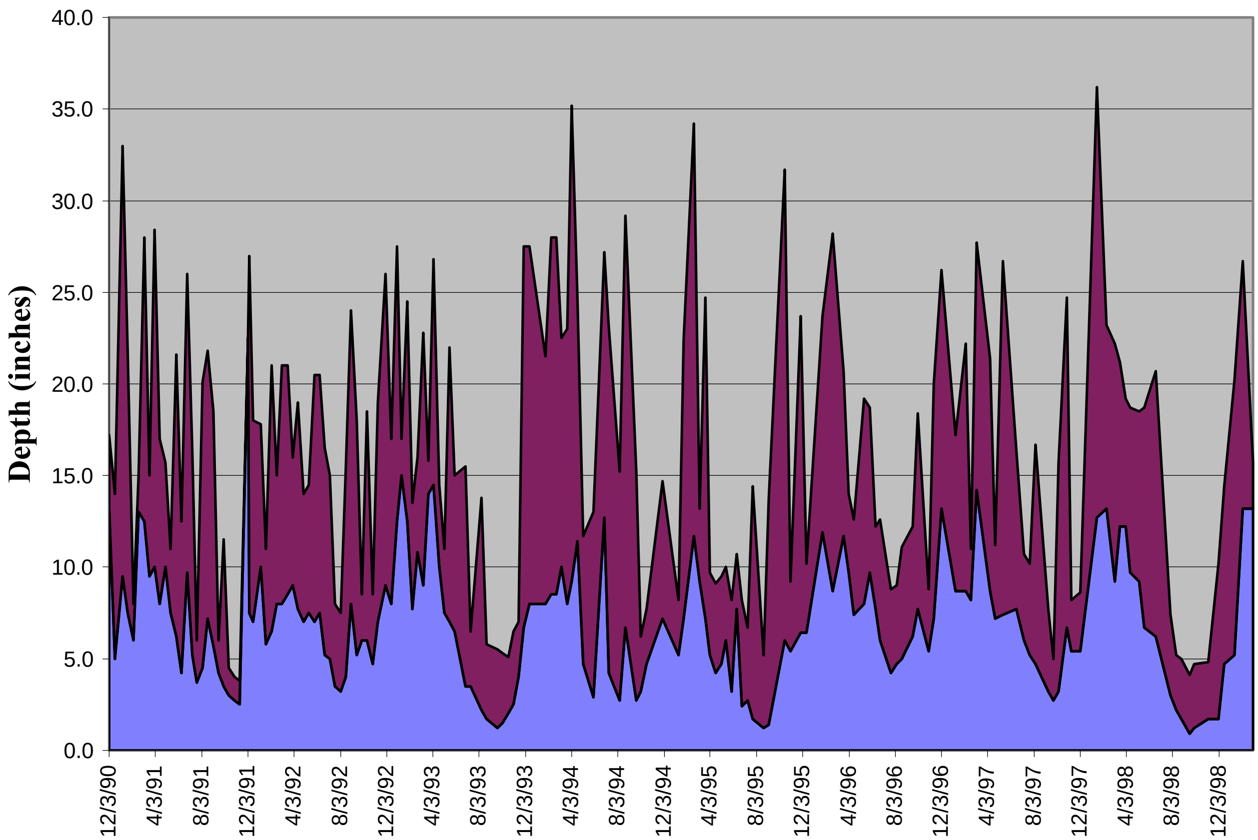
Table 3. Summary of frequency distributions of stormflow TSS concentrations (mg/L), before and after BMP's (Note: the C-categories indicate the "percentage of samples greater than". For example, for C80 = 44, 80% of the samples were greater than 44 mg/L)

	C95	C90	C80	C50	C20	C10	C5	N=
Baseline								
SC	8	15	24	60	182	255	420	92
UT	19	24	40	103	325	860	1700	64
LT	15	20	44	159	1960	3600	6100	101
Interim								
SC	26	30	100	135	340	560	760	19
UT	45	60	81	280	5200	9000	>10000	15
LT	67	70	120	787	5600	6500	9000	22
Post-								
SC	16	28	38	124	365	1350	3500	72
UT	15	20	36	125	2900	8000	>10000	73
LT	15	22	29	94	1050	3500	>10000	66

The channel materials at the study reaches reflected a common trend towards an increase in the relative occurrence of finer materials at downstream sites. The D50 at LT was 92mm, compared to 150mm at UT and 140mm at SC. Topography of the Tellico watershed is largely high gradient, and the farm reach above LT is the highest elevation in the watershed where agricultural activities might be expected. The valley floor first starts to broaden here, reflecting past alluvial deposits. Nevertheless, even the LT site was dominated by cobble (47%) and gravel (37%), with only 7.4% sand and finer.

A rough hydrograph was generated for each site (e.g., Figure 2), based on the measured stream stages on the given dates of site visits, and the estimated peak stages between site visits (i.e., the midpoint between the highest bottle filled, and the next-higher bottle unfilled). This gives a convenient overview of the relative frequency and magnitude of storm events, which were similar during pre- and post-BMP periods. The drought conditions in 1993 and again in 1998 are particularly apparent. The latter drought continued to record low-flow conditions throughout western North Carolina in 1999. Field identification of bankfull stage at LT, corresponded to the depth reading of 28 inches on Figure 2. Although the peak stages are rough estimates (usually +/- 2 to 3 inches), the bankfull stage at LT appears to have been met or exceeded at least six times over the study period. Thus, the LT site was apparently consistent with the bankfull regional curve for North Carolina mountain streams (Harman, et al. 2000), and demonstrated an approximately 1.3-year return interval for bankful discharge.

Figure 2. Baseflow reference depth and stormflow rise (inches) at Lower Tellico.



DISCUSSION

Duke Power Co started monitoring stormflow suspended sediment in 1989 as part of an erosion control assessment program for transmission line construction in the southern Appalachians. Approximately two-thirds of the water pollution in the southern Appalachians is attributable to nonpoint sources (SAMAB 1996); siltation affected 45% of impaired stream miles, and agriculture as a source accounted for 72% of rivers that were impaired. Duke's monitoring program focussed on TSS as a key water quality variable, and single-stage samplers were used to collect representative samples on the rising stage of storm hydrographs, when erosion and worst-case sediment impacts were most likely to occur. The single-stage samplers were inexpensive (< $5 materials per sampler; < $50 total cost per site), allowing us to install monitoring sites on dozens of streams, and sometimes at dozens of sites within a given basin. The program provided prompt feedback to field crews for rapid stabilization of any problem areas indicated after each storm event, or for timely consideration of supplemental aquatic chemistry and/or biological monitoring. At times, such as with Tellico Creek, we could cooperate with other stakeholders to address pollutant source areas that had been identified and localized. Thus, this Tellico Creek study carried the single-stage monitoring protocol beyond our original intent, in part to maximize the utility of the data, and also to further evaluate the suitability of these methods for identifying, quantifying, and localizing nonpoint source (NPS) sediment impacts.

The variability in stormflow suspended sediment concentrations at all study sites may in part be due to the high gradient valley and the channel types, and the variety of sources of fines. The channel itself did not serve as a moderating "reservoir" for fines. While single-stage suspended sediment samplers are most effective for collecting medium sands and finer particles, the total sand fraction of the substrates in the study area ranged from just 3.4 % sand and finer at UT to 7.4% at LT. Fines delivered to the stream during storm events were expected to remain in suspension throughout the study area, and relatively well-mixed throughout the water column. Point samples collected on the rising stage should therefore accurately reflect actual changes in stormflow sediment concentrations. In some cases of extreme events during this study, oversampling was likely, due to excessive velocities, and in several cases, the sampling equipment was simply destroyed in the flood.

The increased TSS above the farm was primarily responsible for the increased average concentration at LT. Activities above the SC and UT sites that are likely responsible for the increased sediment transport include road maintenance, development activities on private property, and small-scale logging, in addition to natural disturbances such as fallen trees and landslides. The unmonitored 2.4% of LT discharge that came from a small tributary, which drained road runoff and ran through the former pasture, may also have been an important sediment source. B3 channels, such as Tellico and Sugar Cove, are considered to have a low sensitivity to disturbance, low sediment supply, low streambank erosion potential, and excellent recovery potential (Rosgen 1996). With agriculture and possibly road runoff as the primary sediment sources, this may also indicate why there was no relationship between stormflow TSS concentrations and discharge (R-squared = 0.01), before or after the BMP's. Tellico Creek is a good subject for the comment by NCASI (1999) that "discharge is not necessarily a good predictor of sediment transport rates. Sediment rating curves tend to be most effective for bedload transport in sand-bedded streams, and are less appropriate in situations where the entrainment of particles is not readily predictable and sediment transport rates are highly dependent on sediment inputs from outside the immediate channel".

Catastrophic rainfall and flooding in April 1994 obliterated several sampling stations; caused landslides, road slumping, high runoff, and extreme headwater channel erosion; and resulted in record high TSS concentrations and sediment transport throughout both the experimental watershed and the paired control watershed. Point samples of TSS in Tellico Creek exceeded 100,000 mg/L. Sugar Cove Creek, the paired control stream, reached nearly 45,000 mg/L TSS, compared to a maximum of 870 mg/L and a mean of 126 mg/L among 92 storm samples over the preceding 42 months. Lower Tellico Creek (below the farm study area), served as a deposition reach in which the streambed aggraded about six inches. Although such natural disaster-type events may be relatively infrequent, they certainly hold the potential to greatly exacerbate, if not exceed, the damage from existing anthropogenic impacts. State and federal agencies which compile lists of sources or causes of sediment impacts, should consider including a "natural" category, to keep impacts from human activities in their proper perspective.

CONCLUSIONS

Construction BMP's and agricultural BMP's were both shown to be effective in reducing stormflow sediment concentrations. However, the reduction in farm-source TSS was more than offset by increased TSS from above the farm. The apparent deterioration in water quality upstream from the farm, even in our paired control watershed, should remind us that we will never achieve a restoration of desired water quality, if we do not give equal and ongoing attention to prevention of future water quality impacts. Although stormflow sediment concentrations were extremely variable, long-term use of single-stage samplers seems to be a very cost-effective means of monitoring suspended sediment, localizing sediment source areas within a watershed, and assessing the effectiveness of erosion control efforts and best management practices.

