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February 9, 2011

Karen A. O'Haire
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Dear Ms. O'Haire:

Thank you for your letter dated December 14, 2010, requesting additional information from the Department of Forestry and Fire Protection (CAL FIRE), and other petitioners for the State Water Resources Control Board (State Water Board) review of petitions A-2029, A-2029(a), and A-2029(b). Specifically, your correspondence requests additional evidence, such as water quality monitoring results or studies, on the following:

1. The waiver's specific conditions to control sediment discharges (i.e., Road Management Plans, Erosion Control Plans/or Sediment Prevention Plans);

2. The waiver's specific conditions for control of thermal discharges (i.e., riparian shade canopy retention standards), and;

3. Sediment and thermal discharges from timber operations conducted under Nonindustrial Timberland Management Plans (NTMPs) covered under North Coast Regional Water Quality Control Board (North Coast Water Board) Order No. R1-2004-0016, Categorical Waiver of Waste Discharge Requirements for Discharges related to timber harvest activities on Non-Federal Lands in the North Coast Region, adopted June 23, 2004.

Due to the abundance of information, CAL FIRE appreciates the 30-day extension provided by the State Water Board so that the additional evidence requested could be collected and compiled. The information is included on the attached compact disc (CD) in three files:

1. Water quality related monitoring reports and studies specific to the control sediment discharges from timber harvesting, including road management measures, erosion control, and sediment prevention;

2. Riparian shade canopy retention requirements monitoring and studies for the protection of stream water temperatures (i.e., control of thermal discharges), and;

3. Sediment and thermal discharge enforcement actions taken on timber operations under NTMPs covered under North Coast Water Board Order No. R1-2004-0016, Categorical Waiver of Waste Discharge Requirements for Discharges related to timber harvest activities on Non-Federal Lands in the North Coast Region, adopted June 23, 2004. These include both NTMP-related Notices of Violation (NOVs) issued by CAL FIRE and NTMP-related Orders adopted by the North.Coast Water Board indicating NTMP-related North Coast Basin Plan violations (Orders).

In addition, per your request, CAL FIRE has provided two files on the attached CD with factual information regarding each NTMP approved by CAL FIRE in the North coast Region.

CAL FIRE has provided monitoring data both from within California and from other western states with similar watershed conditions to California. The California data provided relates specifically to the efficacy of the Forest Practice Rules under which NTMPs operate. The large body of monitoring information and relevant studies included under items one and two above demonstrate that: (1) The rate of compliance with the California Forest Practice Rules (FPRs) (i.e., proper implementation) designed to protect water quality and aquatic habitat (including riparian shade canopy retention) is generally high, and (2) the FPRs are highly effective in preventing erosion, sedimentation and sediment transport to watercourse channels when properly implemented.

We know of no comprehensive monitoring work or studies that demonstrate the need for further regulation of NTMPs to protect water quality, as prescribed in the Waiver adopted June 4, 2009 by the North Coast Water Board. Neither CAL FIRE's enforcement records nor the enforcement records of the North Coast Water Board indicate that there were significant sediment and/or thermal discharge problems associated with timber harvesting conducted under approved NTMPs that were subject to both the FPRs and the Waiver adopted by the North Coast Water Board on June 23, 2004.

Our appeal is supported by both the monitoring information and studies available on the subject and included on the attached compact disc. Our monitoring results and supported-research studies have demonstrated that the FPRs are highly effective in protecting water quality when properly implemented.

The studies provided on the CD are not an exhaustive compilation, but is a broad representation of the current available science. Further, we do not know of any pertinent and comprehensive monitoring studies that are inconsistent with this conclusion, nor to the best of our knowledge, have any such studies been cited by North Coast Water Board staff.

Please call me at (916) 653-4153 if you have any questions.

Sincerely,

Ginevra K. Chandler

Ginevra K. Chandler, Esq.
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cc: see attached list

Attachment: one compact disc

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Effects of Forest Management on Streamflow, Sediment Yield, and Erosion, Caspar Creek Experimental Watersheds

Elizabeth Keppeler, Jack Lewis, Thomas Lisle

Abstract

Caspar Creek Experimental Watersheds were established in 1962 to research the effects of forest management on streamflow, sedimentation, and erosion in the rainfall-dominated, forested watersheds of north coastal California. Currently, 21 stream sites are gaged in the North Fork (473 ha) and South Fork (424 ha) of Caspar Creek. From 1971 to 1973, 65% of the timber volume in the South Fork was selectively cut and tractor yarded, and from 1985 to 1991, 50% of the North Fork basin was harvested, mostly as cable-yarded clearcuts. Three unlogged tributaries serve as controls.

Annual suspended sediment loads changed 331% after logging the South Fork compared to 89% for the North Fork and -40% to 269% for North Fork subwatersheds. In clearcut units, storm peaks increased as much as 300%, but as basin wetness increased, percentage peak flow increases declined. Flow increases are explained by reduced transpiration and interception. Ongoing measurements show a return to pre-treatment flow conditions approximately 12 years post-harvest, but sediment yields have yet to recover.

Landslides are predominantly associated with roads, landings, and tractor skid trails in the South Fork watershed and windthrow in the North Fork watershed.

Keywords: peak flow, sediment, erosion, landslides, timber harvest

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Introduction

For more than four decades, researchers have investigated the effects of forest management on streamflow, sedimentation, and erosion in the Caspar Creek Experimental Watersheds of north coastal California. The California Department of Forestry and Fire Protection and the USDA Forest Service, Pacific Southwest Research Station, began a simple paired watershed study in 1962 with the construction of weirs on the two major Caspar Creek tributaries, the North Fork and the South Fork. Initially, this partnership was born out of necessity. The research station was charged with evaluating harvest impacts in major timber production regions, but the National Forest system lacked significant ownership within the coast redwood Douglas-fir forest type. The Jackson Demonstration State Forest, comprised of nearly 20,000 ha of second-growth forest, met this need, and a successful, long-standing partnership was begun. As management practices have evolved, so, too, have the research questions and technologies. Today, researchers operate 21 gaging stations within the experimental watersheds and utilize state-of-the-art data loggers programmed with sophisticated sampling algorithms, instream turbidimeters, and automated pumping samplers to measure discharge and sediment transport. Additional investigations of the processes important to hydrologic and ecosystem function are emphasized. The Caspar Creek Experimental Watersheds have produced a wealth of data and an extensive library of scientific publications used to guide natural resource management policy.

Methods

Site

The Caspar Creek Experimental Watersheds are located about 7 km from the Pacific Ocean and about 10 km south of Fort Bragg in northwestern California at 39°21'N 123°44'W (Figure 1). Uplifted marine terraces incised by antecedent drainages define the youthful and highly erodible topography. Hillslopes are steepest near the stream channel and become gentler near the broad, rounded ridgetops. About 35% of the basins' slopes are less than 17 degrees, and 7% are steeper than 35 degrees. Elevation ranges from 37 to 320 m.

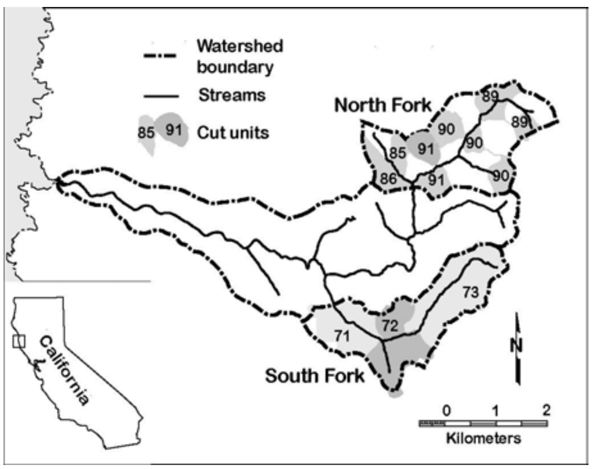

Figure 1. Caspar Creek Experimental Watersheds.

Soils are well-drained clay-loams, 1 to 2 meters in depth, derived from Franciscan greywacke sandstone and weathered, coarse-grained shale of Cretaceous age. Hydraulic conductivities are high and subsurface stormflow is rapid, producing saturated areas of only limited extent and duration (Wosika 1981).

The climate is typical of low-elevation coastal watersheds of the Pacific Northwest. Winters are mild and wet, characterized by periods of low-intensity rainfall delivered by the westerly flow of the Pacific jet stream. Snow is rare. Average annual precipitation is 1170 mm. Typically, 95% falls during the months of October through April. Summers are moderately warm and dry with maximum temperatures moderated by frequent coastal fog. Mean annual runoff is 650 mm.

Like most of California's north coast, the watersheds were clearcut and broadcast burned largely prior to 1900. By 1960, the watersheds supported an 80-year old second-growth forest composed of coast redwood (*Sequoia sempervirens* (D.Don) Endl.), Douglas-fir (*Pseudotsuga menziesii (*Mirb.) Franco), western hemlock *(Tsuga heterophylla* (Raf.).Sarg.), and grand fir (*Abies grandis* (Dougl. ex D.Don) Lindl.). Forest basal area was about 700 m^3 ha^{-1}.
Anadromous fish, including both coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) inhabit the North Fork and the South Fork of Caspar Creek and are protected by state and federal endangered species regulations.

Study design

The Caspar Creek study is a classic paired watershed design where one or more gaged catchments are designated as controls and others are treated with road building, logging, and other timber management practices. After a calibration period wherein a statistical relationship between the catchments is defined, any subsequent change is inferred to be a treatment effect.

The 473-ha North Fork of Caspar Creek and the 424-ha South Fork of Caspar Creek have been gaged continuously since 1962 using 120° V-notch weirs widening to concrete rectangular sections for high discharges. During the early 1980s, three rated sections were constructed upstream of the North Fork weir and 10 Parshall flumes were installed on North Fork subwatersheds with drainage areas of 10 to 77 ha.

Stream discharge was initially recorded using mechanical chart recorders. These were replaced in the mid-1980s with electronic data loggers equipped with pressure transducers. Subsequent upgrades have been implemented as technology has progressed. Early suspended sediment estimates were derived from sediment rating curves, manual depth-integrated sampling, and fixed stage samplers (Rice, et al. 1979). Statistically based sampling algorithms that trigger automated samplers were utilized beginning in the 1980s (Lewis, et al. 2001). In addition, an annual survey of sediment accumulation in the settling basin upstream of each weir has been made since 1963.

Erosion measurements include periodic field surveys to document the location, size, and disposition of landslides. Erosion features greater than 7.6 m^3 (10 y^3) have been recorded annually since 1986. Erosion has on occasion been sampled at a finer scale using erosion plots (Rice et al. 1979, Rice 1996).

Treatment phase I: selection harvest with tractor yarding

After establishing a calibration relationship between the North Fork and the South Fork (1963 to 1967), a main-haul logging road and main spurs were built in the South Fork. The road right-of-way occupied 19 ha, from which 993 m^3 ha^{-1} of timber was harvested. The entire south Fork watershed was logged and tractor yarded between 1971 and 1973 using single-tree and small group selection to harvest 65% of the stand volume. Roads, landings, and skid trails covered approximately 15% of the South Fork watershed area (Ziemer 1981).

Treatment phase II: clearcutting with skyline-cable yarding

A study of cumulative effects began in 1985 in the North Fork watershed. Three gaged tributary watersheds within the North Fork were designated as controls while seven were designated for harvest in compliance with the California Forest Practice Rules in effect in the late 1980s. Two units (13% of the North Fork watershed) were clearcut in 1985-86 and excluded from the cumulative effects study. However, this harvest affects all subsequent analyses of North Fork weir data. After a calibration period between 1985 and 1989, clearcut logging began elsewhere in the North Fork in May 1989 and was completed in January 1992. Clearcuts occupied 30-99% of treated watersheds and totaled 162 ha. Between 1985 and 1992, 46% of the North Fork watershed was clearcut, 1.5% was thinned, and 2% was cleared for road right-of-way (Henry 1998).

In contrast to the harvest treatment of the South Fork in the 1970s, stream-buffer rules mandated equipment exclusion and 50% canopy retention within 15 to 46 m of watercourses providing aquatic habitat or having fish present. Most of the yarding (81% of the clearcut area) was accomplished using skyline-cable systems. Yarders were situated on upslope landings constructed well away from the stream network. New road construction and tractor skidding was restricted to ridgetop locations with slopes generally less than 20%. Four harvest blocks, 92 ha total, were broadcast burned and later treated with herbicide to control competition (Lewis, et al. 2001). Pre-commercial thinning in 1995, 1998, and 2001 eliminated much of the dense revegetation and reduced basal area in treated units by about 75%.

Results

Storm peaks

Ziemer (1981) analyzed peak discharges from 174 storm peaks occurring between 1963 and 1975 and later (1998) expanded upon this analysis with data collected through 1985. This analysis detected no significant increases in storm peaks following selection harvest of 65% of the South Fork watershed stand volume *except* within the smallest flow classes (recurrence interval less than 0.125 year). Early fall peaks increased by about 300%, but these were small storm events.

Lewis et al. (2001) analyzed the peak flow response to clearcutting in the North Fork using 526 observations representing 59 storms on 10 treated watersheds. After logging, eight of the 10 tributary watersheds experienced increased storm peaks ($p < .005$). In clearcut units, storm peaks increased as much as 300%, but most increases were less than 100%. The largest increases occurred during early season storms. As basin wetness increased, percentage peak flow increases declined. In the larger, partially clearcut North Fork watersheds, smaller peak flow increases were observed. Under the wettest antecedent moisture conditions of the study, increases averaged 23% in clearcut watersheds and 3% in partially clearcut watersheds. The average storm peak with a 2-year return period increased 27% in the clearcut watersheds (Ziemer 1998) and 15% in the partially clearcut watersheds. Ongoing measurements show a return to pre-treatment flow conditions approximately 12 years post-harvest and minimal response to the pre-commercial thinning (Figure 2).

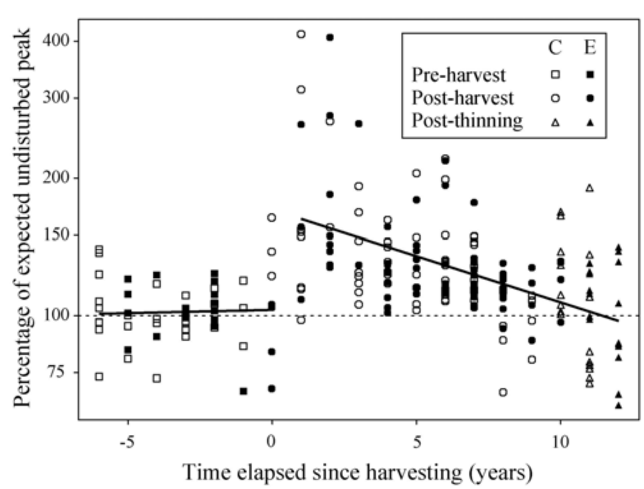

Figure 2. Peak flows observed in North Fork clearcut units C and E from 1986 through April 2003. Reduced transpiration resulting in wetter soils in logged units explains some of the observed increases in streamflow. In addition, recent research at Caspar Creek has documented significant increases in net precipitation within clearcut areas due to reduced canopy interception. Under forested conditions, canopy interception is significant even during the wettest mid-season storms. Preliminary results show that, annually, about 20% more precipitation is delivered to the forest floor after logging.

Sediment loads

Sediment load estimates for the North Fork and South Fork are the sum of the sediment deposited in the weir pond and the suspended load measured at the weir. Comparison of sediment loads produced following the 1971-73 harvest of South Fork and the 1989-92 harvest of North Fork must be made cautiously. Improved and more intensive sampling methods greatly enhance the accuracy of load estimates for the latter study. And large landslides in the North Fork in 1974 and 1995 strongly influence the comparison.

On the South Fork, the suspended sediment loads increased 335% after road building and averaged 331% greater during the 6-year period after tractor yarding. Annual sediment load (including suspended and pond accumulations) increased 184% for the 6-year post-harvest period 1972-1978, returning to pretreatment levels in 1979 (Lewis 1998).

Using the South Fork as the control basin for logging the North Fork, no significant change in annual sediment load was detected after clearcutting 48% of the watershed area. However, analyses using tributary controls were more illuminating. Suspended sediment loads changed 89% at the North Fork weir, primarily due to one landslide in 1995, and –40% to 269% at other gaged locations. The mean annual sediment load increased 212% (262 kg ha^{-1}yr^{-1}) in clearcuts and 73% (263 kg ha^{-1}yr^{-1}) in partially clearcut watersheds. Recent data analysis suggests that sediment loads in North Fork tributaries remain elevated through water year 2002, more than a decade after harvest (Figure 3).

Erosion

Increased sediment loads in the South Fork following road building and tractor harvest are explained by increased sediment delivery to stream channels (Rice 1979). Road building and bridge construction within the riparian zone directly impacted much of the perennial stream. The following winter, 36 discrete landslides were documented along the newly constructed road—17 delivered an estimated 822 m^3 to the stream and 19 deposited 382 m^3 along the road

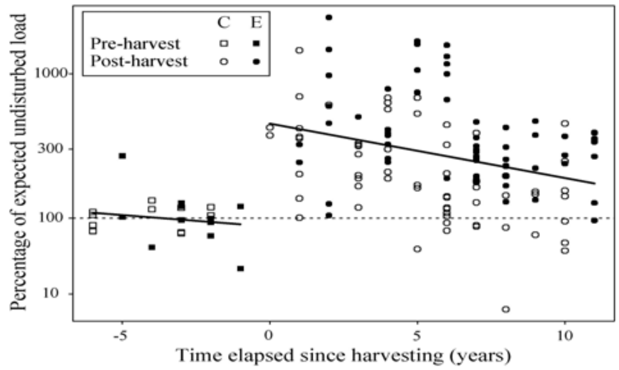

Figure 3. Sediment loads observed in North Fork clearcut units C and E from hydrologic year 1986 through 2002.

surface (Krammes and Burns 1973). Aerial photos of South Fork Caspar taken in 1975 portray 66 recently active landslides. Of these, all but three are associated with roads, landings, or skid trails (Cafferata and Spittler 1998). A field survey of landslides conducted in 1976, three years after tractor harvest was completed on the South Fork, recorded 99 discrete erosion features as small as 4.2 m^3 (150 ft^2). Landslides displaced approximately 189 m^3 ha^{-1} of material (Tilley and Rice 1977). Of these, 85% were associated with roads, landings, or skid trails. In 1994, this survey was repeated documenting 10 additional or re-activated landslides displacing 1515 m^3 of material. Only two of these were not road-related. Another episode of road-related landsliding was observed in the mid-1990s as stream crossing failures became more common. Of the 38 South Fork landslides documented between 1994 and 2003, 89% are road, landing, or skid trail related. These more recent landslides displaced 5804 m^3 and delivered 3503 m^3 to the stream channel. An aging system of logging roads and skid trails continues to deliver sediment to the stream channel.

North Fork sediment load increases were correlated to flow increases and, to a lesser degree, the length of intermittent channels logged or burned (Lewis et al. 2001). Increased erosion is attributed to increased gullying of headwater channels. Field investigations documented gullying and bank erosion in unbuffered channels subjected to intense broadcast burns and logging disturbance.

The annual inventory of failures exceeding 7.6 m^3 suggests that post-harvest erosion and sediment delivery mechanisms are quite different in the North Fork than were documented in the South Fork (Table 1). North Fork windthrow plays a far greater role in soil displacement, but delivers less displaced sediment to stream channels. Of 145 erosion features documented post-harvest (1990-2003), 84 were windthrow-related and only 10 were road-related. Uncut areas of the North Fork are included in this tally because these areas were impacted by edge-effect windthrow and new road construction. Windthrow displaced 2240 m^3 but delivered only 27% of this sediment. Clearcutting left adjacent timber stands and riparian buffers vulnerable to windthrow, but relatively little of the sediment displaced by uprooted trees was delivered to the stream. In contrast, road-related landslides on the North Fork delivered about half of the 3264 m^3 volume displaced. Most of these, including the largest (2012 m^3), are associated with the pre-existing mid-slope road that spans the north side of the watershed. This road was constructed circa 1950 to the standards of the time.

Table 1. Comparison of post-harvest Erosion features inventoried on the North Fork and South Fork.

Erosion Features	**South Fork**	**North Fork**
6-year post-harvest[1]		
Total number	99	81
Volume (m^3)	80046[2]	7285
Delivered Volume (%)	na	39%
Road-related number	85	6
Volume (m^3)	na	533
Delivered Volume (%)	na	8%
Windthrow-related number	na	45
Volume (m^3)	na	1204
Delivered Volume (%)	na	25%
1990-2003		
Total number	38	145
Volume (m^3)	5804	11878
Delivered Volume (%)	61%	45%
Road-related number	34	10
Volume (m^3)	5556	3264
Delivered Volume (%)	63%	52%
Windthrow-related number	5	84
Volume (m^3)	316	2240
Delivered Volume (%)	20%	27%

[1]1971-1976 on South Fork, 1990-1995 on North Fork. [2]Reported as 100 yd^3 $acre^{-1}$ (Tilley and Rice 1977).

Most of the erosion features discussed above are smaller than 76 m^3. Of greater concern to land managers is how timber harvest alters the frequency of large landslides. Debris slides account for a major amount of mass wasting within the Franciscan geology of the Caspar Creek region. Such landslides occur infrequently in response to critical rainfall intensities. Clearly, mass wasting increased following tractor harvest of the South Fork, but attempts to discern a post-harvest change in landslide frequency in the North Fork have been inconclusive (Cafferata and Spittler 1998). Twelve large landslides have occurred post-harvest in the North Fork watershed. The two largest occurred in clearcut units more than 10 years after harvest and account for 60% (5617 m^3) of the volume of all post-harvest erosion features. Of the remaining 10, five occurred in harvest units and five in control watersheds. While serving as a control watershed, the North Fork experienced two other large landslides (in 1974 and 1985) that displaced 4568 m^3.

Bawcom (2003) evaluated 50 clearcut units on Jackson Demonstration State Forest including the 10 North Fork Caspar clearcuts. Of 32 recent debris slides larger than 76 m^3, 28 (two of six in North Fork Caspar) were road-related. Most were associated with decades-old roads low on the slope near watercourses. No increase in the rate of landsliding within JDSF clearcuts was detected.

Conclusions

Timber harvest and road building affect runoff processes, sediment yields, and erosion. Caspar Creek studies document increases in peak flows, suspended sediment loads, and erosion after two very different harvest treatments. Response was highly variable between treatments and among individual treated tributaries. California's modern forest practices rules appear to mitigate, but do not eliminate these impacts.

Changes in basin wetness and canopy interception explain post-harvest flow increases. Sediment loads following partial clearcutting were correlated to flow increases. With forest regrowth, flow increases diminish returning to pre-harvest flow conditions after about 10 years. Sediment yields do not appear to recover as quickly and persist at double the pretreatment levels 12 years after harvest.

Erosion and sedimentation from ground extensively disturbed by road building and tractor yarding remain elevated decades after harvest. The present condition of the South Fork watershed is typical of much of the tractor-yarded lands in the redwood region that are entering yet another harvest cycle. It is becoming crucial for landowners, regulatory agencies, and the public to understand the interactions between proposed future activities and prior disturbances. A third phase of Caspar Creek research is being initiated in the South Fork to examine the effects of re-entry on runoff and sediment production from previously tractor-logged redwood forests. Much remains to be learned regarding restoring impacted ecosystems and mitigating impacts from future harvests. The Caspar Creek Experimental Watersheds provide a long-term research resource for furthering this scientific endeavor.

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Understanding the Hydrologic Consequences of Timber-harvest and Roading: Four Decades of Streamflow and Sediment Results from the Caspar Creek Experimental Watersheds

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The Caspar Creek Experimental Watersheds were established in 1962 to study the effects of forest management on streamflow, sedimentation, and erosion in the rainfall-dominated, forested watersheds of north coastal California. Currently, 21 stream sites are gaged in the North Fork (473 ha) and South Fork (424 ha) of Caspar Creek. From 1971 to 1973, 65% of the timber volume in the South Fork was selectively cut and tractor yarded, and from 1985 to 1991, 50% of the North Fork basin was harvested, mostly as cable-yarded clearcut. The South Fork logging resulted in annual suspended sediment load increases exceeding 300%. Mass-wasting has been predominantly associated with roads, landings, and tractor skid trails in the South Fork. Accelerated mass-wasting and renewed sediment mobilization in the South Fork have occurred since 1998. Peak flow increases detected following North Fork logging are attributable to reduced canopy interception and transpiration. These recovered to pretreatment levels about 10 years after logging, followed by renewed increases from pre-commercial thinning. Annual sediment loads increased 89% in the partially clearcut North Fork and 123% to 238% in 4 of 5 clearcut sub-basins. Twelve years after logging, elevated storm-event sediment yields persist in some clearcut tributaries.

Keywords: *experimental watershed studies, road effects, sediment yield, peak flows, erosion, timber harvesting*

Introduction

For more than four decades, researchers have investigated the effects of forest management on streamflow, sedimentation, and erosion in the Caspar Creek Experimental Watersheds. The California Department of Forestry and Fire Protection and the USDA Forest Service, Pacific Southwest Research Station, began a simple paired watershed study in 1962 with the construction of weirs on the two major Caspar Creek tributaries, the North Fork (NFC) and the South Fork (SFC). Today, researchers operate 21 gaging stations within the experimental watersheds and use data loggers programmed with sophisticated sampling algorithms, instream turbidimeters, and automated pumping samplers to measure water and sediment discharge. Although much of this research is devoted to quantifying the impacts of modern forest management, it also provides valuable data on hydrologic recovery and the lingering effects of more than a century of timber harvest and roading in the Caspar Creek basin.

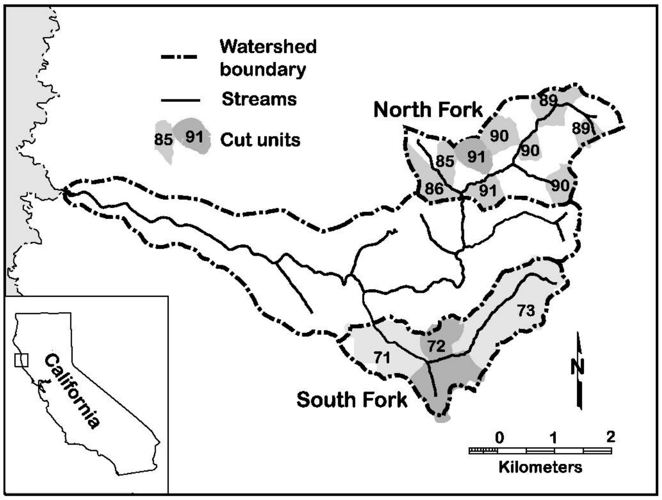
Methods

Site

The Caspar Creek Experimental Watersheds are located about 7 km from the Pacific Ocean and about 10 km south of Fort Bragg in northwestern California at lat 39°21´N, long 123°44´W (Figure 1). Uplifted marine terraces incised by antecedent drainages define the youthful and highly erodible topography with elevations ranging from 37 to 320 m. Hillslopes are steepest near the stream channel and become gentler near the broad, rounded ridgetops. About 35% of the basins' slopes are less than 17 degrees, and 7% are steeper than 35 degrees. Soils are well-drained clay-loams, 1 to 2 meters in depth, derived from Cretaceous Franciscan Formation greywacke sandstone and weathered, coarse-grained shale.

M Furniss, C Clifton, and K Ronnenberg, eds., 2007. *Advancing the Fundamental Sciences: Proceedings of the Forest Service National Earth Sciences Conference, San Diego, CA, 18-22 October 2004*, PNW-GTR-689, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Figure 1. Caspar Creek Experimental Watersheds.



The climate is typical of low-elevation coastal watersheds of the Pacific Northwest. Winters are mild and wet, characterized by periods of low-intensity rainfall delivered by the westerly flow of the Pacific jet stream. Snow is rare. Average annual precipitation is 1,170 mm. Typically, 95% of precipitation falls during the months of October through April. Summers are moderately warm and dry with maximum temperatures moderated by frequent coastal fog. Mean annual runoff is 650 mm.

Like most of California's north coast, the watersheds were clearcut and broadcast burned largely prior to 1900. By 1960, the watersheds supported an 80-year-old second-growth forest composed of coast redwood (*Sequoia sempervirens*), Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and grand fir (*Abies grandis*). Forest basal area was about 700 m^3/ha.

Measurements

The 473-ha North Fork of Caspar Creek and the 424-ha South Fork of Caspar Creek have been gaged continuously since 1962 using 120° v-notch weirs widening to concrete rectangular sections for high discharges. During the early 1980s, three rated sections were constructed upstream of the North Fork weir and 10 Parshall flumes were installed on North Fork subwatersheds with drainage areas of 10 to 77 ha.

Stream discharge was initially recorded using mechanical chart recorders. These were replaced in the 1980s with electronic data loggers equipped with pressure transducers. Early suspended sediment estimates were derived from sediment rating curves, manual depth-integrated sampling, and fixed stage samplers (Rice et al. 1979). Statistically based sampling algorithms that trigger automated samplers were used beginning in the 1980s (Lewis et al. 2001). In addition, the sediment accumulation in the settling basin upstream of each weir has been surveyed annually.

Periodic field surveys have documented the location, size, and disposition of landslides and fluvial erosion. Erosion features greater than 7.6 m^3 (10 yd^3) have been inventoried annually since 1986 in the North Fork, and since 1994 in the South Fork.

Treatments

After establishing a calibration relationship between the North Fork and the South Fork (1963 to 1967), a main-haul logging road and main spurs were built in the South Fork. The road right-of-way occupied 19 ha, from which 993 m^3/ha of timber was harvested. The entire South Fork watershed was logged and tractor yarded between 1971 and 1973 using single-tree and small group selection to harvest 65% of the stand volume. Roads, landings, and skid trails covered approximately 15% of the South Fork watershed area (Ziemer 1981). Almost 5 km of the main-haul and spur roads (out of approximately 10 total km) were decommissioned in 1998.

A study of cumulative effects began in 1985 in the North Fork watershed. Three gaged tributary watersheds within the North Fork were designated as controls while seven were designated for harvest in compliance with the California Forest Practice Rules in effect in the late 1980s. Two units (13% of the North Fork watershed) were clearcut in 1985-86. After calibration, clearcut logging began elsewhere in the North Fork in May 1989 and was completed in January 1992. Clearcuts occupied 30-99% of treated watersheds and totaled 162 ha. Between 1985 and 1992, 46% of the North Fork watershed was clearcut, 1.5% was thinned, and 2% was cleared for road right-of-way (Henry 1998).

In contrast to the harvest treatment of the South Fork in the 1970s, state rules mandated equipment exclusion and 50% canopy retention within 15 to 46 m of watercourses providing aquatic habitat or having fish present. Most of the yarding (81% of the clearcut area) was accomplished using skyline-cable systems. Yarders were situated on upslope landings constructed well away from the stream network. New road construction and tractor skidding was restricted to ridgetop locations with slopes generally less than 20% and affected only 3% of the watershed area. Four harvest blocks, 92 ha total, were broadcast burned and later treated with herbicide to control competition (Lewis et al. 2001). Pre-commercial thinning in 1995, 1998 and 2001 reduced basal area in treated units by about 75%.

RESULTS

Streamflow

Previous publications detail the magnitude and duration of streamflow enhancements following timber harvest in the Caspar Creek basins (Ziemer 1981, 1998; Lewis et al. 2001; Keppeler and Lewis, in press). In the North Fork, the average storm peak flow with a two-year return period increased 27% in the clearcut watersheds (Ziemer 1998) and 15% in the partially clearcut watersheds. Ongoing measurements show a return to pre-treatment flow conditions on NFC approximately 10 to 11 years post-harvest except for a renewed response to the pre-commercial thinning. Of particular interest is that even under the wettest antecedent moisture conditions of the NFC study, increases averaged 23% in clearcut watersheds and 3% in partially clearcut watersheds. These results are explained by wetter soils in logged units resulting from reduced transpiration and increases in net precipitation due to reduced canopy interception after clearcutting (Reid and Lewis, in press).

Sediment Loads

Sediment load estimates for the North Fork and South Fork are the sum of the sediment deposited in the weir pond and the suspended load measured at the weir. Comparison of sediment loads produced following the 1971-73 harvest of South Fork and the 1989-92 harvest of North Fork must be made cautiously. Improved and more intensive sampling methods greatly enhance the accuracy of load estimates for the latter study. Large landslides in the North Fork in 1974 and 1995, and the 1985 harvest in the North Fork complicate the analysis.

South Fork suspended sediment loads increased 335% (1,475 kg ha^{-1} yr^{-1}) after road building and averaged 331% (2,877 kg ha^{-1} yr^{-1}) greater during the 6-year period after tractor logging. Annual sediment load (including suspended and pond accumulations) increased 184% for the 6-year post-harvest period 1972-1978 returning to pretreatment levels in 1979 (Lewis 1998).

North Fork annual sediment loads increased 89% (188 kg $ha^{-1}yr^{-1}$) in the partially clearcut watershed and between 123% and 238% (57 to 500 kg ha^{-1} yr^{-1}) in 4 of 5 clearcut basins during the 1990-96 post-harvest period (Lewis 1998). The load decreased by 40% (551 kg ha^{-1} yr^{-1}) in one clearcut basin. Sediment loads in some North Fork tributaries remained elevated through hydrologic year 2003, twelve years after harvest (Keppeler and Lewis, in press).

Although Thomas (1990) reported that SFC sediment concentrations appeared to be returning to pre-treatment levels in the early 1980s, analysis of more recent data suggests renewed sediment mobilization. The 1998 and 1999 pond depositions were the largest on record at SFC, but not exceptional at NFC. A double mass plot of pond accumulations indicates an increase in deposited sediments at SFC relative to NFC starting in 1998. The same is true to a lesser extent in suspended sediments (Figure 2). Regression analysis of SFC versus NFC pond accumulations indicates a significantly higher slope for the period 1998-2004 compared to 1974 -1997.

Since the suspended sediment data have better temporal and spatial resolution, Lewis' 1998 analysis of NFC was extended using storm load data from control tributary gages H and I to investigate whether the relative change in suspended sediment was due to changes in the North Fork, changes in the South Fork, or both. A plot of the percentage departures from the prelogging (1986-1989) regression of NFC versus HI, shows elevated sediment levels for 1993-1998 only (Figure 3). An analogous plot for SFC versus HI shows elevated sediment levels for some small events in 1993-1997, but most consistently for all

Figure 2. Double mass curve of South Fork pond accumulations and suspended sediment relative to North Fork 1986 through 2004.

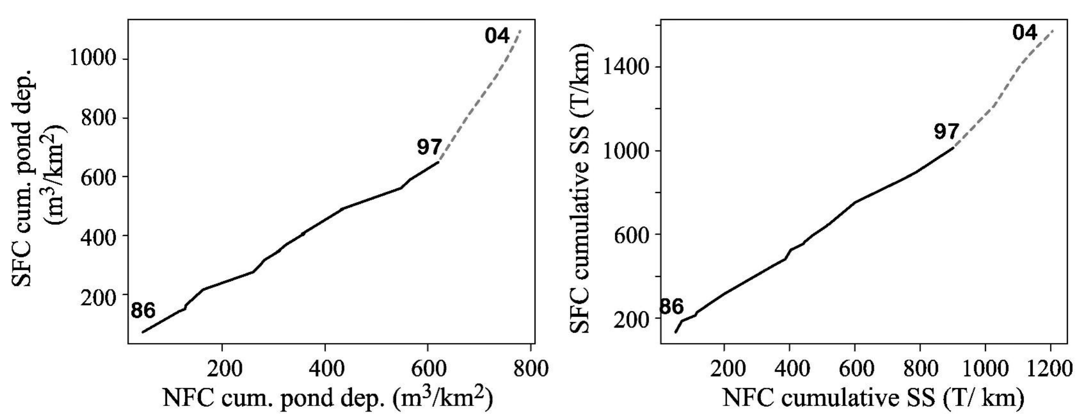
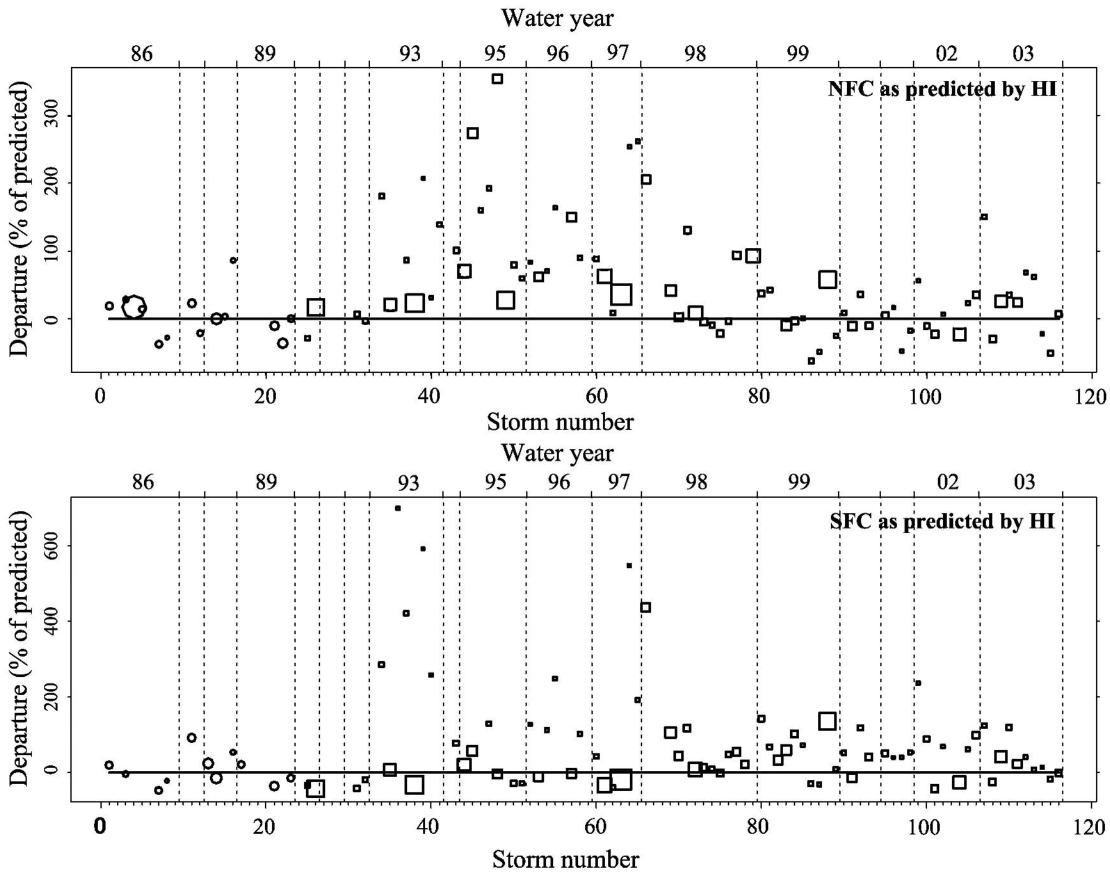


Figure 3. Suspended sediment percentage departures from the 1986-1989 regressions of NFC and SFC versus two untreated North Fork Controls (HI). Marker size is relative to HI storm load.



size events in 1998-2003. Thus the change in relation between NFC and SFC starting in 1998 appears to be a combination of declining loads at NFC and increasing loads at SFC.

To be more rigorous and quantitative about the suspended sediment changes, the gaging records were broken into three periods: NFC prelogging (1986-1989), NFC postlogging (1990-1997), SFC "episode" (1998-2003). These periods included 23, 41, and 52 storm events, respectively. Log-log regression models were fit relating NFC and SFC to HI for the three periods and tested to determine if a unique slope or intercept was appropriate for each period (Figure 4). For the NFC model, a parallel regression model with three intercepts and one slope was adequate. For the SFC model three intercepts and three slopes needed to be retained. Nine post-hoc comparisons of intercept and slope for each period were made. The NFC parallel regression for the 1990-1997 period was significantly different (higher) than either of the other periods, and the SFC 1990-1997 regression had significantly different (lower) slope than the 1998-2003 period. The SFC 1986-1989 slope was similar to the SFC 1998-2003 slope, but did not differ significantly from the 1990-1997 period, possibly due to the smaller number of storms in 1986-1989 compared to 1998-2003 (Table 1).

Table 1. Comparison of regression and intercepts (NFC and SFC) and slopes (SFC only) for three different time periods. Bonferonni's procedure was used to limit the experimentwise error rate to 0.05 which requires setting the pairwise comparison error rates to 0.05/9 = 0.0056. By this criterion, the only significant differences were (1), (3), and (9).

	Comparison	Significance (p)
1	NFC 90-97 intercept to NFC 86-89 intercept	0.000014
2	NFC 98-03 intercept to NFC 86-89 intercept	0.50
3	NFC 98-03 intercept to NFC 90-97 intercept	0.00000032
4	SFC 90-97 intercept to SFC 86-89 intercept	0.039
5	SFC 98-03 intercept to SFC 86-89 intercept	0.096
6	SFC 98-03 intercept to SFC 90-97 intercept	0.40
7	SFC 90-97 slope to SFC 86-89 slope	0.096
8	SFC 98-03 slope to SFC 86-89 slope	0.983
9	SFC 98-03 slope to SFC 90-97 slope	0.00037

Table 2. Summary of post-disturbance Erosion Features greater than 7.6m³.

	South Fork Caspar				North Fork Caspar			
	#	Volume (m³)	m³/ha	Delivery	#	Volume (m³)	m³/ha	Delivery
Post-disturbance[1]	130	65312	154	na	116	6496	14	40%
road-related	115	41706	98	68%	7	424	1	8%
wind-related	5	13985	33	na	74	1466	3	29%
> 1000 m³	18	52120	123	na	1	3605	8	46%
100-1000 m³	38	10507	25	na	5	944	2	26%
HY1990-2004	38	5804	14	61%	147	9121	19	45%
road-related	34	5557	13	63%	10	2495	5	52%
wind-related	4	264	1	21%	86	1732	4	27%
> 1000 m³	0	0	0	na	2	5618	12	52%
100-1000 m³	12	4961	12	62%	5	944	2	26%

[1] Includes hydrologic years 1968-1976 on South Fork, 1990-1998 on North Fork.

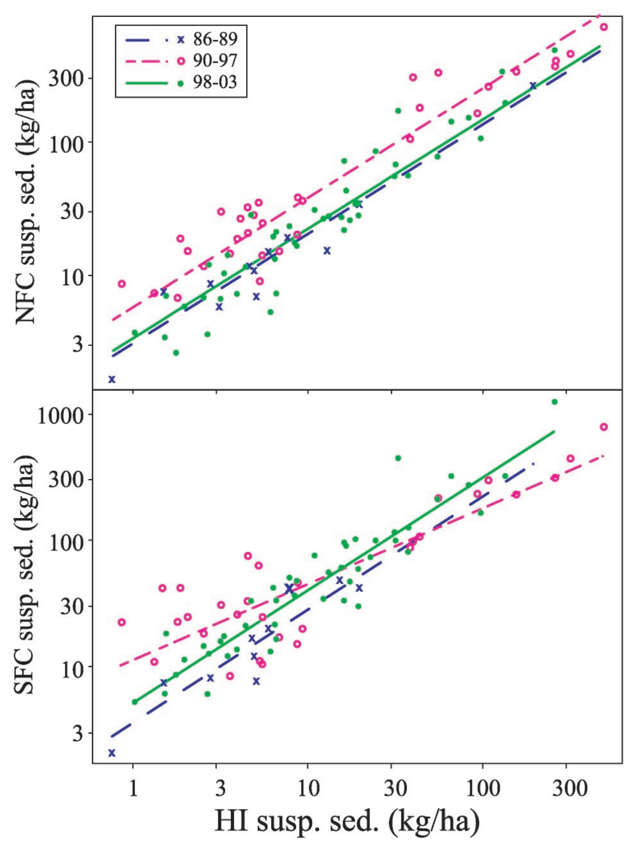
Based on the 1986-1989 relations, the observed sediment from NFC exceeded the predicted sediment by 49% in 1990-1997 and by 12% in 1998-2003. The corresponding numbers for SFC were -20% and +45%, but the only significant difference for SFC was the slope between the last two periods. Based on the 1990-1997 period, the observed sediment from SFC 1998-2003 exceeded the predicted sediment by 36%. If the 1986-1989 and 1990-1997 periods are combined to predict suspended sediment for 1998-2003, the observed sediment from SFC exceeded the predicted sediment by 47% (185 kg ha^{-1} yr^{-1}). (Compared to the 1986-1989 regression, the combined regression predicts lower loads in large storms above ~50 kg/ha at HI.) Thus, it is only during relatively large storm events (> 20 kg/ha at HI) that SFC suspended sediment systematically exceeds the pre-1998 relationship (Figure 4).

Erosion

Erosion inventory data helps to explain sediment load changes. As with sediment loads, improved protocols provide more detailed information on mass-wasting processes than is available for the earlier SFC inventories. Inventories have been more frequent and more intensive on the North Fork since 1986 and the South Fork since 1994. Nonetheless, the contrasts are quite apparent. During the 1970s, the South Fork landscape experienced mass-wasting an order of magnitude greater than that which followed NFC harvesting of the 1990s. SFC erosion has been predominantly related to the roads, landings, and skid trails. In contrast, most North Fork erosion features have been associated with windthrow disturbances and typically displaced and delivered smaller volumes of material. Two large landslides, one related to a preexisting mid-slope road, account for more than half of NFC mass-wasting (Table 2).

Another episode of road-related landsliding commenced in the South Fork in the mid 1990s. Of the 31 SFC landslides documented between 1995 and 2004, 94% are road, landing, or skid trail related. These more recent landslides displaced 4,123 m³ and had an average delivery ratio of 85%. A deteriorating network of logging roads and skid trails continues to deliver sediment to the stream channel and explains much of the recently enhanced SFC sediment production previously discussed. This

Figure 4. Suspended sediment regression results by period for North Fork (NFC) and South Fork (SFC) versus control (HI).



renewed sedimentation may also be a manifestation of the extreme stormflows of 1998 and 1999 and the erosional costs of recent road decommissioning. The 1998 road decommissioning effort removed almost 18,000 m^3 of fill from aging stream crossings, but treatment-related erosion contributed 750 m^3 of sediment.