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LANDSCAPE APPROACH TO SEDIMENT CONTROL

By: Thomas E. Davenport, National Nonpoint Source Pollution Expert, United States Environmental Protection Agency, Chicago, Illinois; Lyn Kirschner, Water Quality Specialist, United States Department of Agriculture, Conservation Technology Information Center, West Lafayette, Indiana

Abstract: States report that about 40 percent of the waters they assessed, do not meet water quality goals. About half of the nation's over 2,000 major water bodies have serious or moderate water quality problems. Among all pollutants afflicting streams and rivers in the United States, sediment is by far the greatest in terms of volume. There are a multitude of anthropogenic sources of sediment that are likely to enter and degrade our nation's water resources. Agriculture in its several forms is by far the largest generator of sediment. The present approach of addressing agricultural sediment sources on a site-by-site basis has not been totally effective and in a number of cases has resulted in further degradation downstream. For environmental purposes we need a landscape approach for comprehensively addressing sediment pollution. This approach must include a combination of activities that promote the prevention, mitigation and treatment of sediment pollution To compliment the landscape approach a new way to prioritize areas within the landscape needs to be adopted that is based upon site-specific conditions and takes in to account the potential for causing off-site impacts. Utilizing the landscape approach and this new prioritization process together, we can focus available resources and implement effective strategies to solve sediment related problems. This paper will a present a landscape approach to sediment management on a watershed basis, focusing on agriculture, and a technique by which to prioritize management efforts based upon potential off-site impacts.

INTRODUCTION

Agriculture is the most widespread nonpoint source of water pollution across the nation. The most severe agricultural related problem is soil erosion resulting in siltation (USEPA, 2000). Nationwide approximately half of the total sediment delivered to lakes and streams is from cropland. The runoff from agricultural lands, carrying large amounts of sediment scours the stream channel, alters the character of the stream, and affects aquatic life impairing functions such as photosynthesis, respiration, growth and reproduction.

Erosion is defined as the detachment and movement of soil or rock fragments by water, wind, ice or gravity. Sediment is defined as solid materials, which have been transported from their place of origin by erosion. There are six principal types of water erosion on cropland: 1) natural, 2) gully, 3) cropland ephemeral gully (mega rill), 4) rill, 5) sheet, and 6) splash. Figure 1 shows the erosion and deposition occurrence in a watershed. Soil erosion has several major consequences: 1) the original sites of the eroded material are degraded, therefore potential productivity is lessened; 2) the sites of deposition of the soil particles are altered physically, chemically and hydrologically, and 3) the chemical and physical nature of the transporting water resource is affected. One of the long-term effects of soil erosion is the entry of phosphorus and pesticides attached to eroded soil particles in waterways. Degradation of productive capacity of agricultural land results in depletions of yield that must be offset with additional inputs of fertilizer.

SOIL EROSION PROCESS

Soil erosion is a natural phenomenon that has occurred since soils were formed. Water is the most widespread agent of cropland erosion and accounts for the bulk of sediment transported (Gottschalk, 1964). Erosion and sedimentation by water involve the processes of detachment, transport, and deposition of soil particles (Foster, 1982). The major force initiating soil detachment is derived from the impact of falling raindrops. Precipitation detaches surface soil particles by raindrop impact and runoff transports particles to points of deposition. Detached soil particles are transported off-site or flushed into surface pores, thus plugging soil pores and reducing soil infiltration capacity. Reduced infiltration results in higher quantities of overland flow at which point energy for transportation of soil particles increases and can be related to the kinetic energy of moving water. The depth and velocity of the moving water increase downslope, as more water is generated by precipitation, until the force applied to the soil by the water is sufficient to exceed the resistance of soil to erosion, then sheet erosion occurs. This generally occurs uniformly from every part of the slope. The greater the kinetic energy of the moving water the more erosive the overland flow. The moving water creates tiny gullies, which are irregularly dispersed. This is called rill erosion. Rill erosion occurs often on soils that are left unprotected by either a crop or crop residue. Where the volume of water is concentrated, the formation of large or small ravines by undermining and downward cutting occurs. This is called gully erosion. While all types of erosion may be serious, the losses due to sheet and rill erosion are the most important from the standpoint of soil quality and crop production.

CONTROL OF SOIL EROSION

The traditional method of controlling agricultural sediment pollution is by the voluntary implementation of various practices that control soil erosion at its source. Most programs used to control agricultural nonpoint source pollution are modeled after the traditional agricultural stewardship efforts that focus on in-field conservation practices based on a landowner/operator’ s interest and operational constraints. It is important to note, the implementation of various conservation practices does not eliminate soil erosion, but lowers soil loss rates thus improving water quality. Physically and economically, it is impractical to eliminate all sediment from surface waters. Resource Management Systems (RMSs) are the combination of conservation practices that can be implemented by individual landowner/operator that would reduce soil erosion to more tolerable losses. While the erosion process can be complex the principles for controlling soil erosion are relatively simple.

RMSs can be divided into two classes. The first, cultural practices are nonstructural in nature. This group includes; conservation tillage systems, crop rotations, filter strips, contouring and strip cropping. The second group consists of structural practices, which usually require off-farm equipment for construction. The practices include: terraces, diversions, grassed waterways, and sediment basins. Economically, the two classes of practices can be distinguished on the basis of both initial and recurring cost. Structural practices are characterized by relatively high initial cost and lower annual cost, representing the cost of maintenance. Cultural practices, such as conservation tillage systems are dependent upon the costs of the annual inputs and equipment requirements.

The traditional planning approach for soil erosion control is to design an RMS that manages the soil to the tolerable soil loss limit (T), taking into account the planned crop rotation and addresses the predominate condition of the individual field. This results in some parts of the field being over managed and others under managed for soil erosion control. Individual RMSs must: control erosion, minimize environmental impact and be practically and economically sound by decreasing the volume and energy of runoff water by increasing crop residues.

FIELD LANDSCAPE

Landscape zones differ from one another based on the consistent pattern formed by their structured elements, and the predominant land management approach needed. The movement of material, energy and organisms between landscape zones is dependent on the movement of water. Landscape analysis considers the spatial juxtaposition and dynamic interaction between potential cropland erosion and runoff processes in the context of the water quality of the landscape for each field. Field landscapes can be divided into general categories based upon topographic characteristics and potential erosion types. Figure 1 landscape zone profile. The upland zone is characterized as being relatively flat less than 2% slope, ideally with permeable, well drained soils and the absence of sheet erosion. The relative magnitude of the eroding forces of sheet erosion is usually less than the resistance of the soil. Raindrop splash erosion does occur in the upland zone. The transition zone has slopes greater than 2%, ideally with well-drained soils. This zone can have a combination of sheet, rill and gully erosion, including those by natural erosion. In addition to raindrop splash, gravity and overland runoff cause erosion. Riparian Zone has a slope of 2% or less and is adjacent to a watercourse or drainage network. Soils generally are poorly drained to hydric and are characterized by being inundated by floodwaters at some interval and/or low permeability. The upland and riparian zones are typically the most productive for rowcrop production.

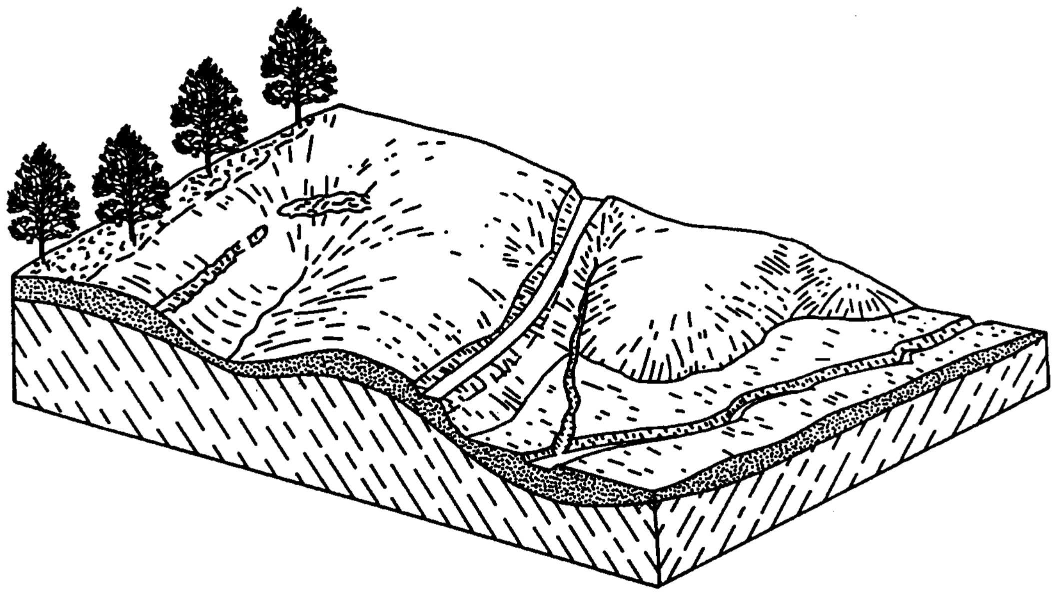

Figure 1: Landscape Zone Profile

Figure 2 shows a watershed with landscape zones. Individual fields can be solely in one zone or multiple zones. There can be up to three landscape zones within a field: upland, transition and riparian.

LANDSCAPE APPROACH

Using site-specific farming, the basic principles of the erosion process and landscape zone characteristics, a landscape approach to addressing soil erosion and sediment problems can be implemented. For each landscape zone a soil erosion/sediment management approach is established. This level of management becomes the minimum that is acceptable for cropland within that zone. Table 1 shows the concept developing management approaches by zone.

Table 1: General management approach by landscape zone.

Landscape zone/management goal.
Upland Zone: source control
Transitional: source control and manage pollutant transport
Riparian: source control, manage pollutant transport and treatment

Source control focuses on reducing the detachment of soil particles by rainfall and overland runoff. The overall goal of an RMS designed for this purpose would be to limit the amount of bare soil exposed to rainfall and runoff. Residue management reduces rain splash erosion and reduces overland flow by encouraging deposition.

Manage pollutant transport focuses on reducing and managing overland runoff. The overall goal of an RMS designed for this purpose would be to reduce the runoff energy, quantity and length of overland runoff. These RMSs either alter the runoff through increased infiltration and/or slowing the flow of water to reduce the velocity.

Treatment focuses the removal or remediation of nonpoint source pollutants. Planning and implementation of RMSs for this purpose focuses on creating sinks for sediment moving off agricultural fields. These sinks must be capable of intercepting sediment and must support one or more of the processes that remove sediment associated pollutants. The two key factors that must be considered are: (1) the capability of a particular area to intercept surface water borne sediment and (2) the activities of different pollutant removal processes. These RMSs are usually considered off-field control techniques. The most commonly used off-field control practices are vegetative filter strips, riparian buffer zones and constructed wetlands.

Traditionally RMSs are designed for predominate field condition and for the expected rotation, different criteria are needed in the landscape approach. The new criteria need to focus on designing RMS that are zone specific and address the maximum sediment production period of an individual rotation rather than the average condition for the rotation and focuses on meeting off-site environmental goals rather than "T" values. Within each zoneRMSs need to be designed so that they alter water runoff, provide cover for soil, and change the water absorption capacity of soil, rain splash energy, and soil structure. Table 2 shows the total amounts of soil erosion and water runoff with varying tillage systems in an Iowa small watershed, natural runoff study.

Table 2: Hydrologic effects of conservation tillage

Tillage	Soil Erosion Tons/Acre	Water Runoff Gallons/Acre
	Year 1	
Moldboard Plow	7.3	43,700
Ridge Till	1.4	21,400
No-Till	0.5	23,500
	Year 2	
Moldboard	23.1	87,600
Ridge-Till	10.1	58,800
No-Till	0.8	40,600
	Year 3	
Moldboard Plow	7.7	37,400
Ridge-Till	3.2	32,000
No-Till	1.6	34,200

CONCLUSIONS

With the advances in precision farming and our increased ability to provide site-specific management a landscape approach to soil erosion control and sediment management is possible. In order to increase our ability to manage and reduce off-site impacts, technical and financial assistance must be target to specific areas. Funding assistance should be limited to addressing implementing RMSs in the riparian and transitional zones, higher rates and greater funding limits need to available for implementation of RMSs in the riparian zone, in order to offset production losses and increased management. In addition to targeting assistance on a landscape basis, field planning should be revised to focus on the maximum sediment periods of a rotation rather the entire rotation, this will ensure that the worst case scenarios are addressed and an environmental margin of safety is applied.

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RETURN OF TWO GRASSED WATERSHEDS TO CROP PRODUCTION

Keith McGregor, Agricultural Engineer, USDA-ARS, Oxford, MS; John Schreiber, Soil Scientist, USDA-ARS, Oxford, MS; Robert Cullum, Agricultural Engineer, USDA-ARS, Oxford, MS; and Joseph Johnson, Agronomist, North Branch of Mississippi Agricultural and Forestry Experiment Station, Holly Springs, MS.

Abstract: Two watersheds at Holly Springs, MS, were returned to row crop production in 1994, thus simulating the similar return of Conservation Reserve Program (CRP) land to row crop production. Runoff and sediment yield was very low before and after the return of the two grassed watersheds to row crop production. One watershed (WS-1) (1.3 ha) had predominantly Lexington silt loam soils on slopes of 5 to 10 percent and Memphis silt loam soils on other parts of the watershed. The other watershed (DB-1) (1.8 ha) had Memphis silt loam and Loring silt loam soils on slopes of 2.5 percent and Providence silt loam soils on slopes of about 7.5 percent. The DB-1 watershed also contained shallow fragipan soils. No-till crops were planted between 6.1-m wide fescue grassed buffer strips in the spring of 1994 on both watersheds. Continuous cotton (*Gossypium hirsutum*) was grown on the DB-1 watershed for four years, but the watershed then was returned to grass because of the development of rills and small gullies. A corn (*Zea mays*)- cotton rotation was grown on the WS-1 watershed. The combination of no-till, cotton-corn rotation, and grassed buffer strips provided good erosion control on the WS-1 watershed. Results from this study will be useful to conservationists and farmers considering returning idle-land to row crop production.

INTRODUCTION

Farmers participating in the Conservation Reserve Program (CRP) have converted erodible, cultivated cropland into grassed areas. As the land became eligible for release, some farmers have returned CRP lands back to crop production. Unless more soil conservation oriented management practices are used with land returned to crop production than used prior to the CRP involvement, the land can be expected to eventually return to its previously erodible state.

Simulated rainfall experiments were used to measure runoff and erosion from former CRP areas in Mississippi, Nebraska, and South Dakota (Gilley and Doran, 1998). The soil loss rates measured immediately following tillage at these locations were similar to values obtained on undisturbed CRP treatments (Gilley and Doran, 1998). Gilley and Doran found that the erosion-reducing effectiveness of the sod, when left in a fallow condition, appeared to have lasted less than one year. They attributed the rapid increase in soil erodibility following tillage to a reduction in surface cover and organic material.

Development of systems utilizing no-till planting into the sod preserves soil benefits gained during CRP years (Broome and Triplett 1998). A combination of tillage and cropping practices that reduces the amount and frequency of tillage as much as possible while maximizing the amount of crop residues left on the soil surface effectively reduces erosion as compared to conventional-till. Crop residues left on the ground surface acts as surface roughness that increases surface retention and enhances infiltration. This residue also reduces flow velocity and reduces the surface area exposed to direct raindrop impact (McGregor, Mutchler, and Römkens 1990; Foster and Meyer 1975. Corn (*Zea mays*) residue amounts are normally higher than residues from small grain, soybean (*Glycine max L.*), and cotton (*Gossypium hirsutum*) (Mannering and Fenster 1977). Shredding increases the total surface area of residue, which is an advantage in covering the ground surface.

A cropping and management factor (C-value) for use in the Revised Universal Soil Loss Equation (RUSLE) is the ratio of soil loss from land cropped under specified conditions to the corresponding loss from tilled continuous fallow land (Wischmeier and Smith, 1978). Some C-values were computed from data measured on erosion plots under natural rainfall at Holly Springs, MS for over 30 years (McGregor et al., 1996). Some examples of tillage practices and cropping systems with low C-values included no-till corn and soybeans in a rotation system, no-till corn for silage and grain, reduced-till corn for grain, and conventional-till corn for silage and grain (McGregor, 1978; McGregor and Greer, 1982; and McGregor and Mutchler, 1983).

Tillage history affected erosion at Holly Springs, even on agricultural land receiving conventional-till (McGregor et al., 1996). Soil loss from no-till cotton after reduced-till soybeans was reduced by 53% as compared to that measured from conventional-till cotton after 11 years of no-till and by 75% as compared to that measured from conventional-till cotton after 11 years of conventional-till. Conventional-till cotton following 11 years of conventional-till produced more than 1.5 times the soil loss measured from conventional-till cotton following 11 years of no-till practices.

Contour strip-cropping, buffer strips, and filter strips have been used in conservation cropping systems. Perennial grassed buffer strips, usually located across the slope but at intervals up the slope, are much narrower than clean-tilled areas (Raffaelle, Jr. et al. 1997). Grassed filter strips usually are located at the base of slopes. Densely vegetated strips may cause deposition of eroded sediment (Renard et al., 1997).

The conservation measures on two grassland watersheds returned to crop production at Holly Springs included no-till cropping in combination with grassed buffer strips. The watersheds were returned to row crop production in 1994 with these implemented conservation measures in order to evaluate the ensuing sediment and agri-chemical losses. Soils at Holly Springs, MS are representative of loess over Coastal Plains material in the Southern Mississippi Valley Silty Uplands land resource area, and represent some of the most erodible soils in the world. This paper addresses the effectiveness of the conservation measures in maintaining low sediment yields on the watersheds following their return to row-crop production.

PROCEDURE

The Upland Erosion Processes and the Water Quality & Ecological Processes Research Units of the USDA National Sedimentation Laboratory (NSL), and the North Mississippi Branch of the Mississippi Agricultural and Forestry Experiment Station began cooperative research in 1992 on two small grassland watersheds (DB-1 and WS-1) at Holly Springs, MS. The watersheds, located on the Experiment Station, were returned to row crop production in the spring of 1994. The WS-1 watershed (1.3 ha) had predominantly Lexington (*Typic Paleudalfs*) silt loam soils on slopes of 5 to 10 percent, and Memphis (*Typic Hapludalfs)* silt loam soils on other parts of the watershed. The DB-1 watershed (1.8 ha) had Memphis silt loam and Loring (*Typic Fragiudalfs*)silt loam soils on slopes of 2.5 percent, and Providence (*Typic Fragiudalfs*) silt loam soils on slopes of about 7.5 percent. A fragipan was very shallow in some parts of the DB-1 watershed. The only row-crop grown on the WS-1 watershed between 1978 and 1994 was no-till corn in 1981. At that time no-till corn was planted in burn-downed fescue and harvested for silage. The DB-1 watershed had been in grass since conventional-till corn had been grown on the DB-1 watershed in 1984. The Brown Loam soils at the station, representative of the severely eroded loess soils of the Southeastern United States, are moderately to relatively high in fertility with high silt and low sand content and are easily eroded by the higher erosive rains typical of the region.