CONCLUSIONS

Timber harvest and road building affect runoff processes, sediment yields, and erosion, but the response is highly variable. Caspar Creek studies document increases in peak flows, suspended sediment loads, and erosion after two very different harvest treatments. California's modern forest practices rules appear to mitigate, but do not eliminate, these impacts.

Erosion and sedimentation from ground extensively disturbed by road building and tractor yarding remain elevated decades after harvest. The present condition of the South Fork watershed is typical of many of the tractor-yarded lands in the redwood region that are entering yet another harvest cycle. Greater understanding of the interactions between proposed activities and prior disturbances is crucial for improved forest management. Thus, a third phase of research is underway to examine the effects of re-entry on the previously tractor-logged South Fork watershed. Much remains to be learned regarding restoring forest ecosystems and mitigating harvest impacts. The Caspar Creek Experimental Watersheds will continue to serve as a resource for furthering this research endeavor.

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Long-term Patterns of Hydrologic Response after Logging in a Coastal Redwood Forest

Elizabeth Keppeler, Leslie Reid, Tom Lisle

Abstract

Experimental watersheds generally provide the only setting in which the more subtle patterns of long-term response to land use activities can be defined. Hydrologic and sediment responses have been monitored for 35 yrs after selective logging and for 16 yrs after clearcut logging of a coastal redwood forest at the Caspar Creek Experimental Watersheds in northwest California. Results show that recovery periods differ for different hydrologic attributes and between the two silvicultural treatments. Total water yield, peakflows, and low flows responded similarly in both settings during the initial post-logging period, but low flows reattained pre-treatment levels more quickly after selective logging. Sediment loads initially recovered relatively quickly after both treatments, but in both cases loads rose once again 10–20 yrs after logging, either because road networks began to fail (South Fork) or because pre-commercial thinning again modified hydrologic conditions (North Fork).

Process-based studies provide the information needed to understand the differing watershed responses. Altered interception after logging provides the primary influence on water yield and peakflow responses, while altered transpiration is largely responsible for the low-flow response. Differences in recovery times between hydrologic attributes and between silvicultural practices may be explained by changes in the relative importance of interception and transpiration and by the long-lasting repercussions of ground disturbance.

Keywords: streamflow, sediment, hydrologic recovery, timber harvest, cumulative watershed effects

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Introduction

Since the installation of stream gaging weirs on the North and South Forks of Caspar Creek in 1962, researchers have been investigating the effects of forest management on streamflow, sedimentation, and erosion under a partnership between State and Federal forestry agencies. As the hydrologic record lengthens following experimental treatments, differing patterns of recovery have become evident. A suite of ongoing process-based studies provides the information needed to understand the contrast in watershed responses. Previous publications detail the range of hydrologic response to the logging treatments. Here, we discuss results from further analyses and provide an updated look at recovery in the Caspar Creek Experimental Watersheds.

Methods

Site

The Caspar Creek Experimental Watersheds are located on the Jackson Demonstration State Forest about 7 km from the Pacific Ocean and about 10 km south of Fort Bragg in northwestern California at 39°21'N 123°44'W (Figure 1). The watersheds are incised into uplifted marine terraces underlain by greywacke sandstone and weathered, coarse-grained shale of late Cretaceous to early Cenozoic age.

Elevations in the watersheds range from 37 to 320 m. Hillslopes are steepest near stream channels and become gentler near the broad, rounded ridgetops. About 35 percent of the slopes are less than 17 degrees and 7 percent are steeper than 35 degrees. Soils are 1- to 2-m-deep, well-drained clay-loams. Hydraulic conductivities are high and subsurface stormflow is rapid, producing saturated areas of only limited extent and duration.

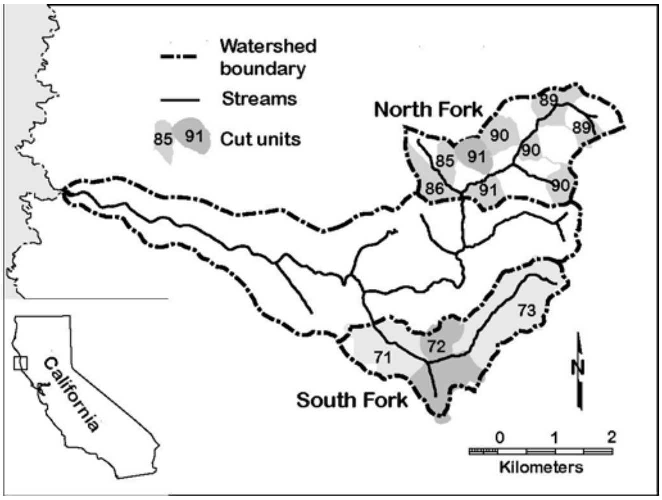

Figure 1. Caspar Creek Experimental Watersheds and 20th century harvest dates.

The climate is typical of low-elevation coastal watersheds of the Pacific Northwest. Winters are mild and wet, characterized by frequent, low-intensity rainstorms interspersed with occasional high-intensity events. About 95 percent of the average annual precipitation of 1,170 mm falls October through April, and snow is rare. Summers are moderately warm and dry, with maximum temperatures moderated by frequent coastal fog. Mean annual runoff is 650 mm.

Like most of California's north coast, the watersheds were clearcut and broadcast burned largely prior to 1900. By 1960, the watersheds supported an 80-year-old second-growth forest with a stand volume of about 700 m^3 ha^{-1}, composed of coast redwood (*Sequoia sempervirens*), Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and grand fir (*Abies grandis*).

Study design

The Caspar Creek study is a classic paired watershed design where one or more gaged catchments are designated as controls and others are treated with road building, logging, and other timber management practices. Statistical relationships are first defined between control watersheds and those to be treated, then post-treatment responses are evaluated as the deviation between observed conditions and those expected based on the pre-treatment calibrations.

The 473-ha North Fork and the 424-ha South Fork of Caspar Creek have been gaged continuously since 1962 using 120° V-notch weirs widening to concrete rectangular sections for high discharges. During the early 1980s, three rated sections were constructed upstream of the North Fork weir and 10 Parshall flumes were installed on tributary reaches with drainage areas of 10 to 77 ha.

Stream discharge was initially recorded using mechanical chart recorders. These were replaced in the mid-1980s with electronic data loggers equipped with pressure transducers. Subsequent upgrades have been implemented as technology has progressed. Early suspended sediment estimates were derived from sediment rating curves, manual depth-integrated sampling, and fixed stage samplers (Rice et al. 1979). Statistically based sampling algorithms that trigger automated samplers were utilized beginning in the 1980s (Lewis et al. 2001). Sediment accumulations in the weir ponds have been surveyed annually since 1963.

South Fork treatment: Selection harvest with tractor yarding

Calibration relationships between the North and South Forks were established for flow and sediment by 1967. That year, right-of-way logging and road construction along the riparian corridor proceeded in the South Fork. The watershed response to roading was monitored for 4 yrs before the remainder of South Fork watershed was logged and tractor yarded between 1971 and 1973. Single-tree and small group selection was used to harvest about two-thirds of the stand volume. Roads, landings, and skid trails covered approximately 15 percent of the watershed area (Ziemer 1981).

North Fork treatment: Clearcutting with skyline-cable yarding

A study of cumulative effects began in 1985 in the North Fork watershed. Three gaged tributary watersheds within the North Fork were selected as controls, while five were designated for harvest in compliance with the California Forest Practice rules. Two additional downstream units (13 percent of the North Fork watershed) were clearcut in 1985–86 and excluded from the cumulative effects study. After the 1985–89 calibration period, clearcut logging began elsewhere in the study area in May 1989 and was completed in January 1992. Clearcuts totaling 162 ha occupied 30–99 percent of treated watersheds. Between 1985 and 1992, 46 percent of the North Fork watershed was clearcut, 1.5 percent was thinned, and 2 percent was cleared for road rights-of-way (Henry 1998).

In contrast to the harvest treatment of the South Fork in the 1970s, watercourse protection rules mandated equipment exclusion and 50 percent canopy retention within 15–46 m of streams containing aquatic organisms. Skyline-cable systems yarded 81 percent of the clearcut area from log landings constructed far from streams. New road construction and tractor skidding was restricted to ridgetop locations with slopes of generally less than 20 percent. Four harvest blocks, 92 ha total, were broadcast burned and later treated with herbicide to control competition (Lewis et al. 2001). Pre-commercial thinning in 1995, 1998, and 2001 eliminated much of the dense regrowth, reducing basal area in treated units by about 75 percent.

Results

Water yield and low flows

Both treatments resulted in increased water yields for a period of 10 yrs or more (Keppeler and Ziemer 1990, Keppeler 1998). When calculated per unit of equivalent clearcut area, the magnitudes of the initial changes were found to be quite similar (Figure 2), but South Fork began to show a trend toward recovery after 7 yrs while North Fork did not. Changes in low flow exhibited a contrasting pattern. Initial changes were similar in the North and South Forks, but South Fork low flows recovered to pre-treatment conditions within 8 yrs of logging, while North Fork low flows had not recovered by year 14.

The contrast in low flow responses between the two experiments probably reflects the difference in silvicultural treatments used. In the South Fork, about a third of the tree canopy remained distributed across the landscape after logging, and the surviving trees no longer had competition for dry-season soil moisture. Under these conditions, actual dry-season transpiration could more closely approach potential transpiration, and the post-logging “excess” of water would contribute to transpiration once root networks expanded. In North Fork clearcuts, no nearby trees could take advantage of the excess water, and this water instead will continue to contribute to dry-season flows until new vegetation is well established on the cut units. In addition, most North Fork clearcut units were later treated with herbicides and pre-commercially thinned, again reducing leaf area and suppressing transpiration.

Water yields, in contrast, are dominated by wet-season flows. After logging at Caspar Creek, the change in foliar interception of rainfall was found to be a stronger influence on the wet-season water balance than was transpiration (Reid and Lewis 2007), as about 22 percent of rainfall is intercepted by foliage in uncut stands (Reid and Lewis 2007). In the case of interception, rates depend more strongly on the amount of canopy removed than on the distribution of remaining trees. The wet-season response—reflected by the water yield—is thus more similar for the two silvicultural strategies than is the transpiration-dependent dry-season response.

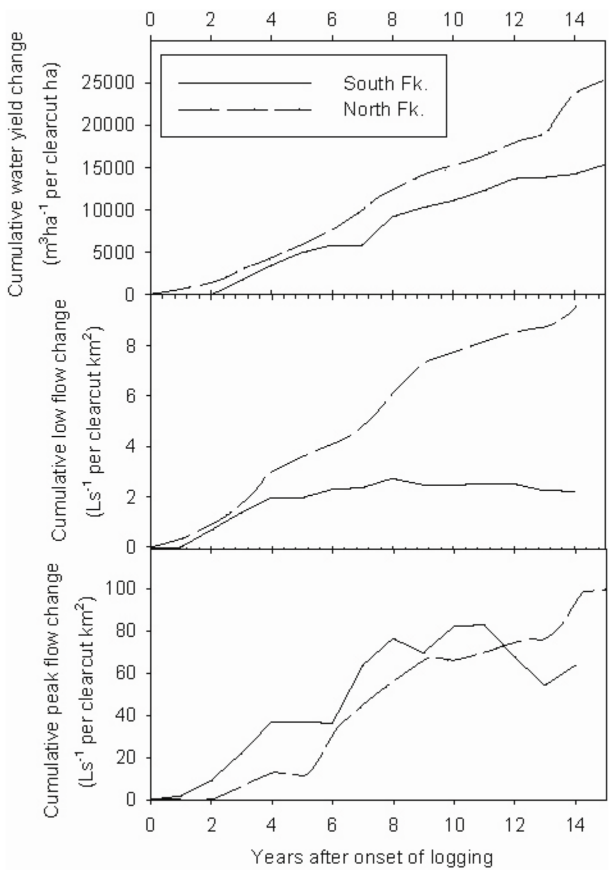

Figure 2. Cumulative change per unit area of clearcut equivalent by time after major logging for water yield, low flow, and peak flow. Minor logging occurred 4 yrs before the major onset in both watersheds and thinning occurred in the North Fork in years 6, 9, and 12.

Peakflows

Changes in major winter peakflows were not initially detected in the dataset from the South Fork, but reanalysis using temporal categories suggested by

North Fork results showed a statistically significant increase between 3 months and 8 years after logging ended. The discharge-weighted average peakflow was 13 percent higher than predicted and the 2-yr storm peak increased 14 percent.

The North Fork study design, wherein five clearcut tributaries and three control tributaries were gaged, yielded a larger dataset. Storm peaks with 2-yr return periods increased an average of 27 percent in the fully clearcut watersheds (Ziemer 1998), and in partially clearcut watersheds the magnitude of the change was proportional to the percentage of the watershed logged (Lewis et al. 2001). Peakflows in clearcut watersheds had nearly reattained pre-treatment levels within about 10 yrs after logging, but pre-commercial thinning then triggered new increases. As of 2007, ongoing measurements in two fully clearcut watersheds indicate that peakflows remain an average of 40 percent above pre-treatment predictions 6 yrs after pre-commercial thinning and 16 yrs after logging (Figure 3).

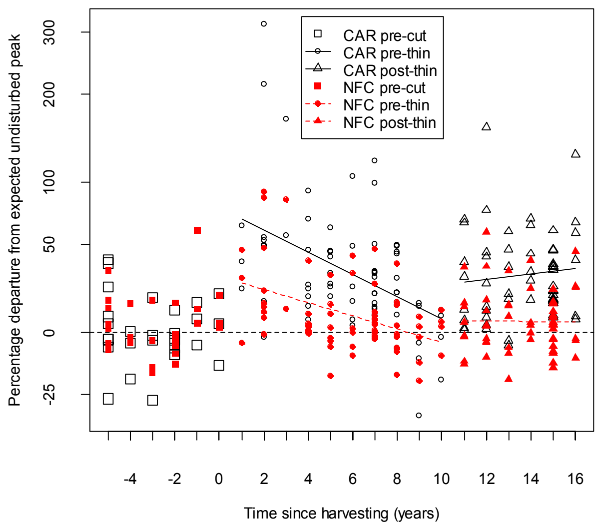

Figure 3. Peakflow departures from predicted in a 26-ha clearcut catchment (CAR) tributary to the 37 percent partially clearcut North Fork (NFC).

Sediment loads

The initial sediment responses following the logging on the South Fork (1971–73) was far greater than that on the North Fork (1989–92). South Fork suspended load more than quadrupled during the 6-yr period after tractor logging, while that in the North Fork roughly doubled during the equivalent post-harvest period (Lewis 1998). In both cases, sediment yields neared or reattained pre-treatment levels by about a decade after logging. In the South Fork, much of the excess sediment production is directly attributed to road-related erosion and mass-wasting (Rice et al. 1979)—problems that were more effectively avoided on the North Fork, where road and skid trail construction was much more limited.

Recent work suggests that an important component of the excess sediment in the North Fork may originate from sources within channels, thus making sediment loads particularly sensitive to logging-related increases in flow. Data from a pair of nested stream gages illustrate the potential importance of in-stream sediment sources. The 27-ha EAG clearcut watershed lies at the headwaters of the 77-ha DOL catchment, which otherwise has not been logged since 1904. Suspended sediment loads measured during storms at the EAG gauge were subtracted from corresponding loads at the DOL flume to estimate the load derived from the unlogged portion of the DOL watershed. These loads were then compared to those expected on the basis of pre-treatment calibrations to control watersheds. The ratio of observed to expected load in the unlogged portion of DOL shows a response similar in initial timing and magnitude to that within the logged watershed upstream (Figure 4). Field observations indicate that bank and headcut erosion in the mainstem DOL channel are the principal sources of sediment in the non-logged portion of the watershed.

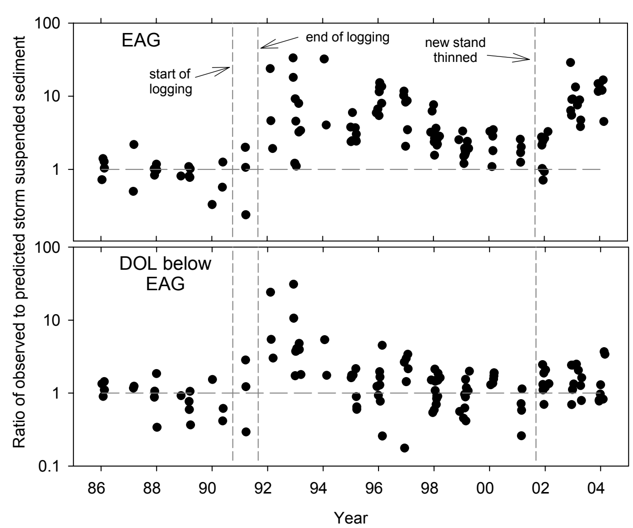

Figure 4. Suspended sediment loads observed in North Fork clearcut EAG and at downstream station DOL from hydrologic year 1986 through 2004.

Although sediment loads in both the South and North Fork watersheds had essentially recovered to pre-treatment levels within a decade of logging (Thomas 1990, Lewis 1998), both subsequently showed renewed increases. On the South Fork, deterioration of the road system contributed to a new period of excess sediment input beginning about 20 yrs after second-cycle logging (Keppeler and Lewis 2007). On the North Fork, pre-commercial thinning 10 yrs after logging again increased runoff and peakflows (Figure 4), triggering renewed channel erosion just as excess loads had nearly recovered. Added to this excess load is the sediment input from a major landslide on a logged slope of the North Fork in 2006.

Discussion

The relative importance of different components of the water balance varies seasonally at Caspar Creek (Figure 5), and those components respond to different silvicultural practices and to post-logging regrowth in different ways. As a result, each seasonally dependent attribute of streamflow demonstrates a unique response and recovery trajectory that is a composite response to a set of changes affecting interception, transpiration, and flow path. Transpiration dominates the water balance during the long dry season, so recovery of dry-season flows would track the recovery of transpiration potential following logging. Peakflows, in contrast, occur during months when the influence of decreased interception after logging is about twice that of transpiration reductions. Water yield, which principally reflects wet-season flows, would also be most strongly influenced by changes in rainfall interception after logging.

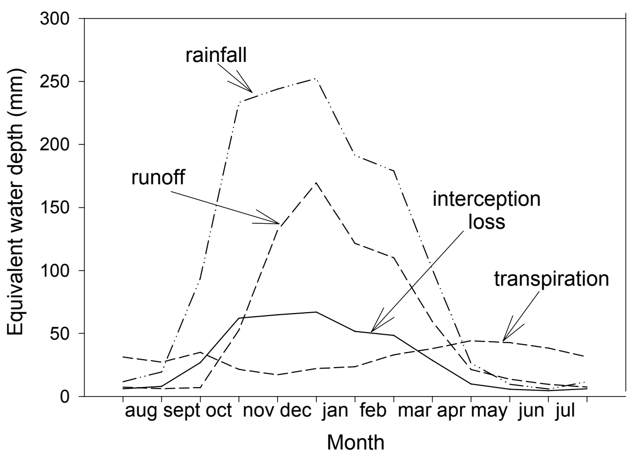

Figure 5. Monthly water balance for forested watersheds, North Fork Caspar Creek (from Reid and Lewis 2007).

Logging-related sediment inputs do not follow a smooth path to recovery. Although much of the initial increase in sediment loads in the North Fork was correlated with increased runoff (Lewis et al. 2001), hydrologic recovery has not translated into a sustained return to pre-treatment sediment loads in the South Fork. In fact, sediment loads 34 yrs after logging are once again nearly equivalent to those in the period immediately following logging. Dry years are now relatively quiescent in terms of sediment production, but years with multiple large storm events generate significant excess sediment.

In the North Fork, increased sediment loads following pre-commercial thinning are large relative to the magnitude of renewed increases in peak flow, suggesting that the new hydrologic conditions are interacting with other changes still present from second-cycle logging. This might be the case, for example, if the new reductions in transpiration and interception are synchronous with the post-logging minimum in root cohesion on hillslopes, or if channel banks already destabilized by the earlier period of increased flow are now subjected to new increases. Additional sediment might also be contributed by remobilization of logging-related sediment that remains in storage in channels downstream of logged areas or that had been trapped behind now decayed logging slash in low-order channels. In each case, new hydrologic changes interact with conditions generated earlier by logging, and the cumulative effect of the interaction is a disproportionate increase in sediment relative to that predicted on the basis of flow effects alone.

Evidence of altered hydrology, in the form of compaction, gullied stream channels, and diversions along abandoned roads and skid trails, persists in Caspar Creek's logged watersheds even as the forest regrows, maintaining an increased susceptibility of the landscape to the effects of major storms. In the North Fork, pre-commercial thinning renewed hydrologic changes, again reducing hillslope stability and contributing to channel adjustments. Through such mechanisms, the potential for enhanced sediment production may be sustained for prolonged periods after logging.

Conclusions

Timber harvest alters forest hydrology by forest canopy reduction and ground disturbances associated with road construction, yarding, and site preparation. Recovery is governed by the rate of revegetation and the more gradual amelioration of ground disturbances and channel re-stabilization. Watershed-scale studies are useful for documenting the hydrologic response over a range of conditions while exploring the cause-and-effect linkages that explain variations in ecosystem response. Long-term studies, such as those at Caspar Creek, are particularly important for disclosing the deviations from recovery trajectories following natural or management-related shifts in vegetation conditions occurring as regrowth proceeds, or as global climatic patterns shift.

Acknowledgments

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Evaluating the Impacts of Logging Activities on Erosion and Suspended Sediment Transport in the Caspar Creek Watersheds[1]

Jack Lewis[2]

***Abstract**: Suspended sediment has been sampled at both the North and South Fork weirs of Caspar Creek in northwestern California since 1963, and at 13 tributary locations in the North Fork since 1986. The North Fork gaging station (NFC) was used as a control to evaluate the effects of logging in the South Fork, in the 1970's, on annual sediment loads. In the most conservative treatment of the data, suspended loads increased by 212 percent over the total predicted for a 6-yr period commencing with the onset of logging. When the roles of the watersheds were reversed and the same analysis repeated to evaluate harvesting in the North Fork under California Forest Practice Rules in the 1990's, no significant increase was found at NFC in either annual suspended or bed load.*

With the advent of automatic pumping samplers, we were able to sample sediment concentration much more frequently in the 1980's. This allowed storm event loads from control watersheds in the North Fork to be used in a new regression analysis for NFC. According to this more sensitive analysis, for the 7-yr period commencing with the onset of logging, the sum of the suspended storm loads at NFC was 89 percent higher than that predicted for the undisturbed condition. The much greater increase after logging in the South Fork is too great to be explained by differences in sampling methods and in water years, and appears to be the result of differences in road alignment, yarding methods, and stream protection zones.

Similar analyses of storm event loads for each of the treated subwatersheds in the North Fork suggested increased suspended loads in all but one of the tributaries, but effects were relatively small or absent at the main stem locations. Of watersheds with less than 50 percent cut, only one showed a highly significant increase. The greater increase in sediment at NFC, compared to other main-stem stations, is largely explained by a 3,600-m^3 landslide that occurred in 1995 in a subwatershed that drains into the main stem just above NFC. Differences among tributary responses can be explained in terms of channel conditions.

Analysis of an aggregated model simultaneously fit to all of the data shows that sediment load increases are correlated with flow increases after logging. Field evidence suggests that the increased flows, accompanied by soil disruption and intense burning, accelerated erosion of unbuffered stream banks and channel headward expansion. Windthrow along buffered streams also appears to be important as a source of both woody debris and sediment. All roads in the North Fork are located on upper slopes and do not appear to be a significant source of sediment reaching the channels.

The aggregated model permitted evaluation of certain types of cumulative effects. Effects of multiple disturbances on suspended loads were approximately additive and, with one exception, downstream changes were no greater than would have been expected from the proportion of area disturbed. A tendency for main-stem channels to yield higher unit-area suspended loads was also detected, but after logging this was no longer the case in the North Fork of Caspar Creek.

Soil erosion and mass movement play major roles in shaping the landscapes that surround us. These processes complement those that build mountains and soils, resulting in landforms such as valleys, ridges, stream channels, and flood plains. Human activities that change the balances between these processes can have consequences that are detrimental to humans and the ecosystems we depend on. Human activities often lead to an acceleration of soil movement, net soil losses from hillslopes, and increases in sediment transport and deposition in stream channels. When soil erosion and mass movement directly damage roads, bridges, and buildings, the costs are immediate and obvious. Direct effects on ecosystem function and site productivity are also serious issues in many areas. Indirect impacts on downstream water quality and stream channel morphology, however, are often of greater concern.

Sediment-laden water supplies reduce the capacity of storage reservoirs and may require additional treatment to render the water drinkable. Sediment in irrigation water shortens the life of pumps and reduces soil infiltration capacity. Water quality is also an important issue for recreational water users and tourism.

Impacts of water quality on fish and aquatic organisms have motivated much of the research being presented at this conference. High sediment concentrations can damage the gills of salmonids and macroinvertebrates (Bozek and Young 1994, Newcombe and MacDonald 1991). High turbidity can impair the ability of fish to locate food (Gregory and Northcote 1993) and can reduce the depth at which photosynthesis can take place. However, suspended sediment is not always detrimental to fish, and indexes based on duration and concentration are unrealistically simplistic (Gregory and others 1993). Turbidity, can, for example, provide cover from predators (Gregory 1993).

If stream channels cannot transport all the sediment delivered from hillslopes, they will aggrade, resulting in increased risks for overbank flooding and bank erosion. It was this sort of risk, threatening a redwood grove containing the world's tallest tree, that motivated the expansion of Redwood National Park in 1978 (U.S. Department of Interior 1981). Accelerated delivery of sediment to streams can result in the filling of pools (Lisle and Hilton 1992), and channel widening and shallowing. Hence, fish rearing habitat may be lost, and stream temperatures often increase. Excessive filling in spawning areas can block the emergence of fry and bury substrates that support prey organisms. Settling and infiltration of fine sediments into spawning gravels reduces the transport of oxygen to incubating eggs (Lisle 1989) and inhibits the removal of waste products that accumulate as embryos develop (Meehan 1974). If aggradation is sufficient to locally eliminate

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surface flows during the dry season, fish can lose access to good upstream habitat or become trapped in inhospitable environments.

How Do Harvest Practices Affect Sediment Movement?

Figure 1 displays some of the mechanisms linking harvest activities with instream sediment transport. It is impossible to show all the potential interactions in only two dimensions, but the figure does hint at the complexity of controls on sediment movement. Timber harvest activities can accelerate erosion primarily through felling, yarding, skidding, building and using roads and landings, and burning.

Felling

Removing trees reduces evapotranspiration and rainfall interception, thus resulting in wetter soils (Keppeler and others 1994, Ziemer 1968). Loss of root strength and wetter soils can decrease slope stability (O'Loughlin and Ziemer 1982, Ziemer 1981). Trees near clearcut edges face increased wind exposure and become more susceptible to blowdown (Reid and Hilton, these proceedings), disrupting soils if trees become uprooted. Addition of woody debris to channels can cause scouring of the banks and channel, but also can reduce sediment transport by increasing channel roughness and trapping sediment (Lisle and Napolitano, these proceedings). The effects of felling upon erosion can be altered by controlling the quantity and the spatial and temporal patterns of cutting.

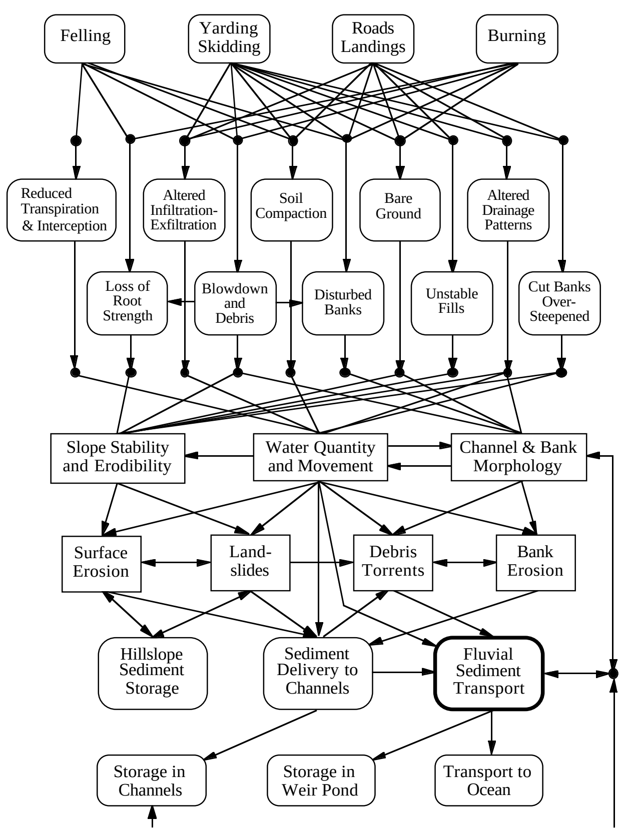

Figure 1—Conceptual diagram showing the major pathways through which logging activities influence fluvial sediment transport.

Yarding and Skidding

Heavy equipment compacts soils, decreasing infiltration and percolation rates and increasing surface water. If vegetation and duff are removed, the underlying soils become vulnerable to surface erosion. The pattern of yarding and skidding can alter drainage paths and redirect water onto areas that may be more likely to erode than naturally evolved channels. Damage from yarding and skidding is controlled primarily by the type of equipment, the care exercised by the equipment operator, timing of operations, landing location, and yarding direction.

Roads and Landings

Roads and landings have similar, but usually more pronounced, impacts as yarding and skidding, and their presence can greatly increase landslide risk. Compaction of the road bed can impede subsurface drainage from upslope areas, resulting in increased pore water pressures (Keppeler and Brown, these proceedings). Road cuts and fills are vulnerable to accelerated runoff and surface erosion, and are particularly vulnerable to slumping, especially on steep slopes or if the fill or sidecast material has not been properly compacted. Although roads and landings may be only a small part of the total forest area, they are responsible for a disproportionate amount of the total erosion (McCashion and Rice 1983, Swanson and Dyrness 1975), often more than half. The erosional impact of roads and landings can be managed through road alignment, design and construction, drainage systems, type and timing of traffic, and maintenance.

Burning

Burning can increase erodibility by creating bare ground, and hot burns can delay revegetation by killing sprouting vegetation. In some cases, burning can accelerate revegetation by releasing or scarifying seeds and preparing a seed bed. Burning in areas with sandy soils can create water-repellent soils and increase surface runoff (DeBano 1979). The effect of burning on erosion depends primarily on the temperature of the burn, soil cover, and soil and vegetation types. Soil moisture, wind, air temperature, humidity, slope steepness, and fuel abundance and distribution are the major factors affecting burn temperatures.

Site Factors

Some sites are particularly vulnerable to mass wasting, and these sites, while occupying a small part of the landscape, have been found to be responsible for a large proportion of the total erosion in northwestern California (Dodge and others 1976, Rice and Datzman 1981). In the Critical Sites Erosion Study, an evaluation of 157 mass failure sites (>153 m^3) and 326 randomly selected control sites from logged areas in northwestern California, Durgin and others (1989) concluded that management and site factors played an equal role in road failures. In contrast, management factors were secondary to site factors on hillslopes. The primary site factors associated with mass failures were steep slopes, noncohesive soils and fill materials, and incompetent underlying regolith. Most failures were associated with the concentration of subsurface water, as evidenced by perennial seeps, poorly drained soils, phreatic vegetation, and locations in swales, inner gorges, and lower slope positions. Previous slope failures were also evident at many of the sites. The primary management factors associated with mass failures were steep or overloaded fill slopes, steep cut banks, and inadequate maintenance of roads and drainage systems. A field procedure for estimating the probability of mass failure was also developed (Lewis and Rice 1990, Rice and Lewis 1991) from the Critical Sites Erosion Study.

Connecting Forest Practices with Water Quality

It is often difficult to identify the causes of erosion. Factors such as increased soil water or reduced root strength are not directly observable. Landslides are normal, stochastic, geomorphic events in many undisturbed areas. Therefore, it may be impossible to show that a landslide in a logged area would not have occurred had the area been treated differently.

There is usually a great deal of uncertainty in determining when and how much sediment from an erosion feature was delivered to a stream channel. And it is even more difficult to determine the origin of suspended sediment that has been measured at a gaging station.

Hence, many studies are correlative and rely on statistics to identify relations between disturbance and water quality. In environmental research, it is difficult to execute an experimental design that permits wide inference. The best designs require randomly assigning the treatments of interest to a large number of similar experimental units. The random assignment reduces the likelihood of associations between treatments and characteristics that might affect the response of some subset of experimental units. When studying a highly variable response such as sediment transport, large sample sizes are needed to detect changes even when the changes are substantial.

When the experimental unit is a watershed, it is usually impractical to randomly assign treatments or monitor a large number of watersheds. Instead, we use watersheds with similar physical characteristics and subject to similar environmental influences, and we repeat measurements before and after treatments are applied, maintaining at least one watershed as an untreated control throughout the study. If the relationship between measurements in the treated and control watersheds changes after treatment, then we can reason that the change is probably due to the treatment, unless some chance occurrence (unrelated to the treatments) affected only one of the watersheds. In reality, we have little control over such chance occurrences. For example, there is no guarantee that rainfall intensities will be uniform over the entire study area.

Such a paired-watershed design can provide a basis for concluding whether a change occurred (Chow 1960, Wilson 1978) and can be used to estimate the magnitude of changes. If chance occurrences can be eliminated, effects can be attributed to the *overall* treatment. If multiple watersheds are included in the design, it may be useful to relate the magnitude of response to disturbances such as proportion of area logged, burned, compacted by tractors, etc. But, without additional evidence, nothing can be concluded about specific causative mechanisms. Conclusions should be *consistent* with the statistical evidence, but cause and effect must be inferred non-statistically, by relating the results to concurrent studies of other responses and physical processes, field observations, and similar observations made elsewhere by others.

Study Area

The Caspar Creek Experimental Watersheds are located about 7 km from the Pacific Ocean in the Jackson Demonstration State Forest, Mendocino County, California (Preface, fig. 1, these proceedings). Until the 1970's, both the 424-ha South Fork and 473-ha North Fork watersheds were covered by second-growth redwood forests, originally logged between 1860 and 1904. Both watersheds are underlain by sandstones and shales of the Franciscan assemblage. Rainfall averages about 1,200 mm yr^{-1}, 90 percent of which falls during October through April, and snow is rare. The location, topography, soils, climate, vegetation, and land use history are described in detail by Henry (these proceedings). The geology and geomorphology are described by Cafferata and Spittler (these proceedings).

Methods

South Fork Treatment

The South Fork of Caspar Creek was roaded in the summer of 1967 and selectively logged in 1971-1973, before Forest Practice Rules were mandated in California by the Z'Berg Nejedly Forest Practice Act of 1973. About 65 percent of the stand volume was removed. In contrast with later logging in the North Fork, 75 percent of the roads in the South Fork were located within 60 m of a stream, all yarding was done by tractor, ground disturbance amounted to 15 percent of the area, and there were no equipment exclusion zones. Details are provided by Henry (these proceedings) and by Rice and others (1979). The North Fork was used as a control watershed to evaluate the effects of logging in the South Fork until the North Fork phase of the study was begun in 1985.

North Fork Treatments

The subwatershed containing units Y and Z (Preface, fig. 2, these proceedings) of the North Fork was logged between December 1985 and April 1986. At the time, this area was thought to have different soils than the remainder of the North Fork, so it was omitted from the study plan that specified logging would begin in 1989. The remainder of the North Fork logging took place between May 1989 and January 1992. Three subwatersheds (HEN, IVE, and MUN) were left uncut throughout the study for use as controls. Henry (these proceedings) summarizes the logging sequence. Briefly, 48 percent of the North Fork (including units Y and Z) was clearcut, 80 percent of this by cable yarding. Tractor yarding was restricted to upper slopes, as were haul roads, spur roads, and landings. Ground disturbance from new roads, landings, skid trails, and firelines in the North Fork amounted to 3.2 percent of the total area. Streams bearing fish or aquatic habitat were buffered by selectively logged zones 23-60 m in slope width, and heavy equipment was excluded from these areas.

Suspended Sediment and Turbidity Measurements

Accurate suspended sediment load estimation in small rain-dominated watersheds like Caspar Creek depends upon frequent sampling when sediment transport is high. Sediment concentrations are highly variable and inconsistently or poorly correlated with water discharge (Colby 1956, Rieger and Olive 1984). Since the 1960's, manual sampling methods have been standardized by the U.S. Geological Survey. However, adequate records are rare because it is inconvenient to sample at all hours of the night and weekends. Errors of 50-100 percent are probably typical when sampling is based on convenience (Thomas 1988, Walling and Webb 1988).

In the South Fork phase of the study from 1963 to 1975, sediment sampling was semi-automated by rigging bottles in the weir ponds at different heights. These *single-stage samplers* (Inter-Agency Committee on Water Resources 1961) filled at known stages during the rising limb of the hydrograph, but the much lengthier falling limb was sampled using DH-48 depth-integrating hand samplers (Federal Inter-Agency River Basin Committee 1952) and, in most cases, was not well-represented. In 1974 and 1975, the number of DH-48 samples was increased greatly and, in 1976, the single-stage samplers were replaced by pumping samplers. The average number of samples collected was 58 per station per year in 1963-1973 and 196 per station per year in 1974-1985.

During the North Fork phase of the study, in water years 1986-1995, the North Fork weir (NFC), the South Fork weir (SFC) and 13 other locations in the North Fork were gaged for suspended sediment and flow (Preface, fig. 2, these proceedings). Pumping samplers were controlled using programmable calculators and circuit boards that based sampling decisions on real-time stage information (Eads and Boolootian 1985). Sampling times were randomly selected using an algorithm that increased the average sampling rate at higher discharges (Thomas 1985, Thomas 1989). Probability sampling permitted us to estimate sediment loads and the variance of those estimates without bias. We also sent crews out to the watershed 24 hours a day during storm events to replace bottles, check equipment, and take occasional, simultaneous, manual and pumped samples. The average number of samples collected in 1986-1995 was 139 per station per year.

In water year 1996, we began using battery-operated turbidity sensors and programmable data loggers to control the pumping samplers at eight gaging stations, and monitoring was discontinued at the remaining seven stations. Although turbidity is sensitive to particle size, composition, and suspended organics, it is much better correlated with suspended sediment concentration than is water discharge. A continuous record of turbidity provides temporal detail about sediment transport that is currently impractical to obtain by any other means, while reducing the number of pumped samples needed to reliably estimate sediment loads (Lewis 1996). However, because these turbidity sensors remain in the stream during measurement periods, they are prone to fouling with debris, aquatic organisms, and sediment, so it was still necessary to frequently check the data and clean the optics. The average number of samples collected in 1996 was 49 per station per year.

Suspended Sediment Load Estimation

The basic data unit for analysis was the suspended sediment load measured at a gaging station during a storm event or hydrologic year. Annual loads were estimated only for NFC and SFC and, to facilitate comparisons with the South Fork study, these were computed by Dr. Raymond Rice using the same methods as in an earlier analysis (Rice and others 1979). This involved fitting sediment rating curves by eye, multiplying the volume of flow in each of 19 discharge classes by the fitted suspended sediment concentration at the midpoint of each class, and summing. As technology has improved over the years, our methods of sample selection have improved. Thus, although the computational scheme for estimating annual loads was repeated in both studies, the sampling bias has changed, and caution must be used when comparing the sediment loads from the two studies.

For estimating storm loads in 1986-1995, the concentrations between samples were computed using interpolations relating concentration to either time or stage. Concentration was first adjusted to obtain cross-sectional mean concentrations using regressions based on the paired manual and pumped samples. For those events in which probability sampling was employed, loads and variances were also estimated using appropriate sampling formulae (Thomas 1985, Thomas 1989). However, Monte Carlo simulations (Lewis and others 1998), showed that the interpolation methods were more accurate (lower mean square error). Based on the variance estimates and simulations, the median error of our estimates for storm events was less than 10 percent.

For estimating storm loads in 1996, concentration was predicted using linear regressions, fit to each storm, of concentration on turbidity. This method produced load estimates with the same or better accuracy than before, while substantially reducing the number of samples collected (Lewis 1996). Time or stage interpolation was employed for periods when turbidity information was unavailable.

Total Sediment Load Estimation

The bedload and roughly 40 percent of the suspended load settle in the weir ponds, and thus are not measured at NFC and SFC. The weir ponds are surveyed annually to estimate total sediment load (suspended plus bedload) by summing the pond accumulations and sediment loads measured at the weirs. Pond volumes are converted to mass based on a density of 1,185 kg m^{-3}. In some of the drier years of record (1972, 1976, 1987, 1991, 1992, and 1994), negative pond accumulations have been recorded. These values may result from settling or measurement errors, but some of the values were too large in magnitude to have resulted from settling alone, so negative values were converted to zero before adding pond accumulations to suspended loads. In the results below, only those that explicitly refer to *total* sediment load include any sediment that settled in the weir ponds.

Erosion Measurements

Starting in 1986, a database of failures exceeding 7.6 m^3 (10 yd^3) was maintained in the North Fork. This inventory was updated from channel surveys at least once a year. Road and hillslope failures were recorded when they were observed, but an exhaustive search was not conducted. Volume estimates were made using tape measurements of void spaces left by the failures, except in a few cases where more accurate survey methods were used. For each failure, crews recorded void volume, volume remaining at the site (starting in 1993), location, distance to nearest channel, and any association with windthrow, roads, or logging disturbance.

Discrete failures such as those included in the failure database are relatively easy to find and measure. In contrast, surface erosion is difficult to find and sample because it is often dispersed or inconspicuous. To obtain an estimate of dispersed erosion sources, erosion plots were randomly selected and measured in each subwatershed. Rills, gullies, sheet erosion, and mass movements were measured on independent samples of road plots and 0.08-ha circular hillslope plots. Road plots consisted of 1.5-m wide bands oriented perpendicular to the right-of-way, plus any erosion at the nearest downslope diversion structure (water bar, rolling dip, or culvert). A total of 175 hillslope plots and 129 road plots were measured. These data were collected for a sediment delivery study and are summarized in a separate report by Rice (1996).

Analyses and Results

Annual Sediment Loads after Logging the South Fork

Linear regressions between the logarithms of the annual suspended sediment loads at the two weirs were used to characterize (1) the relationship of SFC to NFC before the 1971-1973 logging in the South Fork and (2) the relationship of NFC to SFC before the 1989-1992 logging in the North Fork.

The calibration water years used in the South Fork analysis were 1963-1967, before road construction. The sediment load in 1968, after road construction, did not conform to the pretreatment regression (*fig. 2a*), but the data from the years 1969-1971 were not significantly different from the 1963-1967 data (Chow test, $p = 0.10$). In 1968, the increase in suspended load was 1,475 kg ha^{-1}, an increase of 335 percent over that predicted for an undisturbed condition. The years 1972-1978 (during and after logging) again differed from the pretreatment regression. Water year 1977 was missing owing to instrument malfunction. By 1979, the suspended sediment load at SFC had returned to pretreatment levels. The increased suspended load after logging amounted to 2,510 kg $ha^{-1}yr^{-1}$, or an increase of 212 percent over that predicted for the 6-yr period by the regression. (Predictions were corrected for bias when backtransforming from logarithms to original units.) The greatest absolute increases occurred in the years 1973 and 1974, followed by 1975 (*fig. 2b*).

A pair of large landslides (one in each watershed) occurred during hydrologic year 1974, complicating the analysis by Rice and others (1979), where the North Fork's sediment load was adjusted downward because most of the North Fork slide reached the stream, while most of the South Fork slide did not. However, that year did not appear anomalous in my analysis, and I did not make any adjustments. But the unadjusted prediction requires extrapolation of the regression line well beyond the range of the pretreatment

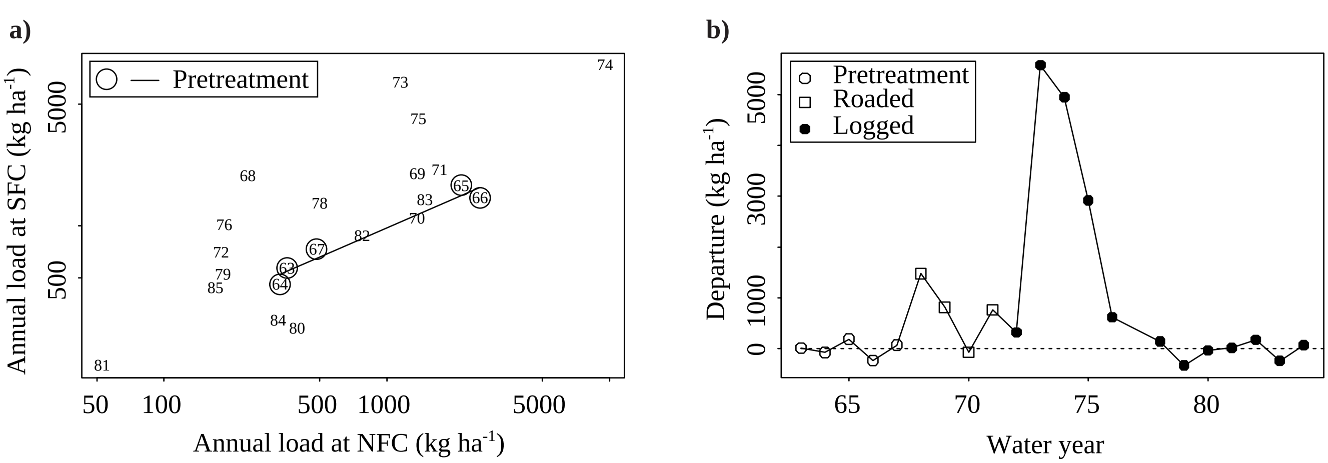

Figure 2—(a) Relation between estimated annual suspended sediment loads at South Fork Caspar Creek (SFC) and North Fork Caspar Creek (NFC) from 1963 to 1985. Pretreatment regression line is fit to the water years before roading and logging activity in the South Fork. (b) Time series of departures from the regression line.

data, so it is still suspect. If the adjustment of Rice and others (1979) is applied in my analysis, the revised increase in suspended sediment load is 2,835 kg $ha^{-1}yr^{-1}$, or an increase of 331 percent over that predicted for the 6-yr period. The adjusted figure reported for the 5-yr period (1972-1976) by Rice and others was 3,245 kg $ha^{-1}yr^{-1}$, an increase of 354 percent over that predicted.

Although no statistically significant logging effect on pond accumulation was detected, regression analysis using total sediment load (including data from 1974) revealed a similar pattern of impacts as that of the suspended load. The increased total sediment load after logging of the South Fork amounted to 2,763 kg $ha^{-1}yr^{-1}$, or an increase of 184 percent over that predicted for the 6-yr period by the regression.

Annual Sediment Loads after Logging the North Fork

The calibration period used in the North Fork analysis includes 1979-1985, the years after the South Fork's apparent recovery, as well as 1963-1967. The years 1986-1989 were not included in the calibration period because the Y and Z units were logged in 1985 and 1986. Applying the Chow test, neither 1986-1989 ($p = 0.43$) nor 1990-1995 ($p = 0.53$) was found to differ significantly from the suspended sediment calibration regression (*fig. 3a*). The (nonsignificant) departures from the regression predictions averaged 118 kg $ha^{-1}yr^{-1}$, amounting to just 28 percent above that predicted for the 6-yr period by the regression (*fig. 3b*). No effect was detected for pond accumulation by itself or total sediment load. For total sediment load, the (nonsignificant) departures from the regression predictions averaged -80 kg $ha^{-1}yr^{-1}$, or 8 percent below that predicted for the 6-yr period by the regression.

The absolute numbers reported in the above and earlier analyses of the South Fork logging (Rice and others 1979) must be viewed with reservation. The suspended load estimates were based on hand-drawn sediment rating curves describing the relation between the concentration of samples collected in a given year to the discharge levels at which they were collected. In several years, samples were not available from all discharge classes, so it was necessary to extrapolate the relation between concentration and discharge to higher or lower unrepresented classes. Also, a majority of the samples from the years 1963-1975 were collected using single-stage samplers that are filled only during the rising limb of hydrographs. In most storm events we have measured at Caspar Creek, the concentrations are markedly higher on the rising limb of the hydrograph than for equivalent discharges on the falling limb (e.g., *fig. 4*). Therefore, the fitted concentrations were likely too high. A plot of estimated sediment loads at NFC against annual water yield for the pre-logging years (*fig. 5*) suggests that there may be a positive bias during the single-stage years. The error associated with this method certainly varies from year to year, depending on the numbers of single-stage and manual samples and their distribution relative to the hydrographs. However, the plot indicates that loads were overestimated by a factor of between 2 and 3 in the range where most of the data occur. A comparison of the annual loads for the years 1986-1995 with annual sums of storm loads (the most accurate) shows very little bias, indicating that bias in the early years resulted mainly from sampling protocols rather than the computational method, which was the same for all years in this analysis.

North Fork Analysis Using Unlogged Subwatersheds as Controls

Because of improved and more intensive sampling methods, the suspended sediment loads for storm events beginning in 1986 are known far more accurately than the annual loads used in the NFC/SFC contrasts presented above. Four unlogged control watersheds were available (HEN, IVE, MUN, and SFC) for the analysis of storm loads. Unfortunately, only one large storm was available before logging. That storm was missed at SFC because of pumping sampler problems. Because of various technical difficulties, not all storms

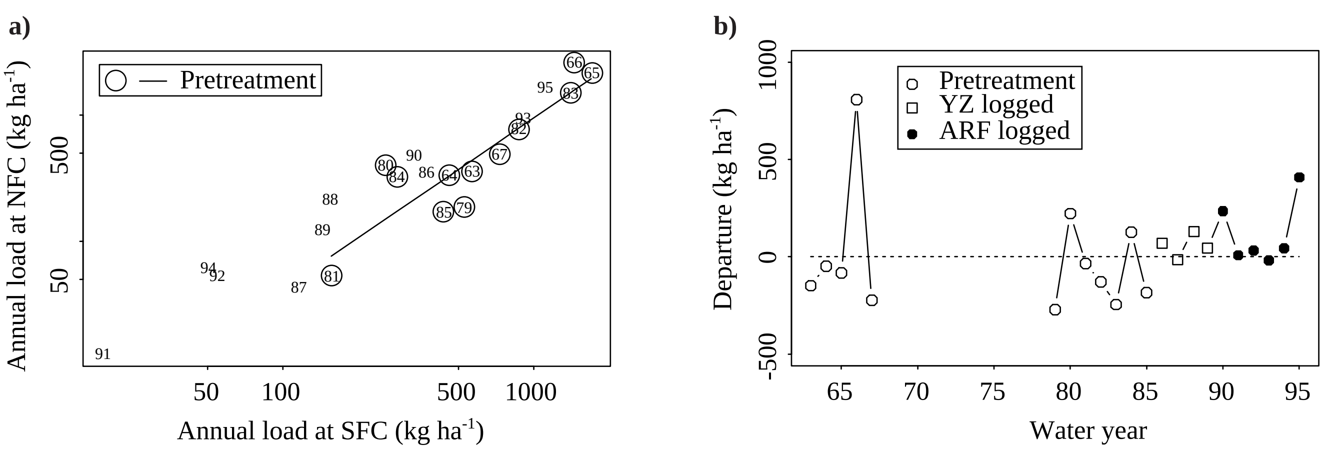

Figure 3—(a) Relation between estimated annual suspended sediment loads at North Fork Caspar Creek (NFC) and South Fork Caspar Creek (SFC) from 1963 to 1967 and 1979 to 1995, excluding years when sediment was elevated following logging in the South Fork. Pretreatment regression line is fit to the water years before roading and logging activity in the North Fork. (b) Time series of departures from the regression line.

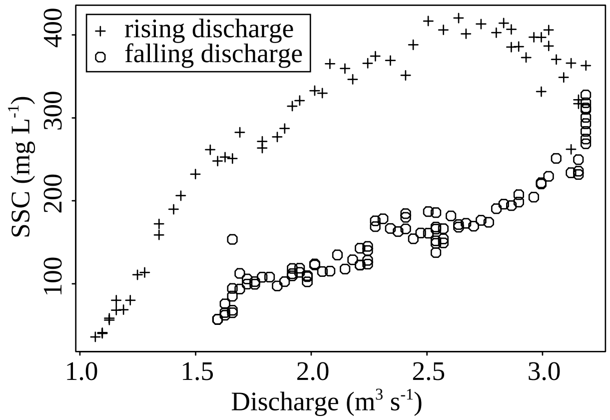

Figure 4—Storm event at lower main-stem station ARF, January 13-14, 1995, with water discharge and laboratory sediment concentrations (SSC) at 10-minute intervals.

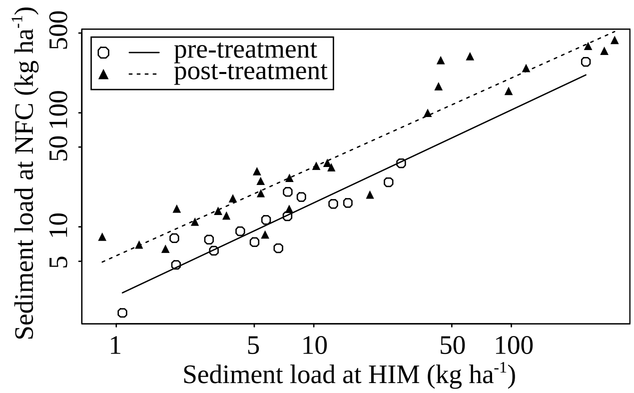

Figure 6—Relation between storm suspended sediment loads at North Fork Caspar Creek (NFC) and HIM control (mean suspended load of unlogged tributaries HEN, IVE, and MUN) from 1986 to 1995. Pretreatment regression line is based on storms in water years 1986-1989, before the major logging activity began.

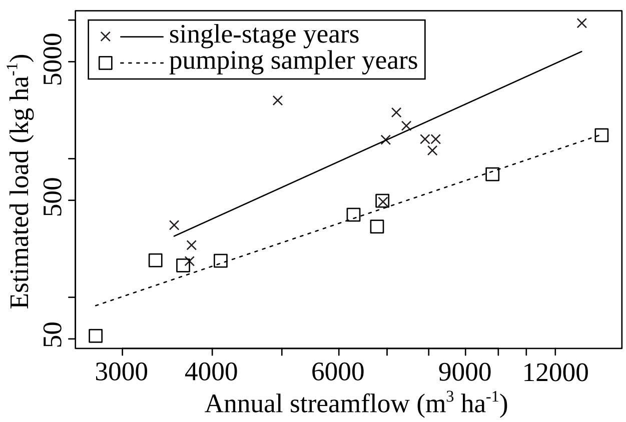

Figure 5—Relations between estimated annual suspended sediment loads and annual streamflow at North Fork Caspar Creek (NFC) prior to logging. Illustrates that load estimates based on sediment rating curves depend systematically on sampling protocols.

were adequately sampled at each station. However, the sample size for analyses was increased by using the mean of available data from the three tributary control watersheds, HEN, IVE, and MUN, in each storm. (SFC was eliminated because it had lower pretreatment correlations with the North Fork stations.) This mean (denoted HIM) provided a pretreatment sample size of 17 storms. The more accurate sediment loads, better controls, and larger sample size gave this analysis greater reliability and increased power to detect changes than the annual load analysis.

A weakness in analyses of logging effects at NFC was the need to use 1986-1989 as a calibration period even though 12 percent of the area had been clearcut. The clearcut area might be expected to somewhat diminish the size of the effect detected. The occurrence of only one large storm event before logging is mitigated by the fact that it was thoroughly sampled at both NFC and the three controls. An average of 59 sample bottles were collected at each of the four stations, and all the standard errors were less than 10 percent of the estimated loads, so there is little doubt about this point's validity.

Figure 6 shows regression lines fit to the suspended storm loads at NFC versus those at HIM before and after logging began in the spring of 1989. There was clearly an increase in suspended loads in small storms after logging began. In large storms there also seems to be an effect, although some post-treatment points are very close to the one large pretreatment point. The Chow test for a change after logging was significant with $p = 0.006$. The increases over predicted load, summed over all storms in the post-treatment period, average 188 kg ha^{-1}yr^{-1}, and amount to an 89 percent increase over background. The storms in this analysis represent 41 percent of the 1990-1996 streamflow at NFC, but carried approximately 90 percent of the suspended sediment that passed over the weir (based on figure 2 of Rice and others 1979).

A 3,600-m^3 landslide that occurred in the Z cut unit (Preface, fig. 2, these proceedings) increased sediment loads at the NFC gaging station starting in January 1995. NFC was the only gage downstream from this slide. The sum of suspended loads from storms preceding the landslide was 47 percent higher (64 kg ha^{-1}yr^{-1}) than predicted. The sum of suspended loads from storms after the landslide was 164 percent higher (150 kg ha^{-1}yr^{-1}) than predicted.

Individual Regressions for Subwatersheds

Similar analyses for each of the subwatersheds in the North Fork (*fig. 7* and *table 1*) indicate increased suspended sediment loads in all the clearcut tributaries except KJE. Sediment loads in the KJE watershed appear to have decreased after logging. The only partly clearcut watershed on a tributary (DOL) also showed highly significant increases in sediment loads. The upper main-stem stations (JOH and LAN) showed no effect after logging, and the lower main-stem stations (FLY and ARF) experienced increases only in smaller storms. Summing suspended sediment over *all* storms, the four main-stem stations all showed little or no change (*table 1*).

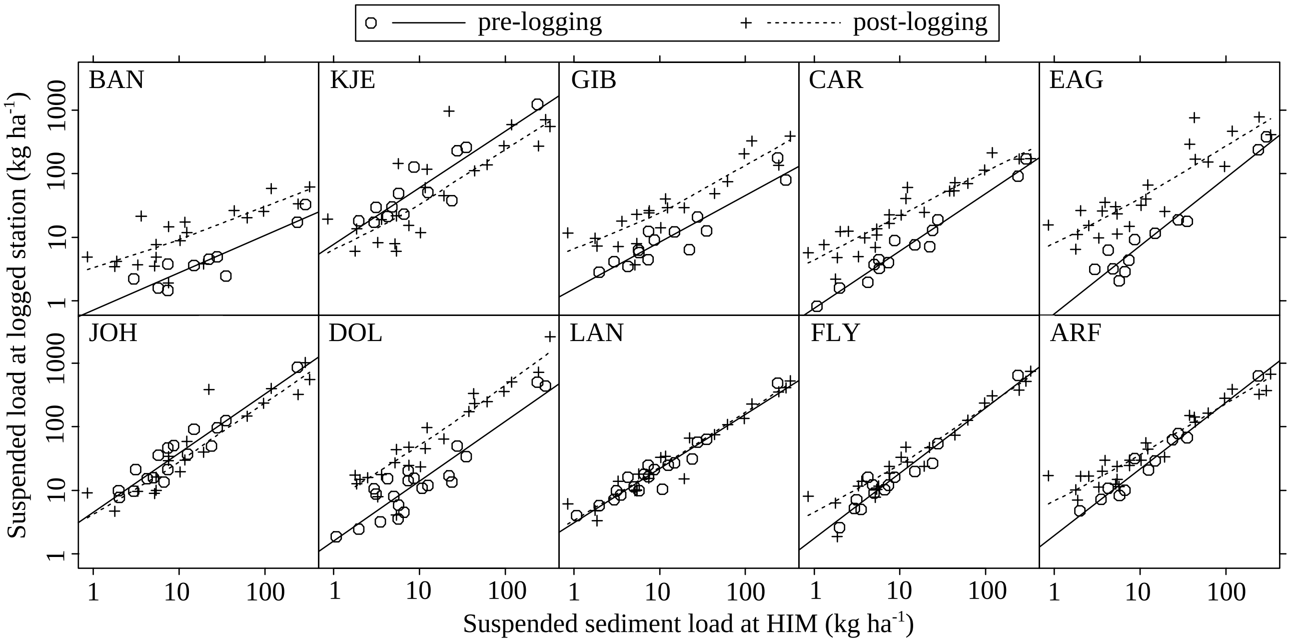

Figure 7—Relations between storm suspended sediment loads at logged subwatersheds in the North Fork and HIM control (mean suspended load of unlogged tributaries HEN, IVE, and MUN) from 1986 to 1995. Pre-logging regression lines are based on pretreatment years that are specific to each subwatershed. Post-logging relations are not assumed to be linear, hence were fitted by locally weighted regression (Cleveland 1993).

Table 1—*Summary of changes in suspended sediment load (summed over storms) after logging in North Fork subwatersheds. Predicted loads are computed from pre-treatment linear regressions between the logarithms of the storm sediment load in the treated watershed and the mean of the storm sediment loads at the control watersheds HEN, IVE, and MUN. Predictions were corrected for bias when back-transforming from logarithmic units. The number of years in the post-logging period varies from 4 to 6, depending upon when the watershed was logged and whether or not monitoring was discontinued in water year 1996.*

Treated watershed	Number of years	Observed ($kg\ ha^{-1}\ yr^{-1}$)	Predicted ($kg\ ha^{-1}\ yr^{-1}$)	Change ($kg\ ha^{-1}\ yr^{-1}$)	Change (%)
ARF	4	505	591	-86	-15
BAN	4	85	28	57	203
CAR	5	240	108	132	123
DOL	5	1130	306	824	269
EAG	5	710	210	500	238
FLY	5	536	555	-19	-3
GIB	4	358	119	239	200
JOH	5	667	865	-198	-23
KJE	5	821	1371	-551	-40
LAN	5	420	400	20	5
NFC	6	465	246	219	89

Aggregated Regression Model for Subwatersheds

To evaluate the relationships between suspended sediment load increases and possible explanatory variables, an aggregated regression model was fit simultaneously to all the subwatershed storms. The model utilized 367 estimated loads from 51 storms when HIM was used as the control or 333 estimated loads from 43 storms when HI (the mean of HEN and IVE) was used. Two regression coefficients were fitted for each watershed. A number of disturbance measures were considered (*table 2*), as well as an area term designed to describe cumulative effects, and a term explaining sediment increases in terms of flow increases. A great deal of effort went into developing a model that would permit valid tests of hypotheses concerning cumulative watershed effects. Therefore, the response model is coupled with a covariance model that describes variability in terms of watershed area and correlation among subwatershed responses as a function of distance between watersheds. These models were solved using the method of maximum likelihood and will be described in detail in a separate publication (Lewis and others 1998).

Departures from sediment loads predicted by the aggregated model for undisturbed watersheds were modest. The median increase in storm sediment load was 107 percent in clearcuts and 64 percent in partly clearcut watersheds. The median annual increase was 109 percent (58 kg $ha^{-1}yr^{-1}$) from clearcut watersheds and 73 percent (46 kg $ha^{-1}yr^{-1}$) from partly clearcut watersheds. The absolute flux values are underestimated somewhat because they include only sediment measured in storms, and no effort has been made to adjust for missing data. However, the major storms have been included, and virtually all of the sediment is transported during storms. Uncertainty due to year-to-year variability is certainly a much greater source of error.

The most important explanatory variable identified by the model was increased volume of streamflow during storms. Storm flow predictions (Ziemer, these proceedings) were based on an aggregated model analogous to that used for predicting sediment loads. The ratio of storm sediment produced to that predicted for an unlogged condition was positively correlated to the ratio of storm flow produced to that predicted for an unlogged condition (*fig. 8*). This result is not unexpected because, after logging, increased storm flows in the treated watersheds provide additional energy to deliver and transport available sediment and perhaps to generate additional sediment through channel and bank erosion.

Whereas individual watersheds show trends indicating increasing or decreasing sediment loads, there is no overall pattern of recovery apparent in a trend analysis of the residuals from the model (*fig. 9a*). This is in contrast with the parallel model for storm flow volume (*fig. 9b*), and suggests that some of the sediment increases are unrelated to flow increases.

Other variables found to be significant were road cut and fill area, and, in models using the HI control, the length of unbuffered stream channel, particularly in burned areas. Under California Forest Practice Rules in effect during the North Fork logging, buffers were not required for stream channels that do not include aquatic life and are not used by fish within 1,000 feet downstream except in confluent waters. As discussed earlier, one must be cautious about drawing conclusions about cause and effect when treatments are not randomly assigned to experimental units and replication is limited. Increases in sediment load in one or two watersheds can create associations with any variable that happens to have higher values in those watersheds, whether or not those variables are physically related to the increases. In this study, the contrast in response is primarily between watershed KJE, where sediment loads decreased, versus watersheds BAN, CAR, DOL, EAG, and GIB. Watershed KJE was unburned and also had the smallest amount of unbuffered stream of all the cut units. Watersheds EAG and GIB were burned and had the greatest amount of unbuffered stream in burned areas. Watershed EAG experienced the largest sediment increases and also had the greatest proportion of road cut and fill area. Because EAG was not unusually high in road surface area, the large road cut and fill area indicates that the roads in EAG are on steeper hillslopes.

There is little field evidence of sediment delivery from roads in

Table 2—Explanatory variables considered in modeling storm sediment loads in North Fork subwatersheds.

Mean unit area suspended load from control watersheds
Excess storm flow volume relative to that of control watersheds
Time since logging completed
Timber removed per unit watershed area
Areas of various disturbances as proportion of watershed:
Cable, tractor yarding
Stream protection zones, thinned areas
Burning (low intensity, high intensity)
Road cuts, fills, running surfaces
Skid trail cuts, fills, operating surfaces
Landing cuts, fills, operating surfaces
Areas of above disturbances within 46 m (150 ft) of a stream channel
Length of impacted stream in above disturbances per unit watershed area
Length of cabled corridors per unit watershed area
Watershed area

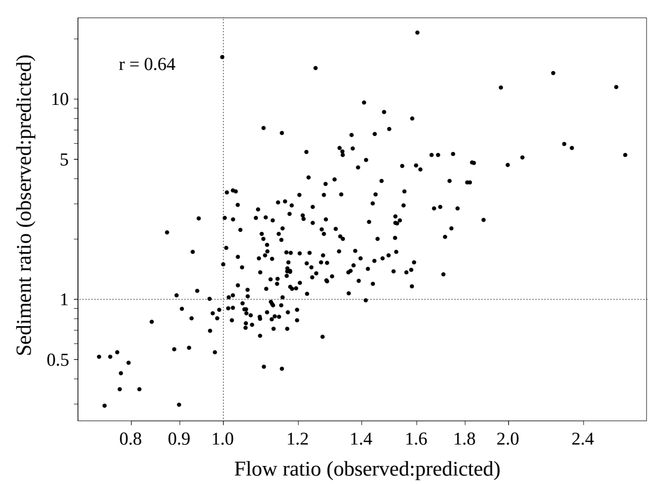

Figure 8—Relation between post-logging ratios of observed to predicted storm flow and suspended sediment load for all North Fork subwatersheds. Predictions are for undisturbed watersheds based on aggregated regression models using HI control (mean response of unlogged tributaries HEN and IVE).

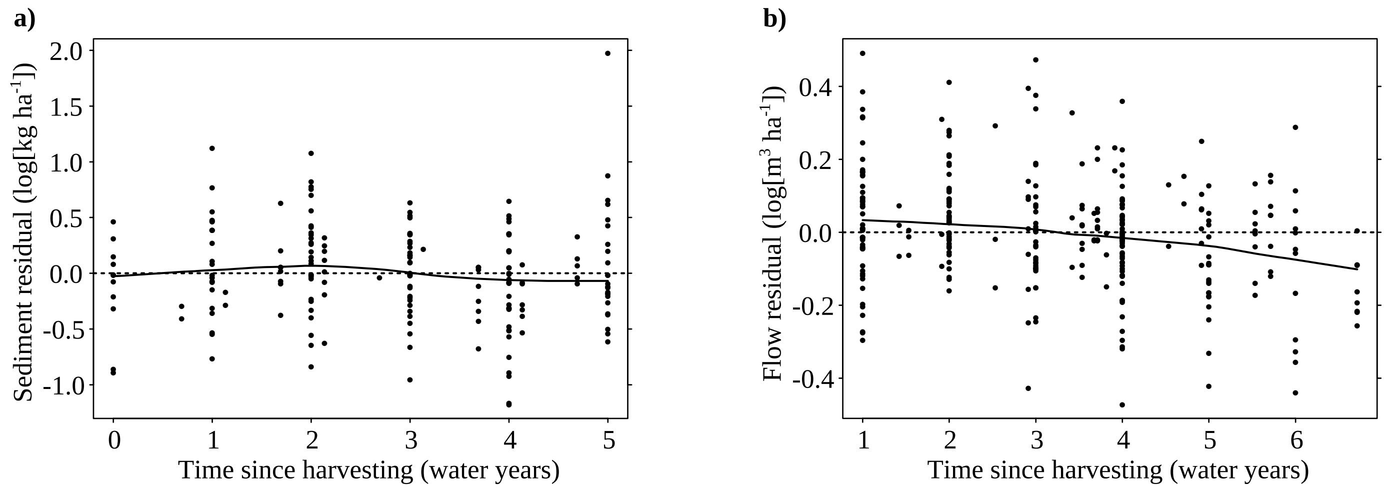

Figure 9—Relation between post-logging residuals from aggregated models and time (difference in water years) since harvesting. (a) model for storm suspended sediment loads, and (b) model for storm flow volumes. Curves were fitted by locally weighted regression (Cleveland 1993).

the North Fork watershed. In the inventory of failures greater than 7.6 m^3, only 8 of 96 failures, and 1,686 of 7,343 m^3 of erosion were related to roads. Nearly all of this road-related erosion was recorded as remaining on-site, and none of the road-related failures occurred in the EAG watershed. Based on the 129 random erosion plots (Rice 1996), the road erosion in EAG was 9.3 m^{3}ha^{-1}, compared to 34.5 m^{3}ha^{-1} for KJE and 16.6 m^{3}ha^{-1} for all roads in the North Fork. Thus it seems that the appearance of road cuts and fills in the model resulted from a spurious correlation.

On the other hand, channel reaches subjected to intense broadcast burns did show increased erosion from the loss of woody debris that stores sediment and enhances channel roughness (Keppeler, electronic communication). And increased flows, accompanied by soil disruption and burning in headwater swales, may have accelerated channel headward expansion, and soil pipe enlargements and collapses observed in watershed KJE (Ziemer 1992) and in EAG, DOL, and LAN.

Based on the 175 random erosion plots in harvest areas (Rice 1996), the average hillslope erosion rates in the burned watersheds EAG and GIB were 153 m^{3}ha^{-1} and 77 m^{3}ha^{-1}, respectively, the highest of all the watersheds. The average rate for the unburned clearcut watersheds BAN, CAR, and KJE was 37 m^{3}ha^{-1}. These figures include estimates of sheet erosion, which is difficult to measure and may be biased towards burned areas because it was easier to see the ground where the slash had been burned (Keppeler, verbal communication). About 72 percent of EAG and 82 percent of GIB were judged to be thoroughly or intensely burned, and the remainder was burned lightly or incompletely. It is unknown how much of this hillslope erosion was delivered to stream channels, but the proportion of watershed burned was not a useful explanatory variable for suspended sediment transport.

The failure inventory identified windthrow as another fairly important source of sediment. Of failures greater than 7.6 m^3, 68 percent were from windthrow. While these amounted to only 18 percent of the failure volume measured, 91 percent of them were within 15 m of a stream, and 49 percent were in or adjacent to a stream channel. Because of the proximity of windthrows to streams, sediment delivery from windthrow is expected to be disproportionate to the erosion volume. Windthrows are also important as contributors of woody debris to channels (Reid and Hilton, these proceedings), and play a key role in pool formation (Lisle and Napolitano, these proceedings). Because woody debris traps sediment in transport, it is unknown whether the net effect of windthrow on sediment transport was positive or negative.

Cumulative Effects

A full explanation of the rationale and methods of testing for cumulative watershed effects is beyond the scope of this paper, and final results on this topic will be reported by Lewis and others (1998). Preliminary results will simply be stated here.

I have considered three types of information that the aggregated model provides about the cumulative effects of logging activity on suspended sediment loads:

1. Were the effects of multiple disturbances additive in a given watershed?
2. Were downstream changes greater than would be expected from the proportion of area disturbed?
3. Were sediment loads in the lower watershed elevated to higher levels than in the tributaries?

The response being considered in all of these questions is the suspended sediment load per unit watershed area for a given storm event. Watershed area was used in the model to represent distance downstream.

The first question may be answered partly by looking at the forms of the storm flow and sediment models. Analyses of the residuals and covariance structures provide good evidence that the models are appropriate for the data, including the use of a logarithmic response variable. This implies a multiplicative effect for predictors that enter linearly and a power function for predictors that enter as logarithms. It turns out that the flow response to logged area is multiplicative, and the sediment response to flow increases is a power function. These effects, however, are *approximately* additive within the range of data observed for watersheds receiving flow from multiple cut units.

The second question was addressed by testing terms formed from the product of disturbance and watershed area. If the coefficient of this term were positive, it would imply that the effect of a given disturbance proportion increases with watershed size. A number of disturbance measures were considered, including road cut and fill area and length of unbuffered stream channels. None of the product terms were found to have coefficients significantly greater than zero, indicating that suspended load increases were not disproportionately large in larger watersheds. To the contrary, the sum of the observed sediment loads at the four main-stem stations were all within 25 percent of the sum of the loads predicted for undisturbed watersheds (*table 1*). Apparently, much of the sediment measured in the tributaries has been trapped behind woody debris or otherwise stored in the channels, so that much of it has not yet been measured downstream.

There is, however, one subwatershed where this second type of cumulative effect may be occurring. Watershed DOL, only 36 percent cut, includes the 100 percent cut watershed EAG, yet the sediment increases (269 percent at DOL versus 238 percent at EAG) have been similar. The increases in DOL seem to be related to channel conditions created in the historic logging (1900-1904) and, possibly, to increased flows from recent logging. At the turn of the century, the channel between the DOL and EAG gaging stations was used as a "corduroy road" for skidding logs by oxen. Greased logs were half-buried in the ground at intervals equal to the step length of the oxen (Napolitano 1996), and an abundance of sediment is stored behind them today (Keppeler, electronic communication). Energy available during high flows may be mobilizing sediment stored behind these logs. In the lower reach, the channel has a low width:depth ratio and is unable to dissipate energy by overflowing its banks. The high banks in this reach would be particularly vulnerable to increased peak flows, and have failed in a number of places in the years since EAG was logged.

The third question was addressed by testing watershed area as a linear term in the model. The coefficient of watershed area was positive ($p = 0.0023$), implying that the response, suspended sediment transport per unit watershed area, tends to increase downstream in the absence of disturbance. This tendency (with the exception of watershed KJE) is apparent in the pretreatment lines fit by least squares (*fig. 10a*), and could be reflecting the greater availability of fine sediment stored in these lower gradient channels. The relevance to cumulative effects is that downstream locations might reach water quality levels of concern with a smaller proportion of watershed disturbance than upstream locations.

To the extent that larger watersheds reflect average disturbance rates and therefore have smaller proportions of disturbance than the smallest disturbed watersheds upstream, one might expect sediment loads downstream to increase by less than those in the logged tributaries, reducing the overall variability among watersheds. In addition, as mentioned before, some of the sediment may be stored for several years before reaching the lower stations. That is what we observed in this study—the post-treatment regression lines (*fig. 10b*) were much more similar among watersheds than the pretreatment lines, and the main-stem stations no longer transported the highest sediment loads relative to watershed area.

Discussion

North Fork versus South Fork

My analysis of the South Fork logging data used a different model than was used by Rice and others (1979). However, the estimated increases in sediment loads were similar. For example, they reported suspended load increases of 1,403 kg $ha^{-1}yr^{-1}$ in the year after road construction and 3,254 kg $ha^{-1}yr^{-1}$ for the 5-yr period after logging. For the same periods, I estimated increases of 1,475 and 2,877 kg $ha^{-1}yr^{-1}$. Reversing the roles of the two watersheds for the later North Fork logging, the same analysis was unable to detect an effect. However, analysis of storm event loads from 1986 to 1996, using smaller subwatersheds within the North Fork as controls that had similar 19th-century logging histories as the whole North Fork, indicated that storm loads at NFC had increased by 188 kg $ha^{-1}yr^{-1}$. When comparing these figures, one should consider the differences between the water years 1972-1978 and 1990-1996, as well as differences in sampling methodologies that could have biased the estimated sediment loads. The mean annual unit area streamflow in the control (NFC) was 63 percent higher in 1972-1978 than that in the control (SFC) in 1990-1996. There is a surprisingly good relation between annual excess sediment load (departures from the pre-treatment regression) and water discharge in each of the studies (*fig. 11*). For equivalent flows, excess sediment loads in the South Fork analysis were six to seven times those in the North Fork analysis. It is probable that the sampling methods in the 1960's and 1970's resulted in overestimation of sediment loads in the South Fork analysis by a factor of 2 or 3. Therefore, comparisons between *relative* increases are more appropriate. Excess suspended load was 212 percent to 331 percent (depending on whether an adjustment is made for the 1974 North Fork landslide) after logging the South Fork, and 89 percent after logging the North Fork, suggesting that the effect of logging on suspended sediment load was 2.4 to 3.7 times greater in the South Fork than in the North Fork. These estimates approximately agree with estimates (Rice 1996) that both erosion and the sediment delivery ratio in the South Fork were about twice that in the North Fork.

Subwatersheds and KJE Anomaly

Analyses of the 10 treated subwatersheds in the North Fork drainage show suspended load increases at the gaging stations located

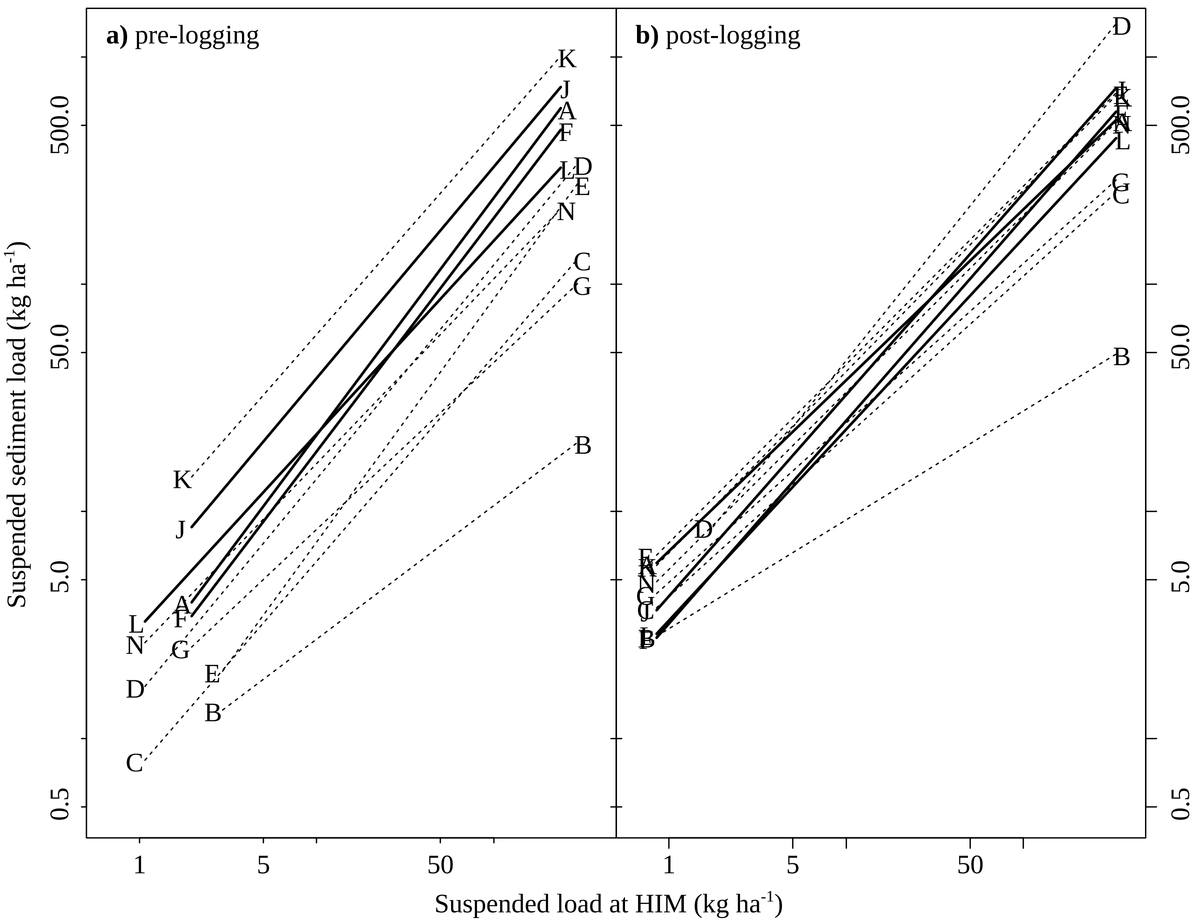

Figure 10—Regression lines for storm suspended sediment loads at treated watersheds in the North Fork, predicted from HIM control (mean suspended load of unlogged tributaries HEN, IVE, and MUN). (a) pre-logging, and (b) post-logging. Solid lines represent main-stem stations and dashed lines represent tributary stations.

immediately below clearcut units with one exception. At KJE, loads have decreased. A possible explanation for this anomaly lies in the tributary channel morphology. The stream channel in the KJE watershed is an extension of the main stem of the North Fork. It is (and, before recent logging, was) more deeply incised than the other tributaries, and it has the lowest gradient of tributaries other than the reach between the DOL and EAG gaging stations. The channel may have taken its gully-like form after the historic logging that took place between 1860 and 1904, when streams and streambeds were used as conduits for moving logs (Napolitano 1996). In any case, KJE had the highest pre-logging (1986-1989) unit area sediment loads of any of the tributaries (*fig. 10a*). Sediment in its channel is plentiful and the banks are actively eroding. It is likely, then, that the pre-logging sediment regime in KJE may have been energy-limited, which is more characteristic of disturbed watersheds. That is, sediment discharge was determined more by the ability of the stream to transport sediment than by the availability of sediment to be transported.

After logging, woody debris was added to the channel, and the number of organic steps in the buffered stream above KJE nearly doubled. Farther upstream, the channel was no longer shaded by the forest canopy and became choked with new redwood sprouts, horsetails, berry vines, and ferns, as well as slash that was introduced during logging. Although small storm flows did increase after logging, it is possible that channel roughness could have increased enough to reduce the energy available for sediment transport. An energy-limited stream would respond to increased sediment supply and reduced energy by reducing sediment transport. On the other hand, tributaries in a supply-limited sediment regime would have responded to a combination of increased sediment supply and reduced energy by increasing sediment transport. At some point, the increased supply probably converted these channels to an energy-limited regime, at which point stream power became the primary factor controlling variation in the increased transport levels. Rice and others (1979) concluded that is what happened after logging in the South Fork.

The aggregated regression for storm flow volumes (Lewis and others 1998; Ziemer, these proceedings) showed that flow increases

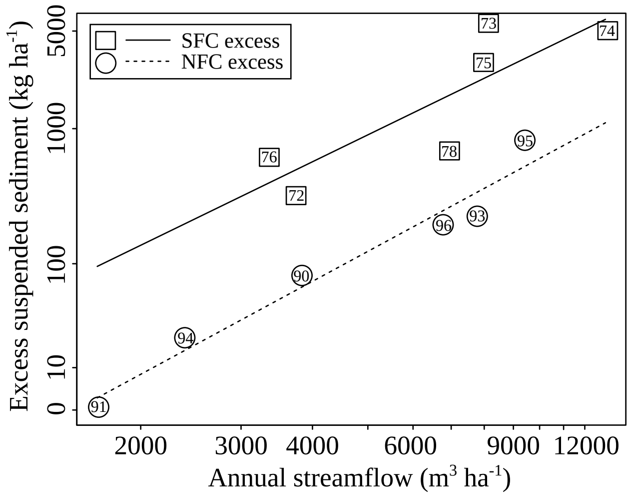

Figure 11—Relations between annual excess suspended sediment and annual streamflow for six years after logging in the South Fork and North Fork. South Fork excess loads are the departures from the pretreatment regression of figure 2. North Fork excess loads are the sums of storm departures from the pretreatment regression of figure 6.

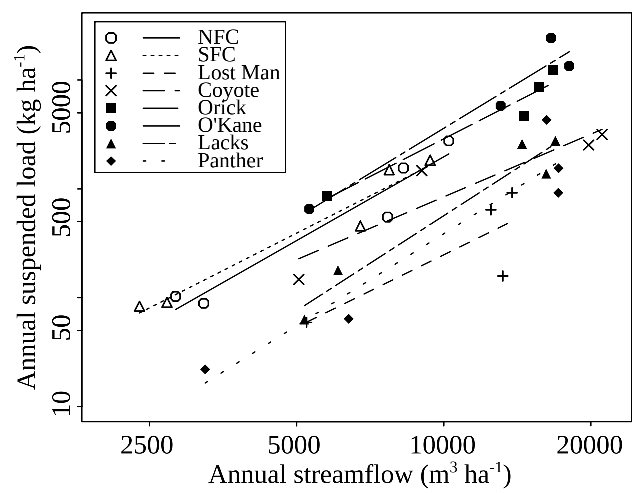

Figure 12—Relation between annual suspended sediment loads and annual streamflow for water years 1992-1996 at North Fork Caspar Creek (NFC), South Fork Caspar Creek (SFC), and 6 gaging stations in the vicinity of Redwood National Park. Caspar Creek sediment loads were divided by 0.6 to account for suspended sediment settling in the weir ponds.

could be largely explained by the proportion of a watershed logged, an antecedent wetness index, and time since logging. The aggregated regression for storm suspended sediment showed that much of the variability in suspended sediment load could, in turn, be explained by the flow increases. The implication is that, after logging, the channels were indeed in an energy-limited regime.

Flow increases accounted for only part of the variability in sediment production. Road systems would typically be expected to account for much of the sediment. However, in this case, roads were relatively unimportant as a sediment source because of their generally stable locations on upper hillslopes far from the stream channels. Field observations of increased bank erosion and gully expansion in clearcut headwater areas indicate that some of the suspended sediment increases were associated with the length of unbuffered stream channels in burned areas and, to a lesser degree, in unburned areas. Further indirect evidence that factors besides flow volume are elevating the suspended loads is that storm flows show a recovery trend, whereas storm suspended loads do not (*fig. 9*). This supports the hypothesis that the sediment regime has changed to one that will support elevated transport levels until the overall sediment supply is depleted. This can happen only after erosion and delivery rates to the channel decline and flows have been adequate to export excess sediment stored in the channels.

Cumulative Effects

Before logging, the larger main stem watersheds generally yielded the highest unit area sediment loads. But the increases after logging were greatest in the tributaries, resulting in a much narrower range of transport, for a given storm size, after logging (*fig. 10*). The North Fork of Caspar Creek is a small watershed (4.73 km^2). To see whether these results might be generalizable to larger watersheds, annual sediment loads for water years 1992-1996 were plotted against annual water yield (*fig. 12*) for NFC, SFC, and six gaging stations on streams in the vicinity of Redwood National Park (RNP). These watersheds were selected because of the high quality of their data and because, like Caspar Creek, they are underlain by the highly erodible Franciscan formation and historically supported mostly redwood forest with varying amounts of Douglas-fir. Caspar Creek receives less rainfall than the RNP watersheds, hence the lower annual flows.

In contrast to Caspar Creek, the RNP main-stem stations (Redwood Creek at Orick, 720 km^2, and at O'Kane, 175 km^2) continue to yield higher sediment loads than the RNP tributaries even after intensive management. Except for Little Lost Man Creek, these watersheds have been heavily logged at various times over the past 50 years, including the 1980's and 1990's. (Ground disturbance from logging in these watersheds was much more severe than that in Caspar Creek.)

The watershed with the lowest sediment loads is the unlogged Little Lost Man Creek (9.0 km^2), which is also the smallest of the RNP watersheds. Lacks Creek (44 km^2), Coyote Creek (20 km^2), and Panther Creek (16 km^2) are high-gradient (4-7 percent) channels in three different geologic subunits of the Franciscan formation (Harden and others 1982). Part of the explanation for the higher sediment loads at the main-stem stations may lie in the greater abundance of fine sediments available for transport in these low gradient (<1 percent) channels. Note that the Caspar Creek main stems are intermediate in both stream gradient (~1 percent) and sediment transport between the RNP tributaries and main stems. Regardless of the cause, if these lower reaches have the poorest water quality, then the incremental effect of an upstream disturbance may be cause for concern whether or not a water quality problem develops at the site of the disturbance. In other words, activities that have acceptable local consequences on water quality might have unacceptable consequences farther downstream when the preexisting water quality downstream is closer to harmful levels.

Cumulative effects considered in this paper were limited to a few hypotheses about water quality that could be statistically evaluated. But cumulative effects can occur in many ways. For example, resources at risk are often quite different in downstream areas, so an activity that has acceptable local impacts might have unacceptable offsite impacts if critical or sensitive habitat is found downstream. For a much broader treatment of cumulative effects see the discussion by Reid (these proceedings).

Conclusions

The main conclusions from these analyses are:

- Improved forest practices resulted in smaller increases in suspended load after logging the North Fork than after logging the South Fork. Increases were 2.4 to 3.7 times greater in the South Fork with roads located near the stream, all yarding by tractor, and streams not protected.
- Much of the increased sediment load in North Fork tributaries was related to increased storm flow volumes. With flow volumes recovering as the forest grows back, these increases are expected to be short-lived.
- Further sediment reductions in the North Fork probably could have been achieved by reducing or preventing disturbance to small drainage channels.
- Sediment loads are probably affected as much by channel conditions as by sediment delivery from hillslopes. The observed changes in sediment loads are consistent with conversion of those channels that were supply-limited before logging to an energy-limited regime after logging.
- The effects of multiple disturbances in a watershed were approximately additive.
- With one exception, downstream suspended load increases were no greater than would be expected from the proportion of area disturbed. To the contrary, most of the increased sediment produced in the tributaries was apparently stored in the main stem and has not yet been measured at the main-stem stations.
- Before logging, sediment loads on the main stem were higher than on most tributaries. This was no longer the case after logging. However, limited observations from larger watersheds suggest that downstream reaches in some watersheds are likely to approach water-quality levels of concern before upstream reaches.

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Trends in Streamflow and Suspended Sediment After Logging, North Fork Caspar Creek[1]

Jack Lewis[2] and Elizabeth T. Keppeler[3]

Abstract

Streamflow and suspended sediment were intensively monitored at fourteen gaging stations before and after logging a second-growth redwood (*Sequoia sempervirens*) forest. About 50 percent of the watershed was harvested, primarily by clear-cutting with skyline-cable systems. New road construction and tractor skidding were restricted to gently-sloping ridge top locations, and watercourse protections were enforced.

Storm peak flows increased as much as 300 percent in clear-cut watersheds, but as antecedent wetness increased, percentage increases declined. In the first five to seven years after logging, the average two-year peak flow increased 27 percent in clear-cut watersheds and 15 percent in partially clear-cut watersheds. Changes in flows are attributable to reduced canopy interception and transpiration. Peak flows and flow volumes had recovered to near-pretreatment levels by about 10 years after logging, when renewed increases occurred from precommercial thinning.

Annual suspended sediment loads in the years following logging increased 123 to 238 percent in four of the five clear-cut watersheds. Loads did not change significantly at most downstream sites as sediment was deposited in the main stem. Channel erosion and changes in storage appear to be important mechanisms for explaining suspended sediment trends at Caspar Creek. Ten years after logging, storm-event sediment yields at one clear-cut tributary were near pretreatment levels, but were elevated again in year 12. At another, yields have remained well above pretreatment levels in the 12 years since harvest.