The watersheds were equipped with Parshall flumes, FW-1 water level recorders, and ISCO pumping samplers. Runoff and sediment measurements began in October of 1992 while the watersheds were still in grass. The grass was kept mowed in 1993. Fescue strips, 6.1-m wide, were drilled in the fall of 1993 on the DB-1 watershed to make buffer strips to use in combination with no-till cotton. No drilling was necessary on the other watershed because of existing fescue sod. Grass between the strips on both watersheds was killed with chemicals to prepare the land for row-crops. The buffer strips were laid out such that the percent grade for the strips was limited to about 1 %, except in the strip that acted as the waterway and carried runoff to the flume for each watershed the grade sometimes was as high as 2 to 3 %. . No-till crops were planted between 6.1-m wide fescue grassed buffer strips in the spring of 1994 on both watersheds. Buffer strips on both watersheds were well established by the time no-till row cropping began in 1994A corn (*Zea mays*)-cotton (Gossypium hirsutum) rotation was grown on the WS-1 watershed. Continuous cotton was grown on the DB-1 watershed for four years, but then was returned to grass because of the development of rills and small gullies. Fertilizer and chemicals were applied as recommended by the Mississippi Agricultural and Forestry Experiment Station.

RESULTS

Rainfall, Runoff, and Sediment Yield: Monthly rainfall, runoff, and sediment yield values from October of 1993 through September of 1998 are presented in Table 1 for both DB-1 and WS-1 watersheds. There were 19 and 18 and 1/2 months, respectively, for the DB-1 and WS-1 watersheds during which measurements were made while the watersheds were in grass and before the return of the watersheds to row crop production. The DB-1 watershed was returned to grass on April 2, 1998, allowing another six months of measurements for grass in 1998.

The average annual rainfall at Holly Springs from 1993 through 1998 water years was 1355 mm, or 17 mm less that the 30–year normal annual rainfall (NOAA, 1993). As expected, the runoff and sediment yield values were low for both watersheds while in grass. The conservation plans implemented on both watersheds were expected to produce low runoff and sediment yield values. However, the values were not expected to be as low as those shown in Table 1. The comparisons between water years and between watersheds for the entire 1993-1998 period can be seen more easily in Table 2, where the summaries of water year data from 1993 through 1998 water years (October – September) are shown. Since the data were presented by water years, the 1994 and 1998 water years reflect runoff and sediment yields for both cotton and grass on the DB-1 watershed.

Table 3 shows the rainfall, runoff, and sediment yield data for the two watersheds arranged in crop years, which allowed comparisons of data for years without grass. The crop years were from May through April. Cotton was planted in early May, while corn was planted in April, so the table uses "approximate crop" years. The average crop year rainfall (1994-1997) was 1397 mm, or 25 mm less than the normal average annual rainfall. Runoff averaged 39 % lower and sediment yield averaged over 200 % greater on the DB-1 watershed as compared to that from the WS-1 watersheds during the 1994-1997 crop years. But the annual runoff and sediment yield values on both watersheds were low. The low values of runoff and sediment yield did not reflect, however, the erosion that was taking place in the DB-1 watershed. Numerous rills and gullies formed in the DB-1 watershed by the end of the summer of 1997. The shallow fragipan soils contributed to the erodibility of the ground surface on the DB-1 watershed. We believe the increased erodibility resulted from reduced infiltration rates, limited water storage capacity, and lower hydraulic conductivities associated with the fragipan soils.

Rills and Gullies

Rills and small gullies formed in DB-1 in 1997, the fourth year of continuous no-till cotton row cropping. Because of the formation of the rills and small gullies, the DB-1 watershed was returned to grass in 1998. The rills and gullies generally originated immediately below the downstream side of the buffer strips (Figure 1). Nine small gullies, with reaches extending in a perpendicular direction to adjacent buffer strips, had maximum depths that averaged 0.2 m with a standard deviation of 0.1 m. The maximum width for each of these nine small gullies averaged 0.6 m with a standard deviation of 0.2 m. The average length of these gullies was 12 m with a standard deviation of 4 m. One gully developed in the original location of the natural drainage-way of the watershed. This gully was about 58 m in length, had a maximum depth of 0.4 m, and a maximum width of 1.2 m.

The formation of the rills and gullies would not be anticipated with the low runoff and sediment yield values that were measured from the DB-1 watershed. However, the formation of the rills and gullies showed that even a combination of no-till crops and buffer strips were not enough erosion protection for the shallow fragipan soils on the moderate to steep slopes of the watershed. The development of the rills and gullies in the fourth year of row-cropping indicated that returning CRP land to row crop production should not be recommended for moderate to steeply sloping land with fragipan soils.

Except for one small area near the edge of the field, there were no serious problems with formation of rills or small gullies in the WS-1 watershed. After 1997, the area from the edge of the field through the rill area was returned to grass. The remainder of WS-1 continued to be managed as before.

Crop and Residue Yields: Cotton yields on the DB-1 watershed averaged 1287, 1772, 1923, and 2120 kgm/ha of seed cotton in 1994, 1995, 1996 and 1997, respectively. Residues left on the ground at harvest on the DB-1 watershed averaged 9.6, 3.2, 6.8, and 6.30 t/ha in 1994, 1995,1996, and 1997, respectively.

Corn yields on the WS-1 watershed averaged 242,000 and 167,000 kgm/ha in 1994 and 1996, respectively. These yields contributed to residues left on the ground at harvest of 14.1 and 20.3 t/ha in 1994 and 1996, respectively. The cotton yields on the WS-1 watershed were 1772 and 3190 kg/ha in 1995 and 1997, respectively. These yields contributed to residues left on the ground at harvest of 4.2 and 6.3 t/ha in 1995 and 1997, respectively.

Future Plans for DB-1 and WS-1 Watersheds (1999-2004): The National Sedimentation Laboratory, the North Mississippi Branch of the Mississippi Agricultural and Forestry Experiment Station, and the Plant and Soil Science Department of Mississippi State University will evaluate the effects of applying poultry litter in the upper areas of the two watersheds on ensuing sediment and agri-chemical losses. The DB-1 watershed will be kept in grass with buffer strips maintained at a higher growth than the rest of the grassed areas to slow the velocity of runoff and to encourage deposition of sediment within the buffer areas. The DB-1 watershed will also make use of narrow stiff-grass hedges that now have been established at the top of the buffer strips and also halfway between adjacent buffer strips. These stiff-grassed hedges will be used to further restrict flow of runoff and to trap sediment. The present WS-1 corn-cotton rotation in combination with 6.1-m wide buffer strips will be maintained because of the opportunity and need for cooperative research on poultry litter applications on both grassed and row-cropped watersheds.

SUMMARY

The combination of no-till, cotton-corn rotation, and grassed buffer strips provided good erosion control on the WS-1 watershed following sod. Runoff and sediment yield values were low for both watersheds while in grass, but also were low after conservation plans were implemented on both watersheds. But the sediment yields for the DB-1 watershed did not reflect the rill and gully erosion within the watershed. The combination of no-till crops and buffer strips did not provide enough erosion control for the shallow fragipan soils on the moderate to steep slopes of the DB-1 watershed. The development of the rills and gullies in the fourth year of no-till row-cropping indicated that returning CRP land to row crop production (even no-till) should not be recommended for moderate to steeply sloping land with fragipan soils. However, CRP land without fragipan soils on moderate slopes can be returned successfully to row-crop production with implemented conservation measures. This conclusion is based on low values of sediment yield and the absence of significant rills and small gullies on the WS-1 watershed. Results from this study should prove useful to conservationists and farmers considering returning idle-land to row crop production.

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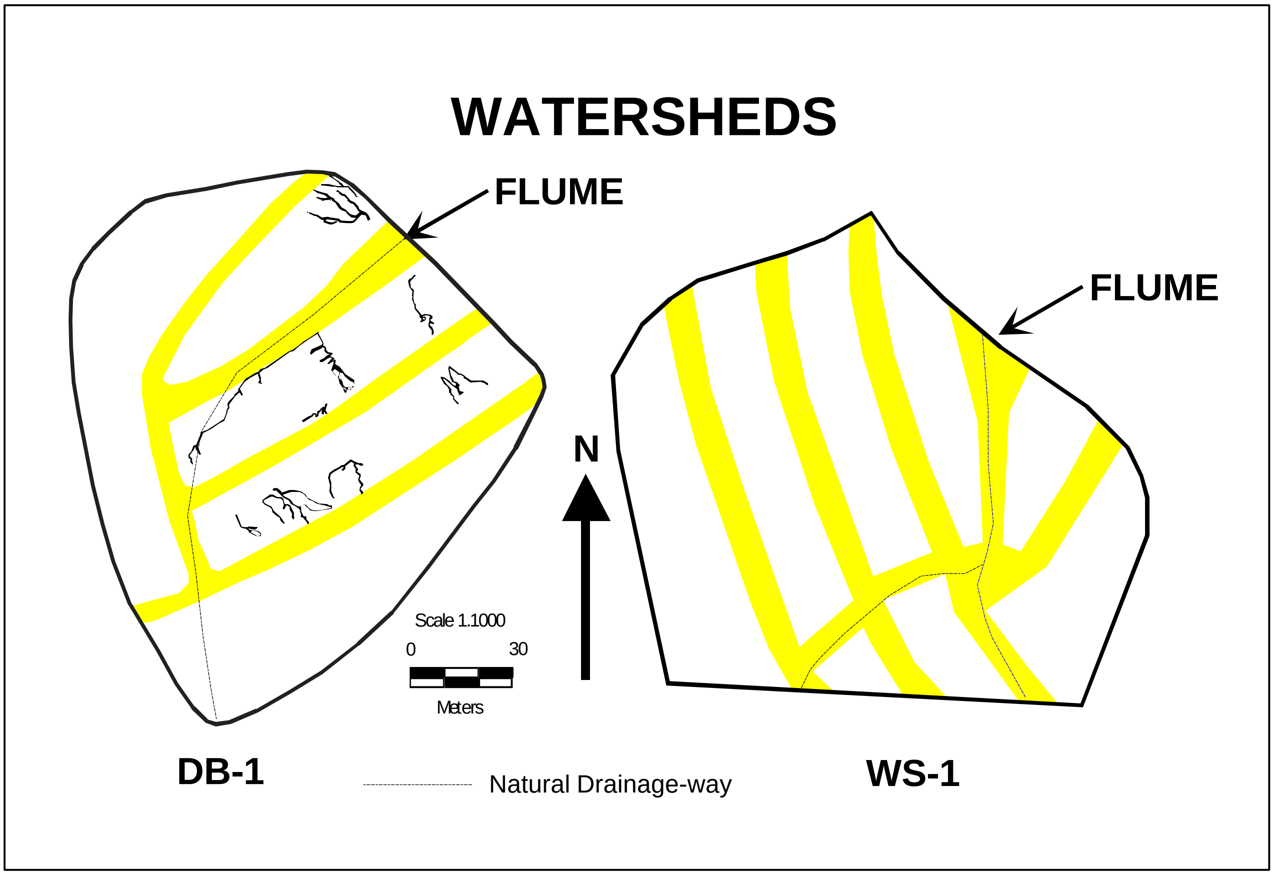


Figure 1. Layout of 6.1 m wide grassed buffer strips on the DB-1 and WS-1 watersheds. Crops were planted between the grassed buffer strips. Rills and small gullies developed in the fourth year after the fourth year of continuous no-till cotton are shown in the DB-1 watershed sketch. A corn -cotton rotation was grown on the WS-1 watershed.