Key words: clear-cutting, logging effects, peak flow, streamflow, suspended sediment,

Introduction

In 1985, a multiple-basin watershed study was initiated in the North Fork of the Caspar Creek Experimental Watershed, in north coastal California. The study is a cooperative effort by the USDA Forest Service, Pacific Southwest Research Station and the California Department of Forestry and Fire Protection to investigate the impacts of harvesting second-growth redwoods under the Z'Berg-Nejedly Forest Practices Act of 1973. Although the logging included large clear cuts (maximum clear-cut size has since been reduced under California rules from 32 to 12 ha), erosional impacts were limited by careful road design and greatly restricted use of

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tractors. The Proceedings of the Conference on Coastal Watersheds: The Caspar Creek Story (Ziemer, 1998) and Lewis and others (2001) reported results of the North Fork study through HY1996 (hydrologic year from August 1, 1995 to July 31, 1996). This paper extends the results through HY2003.

Methods

Study location

The Caspar Creek Experimental Watersheds are located about seven km from the Pacific Ocean and about 10 km south of the town of Fort Bragg in northwestern California. Elevation ranges from 37 to 320 m. Soils in the basin are well-drained clay loams derived from Franciscan sandstone and weathered coarse-grained shale of Cretaceous age.

The climate is typical of low-elevation coastal watersheds of the Pacific Northwest. Winters are mild and wet, characterized by periods of low-intensity (maximum 2.6 cm/hr) rainfall. Snow is rare. Average annual precipitation is 1170 mm. Typically, 95 percent falls during the months of October through April. Summers are moderately warm and dry with maximum temperatures moderated by frequent coastal fog. Mean annual runoff is 650 mm.

Like most of California's north coast, the watersheds were clear-cut and broadcast burned largely prior to 1900. By 1985, the North Fork watershed supported a 100-year-old second-growth forest composed of coast redwood (*Sequoia sempervirens* (D.Don) Endl.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), western hemlock (*Tsuga heterophylla* (Raf.).Sarg.), and grand fir (*Abies grandis* (Dougl. ex D.Don) Lindl.).

Measurements

The North and South Forks of Caspar Creek (draining 473 ha and 424-ha, respectively) have been gaged continuously since 1962 using 120° V-notch weirs widening to concrete rectangular sections for high discharges. In 1985, three rated sections were constructed on the main stem upstream of the North Fork weir, and 10 Parshall flumes were installed on North Fork subwatersheds with drainage areas of 10 to 77 ha. Two of the original redwood Parshall flumes were replaced with fiberglass Montana flumes in HY1999 and 2001.

Since HY1986, stream discharge has been recorded at all gaging stations using electronic data loggers equipped with pressure transducers. From HY1986 to HY1995, suspended sediment was automatically sampled using real-time stage measurements to control a pumping sampler (Thomas 1989). Since HY1996, turbidity is recorded along with stage, and the sampling logic has been altered to use real-time turbidity (Lewis and Eads 2001).

Treatments

Ten areas were designated for harvest in compliance with the California Forest Practice Rules in effect in the late 1980s *(fig. 1)*. Two of these areas (13 percent of the North Fork watershed) were harvested in 1985 and 1986 with the intent of excluding them from the study. However, this harvest affects all subsequent analyses of North Fork weir data. After a calibration period between 1985 and 1989, clear-cut logging began elsewhere in the North Fork in May 1989 and was completed in January 1992. These clear-cuts occupied 30 to 99 percent of treated watersheds and totaled 162 ha. Between 1985 and 1992, 46 percent of the North Fork watershed was clear-cut, 1.5 percent was thinned, and two percent was cleared for road right-of-way. Of the fourteen gaged watersheds in the North Fork, five were clear-cut, three were left as unlogged controls, and six included mixtures of clear-cut and unlogged areas. In HY1996, stream gaging was discontinued at all but two of the clear-cut watersheds, two of the controls, and three of the partially clear-cut watersheds.

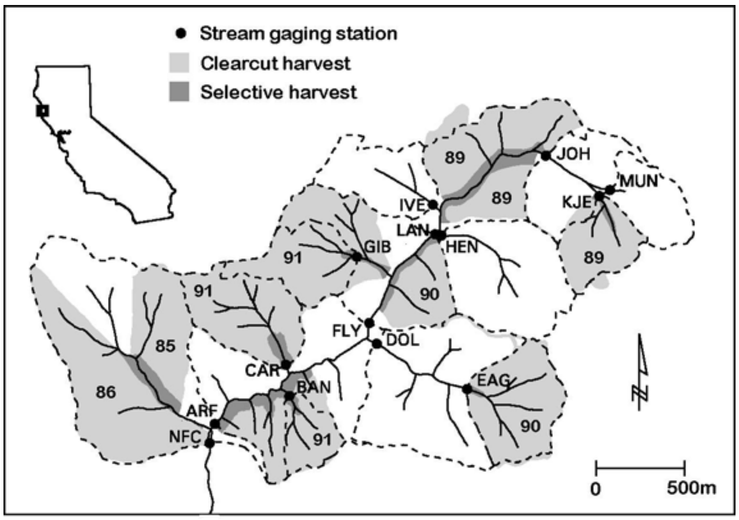

Figure 1—North Fork Caspar Creek gaging stations and harvest units.

Harvest was conducted under stream-buffer rules that mandated equipment exclusion and 50 percent canopy retention within 15 to 46 m of watercourses providing aquatic habitat or having fish present. Most of the yarding (81 percent of the clear-cut area) was accomplished using skyline-cable systems. Yarders were situated on upslope landings constructed well away from the stream network. New road construction and tractor skidding was restricted to ridgetop locations with slopes generally less than 20 percent. Four harvest blocks, 92 ha total, were broadcast burned and later treated with herbicide. Pre-commercial thinning in 1995, 1998, and 2001 eliminated much of the dense revegetation and reduced basal area in treated units by about 75 percent.

Results

Storm peaks

Lewis and others (2001) analyzed peak flow response to clear-cutting in the North Fork using 526 observations from HY1986 to HY1996, representing 59 storms on 10 treated watersheds. After logging, eight of the 10 tributary watersheds experienced increased storm peaks ($p < .005$) relative to those predicted on the basis of the controls for an uncut condition. In clear-cut units, individual storm peaks increased as much as 300 percent, but most increases were less than 100 percent. The largest increases occurred during early season storms. As basin wetness increased, percentage peak flow increases declined *(fig. 4)*. In the larger, partially clear-cut

North Fork watersheds, smaller peak flow increases were observed. Under the wettest antecedent moisture conditions of the study, increases over the first five to seven years after logging averaged 23 percent in clear-cut watersheds and 3 percent in partially clear-cut watersheds. The average increase in storm peak with a two-year return period was 27 percent in the clear-cut watersheds and 15 percent in the partially clear-cut watersheds (Ziemer 1998) for this five to seven year period. While variability is great, ongoing measurements clearly show a recovery to near pre-treatment flow conditions 10 years post-harvest and the suggestion of a renewed response to the pre-commercial thinning *(fig. 2)*.

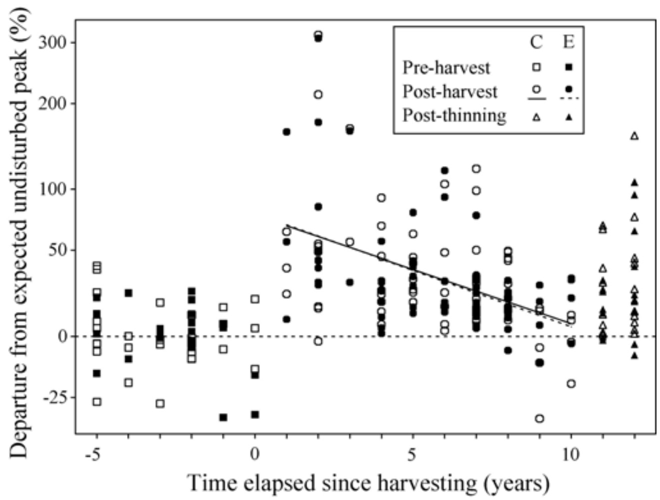

Figure 2—Peak flows observed in North Fork clear-cut units C and E from HY1986 through HY2003. Expected undisturbed peak is based on log-log regressions of pre-harvest peak flows at CAR and EAG on the mean of the corresponding peak flows at control watersheds HEN and IVE.

Wetter soils resulting from reduced transpiration in logged units explain some of the observed increases in streamflow. In addition, because of reduced canopy interception, 28 percent more precipitation is delivered to the forest floor after clear-cut logging in these second-growth redwood stands (Reid and Lewis 2006). Under forested conditions, canopy interception is significant even during the wettest mid-season storms. Loss of interception is therefore expected to maintain wetter soil conditions in logged terrain throughout the rainy season.

Lewis and others (2001) fit an empirical model expressing the HY1986-1996 North Fork peak flows as a function of peaks in the control watersheds, antecedent wetness, proportion of area logged, and time since logging. In this follow-up, a slightly simplified version of that model was refit, using generalized non-linear least squares, to all peak flows before pre-commercial thinning (HY1986-2001).

$$\ln(y_{ij}) = \beta_{0i} + \beta_{1i}\ln(y_{Cj}) + \left[\left(1-\beta_2(t_{ij}-1)\right)c_{ij} + \beta_3 c'_{ij}\right]\left[\beta_4 + \beta_5\ln(y_{Cj}) + \beta_6\ln(w_j)\right] + \varepsilon_{ij} \quad (1)$$

where

y_{ij} = unit area peak flow at treated watershed *i*, storm *j*,

y_{Cj} = mean of unit area peak flows at control watersheds HEN and IVE in storm *j*,

t_{ij} = area-weighted mean cutting age (number of summers passed) in watershed *i* for areas logged in water years preceding that of storm *j*,

c_{ij} = proportion of watershed *i* logged in water years prior to that of storm *j*,

c'_{ij} = proportion of watershed *i* logged in the fall prior to storm *j* (in the same water year)

w_j = wetness index at start of storm *j*, computed from daily streamflow (30-day half-life) at South Fork weir

ε_{ij} = independent normally distributed errors with variance inversely proportional to a power function of watershed area

β_{0i} and β_{1i} are "location" parameters to be estimated for each watershed *i*, and

β_2, β_3, β_4, β_5, and β_6 are parameters describing the effects of the explanatory variables

The first two terms in the model predict the peak flow in the absence of disturbance. The first bracketed term represents vegetation removal and regrowth, and the terms in the second set of brackets are the main effect of vegetation change (β_4) and interactions of vegetation change with storm size and antecedent wetness. The coefficient estimates and their standard errors are given in *table 1*. This model fits the data well (r^2 = 0.95) and residuals are normally distributed with standard error equivalent to 25 percent of the predicted peak.

Table 1—*Parameter estimates for storm peaks and flow volume models.*

		----------Storm peak----------			-------Storm flow volume-------		
Parameter	**Effect**	**Estimate**	**Std error**	**p**	**Estimate**	**Std error**	**p**
β_2	Recovery	0.101	0.0063	<0.0001	0.110	0.0059	<0.0001
β_3	Fall logging	0.447	0.0965	<0.0001	0.876	0.0926	<0.0001
β_4	Vegetation reduction	1.290	0.2596	<0.0001	2.824	0.2287	<0.0001
β_5	Storm size interaction	-0.110	0.0363	0.0025	-0.140	0.0392	0.0004
β_6	Wetness interaction	-0.278	0.0177	<0.0001	-0.298	0.0178	<0.0001

The fitted value of 0.101 for the coefficient β_2 implies recovery of peak flows to pretreatment conditions after 11 growing seasons, in concordance with *figure 2*. A 95 percent confidence interval for β_2 implies recovery in 10 to 12 years. The fitted value of 0.447 for β_3 suggests that the effect on peak flows during the first winter was reduced by about 55 percent because much of the harvest occurred late in the growing season, after substantial transpiration had occurred. The storm size interaction indicates that the proportional increase in peak flows was smaller for larger events, and the wetness interaction indicates that increases in peak flows are greatest during low antecedent wetness conditions.

Model (1) was used to predict peak flows without accounting for the change in cover following thinning. *Figure 3* shows the departures from peak flows predicted by this model for the two clear-cut watersheds, CAR and EAG, that are still being monitored. Departures, e_{ij}, are converted to percentage of predicted peak through the transformation 100exp(e_{ij}). The recovery trend depicted in *figure 2* is not visible in *figure 3* because the model accounts for the recovery. However, the mean post-thinning departure from the predicted peak is 26 percent (the 95 percent confidence interval is 16 to 37 percent). These departures are greatest when antecedent wetness is greatest *(fig. 4)*, suggesting that mechanisms similar to those responsible for increasing peaks after clear-cutting are involved in changing peaks after thinning.

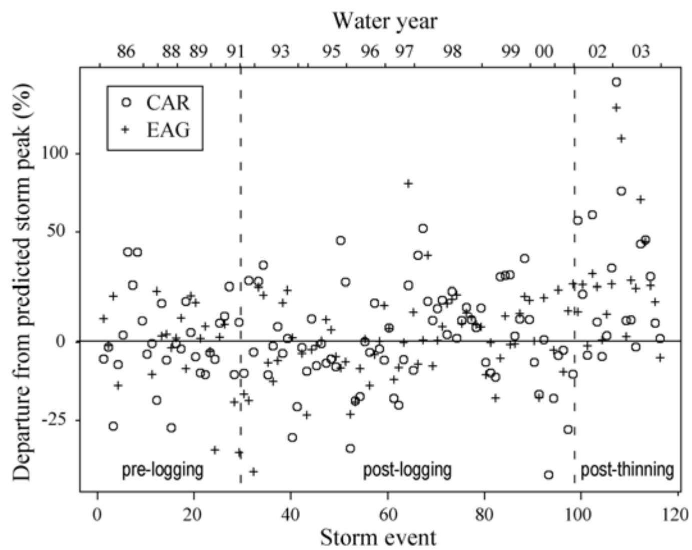

Figure 3—Departures from model (1) predictions of storm peak.

The thinning in watersheds CAR and EAG removed 68 and 84 percent of the crown volumes, respectively. The peaks model permits a test of whether these treatments were equivalent to clear-cutting the same percentage of the watersheds. For the calculations of *figures 3* and *4*, the variable t_{ij} was coded as 10 and 11 years, respectively, for CAR and EAG in HY2002, the winter following thinning. However, if we treat the disturbance as if 68 and 84 percent of the areas were clear-cut in the beginning of HY2002, the area-weighted mean cutting ages t_{ij} should be coded 3.2 for CAR and 1.8 for EAG in HY2002; and the ages in HY2003 should be 4.2 and 2.8 years. Based on this recoding of t_{ij}, the model predicts an average increase of 52 percent in peak flows, suggesting that thinning had half the impact on peak flows of an equivalent harvest by clear-cutting. Such a result is expected if evaporation and transpiration rates are elevated in a thinned stand because of lower aerodynamic resistance to the transport of water vapor as suggested by Calder (1990).

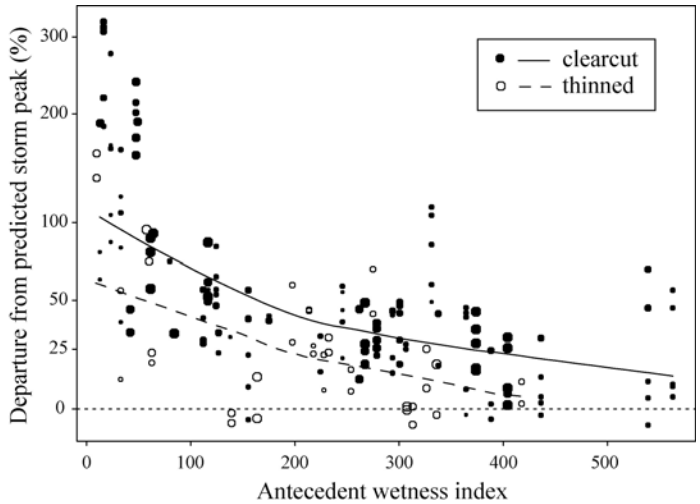

Figure 4—Relation to antecedent basin wetness of (a) clear-cut departures from pretreatment regressions (BAN, CAR, EAG, GIB, and KJE through HY1996), and (b) post-thinning departures from model (1) (CAR and EAG, HY2002-2003). Symbol sizes denote relative storm sizes.

Storm flow volumes

Storm flow volumes were analyzed using the same methods as for peak flows. The results through HY1996, reported by Lewis and others (2001) were similar to peak flow results. In clear-cut units, storm flows increased as much as 400 percent, but most increases were less than 100 percent. The largest increases occurred during early-season storms. As basin wetness increased, percentage increases declined. Under the wettest antecedent moisture conditions of the study, increases averaged 27 percent in clear-cut watersheds and 16 percent in partially clear-cut watersheds over the five to seven year period following harvest. Annual storm runoff volume (sum of storms) increased an average of 58 percent in clear-cut watersheds and 23 percent in partly clear-cut watersheds (the mean percentage harvested was 38 percent). As with peak flows, ongoing measurements show a return to pre-treatment flow volumes approximately 10 years post-harvest, followed by a response to the pre-commercial thinning *(fig. 5)*.

Model (1) also the fits the flow data well (r^2 = 0.94) with normally-distributed residuals and standard error equivalent to 21 percent of the predicted flow volume. The estimated recovery coefficient (*table 1*) suggests return of storm flows to pretreatment condition 10 years after logging, and is consistent with *figure 5*.

The flow model enabled quantification of the impact of pre-commercial thinning at CAR and EAG for 18 events in the two post-thinning years. The mean post-thinning departure from predicted flow volume was 26 percent (the 95 percent confidence interval is 15 to 38 percent) and the total storm flow volume was 19 percent greater than predicted by the model.

When the variable t_{ij} was recoded (as described above for peaks) to represent thinning as an equivalent harvest by clear-cut, the model predicts a mean increase of 53 percent and total increase of 44 percent in storm flow. Compared to an equivalent clear-cut, the mean effect of thinning on storm flows was about half (26/53) and the total effect on storm flows was 43 percent (19/44).

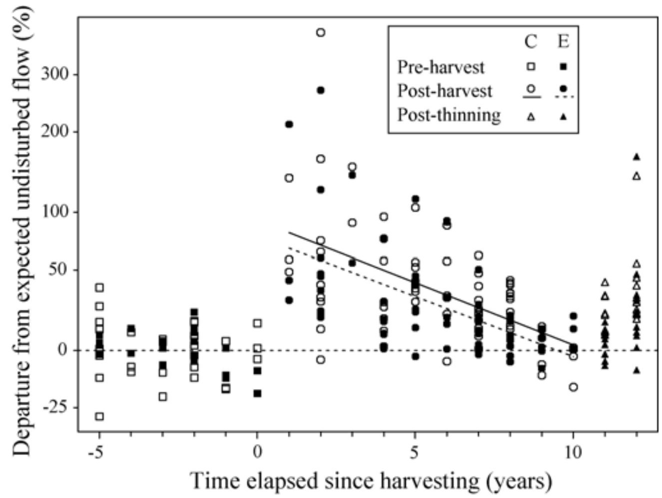

Figure 5—Storm flow volumes observed in North Fork clear-cut units C and E from HY1986 through HY2003. Expected undisturbed flow is based on log-log regressions of pre-harvest flows at CAR and EAG on the mean of the corresponding storm flows at control watersheds HEN and IVE.

Sediment Loads

Suspended sediment loads, summed over post-logging years through HY1996 increased 89 percent at the North Fork weir, primarily due to one landslide that occurred in the 1986 harvest area in 1995. Annual (sum of storms) suspended sediment loads in the years following logging decreased by 40 percent in one clear-cut watershed (KJE) and increased 123 to 238 percent in the other four clear-cut watersheds. Loads did not change significantly at most downstream sites, but at DOL increased by 269 percent. The median estimate of change in annual sediment load was +132 kg $ha^{-1}yr^{-1}$ for five clear-cut watersheds and –19 kg $ha^{-1}yr^{-1}$ for five partially clear-cut watersheds. Increases in sediment loads were greatest during those events with increased storm flows. In clear-cut watersheds where sediment loads increased, the correlations between departures from pretreatment sediment load and storm flow models were 0.66 (BAN to HY1995), 0.70 (CAR to HY2003), 0.62 (EAG to HY2003), and 0.86 (GIB to HY1995). Sediment increases at EAG have been greater than at CAR due to near-channel tunnel collapses. Storm event loads in EAG remained elevated a decade after harvest, while, at CAR, yields were close to the pretreatment level in year 10 (*fig. 6*). Suspended sediment levels from both subwatersheds, especially EAG, increased sharply in year 12 (HY2003), the first above-average runoff year since HY1999. Although sediment levels did not increase the first year after thinning, they certainly may have been influenced by the larger enhanced flows of HY2003. Prolonged impacts from logging in the South Fork (Keppeler and others, 2003) suggest that the episodic nature of sediment releases requires patience regarding conclusions about recovery.

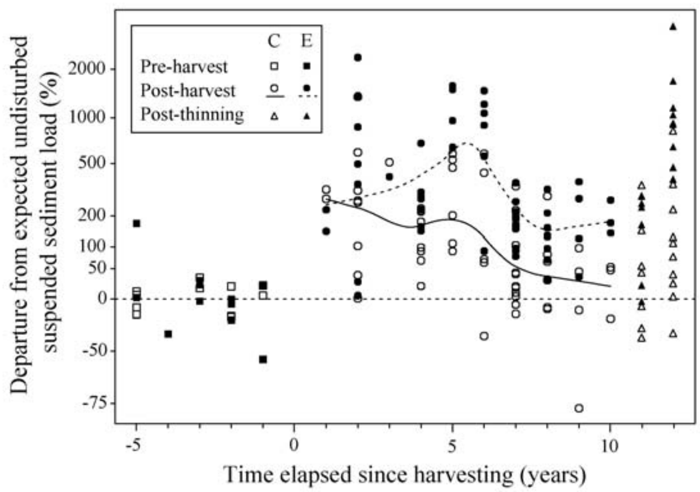

Figure 6—Sediment loads observed in North Fork clear-cut units C and E from hydrologic year 1986 through 2003.

Discussion and Conclusions

Although the variability is great, the impacts of clear-cut logging and forest regrowth on peaks and flows in the North Fork of Caspar Creek are fairly clear and quantifiable. Earlier analyses of selective logging in the South Fork of Caspar Creek (Ziemer 1981) had failed to show significant changes in peak flows, except in the smallest events at the beginning of the rainy season. Those results are not necessarily at odds with the North Fork study and may be attributable in part to differences in silvicultural methods (selective versus clear-cut logging). If thinning is a valid analog for selection cutting, our analysis suggests that the South Fork response should have been smaller than that in the North Fork. In addition, the North Fork analyses were more sensitive because multiple unlogged subwatersheds of the North Fork were available for use as controls. Low variability in the pretreatment relationship is critical to an effective watershed experiment, and the responses in North Fork watersheds slated for treatment were more closely related to North Fork subwatershed responses than to the South Fork response. In fact, the mean of two unlogged subwatersheds provided a better control than any individual subwatershed.

An empirical statistical model describes impacts on flow peaks and volumes in terms of antecedent basin wetness, proportion of area cut, time since logging, and event size. The effect of vegetation removal is greatest when the wetness index is low and diminishes as basin wetness increases. However, no conditions were observed under which the impacts were reduced to zero. The result is not unexpected given that effective rainfall is increased substantially by the loss of canopy interception throughout the rainy season (Reid and Lewis 2006).

A somewhat surprising result is that flow peaks and volumes 10 years after logging were similar to those in 100-year-old redwood forest. Further research will be necessary to understand this result, but it suggests that leaf area recovers very rapidly after harvest, and/or that evapotranspiration rates per unit leaf surface are much greater in younger forests. In fact, evidence suggests both may be true. Crown closure and maximum leaf area in one redwood plantation was attained within 15 years.[4] In riparian Douglas-fir forests of western Oregon, Moore and others (2004) found that a 40-year-old, rapidly growing stand used 3.3 times more water during the growing season than an old-growth stand.

Pre-commercial thinning resulted in smaller flow changes than would have been expected from equivalent clear-cuts. This may be partly related to the influence of canopy structure on evaporation rates. Calder (1990) reported that interception rates in mature spruce forest were almost unchanged after thinning one-third of the stand. He speculated that increased ventilation to lower levels of the canopy could increase evaporation rates. Reduced competition for soil water could also permit increased transpiration by vegetation that remains after thinning.

Variability in suspended sediment yield is much greater than variability in flow. Results are less consistent among clear-cut subwatersheds and much less predictable in downstream watersheds. One North Fork subwatershed that was clear-cut (KJE) experienced a decrease in sediment loads. The others experienced substantial increases. Of the two that are still being measured, neither has returned to pretreatment levels, and one (EAG) is yielding significantly more sediment than the other (CAR). One downstream site (DOL) had larger than expected sediment yields, apparently because of increased channel erosion, while those on the main stem have not experienced elevated sediment yields, apparently because of increased sediment storage. Unusual windstorms in combination with increased wind exposure in stream buffer zones resulted in blowdown that created many new sediment storage sites in the formerly wood-deprived main stem.

The sediment results are less directly extensible to other watersheds than the flow results, because they depend on events and conditions unlikely to be repeated in every coastal watershed. This is especially true as one moves downstream from first and second order streams to locations where channel complexity is greater. The results of the Caspar Creek sediment studies are probably not useful for making quantitative predictions, but they have helped us to understand many controlling factors and links among erosion, sediment delivery, and sediment transport. It has become clear that sediment impacts from regulated logging in the North Fork have been less severe than those from the tractor logging that took place in the South Fork (Keppeler and others 2003), and the research suggests opportunities for further reducing impacts. For example, limiting the rate of harvest in a given watershed would clearly limit increases in peak flows and flow volumes. Sediment yield increases in the North Fork were related to flow increases, so limiting harvest rates should also be effective in limiting sediment impacts. To further limit sediment yields in the North Fork would have required extending streamside protection zones farther upstream, but the incremental benefit of doing so is difficult to quantify, and it probably would not have greatly reduced sediment yields in DOL where much of the channel and bank erosion occurred downstream from the logged watershed (EAG).

Today much of the managed timber-producing area of north coastal California has been logged at least twice and may have experienced heavy impacts from tractor logging and road construction. The condition of the South Fork of Caspar Creek is probably more typical of areas being logged today than was the North Fork. It is becoming crucial for landowners, regulatory agencies, and the public to understand

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the interactions between proposed future activities and prior disturbances. A third phase of Caspar Creek research is being initiated in the South Fork to examine the effects of re-entry on runoff and sediment production from previously tractor-logged redwood forests. Much remains to be learned about restoring impacted ecosystems and mitigating impacts from future harvests.

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Impacts of Logging on Storm Peak Flows, Flow Volumes and Suspended Sediment Loads in Caspar Creek, California

Jack Lewis, Sylvia R. Mori, Elizabeth T. Keppeler, Robert R. Ziemer

ABSTRACT

Models are fit to 11 years of storm peak flows, flow volumes, and suspended sediment loads on a network of 14 stream gaging stations in the North Fork Caspar Creek, a 473-ha coastal watershed bearing a second-growth forest of redwood and Douglas-fir. For the first 4 years of monitoring, the watershed was in a relatively undisturbed state, having last been logged prior to 1904, with only a county road traversing the ridgetops. Nearly half the watershed was clear-cut over a period of 3 years, and yarded primarily using uphill skyline cable systems to spur roads constructed high on the slopes. Three tributaries were maintained as controls and left undisturbed. Four years of data were collected after logging was completed. Exploratory analysis and model fitting permit characterization and quantification of the effects of watershed disturbances, watershed area, antecedent wetness, and time since disturbance on storm runoff and suspended sediment. Model interpretations provide insight into the nature of certain types of cumulative watershed effects.

INTRODUCTION

This paired-watershed study in the North Fork of Caspar Creek was motivated by a desire to understand how a particular logging system affects storm peak flows, flow volumes, and suspended sediment loads in a second-growth coastal redwood forest. The logging system consisted of clear-cutting with streamside buffers, and yarding primarily by skyline to spur roads located on upper slopes and ridges. Primary objectives were to quantify how impacts vary with different levels of disturbance and how the effects of a given disturbance vary downstream. Pursuant to these objectives, a statistical model was developed for a treatment-and-control experimental design involving multiple watersheds. The study was also an opportunity for testing new technologies, and demonstrates two new automated schemes for suspended sediment sampling. Techniques for estimating sediment loads from these samples are tested and applied.

Storm Peaks

Throughout much of the Pacific Northwest, a large soil moisture deficit develops during the dry summer. With the onset of the rainy season in the fall, the dry soil profile begins to be recharged with moisture. In the H.J. Andrews Experimental Forest in the Oregon Cascades, the first storms of the fall produced streamflow peaks from a 96-ha clear-cut watershed that ranged from 40% to 200% larger than those predicted from the pre-logging relationship [Rothacher, 1971; 1973]. In the Alsea watershed near the Oregon coast, Harris [1977] found no significant change in the mean peak flow after clear-cutting a 71-ha watershed or patch cutting 25% of an adjacent 303-ha watershed. However, when Harr [1976] added an additional 30 smaller early winter runoff events to the data, average fall peak flow was increased 122%. In Caspar Creek, Ziemer [1981] reported that selection cutting and tractor yarding of an 85-year-old second-growth redwood and Douglas-fir forest increased the first streamflow peaks in the fall about

300% after logging. The effect of logging on peak flow at Caspar Creek was best predicted by the percentage of area logged divided by the sequential storm number, beginning with the first storm in the fall. These first rains and consequent streamflow in the fall are usually small and geomorphically inconsequential in the Pacific Northwest. The large peak flows, which tend to modify stream channels and transport most of the sediment, usually occur during mid-winter after the soil moisture deficits have been satisfied in both the logged and unlogged watersheds.

Studies of large peak flows in the Pacific Northwest have not detected significant changes after logging. Rothacher [1971, 1973] found no appreciable increase in peak flows for the largest floods attributable to clear-cutting. Paired watershed studies in the Oregon Cascades [Harr et al., 1979], Oregon Coast Range [Harr et al., 1975; Harr, 1976; Harris, 1977], and at Caspar Creek [Ziemer, 1981; Wright et al., 1990] similarly suggested that logging did not significantly increase the size of the largest peak flows that occurred when the ground was saturated.

Using longer streamflow records of 34 to 55 years, Jones and Grant [1996] evaluated changes in peak flow from timber harvest and road building from a set of three small basins (0.6 to 1 km^2) and three pairs of large basins (60 to 600 km^2) in the Oregon Cascades. In the small basins, they reported that changes in small peak flows were greater than changes in large flows. In their category of "large" peaks (recurrence interval greater than 0.4 years), flows were significantly increased in one of the two treated small basins, but the 10 *largest* flows were apparently unaffected by treatment. Jones and Grant [1996] reported that forest harvesting increased peak discharges by as much as 100% in the large basins over the past 50 years, but they did not discuss whether the largest peak flows in the large basins were significantly affected by land management activities. Two subsequent analyses of the same data used by Jones and Grant concluded that a relationship could not be found between forest harvesting and peak discharge in the large basins [Beschta et al., 1997; Thomas and Megahan, 1998].

There are several explanations why relationships between land management activities and a change in storm peaks have been difficult to document. First, the land management activity may actually have no effect on the size of storm peaks. Second, because major storms are infrequent, the range of observations may not adequately cover the range of interest. Third, if the variability in response is large relative to the magnitude of change, it may be difficult to detect an effect without a large number of observations. Fourth, land-use changes in a large watershed are often gradual, occurring over several years or decades. The use of an untreated control watershed whose flows are well-correlated with the treated watershed can greatly increase statistical power, if both watersheds are monitored for an adequate number of years before and after the treatment is applied. The variability about the relation between the two watersheds can be critical. For example, when the South Fork (pre-treatment RMSE = 0.232) was used as the control, no change in peak streamflow was detected at the North Fork Caspar Creek weir after about 50% of the 473-ha watershed had been clear-cut logged. However, when the uncut tributaries within the North Fork (pre-treatment RMSE = 0.118) were used as the controls, an increase in peaks was detected [Ziemer, 1998]. In the analyses described in this paper, uncut tributaries in the North Fork will be used as controls for treated subwatersheds in the North Fork.

Sediment Loads

Paired watershed studies have been utilized to study the effects of logging activities on sediment loads as well as peak flows. Detecting changes in sediment loads is even more difficult than for peak flows, because sediment loads are more variable and more costly to measure. Studies are often dominated by a single extreme event [Grant and Wolff, 1991; Rice et al., 1979; Olive and Rieger, 1991], making the results more difficult to interpret. Most studies have utilized annual sediment loads [Harris, 1977; Rice et al., 1979; O'Loughlin et al., 1980; Grant and Wolff, 1991; Megahan et al., 1995], usually determined by surveys of settling basins behind impoundments. Sediment passing over a spillway is typically determined using sediment rating curves that relate suspended sediment concentration and water discharge.

Only one of these studies has been conducted in the redwood region. Rice et al. [1979] reported the suspended sediment load was 270% above that predicted for 1 year following roading of the South Fork of Caspar Creek, and the debris basin deposit 50% above that predicted. Lewis [1998] estimated an increase of 212% in suspended load in the 6 years following logging of the South Fork, despite a 3300 m^3 landslide contributing directly to the stream in the control watershed.

In the Alsea watershed in coastal Oregon, Brown and Krygier [1971] found a doubling of sediment loads in the year after roading in two different watersheds. In the watershed that was completely clear-cut and burned to the mineral soil the next year, sediment loads increased more than 10-fold the first year, then gradually declined in 7 years to near pretreatment levels [Harris, 1977]. In the watershed that was 25% clear-cut in three small units and remained mostly unburned, the road effect diminished in the second year, and measured increases in loads were not statistically different from the pretreatment relationship. Differences between sediment yields from the two treated watersheds were attributed primarily to the burning.

Sample sizes are necessarily rather limited in analyses using annual loads, an unfortunate situation, considering the variability in response. It is rare to find studies with more than 5 years of pretreatment measurements of sediment on both control and treated watersheds. Exceptions are the experiments in the Alsea [Harris, 1977] and the Silver Creek [Megahan et al., 1995] watersheds, which had 7 and 11 years' pretreatment data, respectively. Many studies have used no pretreatment measurements at all [Plamondon, 1981; O'Loughlin et al., 1980; Leaf, 1970]. These must rely on unproven assumptions about the relation between control and treated watersheds. Post-treatment sample sizes are limited by the rapidly changing conditions that usually follow a disturbance. In analyses based on annual loads, conditions might return to pretreatment levels before enough data are available to demonstrate a change occurred. Even if a change can be detected, it is difficult to establish reasonable bounds on the magnitude of change in the face of such high variability and small sample sizes.

Some paired watershed studies have attempted to look at changes in sediment concentrations. In the Alsea watershed study, an analysis of changes in sediment rating curves was less effective than an analysis of annual loads [Brown and Krygier, 1971]. Such analyses will usually be limited by the inadequacy of models relating sediment concentration to flow. Olive and Rieger [1991] were unable to establish a useful calibration using sediment concentrations, attributing the failure to the highly variable hydrologic environment. Fredricksen [1963] used paired specimens (collected within 1 hour of each other) to analyze changes in the H.J. Andrews concentrations, but found it necessary to discard 8 of 83 data points that represented "unpredictable events" and "sudden movements of soil". Considering the episodic nature of sediment transport, it is not surprising that simul-taneous specimens from adjacent watersheds are poorly related. Such episodic events should probably be focused upon rather than discarded.

Utilizing storm sediment loads circumvents the problems of properly pairing concentration data and permits much larger sample sizes than are possible in analyses of annual loads. Larger sample sizes permit more powerful statistical analyses and construction of confidence limits and prediction limits for responses. Because of the cost of reliably estimating storm loads, studies based upon them are rare. Miller [1984] estimated storm loads from three control and three treated watersheds using pumped specimens triggered at regular time intervals. Although no pretreatment data were collected, the replication of both treatment and control permitted an analysis of variance on storm ranks each year following the treatment. But sampling at regular time intervals will tend to miss peak concentrations in flashy watersheds unless the intervals are very short, in which case more field and lab work is required. In our study we used schemes that increased the probability of sampling during high flows and turbidities.

Cumulative Effects

A great deal of concern has been focused on the cumulative watershed effects of forest harvesting activities. This study design includes multiple gaging stations in the same watershed in order to evaluate cumulative effects. According to the U.S. Council on Environmental Quality's interpretation of the National Environmental Policy Act, a "cumulative impact" is the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions, regardless of what agency…or person undertakes such other actions [CEQ guidelines, 40 CFR 1508.7, issued 23 April 1971]. An activity's importance may depend heavily upon the context of historic and future land use. An infinite variety of interactions is imaginable. We attempt to answer three questions that arise with regard to cumulative watershed effects of logging activities :

1. How are impacts related to the total amount of disturbance? In particular, were the effects of multiple disturbances additive in a given watershed?
2. How do impacts propagate downstream? In particular, were downstream changes greater than would be expected from the proportion of area disturbed?
3. Can activities that produce acceptable local impacts result in impacts that are unacceptable by the same standard at downstream locations? In particular, were sediment loads in the lower watershed elevated to higher levels than in the tributaries?

The scope of these questions is limited here in order to permit scientific investigation. For example, question (2) does not consider that larger watersheds may experience different types of impacts than contributing watersheds upstream, and question (3) does not consider that different standards may be appropriate downstream because different resources may be at risk. Nevertheless, partial answers to these questions can be provided with regards to storm peak flows, flow volumes, and suspended sediment loads through watershed experiments and mathematical modelling.

Environment and History

The Caspar Creek Experimental Watersheds are a pair of rain-dominated forested catchments in the Jackson Demonstration State Forest on the coast of northern California. The 473-ha North Fork and the 424-ha South Fork are both located in the headwaters of the 2,167-ha Caspar Creek, which discharges into the Pacific Ocean near the town of Caspar. Uplifted marine terraces, to 320 m in elevation, are deeply incised by antecedent drainages resulting in a topography composed of steep slopes near the stream channel and broad rounded ridgetops. About one third of the basin's slopes are less than 17° and only 7% are greater than 35°. The watershed receives an average of 1200 mm of rainfall each year, 90% falling in the months of October through April. The forest is composed mainly of redwood (*Sequoia sempervirens* [D.Don.] Endl.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), grand fir (*Abies grandis* [Dougl. ex D.Don] Lindl.), and western hemlock (*Tsuga heterophylla* [Raf.] Sarg.). The well-drained clay loam soils developed in sandstone and shale units of the Franciscan assemblage [Bailey et al., 1964] and are highly erodible.

Streamside landslides, gully erosion, and debris flows are the major erosional processes delivering sediment to the channel system. Soil pipes, common in the unchannelized swales, and steep ephemeral tributaries discharge to the Caspar Creek main stems. Based on debris basin surveys and suspended sediment measurements, the perennial, gravel-bed North Fork channel typically transports about 70% of its sediment load in suspension, and sand rarely exceeds 50% of the suspension. Gravel bars associated with woody debris jams and debris-induced bank erosion furnish the bulk of bedload transported during peak flows. Finer sediments cap the highest gravel bars and are stored in pools for transport during modest storm flows [Lisle and Napolitano, 1998].

Between 1860 and 1904, the old-growth forest in the Caspar Creek watersheds was clear-cut and burned. Log drives were triggered by opening the spillway gates of log crib along the main-stem reaches of both the North Fork and the South Fork, profoundly affecting channel morphology during the earliest logging effort [Napolitano, 1998]. These gave way to semi-mechanized yarding of tributary catchments using railway inclines (tramways) and steam donkeys [Henry, 1998]. A historic stage coach route and a mid-1900's era forest road totaling 11.4 km in length follow the watershed divide along the north and east of the North Fork.

In 1962, Caspar Creek became the site of a paired watershed experiment. In 1968, the South Fork watershed was roaded, and from 1971-1973, it was selectively logged by tractor, while the North Fork watershed was maintained as an undisturbed control [Rice et al., 1979; Ziemer, 1981; Wright et al., 1990; Keppeler and Ziemer, 1990]. In 1985 and 1986, 59 ha of an ungaged tributary basin in the lower North Fork was clear-cut. The present study of cumulative impacts began in 1985 in the 384-ha Arfstein subwatershed (ARF), gaged on the North Fork's main stem just above the confluence with the ungaged tributary (Figure 1). When the stability of ARF's discharge rating equation recently came into question, we decided to use the larger North Fork watershed (NFC) in place of ARF for the analysis of storm peaks and flow volumes. ARF was retained, however, for the sediment analyses because roughly 40% of the suspended sediment settles in a debris basin immediately above the North Fork weir and thus is not measured at the NFC gaging station.

METHODS

Treatment

The treatment design was based on compliance with the California Forest Practice Rules in effect in the late 1980's, except that the proportion of the watershed cut in a 3-year period was atypically high for a watershed of that size. Streams bearing fish or aquatic habitat were protected with selectively logged buffer zones 15 to 46 m in width, depending on stream classification and slope steepness.

Logging began in the headwaters of the North Fork in May 1989 and ended in the lower watershed in January 1992 (Figure 1). Clear-cuts totalled 169 ha (43% of ARF) in blocks of 9 to 60 ha and occupied 30% to 98% of treated subwatersheds. Total logged areas, including timber selectively removed from stream buffer zones, are slightly larger (Table 1). The 60 ha cutblock was composed of two adjacent subwatersheds (CAR and GIB), and an exemption was required from the maximum clear-cut size permitted under California Forest Practice Rules in effect at the time. Of the clear-cut areas, 81% was skyline yarded to landings on spur roads built on the upper hillslopes away from the creeks. Logs only had to be suspended at one point, but in most cases full suspension was achieved by setting the chokers near the middle of the log. This prevented ground dragging except near landings and convex slope breaks. The remaining 19% of the clear-cut area was tractor yarded and was limited to ridgetop areas where slopes were generally less than 20°. In addition, about 34% of the timber was selectively removed from 19 ha of stream buffer zones. New roads, landings, skid trails, and firelines occupied from 1.9% to 8.5% of treated subwatersheds. Four cut units, totalling 92 ha, were broadcast burned following harvest.

Three subwatersheds (HEN, IVE, and MUN) within the North Fork were retained in an unlogged condition for use as controls. In addition, the South Fork watershed, unlogged since 1973, was monitored for possible use as a control.

Gaging Stations

A total of 15 gaging stations were monitored: the North and South Fork weirs (NFC and SFC), four stations on the main stem of the North Fork, and nine on tributaries of the North Fork (Figure 1). The channel control structures at the North and South Fork gaging stations are 120° V-notch weirs with concrete upper rectangular sections. The lowest three main-stem stations (ARF, FLY, and LAN) are rectangular plywood sections, rated by discharge measurements. Each rated section has a natural bottom and a stable downstream sill installed to control bed elevation within the rated section. Discharge at the upper main-stem station (JOH) and the nine tributaries is measured with Parshall flumes. Although the rated sections and flume installations were not designed to guarantee complete capture of subsurface intergravel flows, frequent inspections (before, during, and after storms) were made and regular maintenance was performed at these sites to ensure stable discharge estimates throughout the length of the study. Discharge ratings were validated with new measurements each year, and only station ARF required rating equation changes.

Suspended Sediment Data Collection

Selection At List Time (1986-1995). Selection At List Time (SALT) is a variable probability sampling method similar to PPS (probability proportional to size) sampling with replacement [Hansen and Hurwitz, 1943]. Their estimation formulas are identical. Both methods utilize an auxiliary variable, easily measurable for the entire population, to assign inclusion probabilities to each sampling unit of the population. (We have defined a sampling unit of the sediment population as the suspended sediment load passing a gaged cross-section in 10 min.) The variance is minimized for auxiliary variables that are proportional to the variable of interest. PPS requires enumerating the population and measuring the auxiliary variable on the whole population before sampling. SALT was developed as an alternative to PPS for populations which cannot be enumerated before sampling [Norick, 1969]. SALT inclusion probabilities are computed from an estimate of the auxiliary variable total. Immediately upon measuring each unit's auxiliary variable, a decision is made whether or not to select the sampling unit. The auxiliary variable might be a flow-based prediction of unit yield from a sediment rating curve [Thomas, 1985]. This results in a sampling rate that is proportional to predicted sediment yield. If the discharge and sediment rating curves are power functions of stage (water depth), the sampling rate will also be a power function of stage. In practice, we had to set an upper limit to the sampling rate and modify the parameters of the power function in order to sample small storms as well as large ones [Thomas, 1989].

To implement SALT, at each gaging station we interfaced an HP-71 calculator with an automatic pumping sampler and a transducer mounted in a stilling well. The calculator was programmed to "wake up" every 10 min, read the transducer stage height, calculate the auxiliary variable, and, using the SALT algorithm and a set of stored random numbers, decide whether to sample or not. If the decision was to sample, a signal was sent via an interface circuit board to the pumping sampler, which would then collect a specimen (to avoid ambiguity, the word "sample" is reserved to refer to a selected set of "specimens" or "bottles") from a fixed intake nozzle positioned in the center of the channel. Date, time, stage, and other bookkeeping details were recorded on the calculator for subsequent uploading.

Turbidity-controlled sampling (1996). After 10 years of monitoring, the number of gaging sites was reduced to eight: the North and South Fork weirs (NFC and SFC), two controls (HEN and IVE), one main-stem station (ARF), and three tributary stations (CAR, DOL, and EAG). At that time, SALT and the HP-71's were replaced by a turbidity-controlled sampling system utilizing programmable data loggers and *in situ* turbidity probes. Date, time, stage, turbidity, and sampling information are recorded at 10-min intervals. The nephelometric turbidimeters we are using emit infrared light and measure the amount scattered back to the probe. In lab tests, they respond linearly to sediments of a given size distribution. In the field, with mixed-size sediments present, departures from linearity are usually minor. During each storm event, when certain prespecified turbidity thresholds are reached, the data logger sends a signal to the pumping sampler to collect a concentration specimen. A separate set of thresholds is specified for falling and rising stage conditions. This system reduced sample sizes and field expenses considerably, while still permitting accurate estimation of sediment loads [Lewis, 1996].

Data quality control. Field crews typically visited each gaging station one to three times per 24-hour period during storms to check on flumes and equipment, record manual stage observations, measure discharge at rated cross-sections, and collect depth-integrated suspended sediment specimens. Chart recorders provided back-up data. When problems were encountered with the electronic stage record, they were corrected using observer records or digitized data from back-up chart recorders. In a few instances, portions of discharge records were corrected based on correlation with selected alternate gaging stations. All stage data were coded to indicate the quality of the data.

Storms with poor quality or reconstructed peak data were treated as missing data in the peaks analysis. Storms with 25% or more of the flow volume derived from poor quality stage data were treated as missing data in the flow volumes analysis.

In addition to the suspended sediment specimens collected by the SALT algorithm, auxiliary pumped specimens were manually initiated for comparison with simultaneous depth-integrated DH-48 specimens or to augment the sampling algorithm. On occasions when the HP-71/pumping sampler interface failed and could not be immediately repaired, the sampler was set to collect specimens at fixed time intervals. A total of 21,880 bottles were collected: 19,572 under SALT, 378 under the turbidity threshold algorithm, 1048 auxiliary, 686 depth-integrated, and 196 fixed-time specimens.

Suspended sediment concentration was determined in the laboratory using vacuum filtration. Specimens were coded to indicate such conditions as spillage, organic matter content, low volumes, and weighing errors. Those with serious errors were omitted from the analysis. Those with minor errors were re-examined in the context of the whole storm.

Field crews also noted conditions affecting discharge or sediment data including landslides, windthrow, and culvert blockages and diversions. Post-storm surveys of the watershed stream channels and roads were made to document erosion sources potentially affecting sediment loads.

Storm Definition and Feature Identification

A total of 59 storm events occurred during the 11-year study. Storm events were generally included in the study when the peak discharge at SFC exceeded 0.0016 $m^3 s^{-1} ha^{-1}$ (recurrence interval about 7 times per year). A few smaller peaks were included in dry years. Multiple peak hydrographs were treated as multiple storms when more than 24 hours separated the peaks and the discharge dropped by at least 50% in the intervening period. When multiple peak hydrographs were treated as a single storm, the discharge for the peaks analysis was identified by selecting the feature corresponding to the highest peak at NFC. Thus the same feature was used at all stations, even if it were not the highest peak of the hydrograph at all stations. However, differences in peak discharge caused by this procedure were very small.

The start of a storm was chosen by seeking a point on the hydrograph, identifiable at all stations, where the discharge began to rise. The start times differed by no more than a few hours at the various stations. At the end of a storm, distinctive hydrograph features are more difficult to identify, unless a new start of rise is encountered. We therefore decided to use the same ending time for a given storm at all stations. The ending time was selected by observing the storm hydrograph for all stations and determining either the time of the next storm, the next significant rainfall, or a stable low-flow recession at all hydrographs, usually within about 3 days after the peak. The end of each storm was always well below the quickflow hydrograph separation point described by Hewlett and Hibbert [1967], except when the recession was interrupted by a new storm.

Dependent Variables

The response variables of interest in this study are storm runoff peak (instantaneous discharge), storm runoff volume (total discharge), and storm suspended sediment load (mass of particles greater than 1 micron in diameter). All are expressed on a unit area (per hectare) basis. The runoff variables were derived from the 10-min electronic record of stage and rating equations relating discharge to stage at each station. The computation of sediment loads is more involved and is described in the next section.

Computation of Suspended Sediment Loads

Correction to obtain cross-sectional average concentration. The pumping sampler intakes were oriented downstream and centered in the inclined throat sections of the Parshall Flumes. In the rated sections (ARF, FLY, and LAN), the intakes were similarly oriented at a fixed position about 9 cm off the bed. To determine whether the specimens were starved or enriched because of sampler efficiency or nozzle orientation or position, simultaneous ISCO and DH-48 depth-integrated (equal transit rate) specimens were collected throughout the study. A log-log regression of depth integrated concentration versus fixed intake concentration was developed for each station. Although only six of thirteen regressions differed significantly from the line $y=x$ (experimentwise α=0.05 with Bonferroni [Miller, 1981] adjustment), all fixed intake concentrations were adjusted using the back-transformed regression equations and corrected for bias [Baskerville, 1972] before storm loads were computed.

Load estimation in 1986-1995. Although sediment sampling followed SALT protocol in hydrologic years 1986-1995, we ultimately applied non-SALT methods of estimation to these samples for two reasons:

1. SALT does not provide a way to estimate sediment loads for periods when the sampling algorithm was inoperative due to equipment problems. Other methods can interpolate over such periods and utilize manually-initiated auxiliary specimens and those collected in fixed-time mode.
2. Using computer simulations on intensively collected storm data, other methods were found to have lower mean squared errors than SALT.

Although unbiased estimates of variance are not available for the alternate methodologies, the simulations strongly suggested that SALT variance estimates could be used as very conservative upper bounds on the variance. Two alternate methods were considered. In both of these methods the total load is computed by summing the products of water discharge and estimated concentration over all 10-min periods in the storm. The concentration, c, between adjacent sampled times t_1 and t_2 is modelled as either

1. a linear function of time: $c = c_1 + (t - t_1)(c_2 - c_1) / (t_2 - t_1)$, or
2. a power function of stage: $c = as^b$, where

$$b = \frac{\log c_2 - \log c_1}{\log s_2 - \log s_1}, \quad a = \frac{c_1}{s_1^b} \tag{1}$$

in which the subscripts identify concentrations and stages at times t_1 and t_2. These methods will be referred to as "time interpolation" and "stage interpolation" respectively. Stage interpolation has a better physical basis, but computational difficulties frequently arise when s_1 and s_2 are similar or equal, or when c_1 or c_2 is equal to zero. Therefore, time interpolation was substituted for stage interpolation when the power function defined by a pair of stages and sampled concentrations could not be computed or its exponent was not in the range between 1 and 10. If no specimens had been sampled within 10 hours prior to the start of the storm, the starting sediment concentration was assigned a value of zero and time interpolation was applied. An analogous procedure was followed for the end of the storm. The next section describes simulations leading to the decision that stage interpolation be used for estimating the sediment loads in 1986-1995.

Simulations comparing SALT and interpolation estimators. In addition to the usual SALT sampling, in 1994 and 1995 sediment concentration and turbidity at ARF was sampled at 10-min intervals for five storm events. This data, described in greater detail by Lewis [1996], provided realistic populations with known sediment loads that could be used in simulations to evaluate the performance of different load estimation methods. In addition to these five populations, eight storm populations were available from previous studies on the North Fork of the Mad River in northwestern California: three storms from December 1982, January 1983 and December 1983 [Thomas and Lewis, 1995] and five storms from February 1983 [Thomas and Lewis, 1993]. The Mad River concentrations were derived from turbidity charts and form a smoother, less realistic, time series than the ARF measurements.

In the simulations, 5000 independent SALT samples were selected from each storm event using SALT sampling parameters that were in use at ARF in 1995 and parameters thought to be optimal at Mad River. The sediment load was estimated for each of the 5000 samples using SALT and time and stage interpolation. The simulation results are strictly applicable only to comparing these estimators under a specific SALT sampling protocol.

The simulation results are summarized in Table 2. While SALT was unbiased as expected, it consistently has much higher root mean square error (RMSE) than the interpolated estimators. This can be attributed to the interpolation methods that take advantage of local trends in concentration that SALT ignores. Because the Mad River storm populations were smoother than those from ARF, they indicate a somewhat greater advantage for the interpolated estimators.

While time interpolation appears to have slightly less bias than stage interpolation, the differences in both bias and RMSE are small relative to the loads. Real data differ from these simulated data in that unexpected time gaps are created during unavoidable equipment malfunctions. Stage interpolation is expected to mimic true concentrations better than time interpolation over large time gaps, so the latter method (with the exceptions noted earlier) was chosen for this study during the SALT years (1986-1995).

Quality control for load estimates (1986-1995). Determining which calculated sediment load data were of high enough quality to include in the analysis was a subjective process and involved an examination of plots showing the storm hydrograph, sediment concentrations, and quality codes. The primary considerations were the number of known concentrations (sample size) and their temporal distribution relative to the hydrograph. Out of 51 storms and 15 stations (765 combinations), 74% of the load estimates were judged acceptable. Because sample sizes were in proportion to the size of storm events, most of the discarded loads were from small events. In those events that were retained, the median sample size was 20 and the median standard error from SALT was 14% of the estimated load. Based on the simulations (Table 2) and the fact that SALT estimates did not utilize all the available concentrations, it is likely that the median error from the interpolated estimates is well under 10% of the estimated load.

Load estimation in 1996. With turbidity-controlled sediment sampling in place in 1996, sediment loads were computed using “turbidity rating curves”. Concentration was predicted by linear regressions of concentration on turbidity fit to each storm. This method was shown in simulations [Lewis, 1996], based on the same five ARF populations as shown in Table 2, to produce load estimates with RMSE of 8% or better while sample sizes were reduced to between 4 and 11, depending on storm size and sampling parameters. The interpolation methods used for 1986-1995 would not be as accurate for the generally smaller sample sizes obtained under turbidity-controlled sampling. However, because of intermittent fouling of the turbidity probes with debris and sediment, valid turbidities were not always available. During such periods, if enough concentration measurements were available (and extras were often triggered by false high turbidities), then time or stage interpolation was used. As a last resort, a sediment rating curve derived from nearby data was used to estimate concentrations. Out of 8 stations and 8 storms in 1996, a total of 46 sediment load estimates were judged to be of acceptable quality. The median sample size was 5 from these events.

Derivation of Independent Variables Used in the Analysis

The complete data set included both map-derived and field-derived variables. All disturbance variables were coded as proportions of watershed area. The basic watershed descriptors and variables that were useful in the analyses are shown in Table 1.

Topographic contours and streams were digitized from U.S. Geological Survey 7.5 min quadrangle maps. The mapped stream channels in harvest units were then extended to include all channels showing field evidence of annual scour and/or sediment transport before logging. Watershed boundaries were field-mapped using conventional tape-and-compass surveys, respecting diversions of surface runoff where road drainage structures directed flow into or out of the topographic watersheds. During road maintenance, efforts were made to limit changes in drainage due to ruts and berms. Harvest unit boundaries and roads were surveyed using differentially corrected GPS. All these lines were transferred to GIS coverages from which geographic variables were extracted. Burned areas, stratified into two severity classes, and herbicided areas were transferred to the GIS from field maps. For each variable measured, the area within 150 feet of a stream channel, and the length of channel within the affected area were extracted from the GIS.

The areal extent of ground disturbance from roads, landings, skid trails, firelines, and corridors created by dragging logs up the slope by cable were each determined from exhaustive field transects. The areas within 150 feet of a stream channel, and the number of stream crossings were also recorded for these variables.

Cutting age was calculated as the difference in hydrologic year of a given storm and the hydrologic year an area was logged. For watersheds with areas cut at different times, a weighted average cutting age was calculated using the cut unit areas as weights.

An antecedent wetness index intended to reflect seasonal differences in hydrograph response was derived using mean daily discharges from SFC. The daily discharges were accumulated and decayed using a 30-day half-life, i.e.

$$w_i = Aw_{i-1} + q_i \qquad (2)$$

where w_i denotes the wetness index on day i, and q_i denotes the daily mean discharge at SFC on day i and the constant $A = 0.97716$ satisfies the relation A^{30}=0.5. The decision to use streamflow rather than precipitation to calculate antecedent conditions was based on the assumption that the history of the streamflow response would be a better predictor of streamflow than would the history of rainfall. The response of streamflow to precipitation is delayed as soil moisture deficit is recharged. A half-life of 30 days was selected to smooth the high frequency variation in streamflow, creating an index that would decline significantly only after lengthy dry periods. No optimization was done on the half-life, but it was found that $\log(w_i)$ made a slightly better predictor. The wetness index time series over the 11-year study period is displayed in Figure 2, with solid circles indicating the wetness level at the start of each storm. The wetness index varied from 13 to 150 at the onset of storms occurring in November and December, but assumed the full range from 13 to 562 at the onset of storms occurring in January, February and March. For two storms that occurred in May, the values of the index were 49 and 84.

Statistical Methods

Initially, simple log-log linear regressions were computed for each dependent watershed against selected control watersheds prior to treatment. The Chow test [Chow, 1960; Wilson, 1978] was used to test whether the post-treatment data differed in either intercept or slope from the pre-treatment regressions. Following Bonferroni's procedure [Miller, 1981] for these tests, an experimentwise error rate of 0.05 for 10 tests required setting the nominal α to 0.005 for each test. Because of their limited sample sizes, these tests, while easy to interpret, are not as powerful as models based on all of the data.

Models incorporating all of the watersheds were initially built up in a stepwise fashion using least squares estimation. At each step, residuals were plotted against candidate predictors to select the next variable and the appropriate transformation or form of interaction. Because a non-standard covariance model was employed, models were ultimately fitted using maximum likelihood estimation and selected using a combination of exploratory and diagnostic techniques.

Models for runoff (storm peaks and flow volumes). Consider the following pretreatment model:

$$\log(y_{ij}) = \beta_{0i} + \beta_{1i}\log(y_{Cj}) + \varepsilon_{ij} \tag{3}$$

where
y_{ij} = unit area response (peak or flow) at treated watershed i, storm j,
y_{Cj} = unit area response at control watershed in storm j,
ε_{ij} = non-independent normally distributed errors (see *Covariance Models* below),
and β_{0i} and β_{1i} are "location" parameters to be estimated for each watershed *i*. The log transformations are used in order that ε_{ij} appear to be normally distributed. The pretreatment model can be considered as a special case of the following model:

$$\begin{aligned}\log(y_{ij}) &= \left(\beta_{0i} + \beta_4 D_{ij} + \beta_6 D_{ij}\log(w_j) + \beta_7 D_{ij} a_i\right) \\ &\quad + \left(\beta_{1i} + \beta_5 D_{ij}\right)\log(y_{Cj}) + \varepsilon_{ij}\end{aligned} \tag{4}$$

where
D_{ij} = some measure of disturbance per unit area in watershed *i* at storm *j,*
w_j = wetness index at start of storm *j*,
a_i = drainage area of watershed *i,*
and β_4, β_5, β_6, and β_7 are parameters to be estimated. The log transformation of w_j is not critical, but was found to improve its explanatory value. Wetness enters the equation only as an interaction with D_{ij} because in the absence of disturbance wetness did not affect the relation between y_{ij} and y_{Cj}. As an interaction, it implies that the effect of disturbance on y_{ij} varies linearly with antecedent wetness. The $D_{ij}a_i$ term implies that the disturbance effect also varies linearly with watershed area and it is the key term in this model for detecting a cumulative effect. It describes how watershed impacts propagate downstream and we use it to test the null hypothesis that a unit area disturbance has the same unit area effect in watersheds of all sizes.

The first line of equation (4) permits the intercept of the relation between y_{ij} and y_{Cj} to change following disturbance. The second line, via the $D_{ij}\log(y_{Cj})$ term, permits the slope of that relation to change following disturbance. Equation (4) can be rearranged as

$$\begin{aligned}\log(y_{ij}) &= \beta_{0i} + \beta_{1i}\log(y_{Cj}) + \varepsilon_{ij} \\ &\quad + D_{ij}\left[\beta_4 + \beta_5\log(y_{Cj}) + \beta_6\log(w_j) + \beta_7 a_i\right]\end{aligned} \tag{5}$$

We now model the disturbance term using logged area and cutting age to represent loss of transpiration and interception following logging. Compacted areas such as roads, landings, skid trails, and firelines were not found to be useful predictors. Since relatively little transpiration occurs at Caspar Creek in the fall and winter, we treat areas logged in the fall or winter prior to the occurrence of a storm as special cases. Let

$$D_{ij} = f(t_{ij})(c_{ij}) + g(c'_{ij}) \tag{6}$$

where
t_{ij} = area-weighted mean cutting age (number of summers passed) in watershed *i* for areas logged in water years (defined as Aug.1 - July 31) preceding that of storm *j*
c_{ij} = proportion of watershed *i* logged in water years prior to that of storm *j*, and
c'_{ij} = proportion of watershed *i* logged prior to storm *j* but in the same water year

We model a linear recovery declining from a maximum of unity the year after cutting:

$$f(t_{ij}) = 1 - \beta_2(t_{ij} - 1) \tag{7}$$

where β_2 is a parameter representing the recovery rate, and we assume the effect of newly cut areas depends only on the season they were cut:

$$g(c'_{ij}) = \beta_3^{(k)} c'_{ij} \tag{8}$$

where $\beta_3^{(k)}$ are parameters for the effect of cutting in the fall (k=1) and winter (k=2) immediately preceding storm j. Equation (6) becomes

$$D_{ij} = \left(1 - \beta_2(t_{ij} - 1)\right)c_{ij} + \beta_3^{(k)} c'_{ij} \tag{9}$$

and the complete model is

$$\begin{aligned} \log(y_{ij}) &= \beta_{0i} + \beta_{1i}\log(y_{Cj}) + \varepsilon_{ij} \\ &\quad + \left[\left(1 - \beta_2(t_{ij} - 1)\right)c_{ij} + \beta_3^{(k)} c'_{ij}\right] \\ &\quad \times \left[\beta_4 + \beta_5 \log(y_{Cj}) + \beta_6 \log(w_j) + \beta_7 a_i\right] \end{aligned} \tag{10}$$

To investigate whether unit area response increases downstream independently of disturbance, we can look for a relation between β_{0i} and a_i. Alternatively, we can replace β_{0i} with the linear expression $\beta_0^{(1)} + \beta_0^{(2)} a_i$ and test the hypothesis $H_0{:}\beta_0^{(2)} = 0$. If unit area responses tend to increase downstream, then cumulative impacts might occur where a response threshold of acceptability is exceeded only below some point in the stream network, even though unit area disturbance is no greater in that point's watershed than in watersheds further upstream.

Model (10) is not a linear model because it involves products of the parameters to be estimated. The non-linearity was introduced as a parsimonious way of modelling recovery with time since logging. It avoids introducing separate recovery parameters for each of the terms in equation (4) that involve D_{ij}.

Models for suspended sediment loads. Suspended sediment load from an untreated control watershed was found to be a much better predictor of sediment load at treated watersheds than water discharge at either location. However, the change in storm flow in the treated watershed, relative to that in the control, was found to be the next best predictor in a model for suspended sediment loads. The change in flow, Δq, was formulated two ways:

1. The residual from the flow model with D_{ij} set to zero

$$\Delta q_{ij}^{(1)} = \log(y_{ij}) - \left(b_{0i} + b_{1i}\log(y_{Cj})\right) \tag{11}$$

   where b_{0i} and b_{1i} are estimates of the flow model parameters β_{0i} and β_{1i}.

2. The log of the ratio of the flows between the treated and control watersheds:

$$\Delta q_{ij}^{(2)} = \log\left(y_{ij} / y_{Cj}\right) = \log\left(y_{ij}\right) - \log\left(y_{Cj}\right) \tag{12}$$

The first form makes better sense hydrologically, but treating it as an independent variable may not be statistically legitimate later when estimating precision later on, because it involves parameter estimates from another model. Nevertheless, both forms of Δq were considered. These variables are not useful in a predictive setting because the flows are not known in advance, but the main purpose of these models is explanatory. If prediction is needed, then a third form might be substituted as an approximation to $\Delta q_{ij}^{(1)}$:

3. The predicted change in $\log(y_{ij})$ from equation (10):

$$\Delta q_{ij}^{(3)} = \left[\left(1 - b_2(t_{ij} - 1)\right)c_{ij} + b_3^{(k)} c'_{ij}\right] \times \left[b_4 + b_5 \log(y_{Cj}) + b_6 \log(w_j) + b_7 a_i\right] \quad (13)$$

where the b's are estimates of the β's in equation (10).

After Δq and one or two disturbance variables were included in the model, no further gains were realized in the sediment models by including factors such as antecedent wetness and cutting age. So, unlike the runoff models, the sediment models remain linear in their parameters:

$$\begin{aligned} \log(y_{ij}) = \beta_{0i} + \beta_{1i} \mathrm{bg}(y_{Cj}) + \beta_2 \Delta q_{ij} \\ + \left(\beta_3 + \beta_4 \mathrm{bg}(y_{Cj}) + \beta_5 a_i\right) x_{ij}^{(1)} \\ + \left(\beta_6 + \beta_7 \mathrm{bg}(y_{Cj}) + \beta_8 a_i\right) x_{ij}^{(2)} + \varepsilon_{ij} \end{aligned} \quad (14)$$

where

y_{ij} = unit area sediment load at treated watershed i, storm j,
y_{Cj} = unit area sediment load at control watershed in storm j,
Δq_{ij} = change in flow as defined by (11) or (12) in watershed i, storm j,
a_i = drainage area of watershed i,
$x_{ij}^{(1)}$ = a measure of unit area disturbance in watershed i, storm j,
$x_{ij}^{(2)}$ = a second measure of unit area disturbance in watershed i, storm j,
ε_{ij} = non-independent normally distributed errors (see *Covariance Models* below),

and the β's are parameters to be estimated. The logic behind the interaction terms involving $\log(y_{Cj})\, x_{ij}^{(k)}$ and $a_i x_{ij}^{(k)}$ is the same as in the runoff models. And, as with model (10), we can replace β_{0i} in (14) with the expression $\beta_0^{(1)} + \beta_0^{(2)} a_i$ to investigate whether unit area loads increase downstream independently of disturbance.

Covariance models. The residual covariance was found to depend upon watershed size and location. The correlations decreased with increasing distance between watershed centroids and the variance decreased with increasing watershed size. Serial autocorrelation in the residuals for most watersheds was weak or absent, so responses from different storms were considered independent. The errors were thus assumed to follow a multivariate normal distribution with a covariance matrix for each storm. The dimensions of this square matrix are equal to the number of treated watersheds having good data in that storm. The covariances in the matrix for storm j are modelled as:

$$Cov(\varepsilon_{i_1 j}, \varepsilon_{i_2 j}) = \sigma_{i_1 i_2}^2 = \rho_{i_1 i_2} \sigma_{i_1} \sigma_{i_2} \quad (15)$$

where

$\rho_{i_1 i_2}$ = the correlation between $\varepsilon_{i_1 j}$ and $\varepsilon_{i_2 j}$,
σ_{i_1} and σ_{i_2} = the standard deviations of $\varepsilon_{i_1 j}$ and $\varepsilon_{i_2 j}$
$\varepsilon_{i_1 j}$ and $\varepsilon_{i_2 j}$ = errors for watersheds i_1 and i_2 in storm j

Subscripts j have been omitted from $\rho_{i_1 i_2}$, σ_{i_1} and σ_{i_2} because these terms are assumed to be independent of storm number and are, in fact, modelled upon the errors from all storms. Two models for the correlation $\rho_{i_1 i_2}$ were found to fit the runoff and sediment data.

1. Exponential decline with distance:

$$\rho_{i_1 i_2} = \frac{\exp(-\theta_1 d_{i_1 i_2}) + \theta_2}{1 + \theta_2} \tag{16}$$

where $d_{i_1 i_2}$ is the distance separating watersheds i_1 and i_2, and θ_1 and θ_2 are parameters to be estimated. In this model the correlations decline asymptotically from unity to the value $\theta_2/(1+\theta_2)$.

2. Linear decline with distance:

$$\rho_{i_1 i_2} = \begin{cases} 1, & d_{i_1 i_2} = 0 \\ \theta_1 - \theta_2 d_{i_1 i_2}, & d_{i_1 i_2} > 0 \end{cases} \tag{17}$$

The standard deviations σ_i were modelled as a declining power function of watershed area:

$$\sigma_i = \theta_3 a_i^{-\theta_4} \tag{18}$$

where θ_3 and θ_4 are parameters to be estimated. All peaks models discussed in this paper (other than the least squares fits) employed equations (15), (16), and (18). The flow and sediment models employed equations (15), (17), and (18)

Method of estimation. The parameters of the model were estimated using the method of maximum likelihood [Mood et al., 1974]. The likelihood function is assumed to be the multivariate normal density of the ε_{ij} treated as a function of the β and θ parameters. In practice we minimize the negative of the log likelihood. In this problem, the log-likelihood is equal to the sum of the independent storm-wise log-likelihoods. Thus, the dimension of the multivariate density function is the number of watersheds represented in a given storm, a maximum of 10. The log-likelihood functions and their gradients (derivative vectors) are shown in APPENDIX B. They were programmed in S-Plus [Statistical Sciences, 1995] and FORTRAN, and solved using the S-Plus function *nlminb* (nonlinear minimization subject to bound-constrained parameters). Least squares estimates of the parameters were used as starting guesses in these iterative numeric calculations.

Model size. The inclusion of up to 31 parameters in these models raises questions about overfitting. These questions were addressed by cross-validation (discussed below) after a model was selected, but the proper model size was selected with the objective of minimizing a variant of Akaike's information criterion [Burnham and Anderson, 1998],

$$\mathrm{AIC}_c = -2\log(L) + 2K\left(\frac{n}{n-K-1}\right) \tag{19}$$

where L is the maximum likelihood, K is the number of parameters estimated, and n is the sample size. This criterion is recommended over the unmodified AIC when the ratio n/K is small (less than about 40). The inclusion of the 20 location parameters β_{0i} and β_{1i} is strongly supported by AIC$_c$. Its value increased by 14 to 88 units in the various models when one or two parameters were substituted for either β_{0i} or β_{1i}. Increases of 10 or more AIC units indicate clearly inferior models [Burnham and Anderson, 1998]. Because of the computational time required to fit each model, it was impractical to obtain the likelihoods of all alternative models. For that reason, parameters other than β_{0i} and β_{1i} were evaluated using hypothesis tests based on the normal distribution, and AIC$_c$ was computed only for the more promising candidate models.

Hypothesis testing. Maximum likelihood parameter estimates are approximately multivariate-normally distributed for large samples [Rao, 1973]. The estimated covariance matrix of the estimates was obtained by inverting the observed information matrix, using a finite difference approximation to the Hessian, or matrix of second derivatives of the log-likelihood function [Bishop et al., 1975; McCullagh and Nelder, 1989]. (The observed information matrix is the negative of the Hessian, evaluated at the maximum likelihood estimates.) The standard errors, s_b, of the estimated parameters are the square roots of the diagonal of the covariance matrix. Since the parameter estimates are asymptotically normal, a simple test of the hypothesis H_0: $\beta_i = c$ is provided by observing whether or not the statistic $(b_i - c) / s_b$ is in the rejection zone of the standard normal distribution. The p-values from these hypothesis tests are identified as p_N in this paper. Tests with $p_N < 0.01$ are considered significant in this paper. Tests with $0.01 < p_N < 0.05$ are considered "suggestive" but not conclusive.

Observed change in response. "Observed change" in response was calculated by comparing the observed response, y_{ij}, with an estimate of the expected response, $E(y'_{ij})$, from the same storm and watershed in an undisturbed condition. We define the percentage change in response as

$$p_{ij} = 100\left(\frac{y_{ij} - E(y'_{ij})}{E(y'_{ij})}\right) = 100\left(\frac{y_{ij}}{E(y'_{ij})} - 1\right) \tag{20}$$

The expected undisturbed response, $E(y'_{ij})$, is a function of $E\left(\log(y'_{ij})\right)$:

$$E(y'_{ij}) = \exp\left[E\left(\log(y'_{ij})\right) + \tfrac{1}{2}\sigma_i^2\right] \tag{21}$$

Setting disturbance to zero in either model (10) or (14) above, we have $E\left(\log(y'_{ij})\right) = \beta_{0i} + \beta_{1i}\log(y_{Cj})$. The variances σ_i^2 are a function of θ_3 and θ_4 given by model (18). A nearly unbiased estimator of $E(y'_{ij})$ is given by

$$\hat{y}'_{ij} = \exp\left[b_{0i} + b_{1i}\log(y_{Cj}) + \tfrac{1}{2}\left(\hat{\theta}_3 a_i^{\hat{\theta}_4}\right)^2\right] \tag{22}$$

where b_{0i}, b_{1i}, $\hat{\theta}_3$, and $\hat{\theta}_4$ are the maximum likelihood estimates of β_{0i}, β_{1i}, θ_3, and θ_4, respectively. The term $\tfrac{1}{2}\hat{\sigma}_i^2 = \tfrac{1}{2}\left(\hat{\theta}_3 a_i^{\hat{\theta}_4}\right)^2$ is often called the Baskerville [1972] bias correction. An approximation for p_{ij} that we will call the "observed change in response" is obtained by substituting $\hat{y}'_{ij}$ for $E(y'_{ij})$ in (20):

$$\tilde{p}_{ij} = 100\left(\frac{y_{ij}}{\hat{y}'_{ij}} - 1\right). \tag{23}$$

Of course we are not just interested in the changes in response for the particular values of the explanatory variables encountered during the study. We would like to study the percentage change, p_0, for an arbitrary vector, $\mathbf{x}_0$, of explanatory variables. An unbiased estimator and confidence interval for $E(p_0)$ as well as a prediction interval for p_0 are derived in APPENDIX C. The confidence interval represents the uncertainty of the mean, $E(p_0)$, given $\mathbf{x}_0$. The prediction interval indicates the variability in the individual response p_0, given $\mathbf{x}_0$. Prediction intervals are wider than confidence intervals because they include the variability in the response about its mean value as well as the variability due to uncertainty in the mean itself.

Cross-validation of models. To investigate the possibility that the models were overfitted to the data, ten-fold cross-validation was used [Efron and Tibshirani, 1993]. The data are split into ten groups. Each observation is predicted from a model based on all of the data except that group to which the observation belongs. The RMSE of these predictions is called the cross-validation prediction error and it may be compared with the RMSE of the models fitted with all the data to assess overfitting.

A regression of the observed responses on the fitted values, known as the *calibration*, should have an intercept near zero and slope near unity. The regression of the observed responses on the cross-validated predictions is expected, in general, to have a slope less than one [Copas, 1983]. This phenomenon, known as *shrinkage*, implies that predictions of high or low response values tend to be too extreme. The degree of departure of the calibration slope from unity provides another measure of overfitting.

Because the data were not independent, the cross-validation was repeated using two different methods for splitting the data:

1. Data were randomly divided into groups of equal size.
2. Post-treatment data were omitted systematically, one station at a time.

The latter method does not provide cross-validated predictions for the pre-treatment data, but if all the data from a station, say watershed *i*, are omitted, it becomes impossible to estimate β_{0i} and β_{1i}, which are required to make predictions for that watershed. Nevertheless, the one-station-at-a-time method is probably a more rigorous validation for the inclusion of alternative disturbance variables because it will give higher error rates for models that include variables correlated with the response due to just one or two watersheds.

RESULTS

Storm Peaks

The analysis included 226 pre-treatment and 300 post-treatment observations representing 59 storms on the 10 treated watersheds. For the 226 pretreatment peaks, the control watersheds correlating best with watersheds to be treated were tributaries HEN and IVE, and MUN (Figure 3). The mean of the peaks at HEN and IVE (designated HI), or at HEN, IVE, and MUN (designated HIM), had higher correlations than did peaks from either HEN, IVE, or MUN individually. Because MUN was not monitored the last year of the study, HI was chosen as the control for the peaks analysis.

The Chow tests [Chow, 1960; Wilson, 1978], based on the HI control, revealed strong evidence that post-treatment data differed from pre-treatment regressions. Eight of the 10 watersheds departed ($p < 0.005$) from these regressions after logging commenced. The other two, FLY and LAN on the main stem, had p-values less than 0.05. Departures from the pre-logging regression were greatest in the clear-cut tributaries: BAN, CAR, EAG, GIB, and KJE (Figure 4).

Seasonal patterns in the departures from the pre-treatment regressions were evident in most of the treated watersheds. For example, Figure 5 shows the post-logging departures for watershed EAG plotted against storm number. The largest percentage departures occurred early in the season. These were usually, but not always, relatively small storms. Storms 28 and 29 did not show treatment effects, apparently because logging had just taken place the same winter, so insufficient time had elapsed for soil moisture differences to develop between the controls and the logged area. This exemplifies the situation that necessitated modelling of the disturbance term using equation (9).

To develop an overall model, an intercept and slope for each watershed (equation (3)) was initially fit by least squares. The residuals from this model show a strong interaction between proportion of area logged and antecedent wetness (Figure 6). Area logged includes clear-cut areas and a portion of each buffer zone corresponding to the proportion of timber removed (Table 1). The relation of the residuals with area logged is linear, the slope decreasing from strongly positive with increasing wetness (Figure 6, top row). The relation with log(wetness) is linear, the slope becoming strongly negative with increasing logged area (Figure 6, bottom row). These relations imply a product term is an appropriate expression of the interaction, and the coefficient is expected to be negative. The fact that the average residual increases with different categories of area logged but not with wetness shows that a solo logged area term is needed in the model as well as the interaction product, but a solo wetness term is not. No variables related to roads, skid trails, landings, firelines, burning or herbicide application were found to improve the fit of the linear least squares model that includes logged area and its interaction with wetness. Adding logged area and the wetness interaction to the model, a plot of post-treatment residuals against time after logging (Figure 7) indicates an approximately linear recovery trend in the first 7 years.

When model (10) was fit to the data, the coefficient b_7 on the cumulative effect term did not differ significantly from zero (Table 3, p_N= 0.21). The coefficient b_5 was negative but not highly significant (p_N=0.047), weakly suggesting that the effect of logged area on peak flows tends to diminish in larger storms. The coefficient b_4 on logged area was positive as expected and its interaction with wetness, b_6, was negative as expected. The recovery coefficient, b_2, indicates an average recovery rate of about 8% per year. The null hypothesis for each of the parameters $\beta_3^{(k)}$ is $H_0{:}\beta_3^{(k)} = 1$, because the recovery model assumes a value of unity the year after logging. The coefficient $b_3^{(1)} = 0.59$ (standard error 0.10) indicates a reduced effect from fall logging on peaks in the following winter and $b_3^{(2)} = 0.00$ suggests that the effects of winter logging on peak flow are delayed until a growing season has passed.

There was no indication of a dependency on watershed area in either the coefficients b_{0i} or b_{1i} from model (10). When we replaced β_{0i} in model (10) with the expression $\beta_0^{(1)} + \beta_0^{(2)} a_i$, the coefficient $b_0^{(2)}$ was not significantly different from zero (p_N=0.58), indicating no trend of unit area storm peak with watershed area.

The exponentially declining correlation model (18) was used when solving model (10) for peak flows (with β_7 fixed at zero), and it can be seen to be a reasonable fit (Figure 8). The variance model (18) also seems reasonable (Figure 9). The Box-Pierce test [Shumway, 1988] did not indicate the presence of serial autocorrelation at any of the stations (minimum p=0.089). The residuals conform very well to the normal distribution (Figure 10), as do plots for individual stations (not shown), validating our choice of likelihood function. The lone outlier is from a storm at GIB that produced 2 peaks at all stations except GIB. (The first peak was selected for the storm but was identifiable only as a shoulder of the hydrograph at GIB.) The model fits the data very well (Figure 11). For the regression between observed and fitted values, $r^2 = 0.946$. This compares with $r^2 = 0.848$ for a model with no disturbance variables and $r^2 = 0.937$ for model (3) fit to only the pre-treatment data, so the model fits the post-treatment data as well as the pre-treatment data.

Magnitude of observed changes. Maximum peak flow increases based on equations (22) and (23) were about 300%, but most were less than 100% (Figure 12). The mean percentage increase declined with wetness but was still positive even under the wettest conditions of the study ($w_i > 500$), when it was 23% for clear-cuts but only 3% in partially cut watersheds. Increases more than 100% generally only occurred in clear-cuts under relatively dry conditions ($w_i < 50$) and when peaks in the control were less than 0.0025 $m^3s^{-1}ha^{-1}$ (return period 3-4 times per year). Large increases occurred less frequently as the winters progressed, but increases over 100% did occur in January and February. The mean percentage increase in peak flow declined with storm size and then levelled at an average increase of 35% in clear-cuts and 16% in partially cut watersheds for peaks greater than 0.004 $m^3s^{-1}ha^{-1}$ (return periods longer than 0.5 years) (Figure 13). For a storm size having a 2-year return period, the average peak-flow increase in 100% clear-cuts was 27% [Ziemer, 1998].

Figure 14 shows 95% confidence intervals for the modelled mean response in a 20-ha watershed that has been 50% clear-cut, for two wetness conditions and two cutting ages within the range of our data. The effect of antecedent wetness is a greater influence on the response than time since cutting, although the recovery data only span 7 years. Prediction intervals are much wider than confidence intervals, revealing post-treatment variability that is greater than the treatment effect itself.

Storm Runoff Volume

The analysis included 527 observations representing 59 storms. For the same reasons as in the peaks analysis, HI (the mean of HEN and IVE) was chosen as the control. The modeling results are similar to the peaks analysis results, except that the watershed area interaction b_7 was marginally significant (Table 4, p_N=0.012) and watershed correlations were found to decline linearly with distance, so model (17) was used instead of (16) in the covariance model. For the sake of brevity, the modeling results for storm runoff volume are omitted, and we report only the coefficients (Table 4) and the magnitude of observed changes.

Magnitude of observed changes. The maximum storm runoff volume increase from equations (22) and (23) was 400%, but most were less than 100%. The mean percentage increase declined with wetness but was still positive even under the wettest conditions of the study ($w_i > 500$), when it was 27% for clear-cuts and 16% in partially cut watersheds. Increases more than 100% generally only occurred in clear-cuts under relatively dry conditions ($w_i < 100$) and when runoff volume in the control was less than 250 m^3ha^{-1}. Large increases occurred less frequently as the winters progressed, but in-creases over 100% did occur in January and February. The mean percentage increase in storm runoff volume declined with storm size and then leveled at an average increase of 30% in clear-cuts and 13% in partially cut watersheds for storm runoff greater than 250 m^3ha^{-1}.

Annual storm runoff volume (sum of storms) increased an average of 58% (1119 m^3ha^{-1}) in clear-cut watersheds and 23% (415 m^3ha^{-1}) in partly clear-cut watersheds (Table 5). Based on the complete discharge record at NFC, the runoff volume for the storms included in this analysis represents 41 to 49% of the total annual runoff volume in individual tributaries.

Figure 15 shows confidence intervals and prediction intervals for storm runoff volume in a 20-ha watershed that has been 50% clear-cut, under two wetness conditions and two cutting ages within the range of our data.

Suspended Sediment Loads

The relatively large number of missing observations resulting from quality control screening complicated the selection of controls for the sediment analysis. The use of synthetic controls such as HI and HIM permitted larger sample sizes because these means could be computed from any combination of non-missing controls. Thus the sample size was 376 with the HIM control, but only 333 with the HI control, and less than 300 with HEN or IVE alone. Although HIM control permitted the largest sample size, its correlations tended to be lower than those of HI (Figure 16). We therefore present the analysis twice, once with the HIM control and once with the HI control.

Chow tests [Chow, 1960; Wilson, 1978] for treatment effects at individual stations gave mixed results (Table 6). Only 2 of the tests were significant when HIM was used as the control and 3 were significant with the HI control. The tributaries all had more significant changes than the main-stem stations. Figure 17(top row) indicates that suspended sediment loads increased in all the clear-cut tributaries except KJE, where loads appear to have decreased after logging. The only partly clear-cut watershed on a tributary (DOL) also showed highly significant increases in sediment loads. The upper main-stem stations (JOH and LAN) showed no effect after logging, and the lower main-stem stations (FLY and ARF) experienced increases only in smaller storms. Summing suspended sediment over *all* storms, the four main-stem stations all showed little or no change (Table 7). Sediment loads at the North Fork weir, below ARF, increased by about 89%

per year, mainly as a result of a large landslide in the ungaged subwatershed that enters between ARF and NFC.

Models with HI control. The analysis included 333 observations representing 43 storms. In these models (14), the change in storm flow volume $\Delta q_{ij}^{(1)}$ was found to be the best explanatory variable after sediment load from the HI control, y_{HI}. Figure 18 shows the relation between the post-treatment sediment departures from pretreatment model (3)) and $\Delta q_{ij}^{(1)}$. Since both variables are differences in logarithms, it is convenient to express them as ratios of observed to predicted response, obtained by exponentiating the differences. The linear correlation between the sediment and flow departures is 0.54.

After $\Delta q_{ij}^{(1)}$ is in the model, disturbance variables explain only a very small part of the remaining variation (Figure 19). The length of unbuffered stream channel in clear-cut areas was one of the more useful disturbance variables in the sediment models. Under California Forest Practice Rules in effect during the North Fork logging, vegetation buffers were not required for stream channels that do not include aquatic habitat. The best models were found when this variable was separated into channels in burned clear-cuts and channels in unburned clear-cuts. The variable did not need to be separated, however, in the interaction terms. Thus the model (14) was modified to:

$$\begin{aligned}\log(y_{ij}) = {} & \beta_{0i} + \beta_{1i}\log(y_{(\mathrm{HI})j}) + \beta_2\Delta q_{ij}^{(1)} \\ & + \beta_3 x_{ij}^{(1)} + \beta_4 x_{ij}^{(2)} + \beta_5(x_{ij}^{(1)} + x_{ij}^{(2)})\log(y_{(\mathrm{HI})j}) \\ & + \beta_6(x_{ij}^{(1)} + x_{ij}^{(2)})a_i + \varepsilon_{ij}\end{aligned} \tag{24}$$

where

$x_{ij}^{(1)}$ = length of stream channel in burned clear-cuts, and

$x_{ij}^{(2)}$ = length of stream channel in unburned clear-cuts

To indicate the relative contribution of the various terms in model (24), the increase in residual sum of squares is shown for least squares models after dropping each explanatory variable (Table 8).

The maximum likelihood estimates for model (24) are shown in Table 9. The coefficient estimate b_3 is about 1.8 times b_4, suggesting that streams in burned clear-cuts contribute more sediment than those in unburned clear-cuts. The estimate, b_5, of the storm size interaction is negative, suggesting that the ratio between post-treatment and pretreatment sediment loads diminishes for larger events. The estimate, b_6, of the cumulative effect coefficient in this model was negative and was found marginally significant ($p_N = 0.044$). This interaction in the sediment model only partly offsets the small positive interaction that was noted in the runoff model and is hidden in the term $\Delta q_{ij}^{(1)}$. Other variables being equal, the model still predicts larger unit area sediment loads from larger watersheds (Figure 20). Because of its marginal significance, the β_6 term was dropped from the model for the remainder of this section.

The fitted intercepts b_{0i} from model (24), with β_6 fixed at zero, tend to increase with watershed area (Figure 21), with the exceptions of KJE (K) and JOH (J). This pattern in the intercepts is confirmed by substituting $\beta_0^{(1)} + \beta_0^{(2)}a_i$ for the term β_{0i}. The fitted coefficient $b_0^{(2)}$ is positive and differs significantly from zero (p_N=0.0031). The slope coefficients b_{1i}, are all between 0.8 and 1, except BAN (0.73) and EAG (1.06), and show no trend with area. Thus, ignoring the anomalous KJE and JOH for the moment, the unit area sediment loads from the watersheds prior to disturbance (Figure 22) tend to be highest in the four largest watersheds (ARF, FLY, LAN, and DOL), followed by the tributaries CAR, GIB, and EAG, and are lowest in the smallest watershed BAN.

Although there are signs of positive or negative trends in some individual watersheds, the residuals from model (24) display little if any trend with time (Figure 23). If the anomalous JOH and KJE, which did not show treatment effects, are omitted, hints of a recovery trend disappear entirely.

The covariance model fit rather well for the sediment models based on HI. Correlations declined linearly with watershed separation (Figure 24) and variance declined as a power function of watershed area (Figure 25). The Box-Pierce test [Shumway, 1988] indicated (using an experimentwise error rate of 0.05) the presence of serial autocorrelation at four stations (ARF, BAN, GIB, and KJE) and suggests that we conservatively assess marginally significant terms in the model. The residuals again conform very well to the normal distribution and there is only one outlier (associated with stream bank collapses in EAG). The regression between observed and fitted values has $r^2 = 0.915$. This compares with $r^2 = 0.828$ for a model with no disturbance variables and $r^2 = 0.948$ for model (3) fit to only the pre-treatment data. So the complete model (without the cumulative effects term) explains (0.915 – 0.828) / (0.948 – 0.828) = 72% of the variation introduced by the post-treatment data.

Models with HIM control. This analysis included 376 observations representing 51 storms. In models developed with the HIM control, the log-ratio flow variable $\Delta q_{ij}^{(2)}$ was found to be a better explanatory variable than the flow model residual $\Delta q_{ij}^{(1)}$. The most important disturbance variable in these models is proportion of the watershed occupied by road cuts and fills. The length of stream channel in clear-cuts and the interaction terms in model (24) were not significant when tested in maximum likelihood models with the HIM control. This is partly explained by a high correlation (0.80) between road cut/fills and stream length in burned areas. A negative interaction between road cut/fills and watershed area was marginally significant (p_N=0.037). The maximum likelihood estimates for the model

$$\begin{aligned}\log(y_{ij}) &= \beta_{0i} + \beta_{1i}\log(y_{(\mathrm{HIM})\,j}) \\ &\quad + \beta_2\Delta q_{ij}^{(2)} + \beta_3 x_{ij} + \beta_4 x_{ij} a_i + \varepsilon_{ij}\end{aligned} \qquad (25)$$

where x_{ij} is the proportion of the watershed occupied by road cuts and fills, are shown in Table 10. As with model (24), the interaction only serves to partly offset the positive interaction hidden in the $\Delta q_{ij}^{(2)}$ term, and we do not consider it significant. The trend in intercepts that was seen for model (24) is also present in model (25). Setting β_4 to zero, and substituting $\beta_0^{(1)} + \beta_0^{(2)} a_i$ for β_{0i}, we test $\beta_0^{(2)}$ and again find that it is positive and differs significantly from zero (p_N=0.0023). The residuals from model (25), with β_4 fixed at zero, do not display a significant trend with time since logging.