Table 1. Monthly rainfall, runoff, and sediment yield from DB-1 and WS-1 watersheds during 1993-1998 water years (October through September).

Water Year		RAINFALL (mm)	DB-1[a] Cropping Treatment	RUNOFF (mm)	SED. YIELD (t/ha)	WS-1[a] Cropping Treatment	RUNOFF (mm)	SED. YIELD (t/ha)
1993	**Oct**	119	GRASS	0	0	GRASS	0	0
	Nov	83	GRASS	0	0	GRASS	5	0.002
	Dec	93	GRASS	7	0	GRASS	15	0.004
	Jan	96	GRASS	6	0	GRASS	17	0.004
	Feb	88	GRASS	10	0.002	GRASS	18	0.009
	March	98	GRASS	4	0	GRASS	15	0.002
	April	130	GRASS	13	0.002	GRASS	17	0.002
	May	77	GRASS	1	0	GRASS	2	0
	June	57	GRASS	0	0	GRASS	0	0
	July	107	GRASS	20	0.002	GRASS	4	0
	Aug	156	GRASS	22	0.04	GRASS	22	0.038
	Sept	138	GRASS	3	0	GRASS	8	0
	Sum	**1242**		**86**	**0.046**		**123**	**0.061**
1994	**Oct**	66	GRASS	6	0.002	GRASS	7	0
	Nov	76	GRASS	10	0.007	GRASS	12	0.002
	Dec	160	GRASS	37	0.002	GRASS	37	0.011
	Jan	132	GRASS	19	0	GRASS	39	0.002
	Feb	106	GRASS	13	0.004	GRASS	25	0
	March	209	GRASS	32	0	GRASS	57	0.002
	April	131	GRASS	8	0	CORN, 4/18	14	0
	May	98	COTTON, 5/5	5	0.002	CORN	1	0
	June	273	COTTON	102	0.132	CORN	101	0.009
	July	116	COTTON	23	0.031	CORN	9	0.025
	Aug	100	COTTON	11	0.013	CORN	11	0
	Sept	48	COTTON	0	0	CORN	0	0
	Sum	**1515**		**266**	**0.193**		**313**	**0.051**
1995	**Oct**	101	COTTON	0	0	CORN	2	0
	Nov	100	COTTON	36	0.045	CORN	25	0.011
	Dec	92	COTTON	3	0	CORN	9	0
	Jan	102	COTTON	8	0.016	CORN	9	0
	Feb	60	COTTON	5	0.004	CORN	7	0
	March	102	COTTON	19	0.011	CORN	29	0
	April	90	COTTON	0	0	CORN	0	0
	May	121	COTTON, 5/10	20	0	COTTON, 5/10	9	0
	June	35	COTTON	0	0	COTTON	0	0
	July	189	COTTON	23	0.049	COTTON	24	0
	Aug	175	COTTON	47	0.054	COTTON	55	0.007
	Sept	33	COTTON	0	0	COTTON	0	0
	Sum	**1200**		**161**	**0.179**		**169**	**0.018**

Table 1 (continued).

			DB-1			WS-1		
Water Year		**RAINFALL (mm)**	**Cropping Treatment**	**RUNOFF (mm)**	**SED. YIELD (t/ha)**	**Cropping Treatment**	**RUNOFF (mm)**	**SED. YIELD (t/ha)**
1996	**Oct**	69	COTTON	0	0	COTTON	0	0
	Nov	138	COTTON	13	0	COTTON	19	0
	Dec	156	COTTON	8	0.036	COTTON	27	0
	Jan	101	COTTON	14	0.211	COTTON	16	0
	Feb	59	COTTON	1	0.009	COTTON	0	0
	March	126	COTTON	11	0.025	COTTON	14	0.034
	April	58	COTTON	0	0	CORN, 4/86	9	0
	May	147	COTTON, 5/6	35	0.215	CORN	20	0.029
	June	118	COTTON	33	0.359	CORN	24	0.022
	July	84	COTTON	0	0	CORN	0	0
	Aug	33	COTTON	0	0	CORN	0	0
	Sept	229	COTTON	12	0.049	CORN	27	0
	Sum	**1318**		**127**	**0.904**		**156**	**0.085**
1997	**Oct**	118	COTTON	2	0	CORN	10	0
	Nov	162	COTTON	12	0.339	CORN	25	0
	Dec	116	COTTON	1	0	CORN	19	0.04
	Jan	121	COTTON	2	0.058	CORN	23	0.013
	Feb	164	COTTON	16	0.561	CORN	41	0.04
	March	182	COTTON	8	0.518	CORN	44	0.034
	April	83	COTTON	0	0	CORN	1	0
	May	109	COTTON	0	0	COTTON, 5/6	0	0
	June	189	COTTON	51	0.036	COTTON	46	0.377
	July	27	COTTON	0	0	COTTON	0	0
	Aug	123	COTTON	13	0.083	COTTON	7	0.002
	Sept	167	COTTON	23	0.072	COTTON	41	0.018
	Sum	**1561**		**128**	**1.667**		**257**	**0.524**
1998	**Oct**	77	COTTON	0	0	COTTON	0	0
	Nov	39	COTTON	0	0	COTTON	0	0
	Dec	87	COTTON	1	0	COTTON	12	0
	Jan	161	COTTON	2	0	COTTON	60	0.038
	Feb	165	COTTON	3	0	COTTON	72	0.049
	March	151	COTTON	12	0	COTTON	65	0.148
	April	194	GRASS, 4/2	6	0.087	CORN, 4/2	35	0.054
	May	56	GRASS	0	0	CORN	0	0
	June	71	GRASS	3	0	CORN	12	0
	July	202	GRASS	25	0.027	CORN	29	0.004
	Aug	55	GRASS	1	0.004	CORN	1	0
	Sept	38	GRASS	0	0	CORN	0	0
	Sum	**1296**		**53**	**0.118**		**286**	**0.293**

a. From Jan.-March 22,1998, the ISCO sampler was not working on the DB-1 watershed, so seven runoff events did not have samples taken during this period. However, only one of these storms had runoff > 3 mm, and it only had 12 mm, so sediment yield was assumed negligible. Due to a power failure, a sample was missing for one storm during September of 1996 on the WS-1 watershed, but again the runoff was only 13 mm, so sediment yield was assumed negligible.

Table 2. Rainfall, runoff, and soil loss from DB-1 and WS-1 watersheds during 1993-1998 Water Years (October through September). The DB-1 watershed was planted to continuous no-till cotton in 1994 following grass, while WS-1 was planted in a no-till corn-cotton rotation following grass. The DB-1 watershed was returned to grass in 1998.

		DB-1			WS-1		
Water Year	**RAINFALL (mm)**	**Cropping Treatment**	**RUNOFF (mm)**	**SED. YIELD (t/ha)**	**Cropping Treatment**	**RUNOFF (mm)**	**SED. YIELD (t/ha)**
1993	1242	GRASS	86	0.05	GRASS	123	0.06
1994	1515	COTTON, 5/5	266	0.19	CORN, 4/18	313	0.05
1995	1200	COTTON, 5/10	161	0.18	COTTON, 5/10	169	0.02
1996	1318	COTTON, 5/6	127	0.90	CORN, 4/8	156	0.08
1997	1561	COTTON, 5/6	128	1.67	COTTON, 5/6	257	0.52
1998	1296	GRASS, 4/2	53	0.12	CORN, 4/2	286	0.29

Table 3. Average rainfall, runoff, and soil loss from DB-1 and WS-1 watersheds during 1994-1997 crop years. The DB-1 watershed was planted to continuous cotton during this time, while WS-1 was planted in a corn-cotton rotation.

		DB-1			WS-1		
Crop Year[a]	**RAINFALL (mm)**	**Cropping Treatment[b]**	**RUNOFF (mm)**	**SED. YIELD (t/ha)**	**Cropping Treatment[b]**	**RUNOFF (mm)**	**SED. YIELD (t/ha)**
1994	1282	COTTON, 5/5	212	0.25	CORN, 4/18	203	0.05
1995	1260	COTTON, 5/10	137	0.38	COTTON, 5/10	173	0.04
1996	1557	COTTON, 5/6	121	2.10	CORN, 4/8	234	0.18
1997	1489	COTTON, 5/6	111	0.28	COTTON, 5/6	338	0.69
AVG	**1397**	**COTTON**	**145**	**0.75**	**CORN-COTTON**	**237**	**0.24**

a. The crop year here is defined as May though April of the following year.

b. The cropping treatment is shown with the planting date; however rainfall, runoff, and soil loss shown for May through April of the following year.