Magnitude of observed changes. Sediment load increases were calculated using equations (22) and (23) with the coefficients estimated from model (25). Median increases were 64% in partly clear-cut watersheds and 107% in clear-cut watersheds (Figure 26). Absolute increases were similar in clear-cut and partly clear-cut watersheds (Figure 27). Most of the larger percentage increases in clear-cuts were from small events and equated to relatively minor absolute increases in load. As one would expect, there is a tendency for percentage increases to decrease with storm size, and for absolute increases to increase with storm size. Figure 28 shows 95% confidence intervals and prediction intervals for the sediment model (25), with the area×disturbance interaction, β_4, set to zero. The watersheds are ranked by increasing proportion of road cuts and fills (x_{ij}). The uncertainty in the model and the variability in suspended sediment loads is much greater than for peak flow or storm runoff volume.

Summing storms by year, annual suspended sediment loads increased an average of 212% (262 kg·ha^{-1}yr^{-1}) in clear-cut watersheds and 73% (263 kg·ha^{-1}yr^{-1}) in partly clear-cut watersheds (Table 11). The absolute increases are heavily influenced by outlying data points that tend to occur in wet years (1993 and 1995), while the percentage increases weight all years approximately equally. If the extreme outlier in the partly clear-cut population (Figure 27) is omitted, the mean increase in that category drops to

67% (180 kg·ha^{-1}yr^{-1}). Because of the highly skewed distribution of sediment loads, median increases were much smaller: 109% (59 kg·ha^{-1}yr^{-1}) in clear-cut watersheds and 52% (46 kg·ha^{-1}yr^{-1}) in partly clear-cut watersheds. Based on the complete discharge record at NFC, the storms included in this analysis represent 36 to 43% of the total annual runoff in individual tributaries. However, these storms include roughly 90% of the annual suspended sediment load [Rice et al., 1979].

Cross-Validation of Models for Runoff Peaks, Volumes, and Sediment Loads

Predictions of storm runoff from random 10-fold cross-validation had RMSE only 2 to 3% (peaks) and 4% (volumes) higher than those from the original fitted models, for both pre-treatment and post-treatment responses (Table 12). The systematic cross-validation, omitting the post-treatment data one station at a time, gave RMSE 5% and 7% higher than the apparent post-treatment RMSE from the original runoff peaks and volume models, respectively. The systematically cross-validated RMSE values of 0.1739 and 0.1676 for logarithms of peaks and volumes correspond to prediction errors of about 20% for the untransformed responses. Calibration slopes (for regression of the observed versus predicted runoff) are very close to unity (Table 13) for both peaks and volumes. Both the random and systematic cross-validation calibrations are nearly indistinguishable from $y = x$ on 600 dpi letter-size plots. Both the RMSE and calibration results indicate the models for runoff peaks and volumes are not overfit. Remarkably, they appear to predict independent data nearly as well as the data to which the models were fit.

Predictions of suspended sediment loads from random cross-validation had RMSE 7% (HI control) and 4% (HIM control) higher than those from the original fitted models, for both pre-treatment and post-treatment responses (Table 12). On the other hand, the systematic cross-validation gave RMSE 32% (HIM control) and 50% (HI control) higher than the apparent post-treatment RMSE from the original sediment models. The systematically cross-validated RMSE values of 0.6724 and 0.6966 for logarithms of sediment loads correspond to prediction errors of about 100% for the untransformed responses. Calibration slopes for the sediment models are similar to the original models for the random cross-validation, but the systematic cross-validation has calibration slopes significantly smaller (Table 13), indicating substantial shrinkage in prediction of data from subwatersheds not used in model-fitting. The cross-validations indicate that the sediment models are not likely to predict future sediment loads well, and the associations identified between sediment loads and the disturbance variables in these models may be coincidental.

DISCUSSION

Storm Peaks

The effect of logging second-growth forests on streamflow peaks in Caspar Creek is consistent with the results from studies conducted over the past several decades throughout the Pacific Northwest. That is, the greatest effect of logging on streamflow peaks is to increase the size of the smallest peaks occurring during the driest antecedent conditions, with that effect declining as storm size and watershed wetness increases. However, increases were still apparent even in the largest storm of this study, which had a recurrence interval of 7 years at NFC.

Although the relative increases in peak flows tend to decline as storm size increases, the effects on large storms may still be important when recurrence intervals of a given size peak are considered. The curve for m=2, for example, in Figure 29 shows the increase in peak needed to reach a size that formerly had twice the recurrence interval, based on a curve fitted to the 28-year pre-logging partial duration series at NFC. Equivalently these are the increases necessary to halve the recurrence interval of the peaks that would result from the increased flow regime. Under such a flow regime, the frequency of large peaks of a given size would double, roughly doubling the geomorphic work performed on the channel. For comparison, the increased peak flows observed in this study (Figure 13) have been included in Figure 29, assuming unit-area flow frequencies in the tributaries are the same as at NFC. Although the variability is very great, it appears that the average observed increases in clear-cuts are great enough to roughly halve the recurrence intervals for storm sizes greater than 0.004 $m^3s^{-1}ha^{-1}$ (return periods longer than 0.5 years). Average observed increases in partly cut watersheds were smaller.

Accounting for the amount of watershed disturbance, there was no evidence that either storm peaks or the logging effect on peaks was related to watershed size. Peaks in the smallest drainages tended to have greater responses to logging than in larger watersheds, but this was because the smaller watersheds had greater proportions disturbed. That is the typical pattern because Forest Practice Rules and economics usually limit the amount of intense activity occurring within any given watershed in any year. Therefore, it is possible for entire small first-order watersheds to be logged within a single year. However, as the size of the watershed increases, a smaller proportion of the watershed is likely to be logged in any given year. In the largest watersheds, harvesting may be spread over decades, within which time the earliest harvested areas will have revegetated.

The data from the streamflow, pipeflow [Ziemer, 1992; Keppeler and Brown, 1998], and soil moisture studies [Keppeler et al., 1994] at Caspar Creek all suggest that the peak flow response to logging is related to a reduction in vegetative cover. Reducing vegetative cover, in turn, reduces transpiration and rainfall interception. Since little soil moisture recharge occurs during the spring and summer growing season at Caspar Creek, large differences in soil moisture can develop between logged and unlogged watersheds by late summer because of differences in evapotranspiration. For example, by late summer, a single mature pine tree in the northern Sierra Nevada depleted soil moisture to a depth of about 6 m and to a distance of 12 m from the trunk [Ziemer, 1968]. This single tree transpired about 88 m^3 more water than the surrounding logged area, equivalent to about 180 mm of rainfall over the affected area. In the South Fork of Caspar Creek, the largest changes in peak streamflow after logging were found to be for the first storms after lengthy dry periods [Ziemer, 1981]. Similarly, after logging the North Fork, there was a strong interaction between the proportion of the area logged and watershed wetness that explained differences in streamflow peaks.

Evaporation of rainfall intercepted by the forest canopy can result in a substantial reduction in the amount of water that reaches the ground. Preliminary measurements at Caspar Creek suggest that average rainfall interception is about 20% of gross winter rainfall. Studies elsewhere have also reported that a large portion of annual rainfall is intercepted and evaporated from the forest canopy. For example, Rothacher [1963] reported that under dense Douglas-fir stands in the Oregon Cascades, canopy interception loss averaged 24% of gross summer precipitation and 14% gross winter precipitation. Percentage interception losses are greatest during low-intensity rainfall interspersed with periods of no rain. As with transpiration, rainfall interception can contribute to important differences in antecedent conditions between logged and unlogged watersheds. And during the large high-intensity storms that result in large streamflow peaks, rainfall interception is still important; about 18% of the rainfall from a 96-mm 24-hour storm was intercepted by the forest canopy at Caspar Creek. Differences in interception loss between logged and unlogged areas probably explain most of the observed increases in the larger winter peaks, when transpiration is at its annual minimum.

Road construction and logging were not applied as separate treatments in this study. And, because they are correlated, it is difficult to distinguish their effects statistically. However, soil compaction from roads and timber harvest represents only 3.2% of the North Fork watershed and ranges from 1.9% to 8.5% for the tributary watersheds. Further, roads, landings, and skid-trails in the North Fork are all located near the ridges and well away from any streams. Consequently, roads, soil compaction, and overland flow probably did not produce important changes in peak flow response of the North Fork watersheds. The recovery rate of about 8% per year for storm peaks supports the hypothesis that changes in peak flows are largely controlled by changes in vegetation.

Storm Runoff Volume

Analogous to the storm peaks model, the model for storm flow volumes showed that flow increases could be largely explained by the proportion of a watershed logged, an antecedent wetness index, and time since logging. Logging probably impacted both storm peaks and flow volumes via the same mechanisms: reduction of rainfall interception and transpiration.

Suspended Sediment Loads

The most important explanatory variable identified by the sediment models was increased volume of streamflow during storms after logging. This result is not unexpected because, after logging, increased storm flows in the treated watersheds provide additional energy to deliver and transport available sediment and perhaps to generate additional sediment through channel and bank erosion.

Whereas individual watersheds show trends indicating increasing or decreasing sediment loads, there is no overall pattern of recovery apparent in a trend analysis of the residuals from the model (Figure 23). This is in contrast with the parallel model for storm flow volume, and suggests that some of the sediment increases are unrelated to flow increases.

Other variables found to be significant, depending on the control watersheds used, were road cut and fill area and length of unbuffered stream channel, particularly in burned areas. One must be cautious about drawing conclusions about cause and effect when treatments are not randomly assigned to experimental units and replication is limited. Increases in sediment load in one or two watersheds can create associations with any variable that happens to have higher values in those watersheds, whether or not those variables are physically related to the increases. In this study, the contrast in response was primarily between watershed KJE, where sediment loads decreased, versus watersheds BAN, CAR, DOL, EAG, and GIB. Watershed KJE was unburned and also had the smallest amount of unbuffered stream of all the cut units. Watersheds EAG and GIB were burned and had the greatest amount of unbuffered stream in burned areas. Watershed EAG experienced the largest sediment increases and also had the greatest proportion of road cut and fill area. EAG was not unusually high in road surface area, and the larger road cut and fill area in EAG reflects roads that are on steeper terrain than in the other cut units.

Road systems would typically be expected to account for much of the sediment. During storm events frequent cutbank failures and culvert blockages along the pre-existing North Fork perimeter all-season road (dating back more than half a century) resulted in drainage diversions and sediment input to North Fork tributaries both before and after logging. But there is little field evidence of sediment delivery from the *new* spur roads in the North Fork watershed. In an inventory of failures greater than 7.6 m^3, only 8 of 96 failures, and 1,686 of 7,343 m^3 of erosion were related to roads and none were associated with the new roads. Based on 129 random erosion plots [Rice, 1996; Lewis, 1998] in the North Fork, the road erosion in EAG was 9.3 m^3ha^{-1}, compared to 34.5 m^3ha^{-1} for KJE and 16.6 m^3ha^{-1} for all roads in the North Fork. Thus it seems that the appearance of road cuts and fills in the model resulted from a spurious correlation. The *new* roads were relatively unimportant as a sediment source in the North Fork, probably because of their generally stable locations on upper hillslopes far from stream channels, the use of outsloping and frequent rolling-dips (drains), and negligible rainy season use.

Field evidence suggesting that unbuffered stream channels contributed to suspended sediment loads is more consistent. Channel reaches subjected to intense broadcast burns showed increased erosion from the loss of woody debris that stores sediment and enhances channel roughness. Annual surveys evaluating bank stability, vegetative cover, and sediment storage potential suggest the greatest sediment production and transport potential existed in the burned channel reaches. Bank disturbances from timber falling and yarding were evident in the unburned channels, but slash and residual woody debris provided both potential energy dissipation and sediment storage sites for moderating sediment transport. Increased flows, accompanied by soil disruption and burning in headwater swales, may have accelerated channel headward expansion and soil pipe enlargements and collapses observed in watershed KJE [Ziemer, 1992] and in EAG, DOL, and LAN.

Based on 175 random 0.08-ha erosion plots in harvest areas [Rice, 1996; Lewis, 1998] in the North Fork, total erosion after logging in the burned watersheds EAG and GIB was 153 m^3ha^{-1} and 77 m^3ha^{-1}, respectively, higher than all other watersheds. Total erosion for the unburned clear-cut watersheds BAN, CAR, and KJE averaged 37 m^3ha^{-1}. These figures include estimates of sheet erosion, which is difficult to measure and may be biased towards burned areas because it was easier to see the ground where the slash had been burned. About 72% of EAG and 82% of GIB were judged to be thoroughly or intensely burned, and the remainder was burned lightly or incompletely. It is unknown how much of this hillslope erosion was delivered to stream channels, but the proportion of watershed burned was not a useful explanatory variable for suspended sediment transport. A plausible conclusion is that only burned areas in or adjacent to stream channels contributed appreciable amount of sediment to the streams.

The inventory of failures greater than 7.6 m^3 identified windthrow as another fairly important source of sediment. Of failures greater than 7.6 m^3, 68% were from windthrow. While these amounted to only 18% of the failure volume measured, 91% of them were within 15 m of a stream, and 49% were in or adjacent to a stream channel. Because of the proximity of windthrows to streams, sediment delivery from windthrow would be expected to be high. Windthrows are also important as contributors of woody debris to these channels, and play a key role in pool formation. Because woody debris traps sediment in transport, the net effect of windthrow on sediment transport can be either positive or negative. Woody debris inputs into the channel have been unusually high in the years since logging, partly because of a number of severe windstorms and partly because of the buffer strip design [Reid and Hilton, 1998]. While this has led to substantial bank cutting and channel reworking, the bulk of the increased sediment loads after logging watersheds BAN, CAR, EAG, and GIB has not yet reached the main stem stations FLY and ARF, much of it having been stored in reaches affected by blowdown [Lisle and Napoletano, 1998].

Cumulative effects. We have considered three types of information that the sediment models provide about the cumulative effects of logging activity on (unit area) suspended sediment loads. Keep in mind that the response being considered in all these questions is the suspended sediment load per unit watershed area for a given storm event and that watershed area was used in the model to represent distance downstream.

Question 1. **Were the effects of multiple disturbances additive in a given watershed?** This question may be answered partly by looking at the forms of the storm flow and sediment models. Analyses of residuals and covariance structures provide good evidence that the models are appropriate for the data, including the use of a logarithmic response variable. A logarithmic response implies a multiplicative effect for predictors that enter linearly and a power function for predictors that enter as logarithms. The flow response to logged area in model (10) is multiplicative, and the sediment response to flow increases in models (24) and (25) is a power function because Δq (equations (11), (12)) is equivalent to the log of a ratio. We next examine how much these relations differ from an additive relationship in the range of data we observed.

Consider $E(r_{ij})$, the expected value of the ratio between an observation and its expectation in an unlogged condition. From equations (9) and APPENDIX C, equations (35) and (36),

$$E(r_{ij}) = \exp\left[D_{ij}T_{ij}\right] \tag{26}$$

where $T_{ij} = \beta_4 + \beta_5 \log(y_{Cj}) + \beta_6 \log(w_j) + \beta_7 a_i$. The expected effect of combining two simultaneous disturbances D_1 and D_2 is

$$E(r_{1+2}) = \exp\left[(D_1 + D_2)T_{ij}\right] = E(r_1)E(r_2) \tag{27}$$

where $E(r_1) = \exp[D_1 T_{ij}]$ and $E(r_2) = \exp[D_2 T_{ij}]$ are the expected effects of the individual disturbances. The combined effect departs most from additive when $E(r_1) = E(r_2)$. For example, disturbances that individually would result in 10% and 30% increases in the response produce a combined increase of 43% ($1.10 \times 1.30 = 1.43$), while disturbances that individually would result in 20% increases, produce a greater combined increase of 44% ($1.20 \times 1.20 = 1.44$). If the disturbances were additive the combined increase would be a 40% increase in either case. For more than two disturbances, the departures from additivity can be somewhat greater. In general, multiple disturbances that have a combined effect of r on the response under a multiplicative model will result in a minimum increase of $\log(r)$ in the response under an additive model, where r is defined in the sense of r_{ij} above. (This results from a mathematical limit as the number of equal-magnitude disturbances contributing to the effect r becomes large.)

In the storm flow data, only the main-stem gaging stations received waters from multiple disturbances. The maximum observed increase in storm flow on any main stem gaging station was 118%, but 8 out of 10 increases were under 40% and the median increase was just 16%. Taking the logarithms of 2.18, 1.40, and 1.16, we find that multiple disturbances that could produce these increases in a multiplicative model would produce minimum increases of 78%, 34%, and 15%, respectively, under an additive model. Therefore, in the range of most of the data (increases less than 40%) the disturbance effect on storm flow is approximately additive.

Now we can evaluate the additivity of the disturbance effect on sediment load, since this is expressed mainly through Δq. For this evaluation we fit model {(25),(17),(18)}, but fixing the parameters involving road cuts and fills at zero. Under this model, analogously to equation (26) for the flow model, the expected value of the ratio between an observation and its expectation in an unlogged condition is given by

$$E(r_{ij}) = \exp\left[\beta_2 \Delta q_{ij}^{(2)}\right] = \exp\left[\beta_2 \log\left(\frac{y_{ij}}{y_{Cj}}\right)\right] = \left(\frac{y_{ij}}{y_{Cj}}\right)^{\beta_2} \tag{28}$$

The ratio of y_{ij} and y_{Cj} , the unit area flow volumes in storm j from the treated and control watersheds, is an expression of the increased flow related to tree removal. A plot of equation (28) using the maximum likelihood estimate of 1.514 for β_2 passes through (1,1) and is very nearly linear in the range $0.82 \le y_{ij}/y_{Cj} \le 1.92$, which includes 95% of the observations on the main-stem stations. It follows that the effect of flow on suspended sediment is approximately additive for stations which receive waters from multiple logging units. For example, a flow ratio of 1.40 corresponds to a 66% increase in sediment load, while a flow ratio of 1.80 corresponds to a 143% increase in sediment load. An additive flow effect would produce an increase of 66 + 66 = 132% in sediment load, not much less than 143%. Examples of smaller flow ratios deviate from additivity even less than this example.

So, in the range of data we observed, the effect of disturbance on flow is approximately additive, and the effect of flow on sediment loads is approximately additive. In summary, the mathematical approach indicates that the combined effect of multiple disturbances on sediment loads is very similar to the sum of the effects of the individual disturbances.

Question 2. Were downstream changes greater than would be expected from the proportion of area disturbed? This question was addressed by testing the coefficients of terms formed from the product of disturbance and watershed area. If the coefficient of this term were positive, it would imply that the effect of a given disturbance proportion increases with watershed size. The interactions of those disturbance measures that had explanatory utility in the sediment models were considered, including road cut and fill area and length of unbuffered stream channels. None of the product terms were found to have coefficients significantly greater than zero, indicating that suspended load increases were not disproportionately large in larger watersheds. To the contrary, the sum of the observed sediment loads at the four main-stem stations were all within 25% of the sum of the loads predicted for undisturbed watersheds (Table 7). Channel cross-section measurements indicate 1040 metric tons of net filling in the main stem during the post-logging period [Lisle and Napolitano, 1998]. Much of the logging-related sediment from the tributaries has apparently been deposited in the main stem, especially in reaches affected by blowdowns and in alluvial bars near tributary confluences, and therefore has not reached downstream gages.

There is, however, one subwatershed where this second type of cumulative effect may be occurring. Watershed DOL, only 36% cut, includes the 100% cut watershed EAG, yet the percentage sediment increases have been similar (269% at DOL versus 238% at EAG). Several mechanisms appear to be responsible for the unexpectedly high loads at DOL. In the incised lower reach, bank failures and channel widening have occurred. In addition, a major stream diversion caused by a windthrow resulted in the formation of a major gully eroding 87 m^3 directly into the stream. Sediment is also being released from behind decaying logs that were placed in the channel for skidding by oxen during historic logging. Finally, all these processes would have been augmented by the increased storm flows that followed modern logging.

Question 3. Were sediment loads in the lower watershed elevated to higher levels than in the tributaries? Regardless of the control watersheds used, suspended sediment transport per unit watershed area tended to increase downstream before logging (Figure 21). This tendency may reflect a greater availability of fine sediment downstream in lower gradient channels. If unit area sediment loads increase downstream and result in water quality levels of concern with a smaller proportion of watershed disturbance than upstream locations, then cumulative effects may be said to have occurred, in the sense that activities producing acceptable local impacts resulted in impacts that are unacceptable by the same standard downstream.

To the extent that larger watersheds reflect average disturbance rates and therefore have smaller proportions of disturbance than the smallest disturbed watersheds upstream, one might expect sediment loads downstream to increase by less than those in the logged tributaries. In addition, as mentioned before, some of the sediment may be temporarily stored before reaching the lower stations. Indeed, in this study the post-treatment regression lines were much more similar among watersheds than the pretreatment lines, and the main-stem stations no longer transported the highest unit area sediment loads. However, larger watersheds will not necessarily behave the same way. For example, in geographically similar Redwood Creek in northwestern California, two main-stem gaging stations (175 km^2 and 720 km^2) yield higher sediment loads per unit area than three intensively logged tributaries [Lewis, 1998].

Cumulative effects considered in this paper were limited to a few hypotheses about water quality that could be statistically evaluated. But cumulative effects can occur in many ways. For example, resources at risk are often quite different in downstream areas, so an activity that has acceptable local impacts might have unacceptable offsite impacts if critical or sensitive habitat is found downstream. Different physical processes also tend to dominate upstream and downstream reaches. Channel aggradation may be the biggest problem downstream, while channel scour may be of concern upstream.

Subwatersheds and KJE anomaly. Analyses of the 5 clear-cut tributaries in the North Fork drainage show suspended load increases at all gaging stations located immediately below clear-cut units except at KJE, where loads have decreased. KJE had the highest pre-logging (1986-1989) unit area sediment loads of any of the tributaries (Figure 22), but, after logging, loads were similar to the other logged tributaries (Figure 17).

Prior to logging, the stream channel above KJE was unique. The KJE channel was an active gully with an abundant supply of sediment and the lowest gradient of any of the tributaries. After logging, the number of small debris jams doubled in the buffered channel above KJE, and further upstream the channel contained a large amount of logging debris and dense vegetative regrowth. Thus, opportunities for temporary sediment storage increased, and net energy available for sediment transport may have decreased, despite moderately increased flows, because of the increased channel roughness. The other tributaries were stable, vegetated, steep channels with limited sediment supplies and relatively low unit area sediment loads prior to logging. In these tributaries the increased sediment introduced by logging was readily transported. While this explanation is speculative, response in sediment transport to a disturbance certainly will vary with channel morphology and the relative availability of sediment and energy.

CONCLUSIONS

The main conclusions from these analyses are:

- Models based upon the proportion of watershed area logged, an antecedent wetness index, time since logging, and the responses in unlogged control watersheds explained 95% of the variation in the logarithms of both storm discharge peaks and volumes. Goodness-of-fit is similar for pre-logging and post-logging data, and cross-validation indicates that the models were not overfit to the data.
- Storm discharge peaks and volumes after extended periods with little or no precipitation increased up to 300% and 400% respectively, but most increases were below 100%.
- The effect of logging on storm discharge peaks and volumes declines with increasing regional antecedent wetness, as indexed by a decay function of prior runoff at a control watershed. However, even under the wettest conditions of the study, increases in storm runoff from clear-cut watersheds averaged 23% for peaks and 27% for volumes.
- Relative increases in storm discharge peaks and volumes decline with storm size but were positive even in the largest storms of the study period.
- Average increases in annual storm runoff were 58% from 95-100% clear-cut watersheds and 23% from 30-50% clear-cut watersheds.
- Recovery rates in the first 4-7 years after logging are estimated to be 8% per year for peak flows and 9% per year for storm flow volumes.
- Effects of multiple disturbances on storm discharge peaks and volumes are approximately additive, and there is little evidence for magnification of effects downstream.
- Reduction in rainfall interception and transpiration by forest vegetation is the probable cause of increased storm discharge peaks and volumes following logging.
- Annual sediment loads increased 123-269% in the tributaries, but, at main-stem stations, increased loads were detected only in small storms and had little effect on annual sediment loads. At the North Fork weir, an increase of 89% was caused mainly by a landslide in an ungaged tributary that enters just above the weir.
- Much of the increased sediment load in North Fork tributaries was related to increased storm flow volumes. With flow volumes recovering as the forest grows back, flow-related increases in sediment load are expected to be short-lived.
- The effects of multiple disturbances on suspended loads in a watershed were approximately additive.
- In general, downstream suspended load increases were no greater than would be expected from the proportion of area disturbed. In one tributary, increased flows evidently impacted the channel in an uncut area downstream by mobilizing stored sediment and aggravating bank instabilities, but most of the increased sediment produced in the tributaries was apparently stored in the main stem and has not yet reached the main-stem stations.
- Before logging, sediment loads on the main stem were higher than on most tributaries. This was no longer the case after logging, apparently because sediment exported from tributaries was deposited at temporary storage sites, and smaller proportions of downstream watersheds were disturbed.
- Sediment increases in North Fork tributaries probably could have been reduced by avoiding activities that denude or reshape the banks of small drainage channels.
- Sediment loads are affected as much by channel conditions (e.g. organic debris, sediment storage sites, channel gradient, and width-to-depth ratio) as by sediment delivery from hillslopes.

APPENDIX A. Notation Used in the Text

a_i	Drainage area of watershed *i*
b_i	Estimate of parameter β_i
c_{ij}	Proportion of watershed *i* logged in water years prior to that of storm *j*, and
c'_{ij}	Proportion of watershed *i* logged prior to storm *j* but in the same water year
D_{ij}	Some measure of disturbance per unit area in watershed *i* at storm *j*
$d_{i_1 i_2}$	Distance between centroids of watersheds i_1 and i_2
K	Number of parameters estimated in a model
n	Number of observations used in an analysis
p_{ij}	True (unknown) percentage change in response of watershed *i* in storm *j* as a result of treatment
$\tilde{p}_{ij}$	"Observed" percentage change in response of watershed *i* in storm *j* based on a comparison of y_{ij} and $\hat{y}'_{ij}$
p_0	Percentage change in response, given an arbitrary vector $\mathbf{x}_0$
$\mathrm{p_N}$	Significance level of a hypothesis test based on the normal distribution
$\Delta q_{ij}^{(1)}$	Residual from the flow model (3) containing only β_{0i} and β_{1i}
$\Delta q_{ij}^{(2)}$	Difference between the logarithms of flow in the treated and control watersheds
$\Delta q_{ij}^{(3)}$	Predicted change after logging in the logarithm of storm flow from eqn (10)
t_{ij}	Area-weighted mean cutting age (number of summers passed) in watershed *i* for areas logged in water years preceding that of storm *j*
w_j	Wetness index at start of storm *j*
$x_{ij}^{(1)}, x_{ij}^{(2)}$	Generic measures of unit area disturbance in watershed *i* at storm *j*
$\mathbf{x}_0$	Arbitrary vector of explanatory variables
y_{ij}	Unit area response at treated watershed *i* in storm *j*
y_{Cj}	Unit area response at control watershed in storm *j*
y'_{ij}	Unknown response at watershed *i*, if it had been left untreated, in storm *j*
$\hat{y}'_{ij}$	Estimate of y'_{ij}
β_{0i}, β_{1i}	Location parameters (slope and intercept) to be estimated for each watershed *i*
$\beta_0^{(1)}, \beta_0^{(2)}$	Parameters used to model β_{0i} as a function of a_i
$\rho_{i_1 i_2}$	Correlation between $\varepsilon_{i_1 j}$ and $\varepsilon_{i_2 j}$
$\sigma_{i_1}, \sigma_{i_2}$	Standard deviations of $\varepsilon_{i_1 j}$ and $\varepsilon_{i_2 j}$
ε_{ij}	Error or deviation of y_{ij} from model at treated watershed *i* in storm *j*
$\varepsilon_{i_1 j}, \varepsilon_{i_2 j}$	Errors for watersheds i_1 and i_2 in storm *j*
θ_i	Parameter in covariance model
$\hat{\theta}_i$	Estimate of parameter θ_i

APPENDIX B. Likelihood Function and Gradient

The model for the mean response can be written

$$\mathbf{u} = E(\mathbf{y}) = f(\boldsymbol{\beta}) \tag{29}$$

where $\mathbf{y}$ is an $n\times 1$ response vector and $\boldsymbol{\beta}$ is a $p\times 1$ vector of unknown parameters. The error, $\mathbf{e} = \mathbf{y} - \mathbf{u}$, is modelled as a multivariate normal variable depending on q parameters:

$$\begin{aligned} &\mathbf{e} \sim \mathbf{N}(0, \boldsymbol{\Sigma}) \\ &\boldsymbol{\Sigma} = \mathbf{G}(\boldsymbol{\theta}) \end{aligned} \tag{30}$$

where $\boldsymbol{\Sigma}$ is the $n\times n$ covariance matrix of $\mathbf{e}$ depending on $\boldsymbol{\theta}$, a $q\times 1$ vector of unknown parameters. The elements of $\boldsymbol{\Sigma}$ are paramaterized by equations (15)-(18). The likelihood function and its logarithm are

$$\begin{aligned} &L = (2\pi)^{-n/2}|\boldsymbol{\Sigma}|^{-1/2}\exp\left[-\tfrac{1}{2}(\mathbf{y}-\mathbf{u})^T\boldsymbol{\Sigma}^{-1}(\mathbf{y}-\mathbf{u})\right] \quad \text{and} \\ &\ell = \log(L) = -\tfrac{n}{2}\log(2\pi) - \tfrac{1}{2}\log|\boldsymbol{\Sigma}| - \tfrac{1}{2}(\mathbf{y}-\mathbf{u})^T\boldsymbol{\Sigma}^{-1}(\mathbf{y}-\mathbf{u}) \end{aligned} \tag{31}$$

respectively, where $|\boldsymbol{\Sigma}|$ is the determinant of $\boldsymbol{\Sigma}$. The gradient consists of the partial derivatives of ℓ with respect to $\boldsymbol{\beta}$ and $\boldsymbol{\theta}$:

$$\begin{aligned} &\mathbf{grad} = \left(\frac{\partial\ell}{\partial\beta_1},\cdots,\frac{\partial\ell}{\partial\beta_p},\frac{\partial\ell}{\partial\theta_1},\cdots,\frac{\partial\ell}{\partial\theta_q}\right) \\ &\frac{\partial\ell}{\partial\beta_i} = \frac{\partial\mathbf{u}^T}{\partial\beta_i}\boldsymbol{\Sigma}^{-1}(\mathbf{y}-\mathbf{u}), \qquad i = 1,\cdots,p \\ &\frac{\partial\ell}{\partial\theta_j} = -\frac{1}{2}tr\left(\boldsymbol{\Sigma}^{-1}\frac{\partial\boldsymbol{\Sigma}}{\partial\theta_j}\right) + \frac{1}{2}(\mathbf{y}-\mathbf{u})^T\boldsymbol{\Sigma}^{-1}\frac{\partial\boldsymbol{\Sigma}}{\partial\theta_j}\boldsymbol{\Sigma}^{-1}(\mathbf{y}-\mathbf{u}), \qquad j = 1,\cdots,q \end{aligned} \tag{32}$$

in which $tr(\cdot)$ refers to the trace (sum of the diagonal elements) of the matrix. The partial derivatives, $\partial\mathbf{u}^T/\partial\beta_i$ and $\partial\boldsymbol{\Sigma}/\partial\theta_j$, are model-specific and can be derived from equations (10) and (14)-(18).

APPENDIX C. An Unbiased Estimator, and Confidence and Prediction Intervals for Percentage Change in Response

Let y_0 be the response given an arbitrary predictor vector $\mathbf{x}_0$ and let y_0' be the unknown response for the same storm assuming the watershed were undisturbed. A prediction interval is sought for $p_0 = 100\left[y_0/E(y_0') - 1\right]$, the percentage change in response, and an unbiased estimator and confidence interval are sought for its expectation, $E(p_0)$. It will be convenient to obtain the unbiased estimator and confidence interval first. Since $\log(y_0)$ and $\log(y_0')$ are assumed to be normally distributed,

$$\begin{aligned} &E(y_0) = \exp\left[E\left(\log(y_0)\right) + \tfrac{1}{2}\sigma^2\right] \qquad \text{and} \\ &E(y_0') = \exp\left[E\left(\log(y_0')\right) + \tfrac{1}{2}\sigma^2\right] \end{aligned} \tag{33}$$

Let us denote the ratio of the actual response to its expected undisturbed value by

$$r_0 = \frac{y_0}{E(y_0')} \tag{34}$$

Its expectation is

$$\begin{aligned} E(r_0) &= \frac{E(y_0)}{E(y_0')} \\ &= \frac{\exp\left[E\left(\log(y_0)\right) + \frac{1}{2}\sigma^2\right]}{\exp\left[E\left(\log(y_0')\right) + \frac{1}{2}\sigma^2\right]} \\ &= \exp\left[f_0(\boldsymbol{\beta})\right] \end{aligned} \tag{35}$$

where, for the runoff models (10),

$$f_0(\boldsymbol{\beta}) = \left[\left(1 - \beta_2(t_0 - 1)\right)c_0 + \beta_3^{(k)}c_0'\right] \times \left[\ \beta_4 + \beta_5 \log(y_{C0})\ + \beta_6 \log(w_0) + \beta_7 a_0\right] \tag{36}$$

Since $\mathbf{b}$, the vector of estimates for $\boldsymbol{\beta}$, is asymptotically distributed normal, we have that $f_0(\mathbf{b})$ is asymptotically distributed normal with $E[f_0(\mathbf{b})] = f_0(\beta)$ and unknown variance σ_*^2 [Bishop et al., 1975]. In shorthand, $f_0(\mathbf{b}) \sim N\left(f_0(\boldsymbol{\beta}), \sigma_*^2\right)$ for large samples. The variance σ_*^2 may be approximated using the delta method [Bishop et al., 1975]:

$$\tilde{\sigma}_*^2 = \sum_{i=1}^{p} \sum_{j=1}^{p} \frac{\partial f_0}{\partial b_i} \frac{\partial f_0}{\partial b_j} Cov\left[b_i, b_j\right] \tag{37}$$

The covariances are estimated by the elements of the inverted information matrix [McCullagh and Nelder, 1989]. The information matrix is the negative of the matrix of second derivatives (Hessian) of ℓ with respect to the parameters, $\boldsymbol{\beta}$ and $\boldsymbol{\theta}$.

Let us introduce an estimator $\hat{r}_0 = \exp\left[f_0(\mathbf{b}) - \frac{1}{2}\sigma_*^2\right]$. Its expected value is

$$\begin{aligned} E(\hat{r}_0) &= \exp\left(-\tfrac{1}{2}\sigma_*^2\right) E\left\{\exp\left[f_0(\mathbf{b})\right]\right\} \\ &= \exp\left(-\tfrac{1}{2}\sigma_*^2\right) \exp\left\{E\left[f_0(\mathbf{b})\right] + \tfrac{1}{2}\sigma_*^2\right\} \\ &= \exp\left\{E\left[f_0(\mathbf{b})\right]\right\} \\ &= \exp\left\{f_0(\boldsymbol{\beta})\right\} \\ &= E(r_0) \end{aligned} \tag{38}$$

Hence $\hat{r}_0$ is an asymptotically unbiased estimator for $E(r_0)$, and $100(\hat{r}_0 - 1)$ is an asymptotically unbiased estimator for $100(E(r_0) - 1) = E(p_0)$. In practice, because σ_* is unknown, we replace it with $\tilde{\sigma}_*$ in the expression for $\hat{r}_0$.

Next we will compute a confidence interval for $E(r_0)$, and convert it to a confidence interval for $E(p_0)$. A 100(1-α)% confidence interval for $f_0(\beta)$ is defined by the probability

$$\Pr\left[f_0(\mathbf{b}) - z_{\alpha/2}\sigma_* \le f_0(\boldsymbol{\beta}) \le f_0(\mathbf{b}) + z_{\alpha/2}\sigma_*\right] = 1 - \alpha \tag{39}$$

where $z_{\alpha/2}$ is the $\alpha/2$ cutoff point of the standard normal distribution. Applying the monotone transformation **exp** to all sides of the inequality yields a confidence interval for $E(r_0)$:

$$\Pr\left[\exp\left[f_0(\mathbf{b}) - z_{\alpha/2}\sigma_*\right] \le E(r_0) \le \exp\left[f_0(\mathbf{b}) + z_{\alpha/2}\sigma_*\right]\right] = 1 - \alpha \tag{40}$$

Noting that $E(p_0) = 100(E(r_0) - 1)$, the above confidence interval is readily transformed into a confidence interval for $E(p_0)$.

$$100(1-\alpha)\,C.I.\,\text{for } E(p_0):\ 100\left[\exp\left(f_0(\mathbf{b}) \pm z_{\alpha/2}\sigma_*\right) - 1\right] \tag{41}$$

Since σ_* is unknown, we replace it with $\tilde{\sigma}_*$.

Finally, we will compute a prediction interval for r_0, and convert it to a prediction interval for p_0. Using model (10) and (33), we find

$$\begin{aligned} r_0 &= \frac{y_0}{E(y_0')} \\ &= \frac{\exp\left[\beta_0 + \beta_1 \log(y_{C0}) + f_0(\boldsymbol{\beta}) + \varepsilon_0\right]}{\exp\left[\beta_0 + \beta_1 \log(y_{C0}) + \frac{1}{2}\sigma^2\right]} \\ &= \exp\left[f_0(\boldsymbol{\beta}) + \varepsilon_0 - \tfrac{1}{2}\sigma^2\right] \end{aligned} \tag{42}$$

Since $\varepsilon_0 \sim N(0, \sigma^2)$ and, asymptotically, $f_0(\mathbf{b}) \sim N\left(f_0(\boldsymbol{\beta}), \sigma_*^2\right)$, and they are independent random variables, it follows that $f_0(\mathbf{b}) - \varepsilon_0 \sim N\left(f_0(\boldsymbol{\beta}), \sigma_*^2 + \sigma^2\right)$. Thus

$$\Pr\left[f_0(\mathbf{b}) - z_{\alpha/2}\left(\sigma_*^2 + \sigma^2\right)^{\frac{1}{2}} \le f_0(\boldsymbol{\beta}) + \varepsilon_0 \le f_0(\mathbf{b}) - z_{\alpha/2}\left(\sigma_*^2 + \sigma^2\right)^{\frac{1}{2}}\right] = 1 - \alpha \tag{43}$$

Subtracting $0.5\sigma^2$ and applying the monotone transformation **exp** to all parts of the inequality converts the middle term to r_0, yielding the following prediction interval:

$$100(1-\alpha)\,P.I.\,\text{for } r_0:\ \exp\left(f_0(\mathbf{b}) - \tfrac{1}{2}\sigma^2 \pm z_{\alpha/2}\left(\sigma_*^2 + \sigma^2\right)^{\frac{1}{2}}\right) \tag{44}$$

which is readily transformed to a prediction interval for p_0:

$$100(1-\alpha)\,P.I.\,\text{for } p_0:\ 100\left[\exp\left(f_0(\mathbf{b}) - \tfrac{1}{2}\sigma^2 \pm z_{\alpha/2}\left(\sigma_*^2 + \sigma^2\right)^{\frac{1}{2}}\right) - 1\right] \tag{45}$$

Since σ_* and σ are unknown, we replace them with $\tilde{\sigma}_*$ and $\hat{\sigma} = \hat{\theta}_3 a_0^{\hat{\theta}_4}$, where a_0 is the watershed area.

Confidence and prediction intervals for sediment models (24) and (25) are similar, but $f_0(\mathbf{b})$ is replaced by the linear functions $g_0(\mathbf{b})$ and $h_0(\mathbf{b})$, respectively, where

$$g_0(\mathbf{b}) = \beta_2 \Delta q_0^{(1)} + \beta_3 x_0^{(1)} + \beta_4 x_0^{(2)} + \beta_5 (x_0^{(1)} + x_0^{(2)}) \log(y_{(\mathrm{HI})0}) + \beta_6 (x_0^{(1)} + x_0^{(2)}) a_0 \tag{46}$$

$$\text{and}\quad h_0(\mathbf{b}) = \beta_2 \Delta q_0^{(2)} + \beta_3 x_0 + \beta_4 x_0 a_0 \tag{47}$$

Since these functions are linear, the delta method yields the exact variance, but, as before, the covariance matrix of **b** must be estimated from the observed information matrix, so σ_*^2 is still only known approximately.

Acknowledgments. This research was a result of a cooperative effort of the California Department of Forestry and Fire Protection and the USDA Forest Service, Pacific Southwest Research Station. The study design was a result primarily of the efforts of Raymond Rice, who was the Principal Investigator until his retirement in 1989. Robert Thomas designed the SALT sampling algorithm and Rand Eads designed the hardware/software interface used in the field implementation. In addition, the authors are grateful to Dave

Thornton and the many individuals, too numerous to mention here, who spent thousands of hours in the field (often during hazardous storm conditions), in the laboratory, and in the office to ensure the best possible data quality.

Detailed hydrologic and climatic data collected at Caspar Creek between 1963 and 1997 are available on compact disk from the authors.

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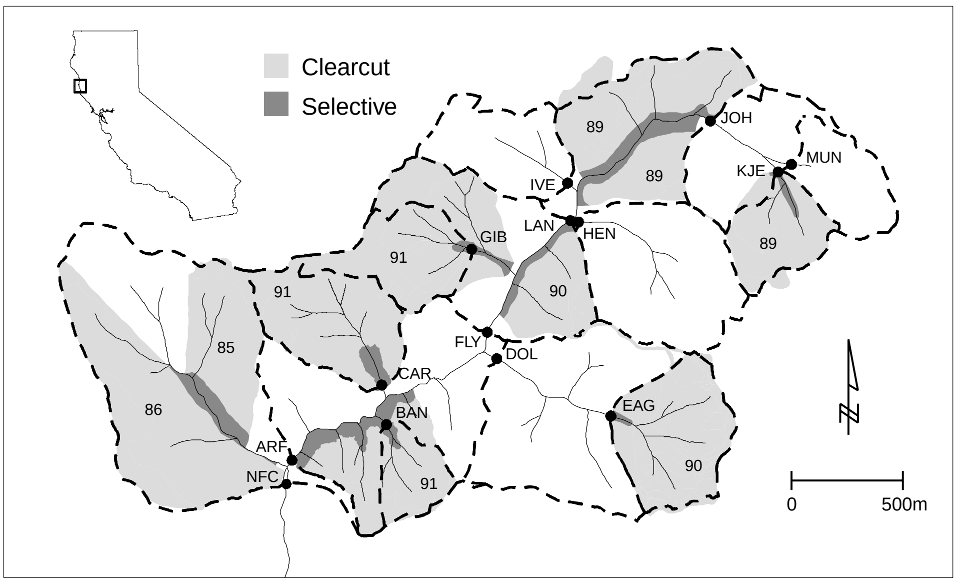

Figure 1. North Fork Caspar Creek. Gaging stations are identified by 3-letter abbreviations and dots, subwatershed boundaries by dashed lines, and logged areas by shading. Inset locates Caspar Creek within California.

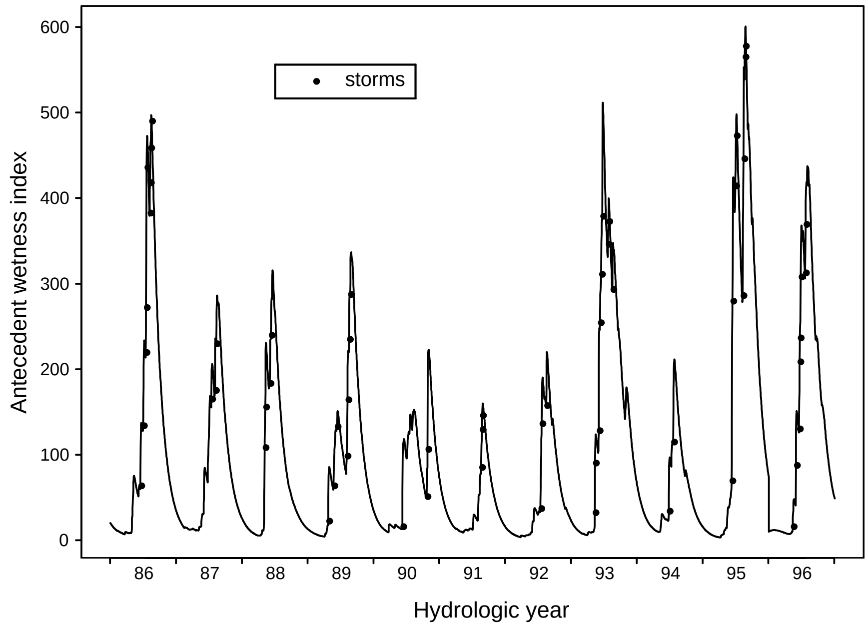

Figure 2. Antecedent wetness index (equation (2)) and temporal distribution of storms for the period of study (1986-1996). Solid circles indicate the wetness level at the start of each storm.

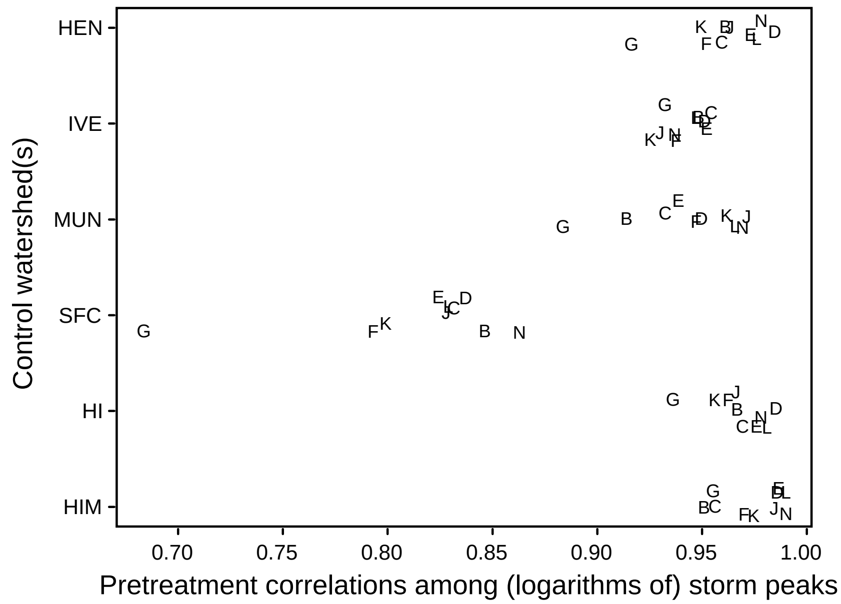

Figure 3. Pretreatment correlations between logarithms of storm peak at treated watersheds and alternative control watersheds. Letters designate watersheds (e.g. G is watershed GIB). Random noise has been added to the vertical plotting positions to improve readability.

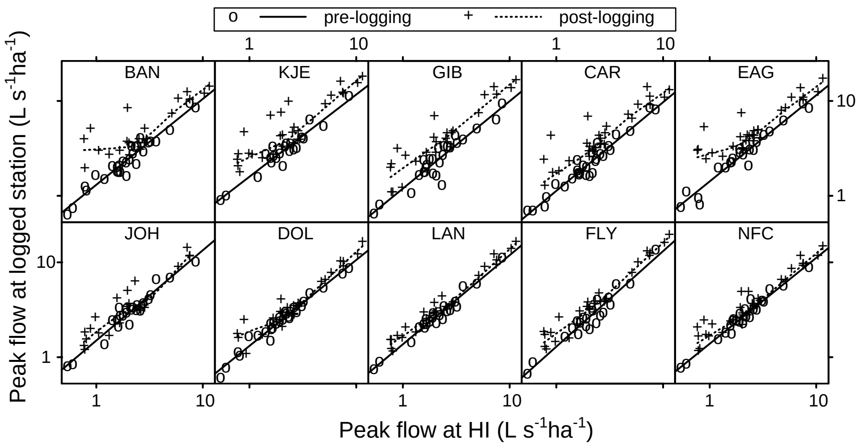

Figure 4. Relation between peak streamflow in the 10 treated tributaries in the North Fork of Caspar Creek, and that of the HI control. Post-logging relations were fitted by locally weighted regression [Cleveland, 1979]. The top row represents 95-100% clear-cut watersheds.

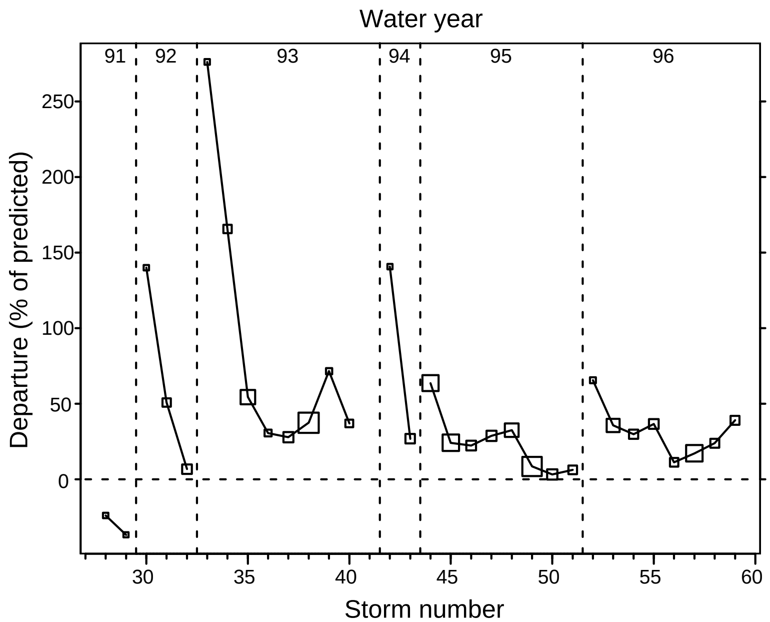

Figure 5. Post-logging departures of storm peaks (as percentage of predicted) at watershed EAG from those predicted from pretreatment regression on HI control. Axes are logarithmic. Symbol sizes indicate relative size of storm peak at HI control. Vertical dotted lines separate water years. About half the watershed was winter-logged before storm 28 and logging was completed by storm 30.

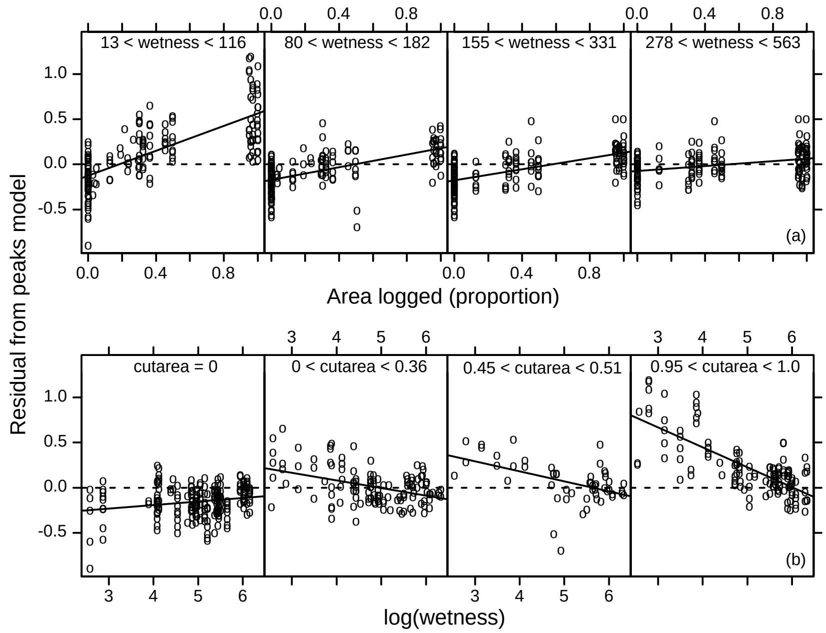

Figure 6. Conditioning plots of residual from storm peaks model (3) and interaction between area logged and antecedent wetness index with (a) wetness index fixed in each frame, and (b) proportion of area logged fixed in each frame.

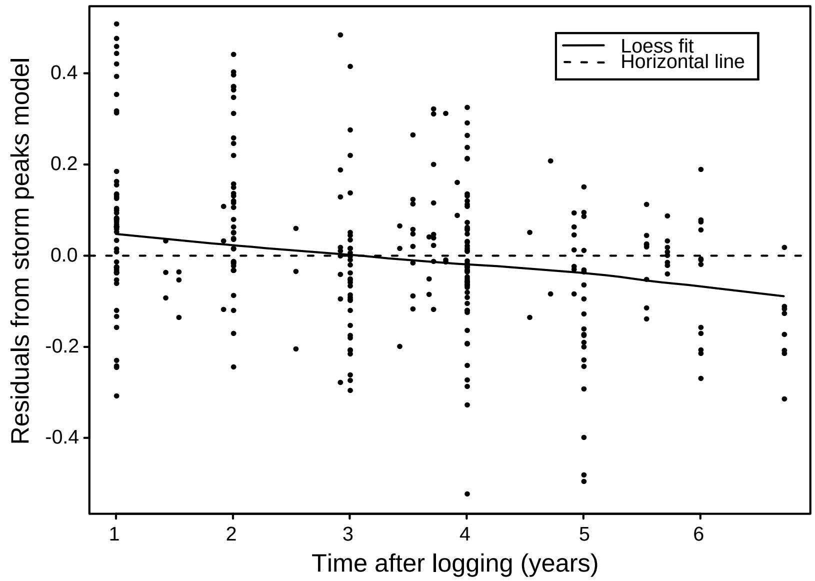

Figure 7. Relation between storm peak residuals and time after logging. Curve is fit by loess method [Cleveland, 1979]. Residuals are from least squares fit to the model $\log(y_{ij}) = \beta_{0i} + \beta_{1i}\log(y_{Cj}) + \beta_4 D_{ij} + \beta_6 D_{ij}\log(w_j) + \varepsilon_{ij}$.

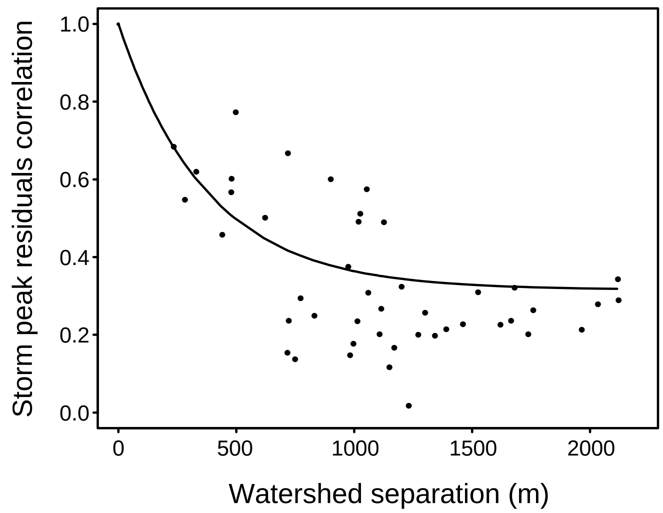

Figure 8. Relation between storm peak residuals correlation and distance between watershed centroids. Residuals are from maximum likelihood fit to storm peak model {(10),(16),(18)}. Curve depicts equation (16), with estimated parameters $\hat{\theta}_1$ and $\hat{\theta}_2$.

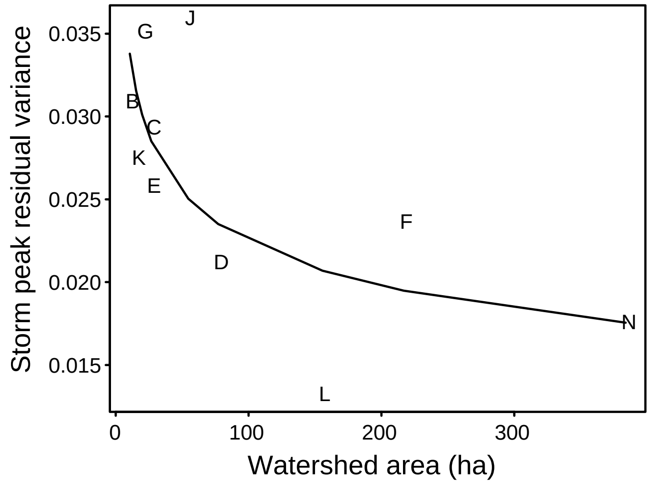

Figure 9. Relation between variance of storm peak residuals and watershed area. Residuals are from maximum likelihood fit to storm peak model {(10),(16),(18)}. Curve depicts equation (18) with estimated parameters $\hat{\theta}_3$ and $\hat{\theta}_4$. Letters designate watersheds (e.g. G is watershed GIB).

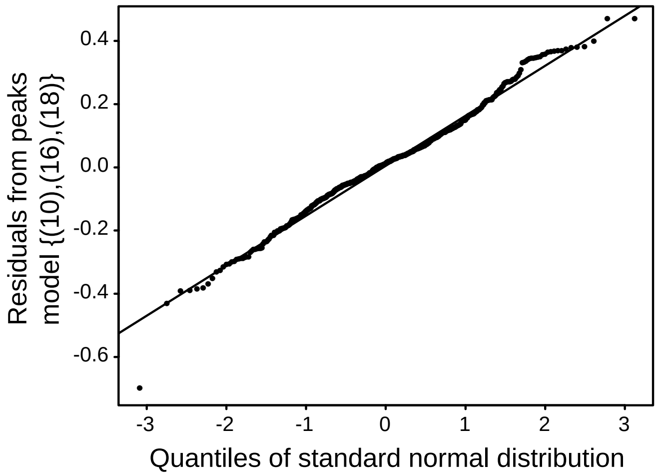

Figure 10. Normal quantile plot of residuals from storm peak model {(10),(16),(18)}. Line is least squares fit.

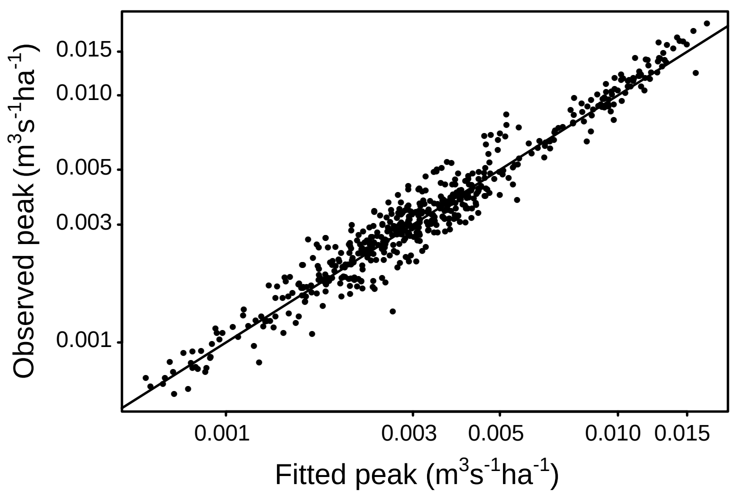

Figure 11. Observed storm peaks versus fitted values from model {(10),(16),(18)}. Line is $y = x$.

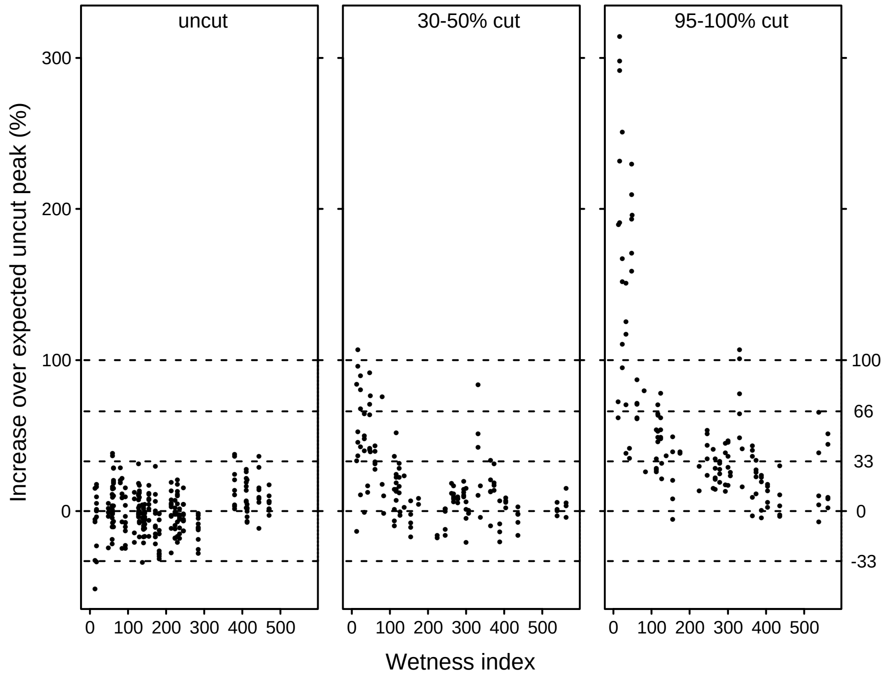

Figure 12. Percentage increase over expected uncut storm peak as related to antecedent wetness index for uncut (before treatment), partly (30-50%) clear-cut, and (95-100%) clear-cut watersheds. Bias-corrected predictions are from model {(10),(16),(18)} with disturbance set to zero.

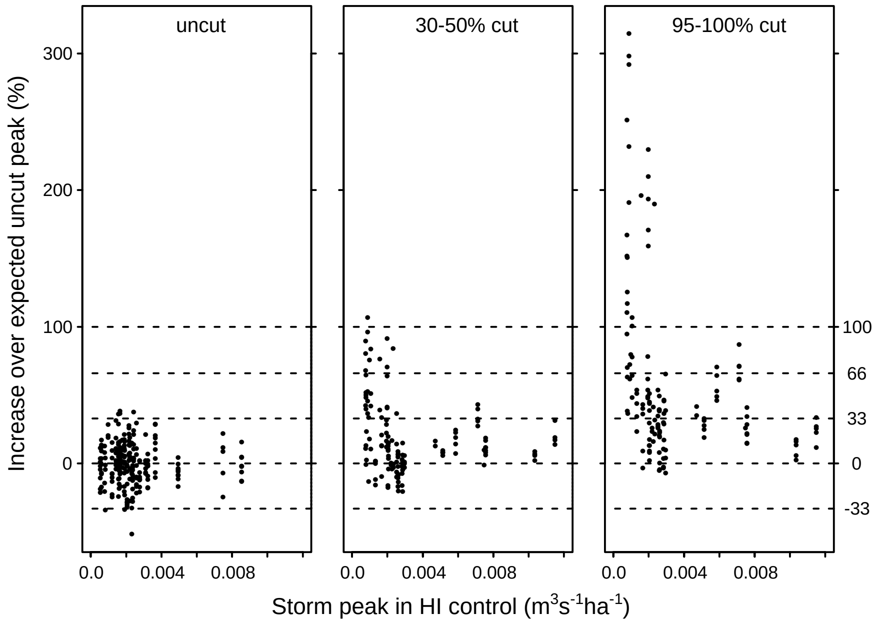

Figure 13. Percentage increase over expected uncut storm peak as related to peak size in the HI control for uncut (before treatment), partly (30-50%) clear-cut, and (95-100%) clear-cut watersheds. Bias-corrected predictions are from model {(10),(16),(18)} with disturbance set to zero.

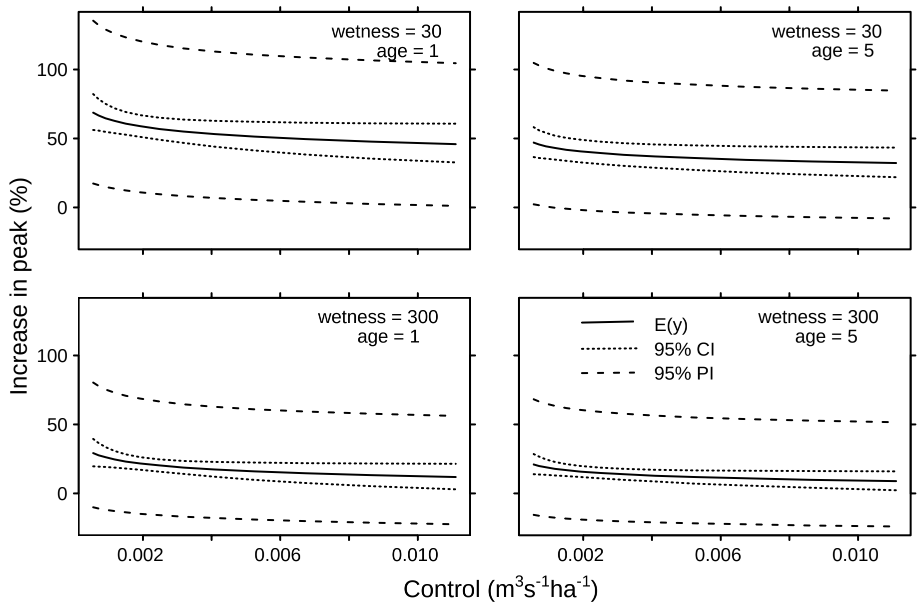

Figure 14. The effect of wetness and age after cutting on predictions from storm peak model {(10), (16),(18)} after clear-cutting 50% of a 20 ha watershed. Expected increases and 95% confidence (CI) and prediction (PI) intervals are shown for two levels of antecedent wetness 1 and 5 years after cutting.

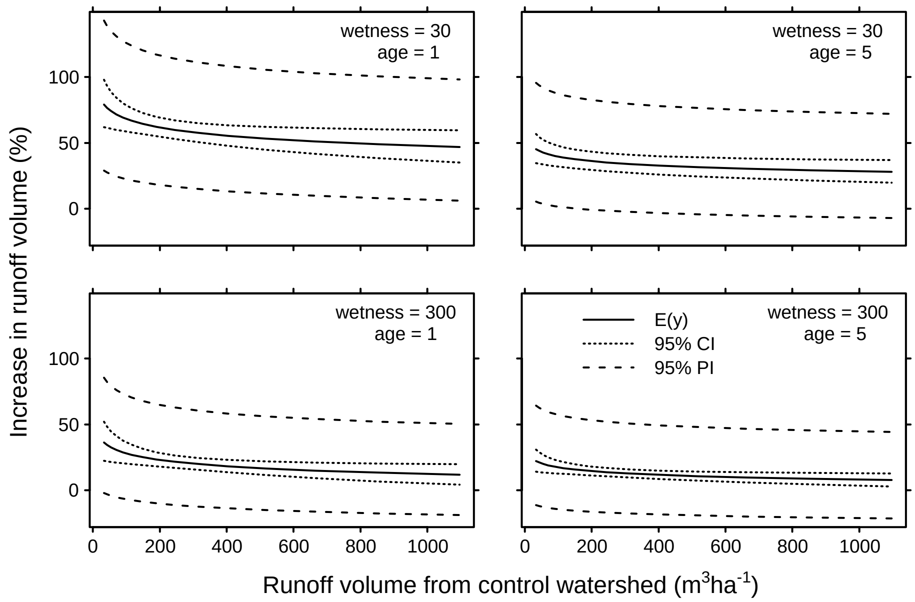

Figure 15. The effect of wetness and age after cutting on predictions from storm runoff volume model {(10),(17),(18)}, after clear-cutting 50% of a 20 ha watershed. Expected increases and 95% confidence (CI) and prediction (PI) intervals are shown for two levels of antecedent wetness 1 and 5 years after cutting.

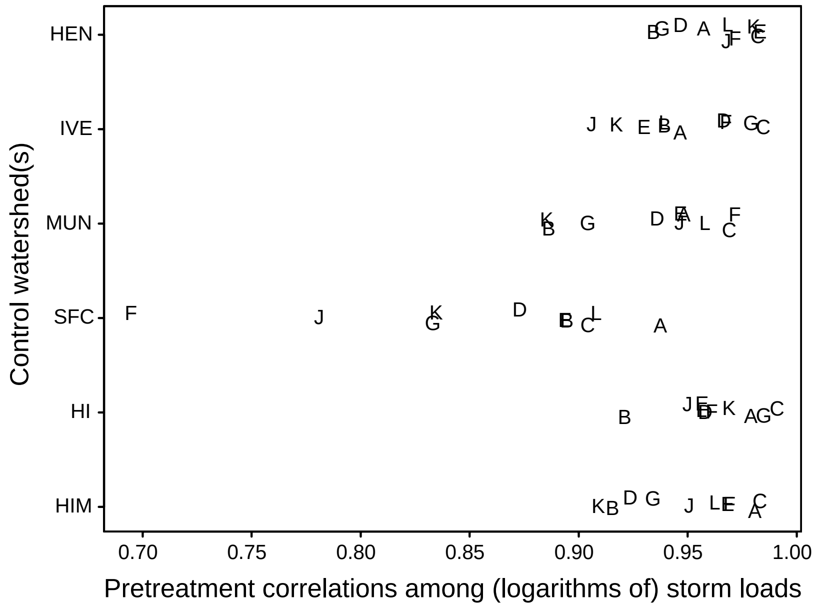

Figure 16. Pretreatment correlations between logarithms of storm sediment load at treated watersheds and alternative control watersheds. Letters designate watersheds (e.g. G is watershed GIB). Random noise has been added to the vertical plotting positions to improve readability.

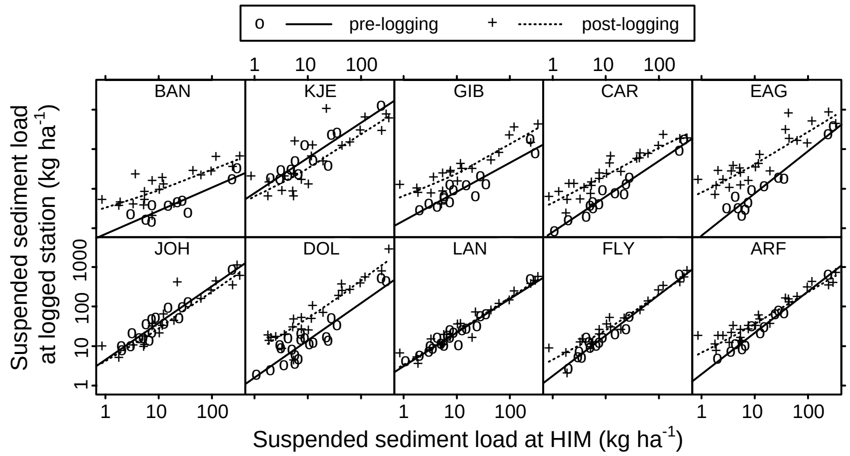

Figure 17. Relations between storm suspended sediment loads at logged subwatersheds in the North Fork and the the HIM control from 1986 to 1995. Post-logging relations were fitted by loess method [Cleveland, 1979]. The top row represents 95-100% clear-cut watersheds.

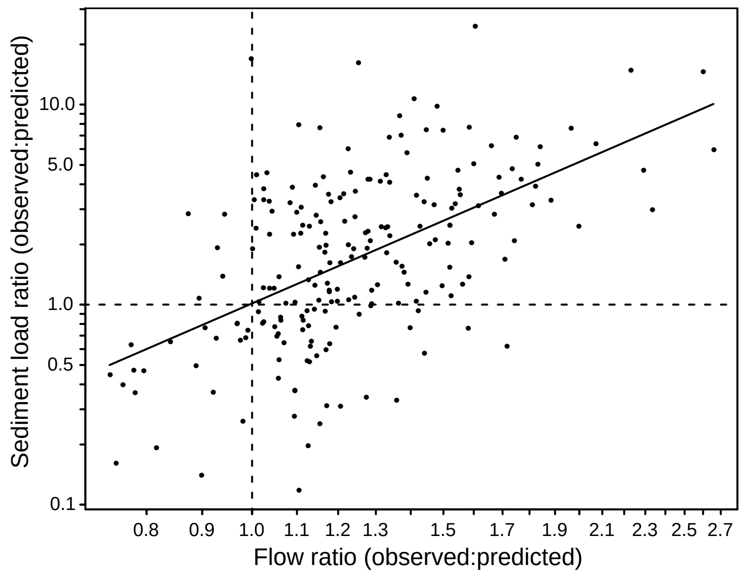

Figure 18. Relation between post-treatment sediment load departures from pretreatment relationship (3) and flow departures $\Delta q_{ij}^{(1)}$. Departures are expressed as the ratio of observed to predicted response.

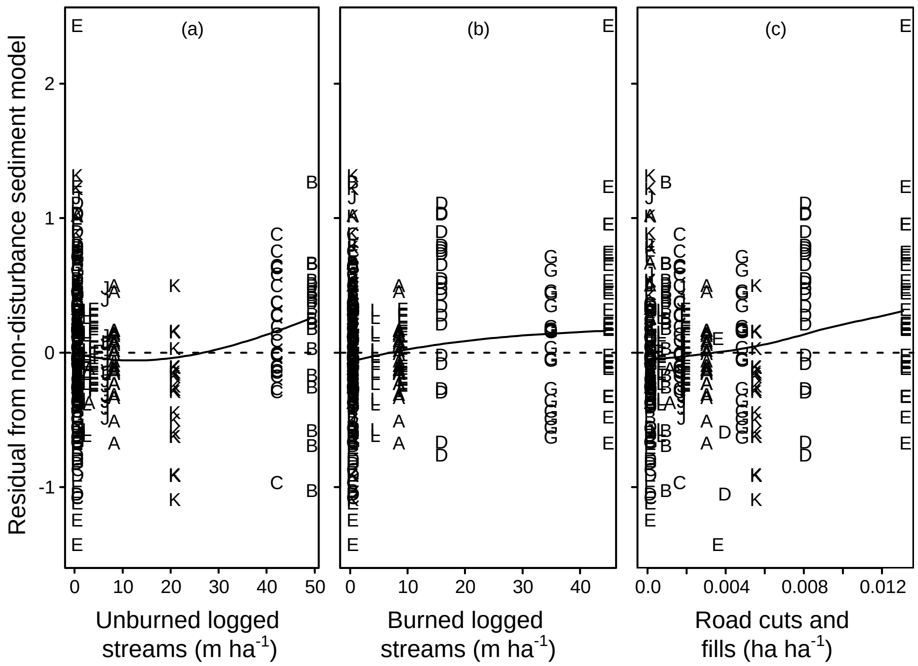

Figure 19. Relation between sediment load residuals and disturbance per unit watershed area. Curves are fit by loess method [Cleveland, 1979] to least squares residuals from the model: $\log(y_{ij}) = \beta_{0i} + \beta_{1i}\log(y_{(\mathrm{H\,I})\,j}) + \beta_2 \Delta q_{ij}^{(1)} + \varepsilon_{ij}$. Disturbance variables shown are (a) length of stream in burned clear-cut areas, (b) length of stream in unburned clear-cut areas, and (c) road cut and fill area. Letters designate watersheds (e.g. G is watershed GIB).

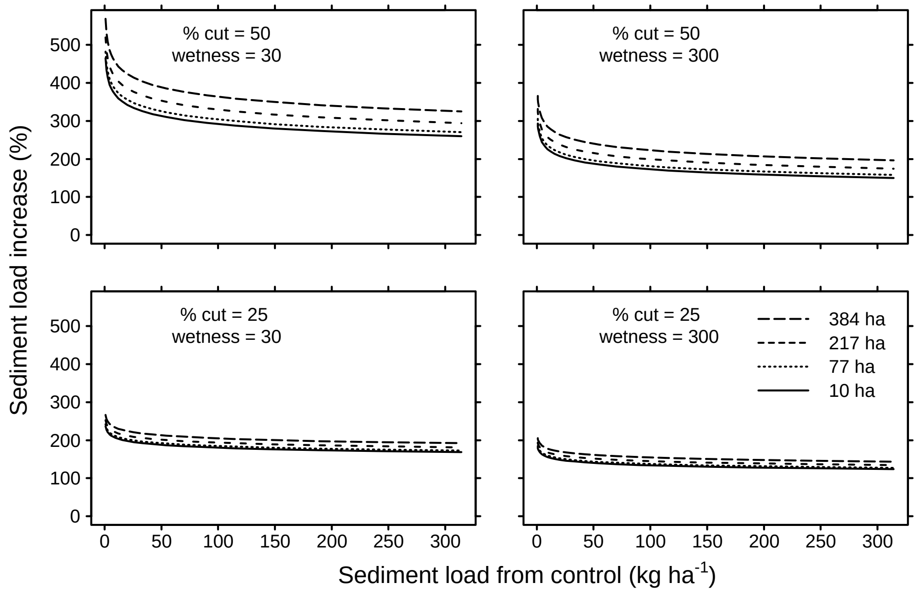

Figure 20. Effect of watershed area on predictions from sediment model {(24),(17),(18)} for two levels of cutting and two levels of antecedent wetness. Watershed areas are those of ARF, FLY, DOL, and BAN (Table 1). Predictions are for first year after cutting with $x_{ij}^{(1)} = x_{ij}^{(2)} = 12 \text{ m ha}^{-1}$.

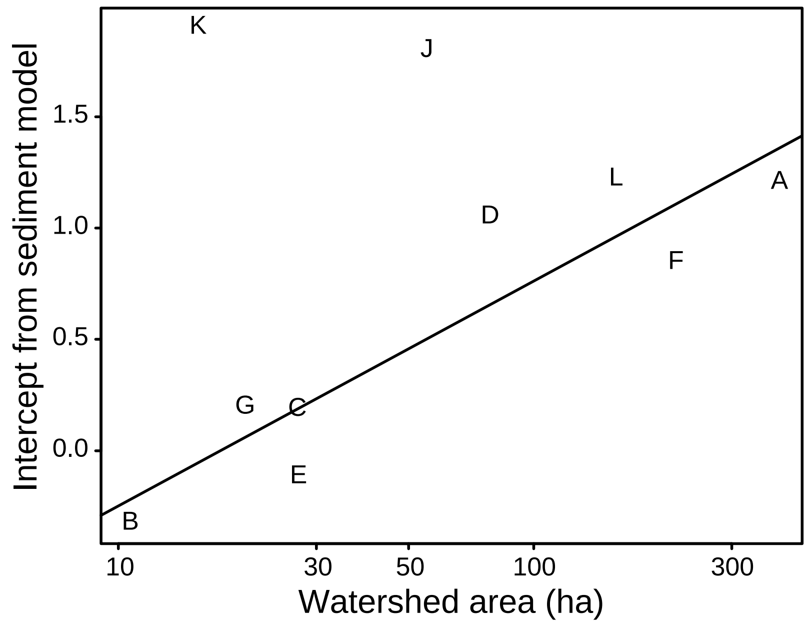

Figure 21. Relation between watershed area and fitted intercepts b_{0i} from model {(24),(17),(18)}, with β_6 fixed at zero. Watersheds JOH (J) and KJE (K) are omitted from regression. Letters designate watersheds (e.g. G is watershed GIB).

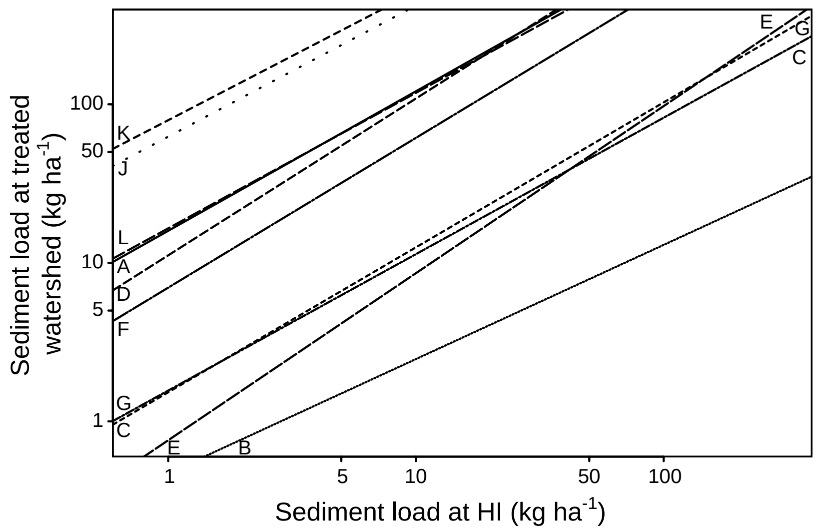

Figure 22. Regression lines for each watershed based on intercepts b_{0i} and slopes b_{1i} of sediment model {(24),(17),(18)}, with β_6 fixed at zero. Letters designate watersheds (e.g. G is watershed GIB).

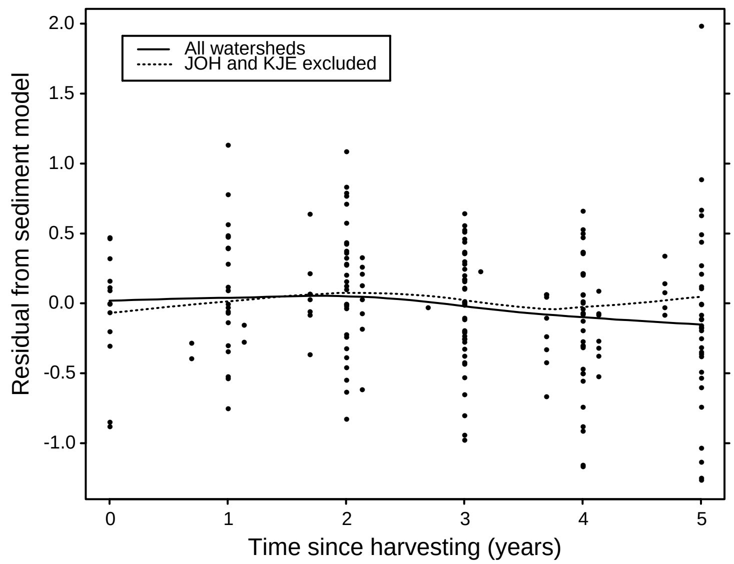

Figure 23. Relation between residuals from sediment model {(24),(17),(18)} and time after logging. Curves are fit by loess method [Cleveland, 1979], with and without the anomalous watersheds JOH and KJE.

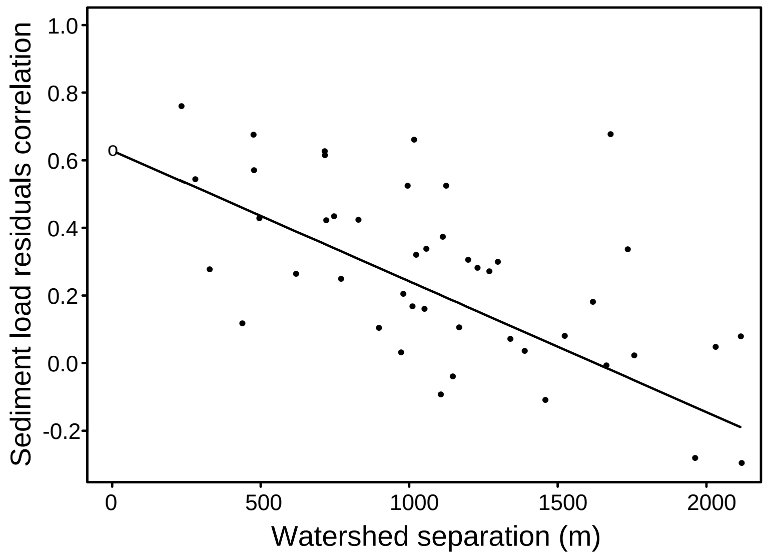

Figure 24. Relation between sediment residuals correlation and distance between watershed centroids. Residuals are from maximum likelihood fit to sediment model {(24),(17),(18)}. Curve depicts equation (17), with estimated parameters $\hat{\theta}_1$ and $\hat{\theta}_2$.

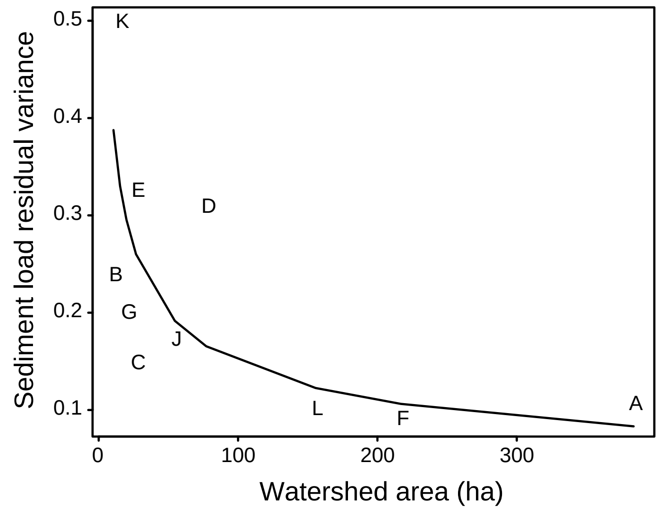

Figure 25. Relation between variance of sediment residuals and watershed area. Residuals are from maximum likelihood fit to model {(24),(17),(18)}. Curve depicts equation (18) with estimated parameters $\hat{\theta}_3$ and $\hat{\theta}_4$. Letters designate watersheds (e.g. G is watershed GIB).

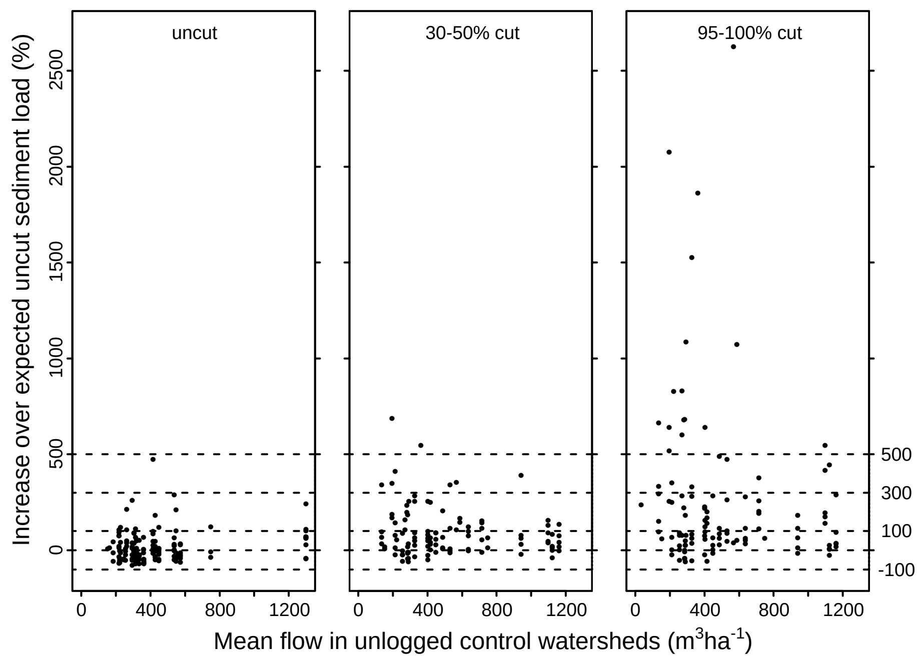

Figure 26. Percentage increase over expected uncut storm sediment load as related to mean of storm runoff volume in HIM control watersheds for uncut (before treatment), partly (30-50%) clear-cut, and (95-100%) clear-cut watersheds. Bias-corrected predictions are from model {(25),(17),(18)} with disturbance set to zero.

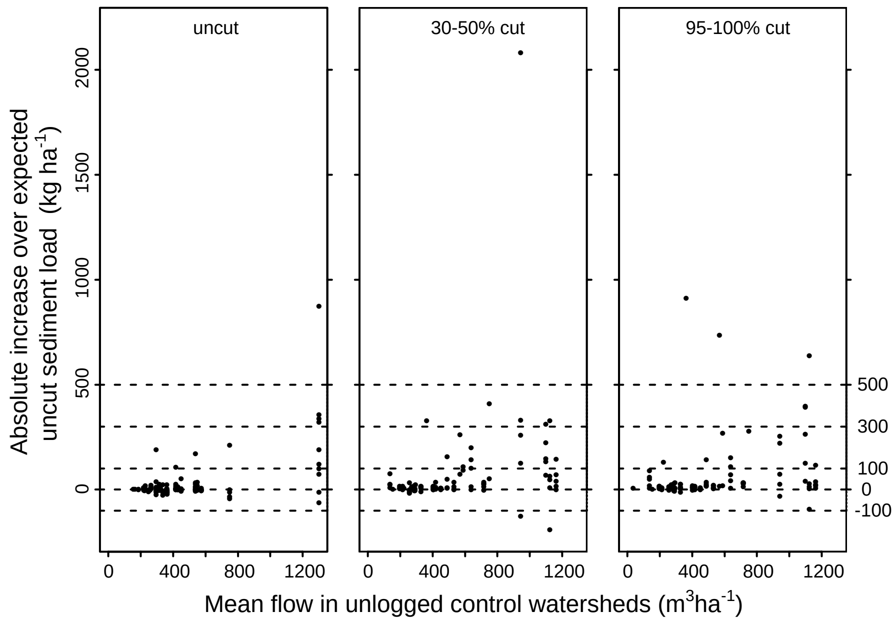

Figure 27. Absolute increase over expected uncut storm sediment load as related to mean of storm runoff volume in HIM control watersheds for uncut (before treatment), partly (30-50%) clear-cut, and (95-100%) clear-cut watersheds. Bias-corrected predictions are from model {(25),(17),(18)} with disturbance set to zero.

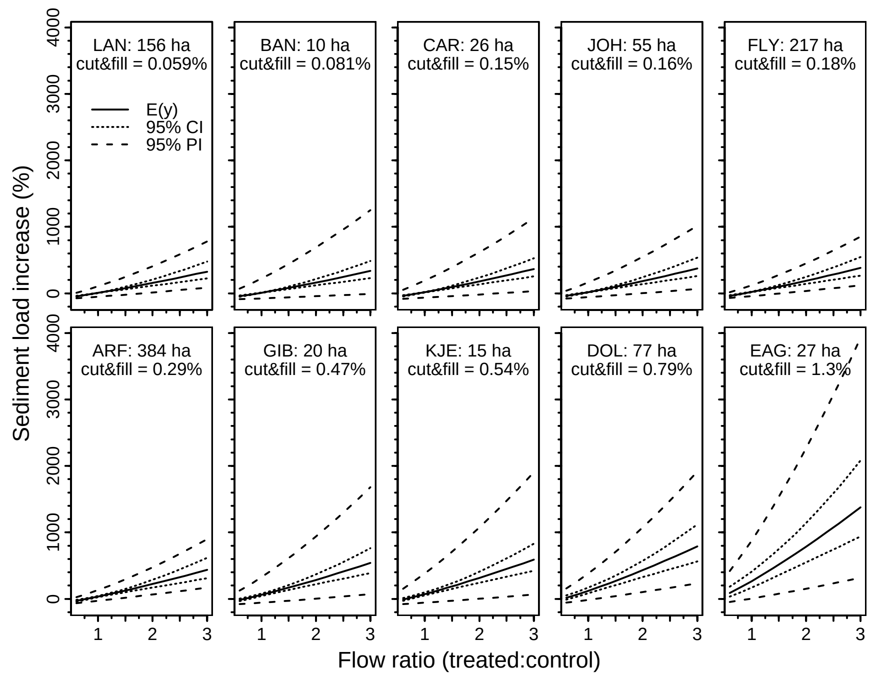

Figure 28. Predictions of sediment load as a function of flow ratio ($\Delta q_{ij}^{(2)}$) based on sediment load model {(25),(17),(18)}, with area interaction term for cumulative impacts (β_4) fixed at zero. Expected increases and 95% confidence (CI) and prediction (PI) intervals are shown for each treated watershed, ordered by proportion of the watershed occupied by road cuts and fills.