CHATTOOGA RIVER WATERSHED ECOLOGICAL/SEDIMENTATION PROJECT

By Bruce A. Pruitt[1], Watershed Hydrologist; Dave L. Melgaard[2], Aquatic Ecologist; Hoke Howard[1], Aquatic Ecologist; Morris C. Flexner[2], Fluvial Geomorphologist; Anthony S. Able[2], Geologist

[1]U.S. Environmental Protection Agency, Science and Ecosystem Support Division, Athens, GA, 980 College Station Rd., Athens, Georgia, 30605, (706) 355-8713, fax (706) 355-8726, pruitt.bruce@epa.gov; [2]U.S. Environmental Protection Agency, Water Management Division, SNAFC, 61 Forsyth St. SW, Atlanta, GA 30303

Abstract As an integral part of the comprehensive water quality investigation of the Chattooga River watershed, an ecological and sedimentological study was conducted on selected stream reaches within the study area. The objective of this study was to conduct a sediment yield study and determine if sediment was a primary cause of physical and biological impairment to streams within the watershed. As result of this study, accelerated sedimentation has been identified to be the leading determinant in loss of habitat and reduction in bedform diversity within the study area. Good correlation was observed between aquatic ecology and normalized total suspended solids (TSS) data. Based on overlaying the biological index on TSS normalized to discharge/mean discharge, TSS concentrations greater than 284 mg/l adversely affected aquatic macroinvertebrate community structure. However, based on historic regional suspended-sediment concentrations, a normalized TSS concentration of 58 mg/l or less during storm flow provides an adequate margin of safety and is protective of aquatic macroinvertebrates in the Blue Ridge physiography. Corresponding turbidity limits of 69 and 22 NTU established the threshold of biological impairment and margin of safety, respectively. Previously, a similar turbidity of 25 NTU has been recommended for stream restoration management plans. Relative to reference streams, impaired streams yielded higher bedload and suspended load. The results of this study showed that road density and associated sediment sources accounted for 51% of the total sediment loading.

INTRODUCTION

In response to issues included in the settlement of the Georgia Total Maximum Daily Load (TMDL) lawsuit, EPA was required to conduct an evaluation of the Chattooga River watershed to determine if waters within the watershed were not meeting designated uses (Sierra Club, Georgia Environmental Organizations, Inc., Coosa River Basin Initiative, Inc., Trout Unlimited, and the Ogeechee River Valley Association, Inc., Versus: U.S. Environment Protection Agency (EPA); Carol Browner, Administrator, EPA and John Hankinson, Regional Administrator, EPA Region 4). For those waters not meeting designated uses, EPA was required to determine the cause of non-support and develop the appropriate TMDL.

Sedimentation has been reported to be the leading determinant in loss of habitat and reduction in bedform diversity within the study area. The State of Georgia is initiating a statewide effort and geographic calibration of reference conditions for assessing the ecological status of its water resources using biological assessment. However, the effort has not been completed. As an interim solution, it was necessary to develop reference conditions at the scale of the Chattooga Basin. The objective of this study was to conduct a sediment yield study and determine if sediment was a primary cause of physical and biological impairment to streams within the watershed. The results were correlated with aquatic ecological data to develop an overall condition of the watershed.

Setting The Chattooga River watershed, located in northeast Georgia, northwest South Carolina, and southwest North Carolina, has a total drainage area of approximately 180,000 acres, and is entirely within the Blue Ridge Ecoregion. Land cover within the watershed is primarily forested, with some areas of commercial development, urban and residential use, and agriculture. Although the average "forested" land cover within the watershed is greater than 96%, there has been concern that gradual increases in sediment inputs to streams may be causing ecological impairment. Consequently, EPA Region 4 began an evaluation of water quality conditions within the Chattooga River watershed, and how they may have changed due to forestry or forestry-related practices. To accomplish this, sampling and analysis was undertaken in 1997-2000 by U.S. EPA Region 4 for biological and habitat quality, channel morphology, selected water chemistry, and sediment yield.

METHODS

Aquatic Ecology A total of 3 reference sites and 56 other sites were sampled from six subwatersheds: Headwaters (n = 14), Lower Chattooga (n = 3), Middle Chattooga (n = 10), Stekoa Creek (n = 7), West Fork (n = 11), and Warwoman Creek (n = 11). Biological sampling methods were focused on benthic macroinvertebrates and used modified rapid bioassessment protocols (RBP) (Plafkin et al. 1989, Barbour et al.1999, and U.S. EPA' s Region 4, Ecological Assessment Branch-*Draft* Standard Operating Procedures 1999). Reference sites were selected prior to initiation of sampling based on habitat condition, *in situ* water chemistry and surrounding land use. Reference sites R1 and R2 were located in the Chattooga River watershed and reference site R3 was on the upper Chattahoochee River outside of the Chattooga watershed. It was determined that the reference sites were representative of least-impaired conditions of the Blue Ridge Ecoregion. Data for all 59 stations were analyzed using a multimetric approach, in agreement with the recommendations of U. S. EPA (Gibson et al. 1996). From the raw data, 17 metrics were calculated including: total taxa, number of Ephemeroptera , Plecoptera, and Trichoptera (EPT) taxa , number of clinger taxa (clingers),percent clingers, percent most dominant taxon, percent 2nd dominant taxa, percent tolerant organisms, number of intolerant taxa, percent diptera, percent Chironomidae, percent EPT, North Carolina Biotic Index (NCBI), percent collectors, percent filterers, percent scrapers, percent shredders, and percent predators.

From the original list of 17 metrics, five were selected that had the greatest ability to detect impairment, determined by examining the position of the *a priori* reference sites to the overall distribution of metric values. For the most appropriate metrics, scoring criteria were determined based on the 95th percentile of all metric values for those metrics that decrease with impairment (Barbour et al.1999). For those that increase with impairment, the 5th percentile was used. This approach was used since there were no *a priori* impaired sites against which to calibrate. Each metric was scored according to its relation to the 95th (or 5th) percentile standard (Table 1). Eighty-five percent (85%) of the area below the 95th percentile standard (or 15% above the 5th percentile) was equally divided into four ranges and each range is given a numeric value of 0, 2, 4, or 6. A score of zero was the farthest away from the percentile standard (i.e., zero was most unlike the best attainable conditions and 6 was the score closest to the percentile standard). One exception was the "North Carolina Biotic Index" (NCBI), for which the scoring criteria developed by Lenat (1993) were used.

Table 1. Table of metrics and percentile distribution for each.

Metric	Min	05th	Median	95th	Max	Percentile Standard	Expected Response to Stressors
EPT taxa	3	10	15	21	25	95	Decrease
% EPT	27.9	36.7	66.7	85.0	95.4	95	Decrease
% 2 dominant taxon	19.2	22.0	30.0	52.8	65.4	5	Increase
NCBI	2.6	2.7	4.1	5.6	6.2	5	Increase
Clinger taxa	7	7	17	23	24	95	Decrease

A final biological index was assigned to each site based on a simple sum of the scores for the five metrics. An assessment rating was then assigned by dividing the range of the overall index scores into 5 categories. Narrative descriptions of the assessments correspond to:

- **Very Good** - best attainable conditions indicating no impairment to the aquatic community;
- **Good** - close to best attainable conditions but at risk and possibly influenced by limited stressors;
- **Fair** - some biological impairment observed, due to minor stressor input;
- **Poor** - substantial impairment of stream biota observed, due to moderate stressor input; including habitat degradation;
- **Very Poor** - severe impairment of stream biota observed, due to major stressor input, including habitat degradation.

Sediment Sampling Seventeen stream reaches were selected for storm flow investigations based on the following criteria: (1) relative degree of biological impairment as measured using RBP; (2) position within the watershed; (3) relative geomorphic condition; and (3) access logistics. The storm flow investigations were conducted during three storm events (March 28-30, 1998, June 15-17, 1999 and March 16-17, 2000). Prior to storm flow sampling, tape downs were established and appropriate cross-sections for gaging and sediment collection were identified. Base flow discharge and sediment samples were collected prior to the storm initiation. Precipitation was measured at Clayton, Georgia for response planning and rapid deployment of sample teams during the storm flow study. In addition, several rain gages were strategically deployed within the watershed to address rainfall distribution. Also, stream stage was monitored in Stekoa Creek at Clayton for response planning.

A total of 58 observations were made across the 17 stations. *In-situ* measurements at each station included tape downs (start and finish), stream discharge, turbidity, and collection of suspended and bedload sediment. Stream discharge was gaged simultaneously with sediment collection. Water column samples were collected using a depth integrating suspended hand-line sampler (US DH-59). Field turbidity was determined *in-situ* at ambient air conditions using a HACH™ Model 2100P Turbidity Meter. Turbidity was field determined for future use by EPA Region IV and state water quality personnel as a rapid means of identifying potential sediment impaired streams ("red flags"). Consequently, sample temperature was not adjusted prior to measuring turbidity. Laboratory determination of total suspended solids (TSS) and total dissolved solids (TDS) followed USEPA Methods 160.2 and 160.1, respectively. Whole samples were filtered for TSS analysis. Because the TSS data were produced without subsampling, they should be directly comparable to suspended-sediment concentration data (SSC) (Gray et al. 2000 and personal communication with John Gray, USGS). Bedload sediment samples were collected utilizing a 6-inch cable suspended bedload sampler or a 6-inch wading type bedload sampler, transported to the laboratory in 1-liter containers, and processed for particle size determination (PSD) in the laboratory using the EPA-SESD wet sieve method (SESD-EAB Draft SOP, Jan. 99). The procedure was followed with the exception of the silt/clay separation step that was not required since the samples were collected in coarse Nitex™ mesh bags (250 :m).

Laboratory results of dry-weight, bedload samples (M_b, grams) were converted to bedload transport rate (Q_b, tons/day) by the following equation (Edwards and Glysson 1988):

$$Q_B = K(W_T/T)\, M_T \qquad (1)$$

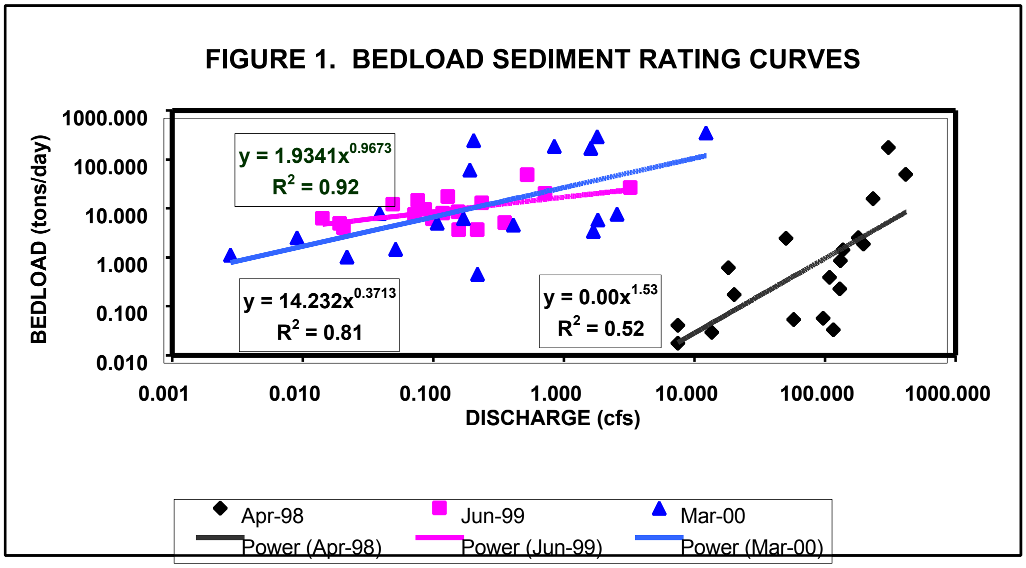
where Q_B = bedload discharge (tons/day);
K = converts grams/second/foot to tons/day/foot
W_T = wetted surface (ft);
T = total time sampler on bottom (seconds);
M_T = total mass of samples (grams)

Regression relationships were tested against ANOVA at a 95% confidence level. Consequently, unless otherwise noted hereafter, significance was determined at $\alpha = 0.05$, based on a t-test using advanced regression.

RESULTS

Aquatic Ecology Biological conditions in most streams sampled in this study show little or no impairment. Seventy-eight percent (78%) of the sites were rated as "very good" (22 sites) or "good" (24 sites). Since greater than 96% of the watershed land cover is classified as forested, this result was expected. Streams rated as "good" (41% of all stream sites sampled) are defined as possibly being influenced by some stressors. Eleven sites (19%) were rated as "fair", and two sites (3%) were rated as "poor". No sites were rated as "very poor". Although some sedimentation, or the habitat effects of sedimentation, may have been evident at many sites, a negative biological response was not always evident. The sedimentation also may not have reached a level that would cause a biological response. Due to the fact that this project used multihabitat sampling of benthic invertebrates, samples were taken from some stream subhabitats that were not adversely affected by sediment deposition resulting in habitat loss. The three reference sites had high biological scores: 24, 22, and 28, respectively, out of a maximum possible score of 30. The most degraded biological community was observed in the Stekoa Creek subwatershed. This subwatershed has a higher percentage of bare land and less forest cover than other subwatersheds in the Chattooga River basin. Consequently, none of the sample stations were rated as "very good" (i.e., zero out of seven stations). Two stations were rated "good", four stations were rated as "fair", and one station was rated as "poor".

Bedload Sediment Bedload over the three storm events averaged 13.32 tons/day (range 0.02-176.96 tons/day, standard deviation = 41.28). Median bedload particle sizes (D_{50}) ranged from fine sand to very coarse sand. Bedload accounted for only 14 percent of the total sediment load (on average). By plotting bedload against discharge, bedload sediment rating curves for each of the three storm events were created (Figure 1). Relatively good regression coefficients were observed within each storm event. However, regressed slopes varied between storm events.

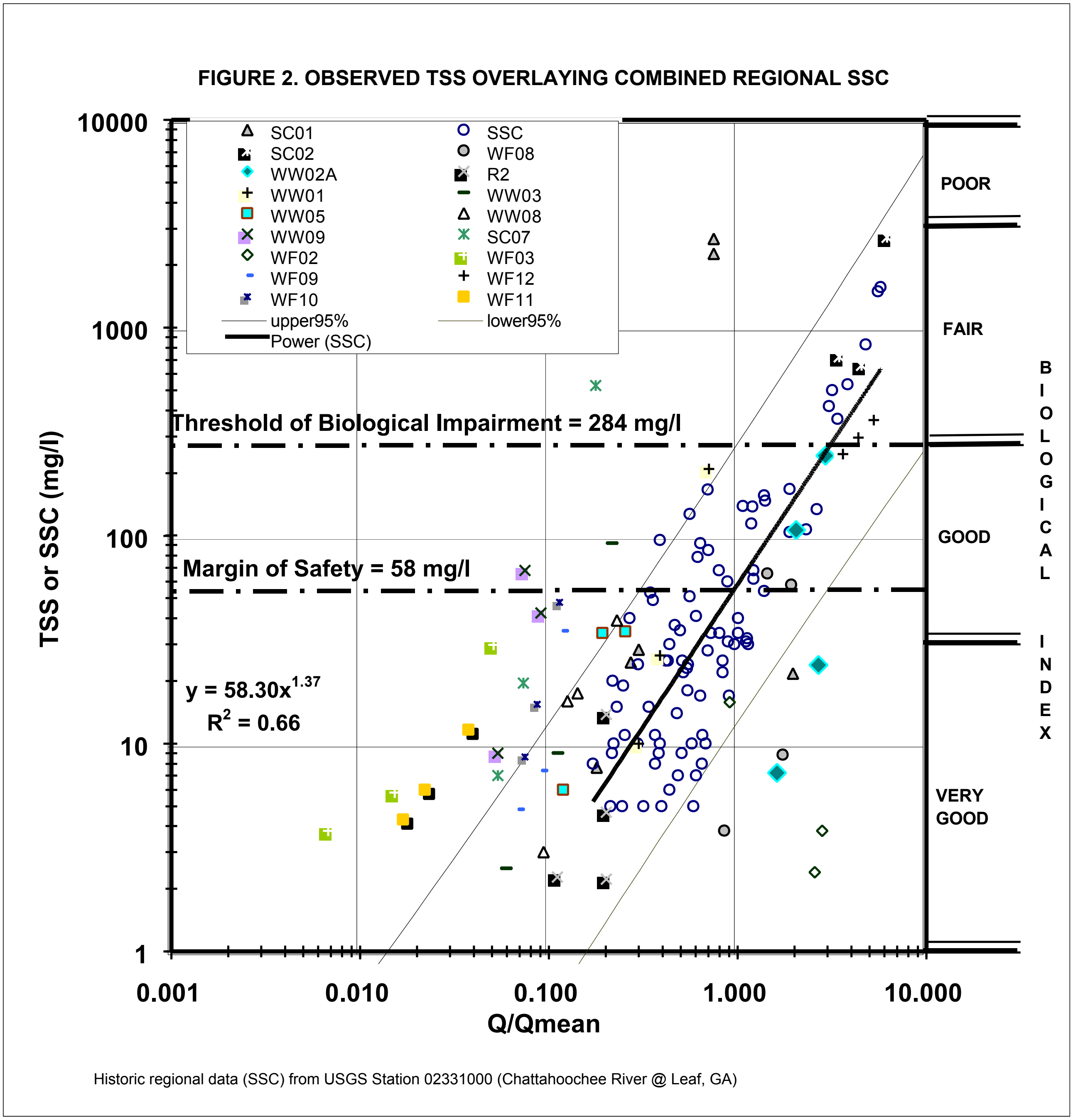


<u>Suspended Sediment (Regional)</u> Regional SSC data, compiled from the United States Geologic Survey records (Perlman 1984), were regressed against discharge normalized to mean discharge (Q/meanQ) (Holmbeck-Pelham and Rasmussen 1997). The USGS stream station utilized in development of the regional sediment curve was the Chattahoochee River near Leaf (Station no. 02331000) for the period of record, 1958 - 1984. TSS data from the Soque River station near Cornelia (02331250) and the Chestatee River near Dahlonega (02333500) were not used due to the difference in slope of the regression as compared to the Chattahoochee River station in the former and shift upward in the regression of the latter. An improvement was observed in the regression coefficient from 0.54 to 0.66 and, consequently, confidence in using the regional data set improved as a reference. In addition, SSC data from the Chattahoochee River was the most protective as compared to the other two datasets. Regional SSC (from the Chattahoochee River) regressed against Q/Qmean was observed to be significant (R^2=0.66, log transformed), given by (Figure 2):

$$\text{TSS or SSC} = 58.3(Q/Qmean)^{1.37} \qquad (2)$$

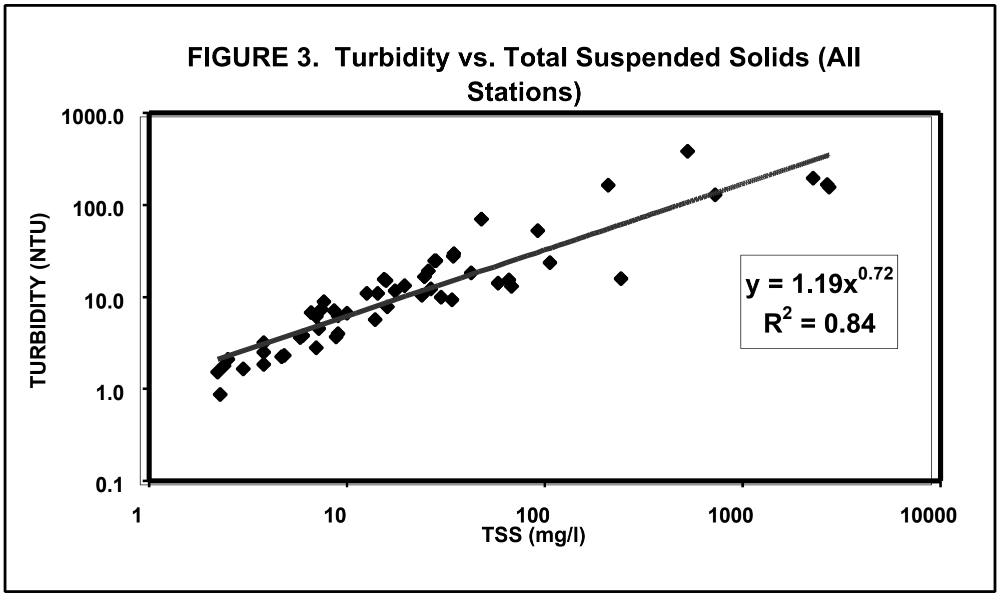
<u>Suspended Sediment (this study)</u> TSS over the three storm events averaged 85.3 tons/day (range 0.0002-3136.2 tons/day, standard deviation = 418.0). TSS accounted for the majority (86 %) of the total sediment load over the three storm events (on average). TSS, collected by vertical integration of the water column, was regressed against discharge (Q) and was observed to be highly variable between stations during the same storm event and between different storm events. In contrast, the log transformed relationship between TSS and NTU was significant (Figure 3). TSS data were compared against regional SSC by overlaying the two and constructing 95% confidence bands (Figure 2). Six stations, SC01, SC07, WW09, WF03, WF10 and WF11, were observed above the upper 95% confidence band (i.e., 6 out of the 17 stations during the three stormflow investigations). In general, data points that plot above the upper 95% confidence band are indicative of higher than "normal" concentrations of TSS for a given discharge to mean discharge. Other stations were observed to be below or within the normal range of the regional SSC data set. In addition, three stations, WW02A, WF02, and WF08, were below the lower 95% confidence band.

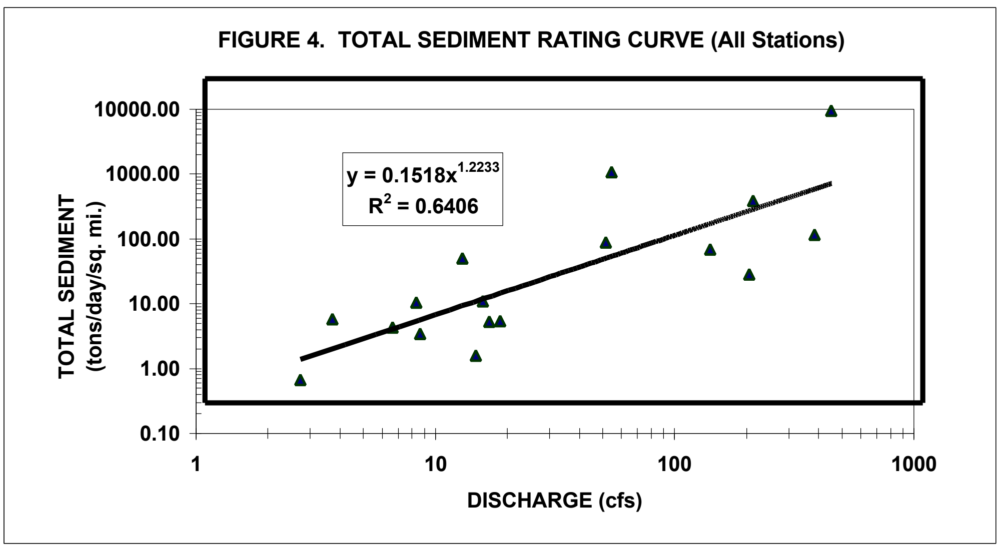
<u>Total Sediment</u> Bedload and TSS loadings were combined into total sediment load and plotted against discharge (Figure 4). Total loads were also plotted against road density (road length / corresponding drainage area) (Figure 5). Road density ranged from zero (R2 - Addie Branch, reference) to 6.60 (SC01 - Stekoa Creek. Road density represents the net impacts of road construction and maintenance, interception of subsurface interflow, routing of other non-point sources to the stream, and entrainment, mobilization, and transport of sediment to the stream. In contrast to drainage density, a significant increase in peak total loads in response to road density was observed at the two Stekoa Creek stations (SC01 and SC02).



CONCLUSIONS

Good correlation was observed between the biological index and normalized total suspended solids (TSS) data (Figure 2). TSS concentration normalized to discharge/mean discharge greater than 284 mg/l adversely affected biological community structure. However, based on regional suspended-sediment concentrations, a normalized TSS concentration of 58 mg/l or less during storm flow provides an adequate margin of safety and is protective of aquatic macroinvertebrates in the Blue Ridge physiography. Furthermore, corresponding turbidity limits from the above TSS estimates can be calculated from the NTU versus TSS relationship (Figure 3) as 69 and 22 NTU for the threshold of biological impairment and margin of safety, respectively.



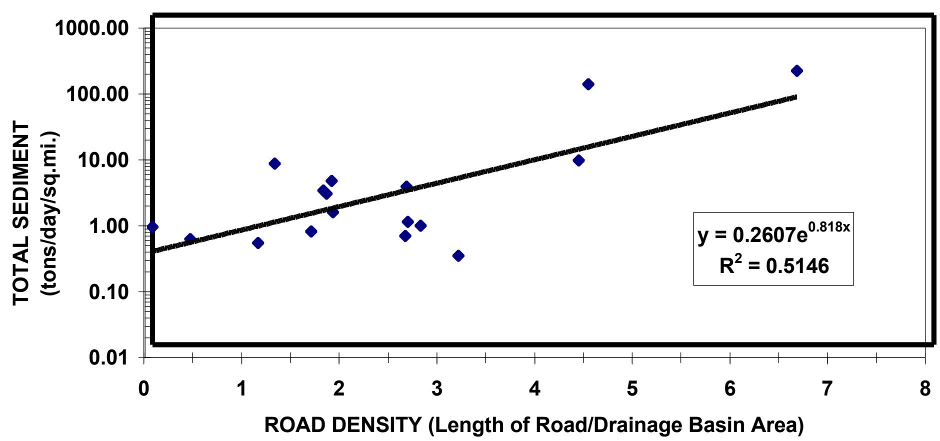


Relative to the reference stream (R2), impaired streams yielded higher bedload and suspended load. Based on the results of this study and comparison against regional sediment data, Stekoa Creek (SC01 and SC07) exhibits greater than "normal" suspended sediment loads. TSS concentrations from Addie Branch (R2) were within or below "normal" regional TSS concentrations. Total storm flow sediment load and peak total sediment loads did not increase significantly with drainage density. Increased sediment loads were correlated with an increase in road density. Road density and associated sediment sources accounted for 51% of the total sediment loading. Assuming that every road has at least one road ditch, road density nearly doubled the effective drainage density at the Stekoa Creek stations. The condition of the macroinvertebrate community of Stekoa Creek is rated as "fair" and is evidence of the impact of the accelerated sediment loads in the stream at stations SC01, SC02, and SC07.

DISCUSSION

Presently, several states are evaluating their water quality standards to include narrative or numeric turbidity and/or TSS standards. For example, Georgia has recently enacted a narrative standard for turbidity that is based on "visual contrast in a water body due to man-made activity" (DNR 2000). In addition, Alabama and Florida use 50 and 29 NTU above background, respectively; South Carolina allows a increase of ten percent above background; North Carolina uses 10 NTU for trout streams, 50 NTU for non-trout streams, and 25 NTU for non-trout lakes; Tennessee uses a standard that does not allow any material effect on fish or aquatic life (Kundell and Rasmussen 1995). Holmbeck-Pelham and Rasmussen (1997) recommended a reduction in average turbidities to below 25 NTU for stream restoration plans in Georgia. In addition, a turbidity of 25 NTU was recommended by the Georgia Board of Regents' Scientific Panel as an instream turbidity standard (Kundell and Rasmussen 1995). Also, the report cited a TSS concentration of 80 mg/l as a threshold between moderate and low levels of protection for fish and aquatic invertebrates (NAS 1972).

FIGURE 5. PEAK TOTAL SEDIMENT LOAD DURING STORM EVENT



Similar findings were observed in this study. TSS concentrations greater than 284 mg/l resulted in biological impairment of macroinvertebrate communities. Also, TSS concentrations of 58 mg/l or less during storm flow provided an adequate margin of safety and were protective of aquatic macroinvertebrates in the Blue Ridge physiography. Furthermore, corresponding turbidity limits of 69 and 22 NTU established the threshold of biological impairment and margin of safety, respectively.

A relationship between TSS and turbidity (NTU) can be developed within a specific hydro-physiography. Turbidity can be used as a surrogate to TSS with the following assumptions and cautions: 1) the relationship between TSS vs. NTU is hydro-physiography specific; 2) turbidity includes inorganic and organic constituents including phyto- and zooplankton which can be extreme during the growing season; and 3) stream discharge and/or stage should be measured at the time of turbidity measurements and compared against a regional regression curve.
A biological endpoint is critical to addressing stream condition and beneficial uses. An index of biological integrity overlaying a sliding, sediment scale (concentration or load) is recommended. Additional surrogates need to be developed and tested between bedload versus embeddedness (MacDonald et al. 1991), bedload versus one-third lower bar (Rosgen 1996), and sediment load versus Pfankuch (1975) or RBP habitat assessments (Plafkin et al. 1989).

The relationship between suspended-sediment concentration and total suspended solids needs to be established for specific physiographies. In addition, in physiographies with high concentrations of clay particle sizes, filtration of the whole sample needs to be explored *in lieu* of withdrawing the supernatant using a J-tube.

The findings of this study emphasize the importance of incorporating aquatic ecological assessments into addressing the effects of accelerated sedimentation and deposition within a watershed. Biological endpoints (e.g., clinger-burrower ratio) can be directly applied to designate beneficial uses such as fishing and recreation. Consequently, comprehensive aquatic ecological studies are a critical component of identifying reference stream reaches and determining whether designated or beneficial uses are being met. Additional research should focus on developing fisheries and aquatic macroinvertebrate indices that are sensitive to impacts caused by accelerated sedimentation.

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