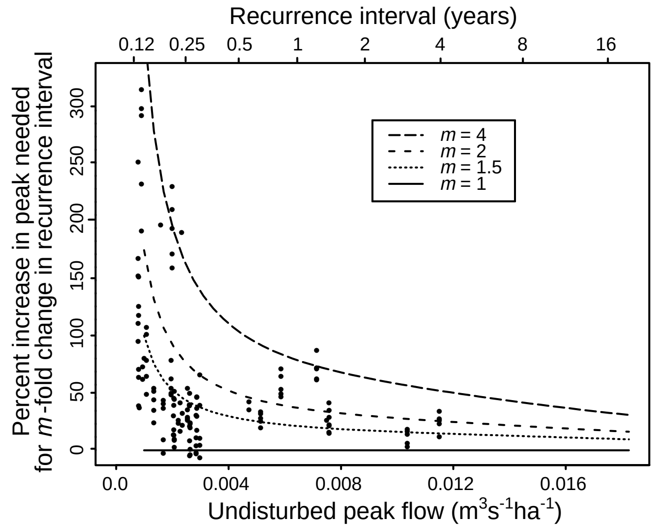

Figure 29. The curve shows the percentage increase in peak flow necessary to reach a size that formerly had 1 to 4 times the recurrence interval. The data points are from Figure 13 (third frame), representing the observed percentage increases in storm peak flow (based on the HI control, plotted on the abscissa) in 95-100% clear-cut watersheds.

Table 1. Basic watershed data and percentage in various conditions. Cut area includes portions of stream buffer zones corresponding to the proportion of timber volume removed.

Water-shed	Area (ha)	Cut area	Cable	Trac-tor	Road+ Lndg	Total Bare	Total Burnt	Dates logged
ARF	384	45.5	35.1	7.1	1.8	2.9	24.0	Spr89-Win92
BAN	10	95.0	77.3	13.4	2.6	3.2	0.0	Fall91
CAR	26	95.7	2.1	9.2	2.8	4.4	0.0	Fall91-Win91
DOL	77	36.4	27.4	5.9	2.5	3.7	33.9	Fall90-Fall91
EAG	27	99.9	79.0	15.4	4.9	8.5	97.8	Fall90-Fall91
FLY	217	45.4	34.6	7.6	1.6	3.0	30.4	Spr89-Sum91
GIB	20	99.6	54.9	39.4	4.2	7.9	98.2	Spr91-Sum91
HEN	39	0.0	0.0	0.0	0.0	0.0	0.0	
IVE	21	0.0	0.0	0.0	0.0	0.0	0.0	
JOH	55	30.2	26.4	1.3	2.0	2.1	0.1	Spr89-Fall89
KJE	15	97.1	85.2	3.9	6.5	6.9	0.0	Spr89-Fall89
LAN	156	32.2	27.8	1.9	1.0	1.9	20.3	Spr89-Spr90
MUN	16	0.0	0.0	0.0	0.0	0.0	0.0	
NFC	473	12.7	38.6[a]	7.6[a]	2.0[a]	3.2[a]	19.5[a]	Spr85-Spr86
		+36.9	38.6	7.6	2.0	3.2	19.5	Spr89-Win92

[a] not measured; assumed equal to Spr89-Win92 disturbance proportions

Table 2. Comparison of suspended sediment load estimation by time interpolation, stage interpolation, and SALT algorithms. The load was estimated for 5000 simulated SALT samples from each storm event.

				Percent RMSE			Percent Bias		
Station	Start of Storm	Load (kg/ha)	$\bar{n}$	Time Interp	Stage Interp	SALT	Time Interp	Stage Interp	SALT
ARF	950109	178.6	21.2	6.0	6.7	12.2	-2.3	-2.8	0.1
ARF	950113	123.6	22.9	2.8	3.4	8.2	-1.6	-2.0	0.1
ARF	950308	122.4	32.6	4.1	4.1	7.6	-0.3	-0.5	0.0
ARF	950108	99.2	8.6	14.2	14.6	19.8	-6.0	-7.2	-0.0
ARF	940216	33.6	16.5	7.0	6.7	10.0	-3.7	-3.5	-0.2
Mad	821214	846.3	41.8	2.1	1.8	10.0	0.0	-0.3	-0.1
Mad	830209	527.2	36.0	4.2	4.1	13.8	0.4	-1.3	0.1
Mad	830117	198.0	40.8	2.2	2.6	7.2	-0.4	-0.9	0.1
Mad	830225	134.4	22.9	7.8	7.6	19.3	-1.6	-2.6	0.3
Mad	831223	42.8	18.1	5.8	5.4	13.6	-2.7	-2.7	0.0
Mad	830221	33.2	15.7	7.5	8.1	16.1	-4.0	-4.9	-0.3
Mad	830212	27.2	14.0	8.1	7.4	16.2	-3.2	-3.9	0.0
Mad	830218	25.4	14.1	14.7	15.1	22.3	-3.4	-4.2	0.0

Table 3. Maximum likelihood parameter estimates for storm peaks model {(10),(16),(18)}, excluding β_{0i} and β_{1i}. p_N is normal probability value for H_0: $\beta = 0$.

Parameter	Effect	Estimate	Standard Error	p_N
β_2	Recovery	0.0771	0.0183	<0.0001
$\beta_3^{(1)}$	Fall logging	0.5939	0.0996	<0.0001
$\beta_3^{(2)}$	Winter logging	0.0000	0.2843	1.0000
β_4	Amount logged	1.1030	0.3409	0.0012
β_5	Storm size interaction	-0.0963	0.0484	0.0468
β_6	Wetness interaction	-0.2343	0.0251	<0.0001
β_7	Watershed area interaction	3.553E-4	2.861E-4	0.2142
θ_1	Correlation shape parameter	2.809E-3	6.188E-4	<0.0001
θ_2	Correlation limit parameter	0.4698	0.1564	0.0027
θ_3	Variance magnitude	0.2285	0.0242	<0.0001
θ_4	Variance shape	-0.0937	0.0238	0.0001

Table 4. Maximum likelihood parameter estimates for storm runoff model {(10),(17),(18)}, excluding β_{0i} and β_{1i}. p_N is normal probability value for H_0: $\beta = 0$.

Parameter	Effect	Estimate	Standard Error	p_N
β_2	Recovery	0.0912	0.0143	<0.0001
$\beta_3^{(1)}$	Fall logging	0.8117	0.0910	<0.0001
$\beta_3^{(2)}$	Winter logging	-0.196	0.225	0.3843
β_4	Amount logged	2.3054	0.2646	<0.0001
β_5	Storm size interaction	-0.1103	0.0467	0.0181
β_6	Wetness interaction	-0.2362	0.0236	<0.0001
β_7	Watershed area interaction	6.481E-4	2.578E-4	0.0119
θ_1	Correlation intercept	0.6697	0.0587	<0.0001
θ_2	Correlation slope	-1.898E-4	4.962E-5	0.0001
θ_3	Variance magnitude	0.1987	0.0190	<0.0001
θ_4	Variance shape	-0.0873	0.0209	<0.0001

Table 5. Percentage and absolute departures from predicted annual storm runoff volume (sum of storms).

	Uncut	30-50% Clearcut	95-100% Clearcut
Mean (%)	2	23	58
Median (%)	2	19	51
Mean ($m^3\ ha^{-1}\ yr^{-1}$)	54	415	1119
Median ($m^3\ ha^{-1}\ yr^{-1}$)	29	387	1050

Table 6. Chow test [Chow, 1960; Wilson, 1978] significance levels for hypothesis of no change in suspended sediment load after logging.

Watershed	HI control	HIM control
ARF	0.1649	0.0215
BAN	0.0128	0.0292
CAR	0.0000*	0.0001*
DOL	0.0198	0.0093
EAG	0.0056	0.0013*
FLY	0.3528	0.0955
GIB	0.0002*	0.0096
JOH	0.0983	0.0476
KJE	0.0026*	0.0384
LAN	0.8018	0.2453

* significant at nominal α = 0.005 (experimentwise error rate = 0.05)

Table 7. Summary of changes in suspended sediment load (summed over storms) after logging in North Fork subwatersheds. Predicted loads are computed from pre-treatment linear regressions between the logarithms of the storm sediment load in the treated watershed and control HIM, the mean of the storm sediment loads at watersheds HEN, IVE, and MUN. Predictions were corrected for bias when back-transforming from logarithmic units. The number of years in the post-logging period varies from 4 to 6, depending upon when the watershed was logged and whether or not monitoring was discontinued in water year 1996.

Treated watershed	Number of years	Observed (kg $ha^{-1}yr^{-1}$)	Predicted (kg $ha^{-1}yr^{-1}$)	Change (kg $ha^{-1}yr^{-1}$)	Change (%)
ARF	4	505	591	-86	-15
BAN	4	85	28	57	203
CAR	5	240	108	132	123
DOL	5	1130	306	824	269
EAG	5	710	210	500	238
FLY	5	536	555	-19	-3
GIB	4	358	119	239	200
JOH	5	667	865	-198	-23
KJE	5	821	1371	-551	-40
LAN	5	420	400	20	5
NFC	6	465	246	219	89

Table 8. Increase in residual sum of squares after dropping variables from least squares fit to model (24).

Coefficient	Variable	SS Reduction
β_2	Change in flow	25.33
β_3	Burned stream channel	10.21
β_4	Unburned stream channel	3.51
β_5	Storm size interaction	1.62
β_6	Watershed area interaction	0.62

Table 9. Maximum likelihood parameter estimates for suspended sediment load model {(24),(17), (18)}, excluding β_{0i} and β_{1i}. p_N is normal probability value for H_0: $\beta = 0$. Control is HI, the mean sediment load from watersheds HEN and IVE.

Parameter	Effect	Estimate	Standard Error	p_N
β_2	Change in flow	1.3276	0.1609	<0.0001
β_3	Stream length, burned	0.0376	0.0057	<0.0001
β_4	Stream length, unburned	0.0204	0.0053	0.0001
β_5	Storm size interaction	-0.0051	0.0017	0.0031
β_6	Watershed area interaction	-3.316E-5	1.649E-5	0.0443
θ_1	Correlation intercept	0.6222	0.0846	<0.0001
θ_2	Correlation slope	-3.802E-4	9.218E-5	<0.0001
θ_3	Variance magnitude	1.0841	0.1565	<0.0001
θ_4	Variance shape	-0.2286	0.0338	<0.0001

Table 10. Maximum likelihood parameter for suspended sediment load model {(25),(17),(18)}, excluding β_{0i} and β_{1i}. p_N is normal probability value for H_0: $\beta = 0$. Control is HIM, the mean sediment load from watersheds HEN, IVE, and MUN.

Parameter	Effect	Estimate	Standard Error	p_N
β_2	Flow increase (log ratio)	1.3564	0.1414	0.0000
β_3	Road cut and fill area	107.11	13.071	0.0000
β_4	Watershed area interaction	-0.1822	0.0872	0.0367
θ_1	Correlation intercept	0.6848	0.0643	0.0000
θ_2	Correlation slope	-3.949E-4	7.618E-5	0.0000
θ_3	Variance magnitude	1.1839	0.1473	0.0000
θ_4	Variance shape	-0.2330	0.0290	0.0000

Table 11. Percentage and absolute departures from annual (sum of storms) sediment load predicted from HIM control. Parenthesized values omit outlier in middle frame of Figure 27.

	Uncut	30-50% Clearcut	95-100% Clearcut
Mean (%)	35	73 (67)	212
Median (%)	15	52	109
Mean (kg ha^{-1} yr^{-1})	65	263 (180)	262
Median (kg ha^{-1} yr^{-1})	1	46	59

Table 12. Apparent and cross-validated RMSE for model predictions.

Data Omitted	Data Predicted	Model			
		Peaks[a]	Volume[b]	Sed (HI)[c]	Sed (HIM)[d]
None	All	0.1589	0.1426	0.4584	0.5046
10% at random	All	0.1633	0.1483	0.4900	0.5238
None	Post-treatment	0.1654	0.1560	0.4644	0.5094
10% at random	Post-treatment	0.1692	0.1623	0.4948	0.5291
Systematic by station	Post-treatment	0.1739	0.1676	0.6966	0.6724

[a] model {(10),(16),(18)}, HI control, $\beta_7 = 0$

[b] model {(10),(17),(18)}, HI control, $\beta_3^{(2)} = 0$

[c] model {(24),(17),(18)}, HI control

[d] model {(25),(17),(18)}, HIM control

Table 13. Regression slope of observed versus predicted response.

Data Omitted	Data Predicted	Model			
		Peaks[a]	Volume[b]	Sed (HI)[c]	Sed (HIM)[d]
None	All	1.0039	1.0103	1.0012	0.9986
10% at random	All	1.0028	1.0047	0.9920	0.9947
None	Post-treatment	1.0077	1.0103	0.9921	0.9651*
10% at random	Post-treatment	1.0085	1.0020	0.9825	0.9611*
Systematic by station	Post-treatment	1.0014	0.9998	0.8601**	0.8775**

[a] model {(10),(16),(18)}, HI control, $\beta_7 = 0$

[b] model {(10),(17),(18)}, HI control, $\beta_3^{(2)} = 0$

[c] model {(24),(17),(18)}, HI control

[d] model {(25),(17),(18)}, HIM control

* $0.01 < p < 0.05$ for one-sided test of H_0: slope=1 (with H_A: slope<1)

** $p < 10^{-6}$ for one-sided test of H_0: slope=1 (with H_A: slope<1)

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The incidence and role of gullies after logging in a coastal redwood forest

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A B S T R A C T

The distribution and morphological characteristics of channels were mapped in a redwood forest at Caspar Creek, California, USA, to evaluate the extent to which recent logging has influenced channel conditions in the area. In the North Fork Caspar Creek watershed, second-cycle logging of the early 1990s appears to have triggered increased coalescence of discontinuous gullies within clearcut tributary watersheds, and upstream channel limits in logged watersheds are now located significantly farther upslope than in control watersheds. The magnitudes of observed increases in peakflow after logging are consistent with the change in drainage density. Relations between channel morphological variables and indices of stream power are less well-defined in logged watersheds than in controls, suggesting that logging may have led to disruption of previously established channel forms. Correlations between suspended sediment yields and indices of gully erosion suggest that in-channel erosion associated with hydrologic change is an important source of post-logging sediment at Caspar Creek. Common sediment-control measures, such as use of riparian buffer strips and reduction of road surface erosion, would not be effective for reducing sediment input from this source.

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1. Introduction

Gullying has been triggered by increased runoff and reduced vegetation cover in many areas and is of great concern to land managers (Valentin et al., 2005). Gullies contribute to loss of fertile soils, disrupt transportation networks, and depress water tables in floodplains; and they create persistent sources of sediments that reduce water quality, fill reservoirs, aggrade downstream channels, and impair aquatic habitats. Gullying can increase sediment yields both through incision of existing channels and by expansion of drainage networks into previously unchanneled swales. In the latter case, channel incision may increase transport capacity through the swale, providing hillslope-derived sediment more direct access to downstream channels (Reid, 1989).

Most studies of gullies have been carried out in agricultural lands, grasslands, or arid areas, where the features are most visible and where they most directly challenge land-management activities; gullies in forests are less commonly encountered. Gullying occurred in some areas after conversion of forest to pasture or agriculture (e.g., Gábris et al., 2003; Parkner et al., 2006). In several areas, however, gullies have been identified as major sediment sources within unconverted forest lands. Often forest gullies are associated with road drainage or with areas compacted by logging equipment (e.g., Weaver et al., 1995; Croke and Mockler, 2001), but in other cases gullying appears to be a more generalized response to forest management (Heede, 1991) or to temporary vegetation changes upslope of the forested area (Vanwalleghem et al., 2003) or is an inherent feature of the forested setting (Parkner et al., 2007).

In part because of the long duration of forest management cycles and the inability to detect forest gullies on historical aerial photographs, the incidence of forest gullies and their relation to forest management remain poorly understood, particularly in settings where overland flow is uncommon. Observed changes in gully activity with forest conversion and subsequent reforestation (e.g., Gábris et al., 2003; Parkner et al., 2006) indicate that catchment vegetation can influence gullying and that those influences vary by vegetation type. Forest vegetation commonly has larger roots and produces larger quantities of coarse litter than other vegetation types, and forests provide woody debris that retains sediment on hillsides and in small channels (Maser et al., 1988). Vegetation type is also expected to influence runoff volumes (Bosch and Hewlett, 1982), peakflow magnitudes (Guillemette et al., 2005), and snowmelt timing (Winkler et al., 2005), each of which could influence gully erosion rates during short-term perturbations in vegetation cover. Vegetation can also regulate gully activity at a smaller scale. For example, Molina et al. (2009) have shown that herbaceous and shrubby vegetation on gully floors is effective in trapping sediment.

During recent forestry planning efforts in NW California, unanswered questions were raised concerning the role of management-related channel erosion in sediment production. This paper presents the results of a study implemented in the Caspar Creek Experimental Watersheds of north coastal California, USA, to (i) evaluate the distribution of incised channels and associated headcuts in the area; (ii) assess the relative importance of gully erosion as a sediment

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source there; and (iii) identify potential influences of forest land management on the extent and character of the gullied reaches.

2. Study area

The Caspar Creek Experimental Watersheds (N39°21′ W123°44′) include the 424-ha South Fork and 473-ha North Fork tributaries of Caspar Creek, which drains to the Pacific Ocean 10 km south of Fort Bragg, CA (Fig. 1). The Caspar Creek catchment is deeply incised into a flight of uplifted marine terraces formed over more than 300,000 years on Franciscan sandstone and shale (Merritts et al., 1991). Elevation in the experimental watersheds ranges from 37 to 320 m, with hillslopes steepest near the stream channel and becoming gentler near the broad, rounded ridgetops. About 35% of the slopes are lower than 17° and 7% are steeper than 35°. Longitudinal channel profiles are generally concave, but many include low-gradient reaches (1° to 5°) immediately followed by relatively short steeper reaches (5° to 20°). Mainstem channels and many tributary segments are bordered by narrow valley flats. Radiocarbon dating of charcoal in 3- to 4-m-deep valley fills in upper reaches of the North Fork suggests that deposition began about 7000 ^{14}C YBP and proceeded episodically through the middle to late Holocene (Steven Reneau, Los Alamos National Laboratory, personal communication, 1989).

The climate is typical of coastal watersheds in the region: winters are mild and wet, while summers are mild and dry. About 95% of the average annual precipitation of about 1200 mm occurs in October through April, and most rain falls in storms of long duration but low intensity. Snow is uncommon.

Soils are predominantly gravelly to sandy loams with a typical depth of 1.5 m. The upper meter of the most common soils includes 25 to 50% gravel and cobble and 30 to 50% clay and silt, and has a bulk density of 1.4 to 1.6 Mg m^{-3} (Wosika, 1981). Subsurface stormflow is rapid, and saturated areas are uncommon and drain quickly after storms. Soil pipes are present in most swales and transport a substantial discharge of storm flow to low-order channels (Albright, 1991). Channel flow often becomes spatially intermittent within a few weeks of a winter storm in catchments smaller than 20 ha (L. Keppeler, USFS, personal communication, 2008).

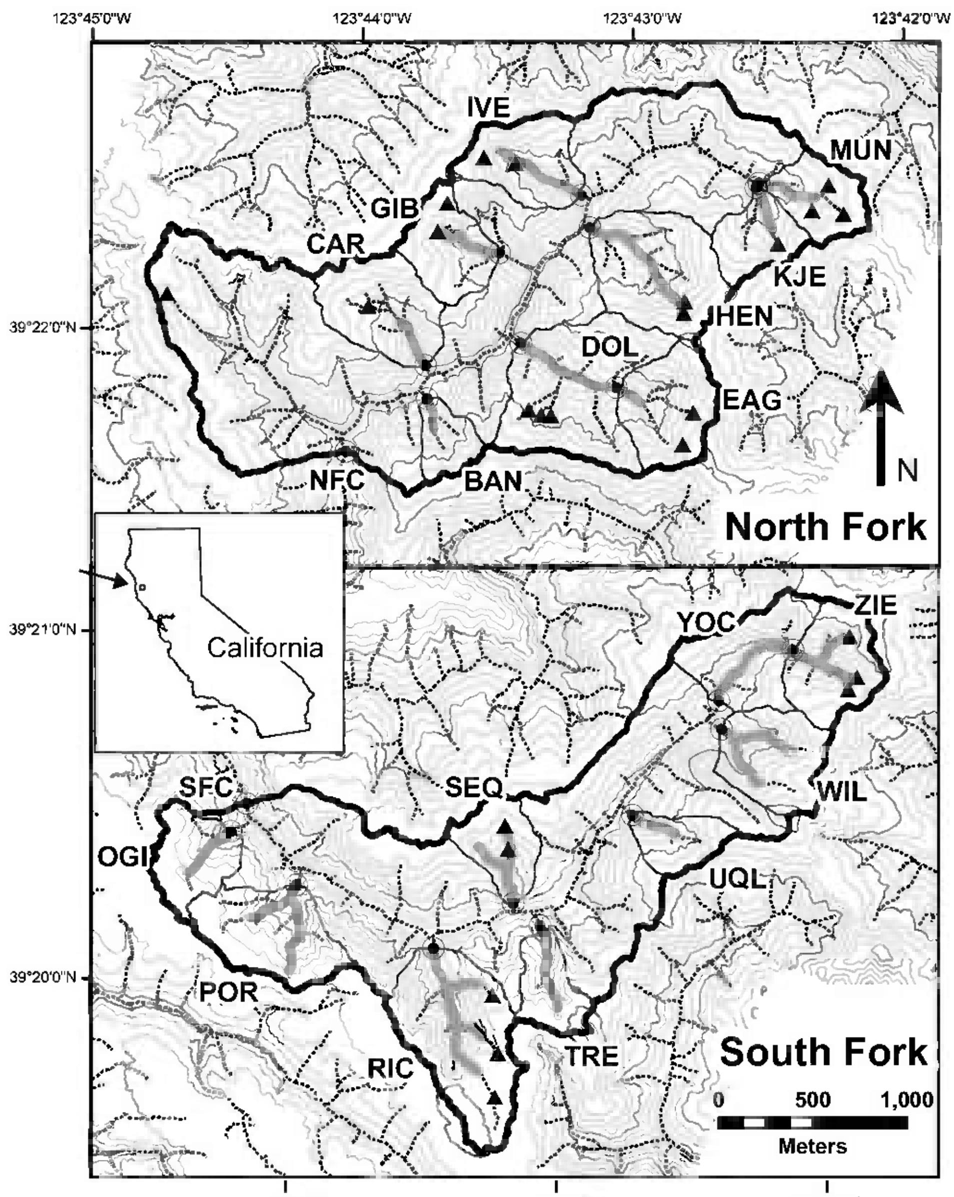

Fig. 1. The Caspar Creek experimental watersheds. Tributaries in which headcuts were mapped are shown in gray, dots indicate location of gaging stations, and triangles indicate locations of mapped channel limits.

Both experimental watersheds are densely forested with second- or third-growth stands dominated by coastal redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*). The entire study area was first logged between 1860 and 1904 using cleared-out channels as routes for log transport (Napolitano, 1998). The low incidence of old-growth wood in today's channels reflects this use, presenting a contrast to the high volumes of very large woody debris present in channels draining old-growth redwood forests of the region (e.g., Keller et al., 1995).

The Caspar Creek Experimental Watersheds were established in 1961 as a long-term research site run jointly by the California Division of Forestry and Fire Protection and the U.S. Forest Service (USFS) Pacific Southwest Research Station. Weirs and gaging stations were constructed by November 1962 on the North and South Forks. After an initial period of calibration between the two watersheds, roads were constructed in the South Fork in 1967, and nearly two-thirds of the stand volume was selectively logged from 1971 to 1973. Tractors commonly dragged logs down tributary valleys, once again severely disturbing some channels.

Twelve years after South Fork logging, thirteen additional gaging stations were constructed in tributaries and along the main stem of the North Fork. The new stations underwent calibration for 4 years. Thirty-seven percent of the North Fork watershed was then roaded, clearcut, and cable-yarded from ridgetops between 1989 and 1992. Selectively logged buffer strips were left around channels with drainage areas of greater than about 10 to 12 ha, and three subcatchments (HEN, IVE, and MUN, Fig. 1) were left as mature second-growth controls within the North Fork watershed. All watersheds but BAN and HEN included some roads constructed long before calibration began, and both new and old roads were located near ridges.

Both episodes of experimental logging resulted in increased runoff and sediment yield, followed by substantial recovery within a decade of logging (Ziemer, 1998). In the North Fork watersheds, increases in sediment production were correlated with increases in storm runoff (Lewis et al., 2001), which were attributable largely to reduced rainfall interception and transpiration after logging (Reid and Lewis, 2007). Ten new gaging stations were constructed on South Fork tributaries in 2000 in preparation for a third experiment.

Stands representing three broad categories are now present in the North and South Fork watersheds. In North Fork control watersheds, the overstory is dominated by 100- to 140-year-old redwoods and Douglas-firs with diameters at breast height (DBH) of about 70 to 150 cm. A subcanopy composed of hardwoods (principally tanoak, *Lithocarpus densiflora*) and small conifers is present at most sites, and shrubs, ferns and herbs provide a discontinuous ground cover. Total stand density for stems larger than 12.7 cm DBH is typically on the order of 500 stems/ha, with a basal area of about 100 m^2 ha^{-1} as of 1998. South Fork stands reflect the selection logging of the early 1970s, having a higher density of small trees than North Fork stands but including a similar range of stem sizes. South Fork stands typically have 600 to 1200 stems/ha and basal areas of 65 to 85 m^2 ha^{-1} as of 2009. The North Fork watersheds logged in 1989–1992 were clearcut, leaving only a disturbed and discontinuous cover of shrubs and herbs. Regrowth has been rapid, and herbicides were applied to watersheds EAG and GIB in 1994 and again in 1996 to control growth of shrubs and hardwoods, while logged watersheds BAN, CAR, and KJE were not treated. Logged watersheds were then thinned in 1998 (KJE) or 2001, leaving about 800 to 2000 stems/ha (basal area: 3 to 5 m^2 ha^{-1}) and reducing crown area to about 20% of that present before thinning.

Most tributary segments with catchments larger than 1.9 ha show evidence of active incision characteristic of gullying, such as steep, raw banks, eroding headcuts, quasirectangular cross sections, and low width–depth ratios (Fig. 2). Old-growth roots often span channels or form headcut lips, and many old-growth stumps are now being undermined by gully-bank erosion and are toppling into channels.

Fig. 2. Headcut in MUN tributary, North Fork Caspar Creek. Note knapsack for scale; headcut is 1 m high.

Much of the valley-axis area thus had supported trees for centuries, and gullies have expanded significantly since the trees were cut during first-cycle logging. Either previous gullies were limited in extent or earlier episodes of extensive gullying had stabilized by about 400 years ago. Many gullies in both confined reaches and valley flats excavate saprolite, also suggesting that the current extent of gullying is unprecedented.

At gully headcuts, the stream typically flows over a lip reinforced by a large root or piece of woody debris and scours a plungepool below. Several modes of headcut erosion are observed in the area: (i) gradual backwasting of the exposed face through ravel, spalling, and tractive erosion, (ii) sapping or tunnel erosion, with rapid retreat occurring when the upstream tunnel enlarges enough that a portion of the roof collapses, (iii) block failures induced by undercutting from backwasting, plungepool erosion, or sapping, and (iv) rapid tractive erosion when a reinforced lip loses its influence, causing a sudden drop in base level to the upstream reach. Most headcuts and associated plungepools show complex forms modified by woody debris, roots, bedrock, boulders, and in-place or toppled bank vegetation; few exhibit the regular morphology typical of grassland gullies. Little vegetation other than moss is present in growth position on gully floors.

3. Methods

The study relies on four kinds of data: (i) measurements of gully distribution and characteristics recorded during field surveys carried out during 2000–2002 and 2006–2008; (ii) measurements of process rates monitored within a subset of the gullies; (iii) sediment gaging records from gullied tributaries; and (iv) information from earlier studies of surface erosion and landslide distribution in the area.

3.1. Gully distribution and dimensions

Channel characteristics were mapped along the main axes of 16 tributaries from near the mouth of the channel to a point above which the drainageway is no longer primarily in the form of an active channel; this aspect of the study is described in more detail by Dewey (2007). Above most of the upstream mapping points in North Fork control channels and South Fork channels are 1 to 4 ha of swale in which either unchanneled reaches predominate or most of the channel is inactive and filled with duff. Additional pipe collapses and discontinuous gullies occur upstream of mapped reaches, and mapped reaches include occasional unincised zones. During this phase of the study, the upstream end of the channel could not be readily found in some clearcut watersheds where channels were obscured by dense regrowth and logging debris. Hillslope gullies and subsidiary tributaries generally were not mapped, but both branches of the main tributary channel were mapped if the dominant fork could not be identified.

In North Fork tributaries, headcut locations and channel width and depth measurements were added to a preexisting channel map. In the South Fork, channels had not previously been mapped, so tributary thalwegs were mapped relative to tapelines established from surveyed benchmarks. Tapeline slope was calculated from surveyed endpoint locations or measured with a hand level; and a stadia rod was used to measure bank height, channel width, and thalweg elevation relative to the tapeline.

Bank-to-bank width and bank-to-thalweg depth were measured at 1074 locations along 3290 m of valley axis in eight North Fork tributaries and at 2124 locations along 5670 m in eight South Fork tributaries. Depth is reported as an average of measurements taken relative to the left and right banks, and depth and width are interpolated linearly between measurement points to allow estimation of average values for 25- and 50-m channel segments. In North Fork tributaries, the product of depth, width, and channel increment length is summed by 2-m channel pixels to estimate channel-segment volumes. South Fork volumes are estimated by summing values for measurement-bounded channel increments. Measurements from 18 surveyed cross sections indicate that the product of thalweg depth and bankfull width overestimates cross-sectional area in these tributaries by a factor of 1.40 (95% confidence interval: ± 0.09), with no significant dependence on width-to-depth ratio, and this value is used as a correction factor for estimating volumes.

Each mapped channel contains many vertical or near-vertical steps in its long profile. These features were classified as headcuts for this study if the step face included some component of material other than woody debris or bedrock. The location and height (relative to the plungepool nadir) of almost all headcuts higher than 0.3 m were recorded, but only headcuts higher than 0.44 m are considered in this analysis in order to ensure a consistent resolution between tributaries. The distribution of recorded heights suggests that 0.26- to 0.44-m headcuts would account for 15 to 20% of the headcuts higher than 0.26 m, or about 7 to 10% of the total headcut height, so exclusion of these features from the data set is not expected to substantially influence conclusions. A 2-m digital elevation model (DEM) was used to estimate drainage area and elevation at each channel pixel in the North Fork, while results from the South Fork are based on information from a 10-m DEM.

To evaluate the extent of the channel network, upstream channel limits were subsequently mapped in 10 forested tributaries and 7 clearcut tributaries in North Fork watersheds and in 8 South Fork catchments (Fig. 1). Because channels are discontinuous in their upstream reaches, the limit was defined for this portion of the study as the upper boundary of the upstream-most channel segment longer than 5 m that has well-defined banks and appears to have carried flow within the previous decade.

We evaluate the distribution of incised channels and associated headcuts by testing for patterns of association between headcut characteristics or channel form descriptors and such factors as drainage area, stream power, and treatment category. Because channel characteristics vary by position in the watershed and because different sections of each watershed were logged or mapped (Table 1), long-profile variations in channel and headcut characteristics must be accounted for if differences in characteristics between treatment groups (i.e., control, recently clearcut North Fork, and selectively logged South Fork watersheds) are to be validly identified. To do so, each channel or headcut characteristic identified for 25- or 50-m reaches within a treatment group was first regressed individually against catchment area, channel gradient, and a stream power index (the product of catchment area and segment gradient, as defined by Montgomery and Dietrich, 1989). If a significant relationship to one of these variables was found, multiple regression was used to determine if treatment is an additional significant variable, with significance assessed at the 0.05 level throughout the sequence of analyses. Comparisons are restricted to data from

Table 1
Mapped tributaries at Caspar Creek; channel attributes are reported here only for the mapped portion of the channel network.

	Gaged area (ha)	Percent logged	Percent roads[a] 1986	1995	Gage record	Area above survey (ha)	Total channel length (m)[b]	Buffer strip length (m)	Surveyed channel length (m)	Surveyed channel volume (m^3)	Surveyed bank area (m^2)	Headcuts >0.44 m
North Fork clearcut 1989–91												
BAN	10	95	0.0	3.5	1985–1995	5.1	875	88	157	63	106	19
CAR	26	96	4.7	6.4	1985–now	14.2	2343	242	284	611	646	15
DOL[c]	77	36	–	–	1985–now	26.6	–	n.a.	586	1243	1174	37
EAG	27	100	0.8	4.6	1985–now	8.9	2406	105	149	339	383	6
GIB	20	100	5.2	7.8	1985–1995	1.2	2115	107	360	312	427	41
KJE	15	97	0.3	6.3	1985–1995	3.7	1179	235	253	1511	880	14
North Fork control												
HEN	39	0	0.1	0.1	1985–now	3.7	3341	n.a.	675	640	821	78
IVE	21	0	7.6	7.6	1985–now	3.4	948	n.a.	494	512	625	53
MUN	16	0	5.2	5.2	1985–1995	6.9	1293	n.a.	330	633	467	21
South Fork selectively cut 1971–73												
OGI[d]	18	50	–	–	2000–now	5.2	–	n.a.	350	337	495	18
POR	32	54	–	–	2000–now	1.7	–	n.a.	780	1207	1135	41
RIC	49	67	–	–	2000–now	1.3	–	n.a.	1144	1356	1477	75
SEQ	17	67	–	–	2000–now	1.2	–	n.a.	668	845	810	45
TRE	14	67	–	–	2000–now	1.8	–	n.a.	489	464	665	33
UQL	13	67	–	–	2000–now	3.4	–	n.a.	269	258	316	42
WIL	26	67	–	–	2000–now	3.5	–	n.a.	699	687	867	94
YOC[c]	53	67	–	–	2000–now	25	–	n.a.	575	645	750	41
ZIE	25	67	–	–	2000–now	1.4	–	n.a.	694	458	706	57

[a] Percent of area in roads and landings; values are approximate due to potential instability of drainage patterns on ridge-top roads.
[b] Includes all sub-tributaries, as estimated from DEMs on the basis of expected headwater catchment area; value listed for DOL is exclusive of length in EAG.
[c] DOL is located downstream of EAG, and YOC is downstream of ZIE; surveyed values listed for DOL and YOC pertain only to the segment of the watershed below the upstream gage.
[d] OGI includes 27% private land that has been selectively logged several times since 1971.

watershed areas of 3 to 30 ha to ensure that samples are available from similar locations in both logged and control watersheds.

3.2. Gully erosion rates

Tributaries YOC, ZIE, and MUN (Fig. 1) were selected for monitoring of headcut changes over a multi-year period using one of three methods. Five headcuts (group I) were surveyed in detail with an electronic total station in the summers of 2000 and 2002. The resulting maps are capable of revealing local changes of more than about 5 cm. At six other headcuts (group II), transects were surveyed first by survey laser and later by rod and tape to measure headcut retreat and bank erosion upstream and downstream of the headcut; survey error is estimated to be about 2 cm. An additional 41 headcuts (group III) were visually checked for obvious retreat, with changes >30 cm expected to be visible. Headcuts were revisited during the summer of 2003 and in February 2004, and additional measurements were made at three rapidly eroding headcuts. Nine headcuts from groups I and II at which benchmarks could be relocated were again surveyed in 2006 and 2008 using an electronic total station. Cross-section measurements from groups I and II also allowed monitoring of bank erosion on 15 near-vertical banks between the summers of 2000 and 2002; 12 of the 15 measurements are from banks within 4 m downstream of a headcut. Flow conditions over the monitoring period ranged widely: the highest flow recorded at IVE during 24 years of monitoring occurred in December 2005, while all other years produced maximum flows with recurrence intervals of less than 2 years.

During the autumn of 2000, paint was applied to roots at the point at which each emerged from a headcut face to allow long-term monitoring of bank recession at several headcuts. Distances between the wall surface and the painted portions of roots were measured at 43 points on four headcut overfalls during July 2008.

In calculations of erosion rates, a bulk density of 1.5 Mg m^{-3} is assumed when converting rates from units of $m^3\,y^{-1}$ to Mg y^{-1}.

3.3. Suspended sediment transport at gaging stations

We use records from nine tributary gaging stations in the North Fork watershed (Fig. 1; Table 1) to evaluate variations in annual suspended sediment yield as they may relate to gullying; results of the suspended sediment study evaluated by storm period are reported elsewhere (Lewis, 1998; Lewis et al., 2001). Automatic sediment samplers at each gage are programmed to sample at predetermined turbidity levels. These sediment measurements allow ongoing recalibration of the turbidity record at each gage in order to produce a continuous estimate of suspended sediment concentration during storms (Lewis, 1996), and transport rates are then computed from discharge and concentration. Measured storms account for 90 to 99% of sediment output in most years (Jack Lewis, USFS, personal communication, 2007), with the highest percentages for years with large flows, so the approach is expected to account for at least 95% of the suspended load during the period analyzed.

For the current study, analysis requires annual loads rather than storm-period loads, so storm loads had to be estimated at several stations during periods with missing record. Loads at individual stations were estimated for missing portions of the pre-logging record using correlations between peakflow and storm sediment load or between loads at nearby gages if peakflow data were missing. Regressions are not expected to be stable during the post-logging period, so missing values after logging were estimated using ratios between the target storm load and temporally adjacent storm loads at stations with records for the missing storm. Imputed values account for about 4% of the total suspended sediment load.

3.4. Other erosion studies

The incidence of landslides and treethrows mobilizing more than 7.5 m^3 has been recorded in the North Fork watershed since 1986 through “large-event surveys” carried out during post-storm channel inspections. Each event is located on a channel map with reference to existing benchmarks, and the scar is diagramed and dimensions measured. Associations with roads or treethrows are noted if present, and the volume of displaced sediment still present on the scar is estimated.

Surficial erosion has also been evaluated in the area. Rice (1996) reported results of void measurements on erosion plots at 175 hillslope sites and 129 road sites distributed randomly through the North Fork watershed after completion of second-cycle logging. Sample units on logged and unlogged hillslopes were circular 0.08-ha plots, while those on roads were 1.5-m-wide segments of the road prism oriented perpendicular to the road centerline. Processes assessed on plots include rilling, sheet erosion, and soil displacement from yarding. Rice weighted erosion plot data according to the proportion of the sub-watershed logged and the road length present in each and reported results as average total erosion per unit area of each sub-watershed.

Rice's results require reevaluation for the current study because several sub-watersheds had experienced different durations since logging at the time of the survey (1995). Given the large reported differences in erosion between logged and unlogged sites (Rice, 1996), most soil displacement appears to be associated with treatment, so Rice's estimates for rates of road and hillslope surface erosion are here recalculated as erosion per year since the onset of construction of new logging roads in the treated watersheds. Rates for the control watersheds are also calculated assuming a 5-year period of visibility rather than the 10-year period implicitly assumed by Rice (1996).

For the calibration period, hillslope surface erosion is estimated simply as the mean of the rates calculated for the post-calibration period in the three control watersheds. This approach cannot be adopted to estimate road-related erosion during the calibration period because road erosion for the post-calibration period in both logged and control watersheds reflects the period of high-intensity use during logging. Results from Reid and Dunne (1984) suggest rates of surface erosion on gravel-surfaced logging roads are about 75 times higher during high-intensity use periods than during periods of light use. Log hauling typically took place over a 4-month period within each logged watershed, and mainline roads in all watersheds would have undergone longer periods of heavy use as neighboring units were logged. Road erosion rates are assumed to be 75 times higher than “background” road erosion rates for a 4-month period during the post-calibration period, and the corresponding rate for light-use periods is then calculated algebraically from the road erosion plot data (i.e., $\textit{plot total} = (0.33\ \textit{year} \times 75x) + [(\textit{post-calibration duration} - 0.33\ \textit{year}) \times x]$, where x = rate for the light-use period). The mean of estimated light-use rates calculated per unit area of road is then applied to the area of roads present during the calibration period in each watershed.

4. Results

4.1. Gully distribution and characteristics

Gully headcuts are common along all mapped valley axes (Fig. 3, Table 1), with some present in catchments of <1 ha. There are 284 mapped headcuts higher than 0.44 m in North Fork tributaries and 446 in South Fork tributaries, indicating average frequencies of 8.6 and 7.9 headcuts per 100 m of channel, respectively. Differences in gully characteristics between logged and control watersheds are not evident to casual observation (e.g., Fig. 3), indicating that any major

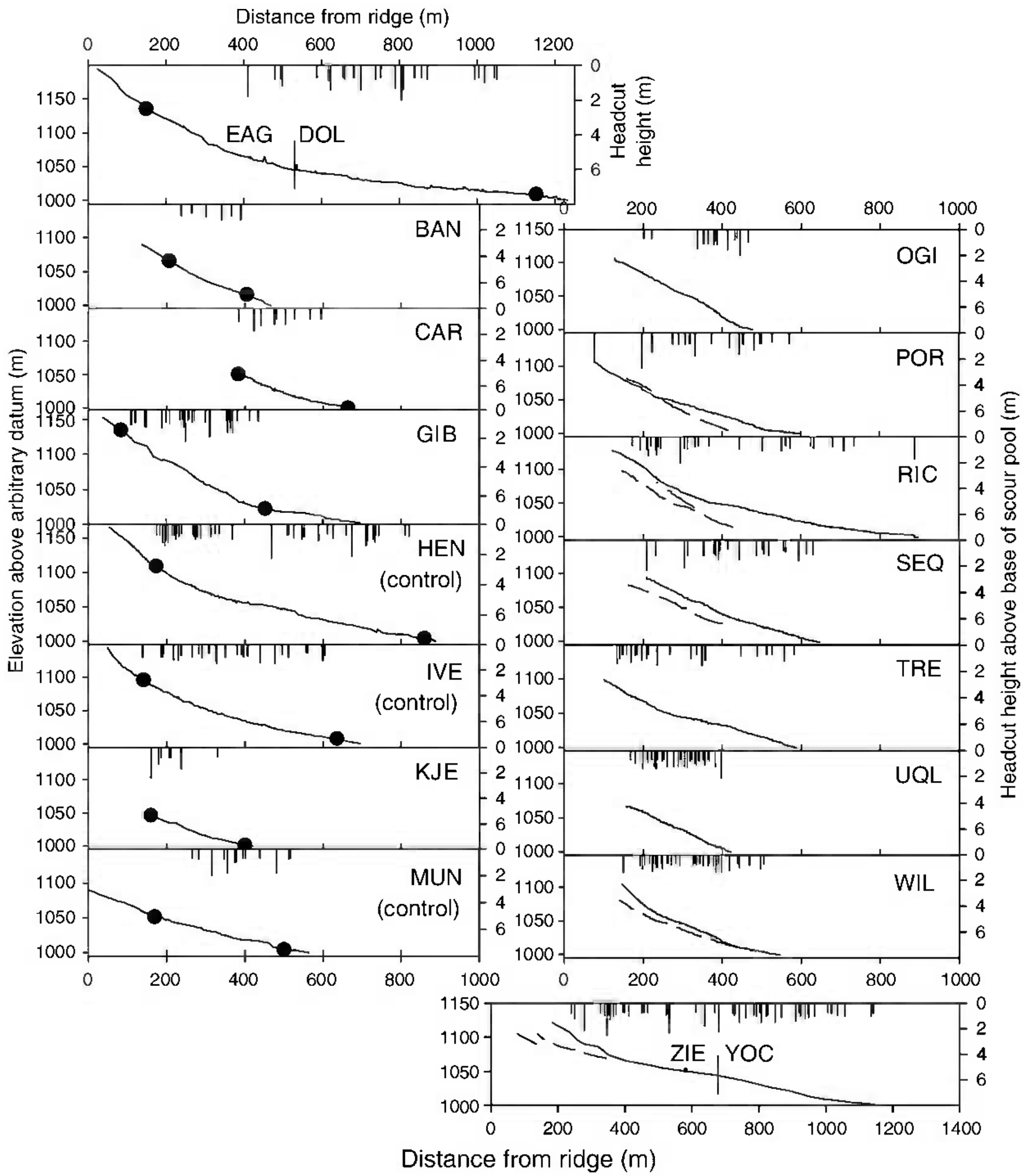

Fig. 3. Surveyed valley-axis profiles in gaged tributaries. Tics indicate the locations and heights of headcuts along the main tributary channel, and dashed lines show profiles of additional mapped branch tributaries. Dots on North Fork profiles indicate the limits of headcut mapping. For clarity, mapped headcuts shorter than 0.64 m are omitted. See Table 1 for watershed descriptions.

incision episode predated second-cycle logging. In particular, the logged KJE catchment was noted to have exhibited particularly well-developed gullies even before the recent logging.

Headcut characteristics are assessed for 50-m channel reaches; analysis of shorter reaches would provide too few headcuts in each sample unit. Headcut height (h) in North Fork tributaries increases weakly but significantly with channel-segment gradient (s) ($\log h = -0.34 + 0.58s$, $p < 0.001$, $r^2 = 0.064$), and the relation shows no significant difference between logged and control tributaries. In contrast, mean headcut spacing decreases significantly with increasing gradient (Fig. 4), and headcuts in control watersheds are spaced significantly more closely than in logged watersheds. Although variance is high, control watersheds show a tendency for the proportion of channel elevation drop accounted for by headcuts to increase with increasing catchment area (Fig. 4C), while logged watersheds show no significant increase (Fig. 4D). For channel segments with catchment areas larger than 10 ha, control watersheds show the mean proportion of elevation drop from headcuts to be 0.53 ± 0.09, compared to 0.26 ± 0.09 in logged watersheds.

Sample frequencies allow assessment of channel characteristics in reaches shorter than those required for headcut assessments. Comparisons of morphological attributes in 25-m channel reaches in control and logged watersheds show patterns similar to that of Fig. 4D: the systematic variations in width, depth, and cross-sectional

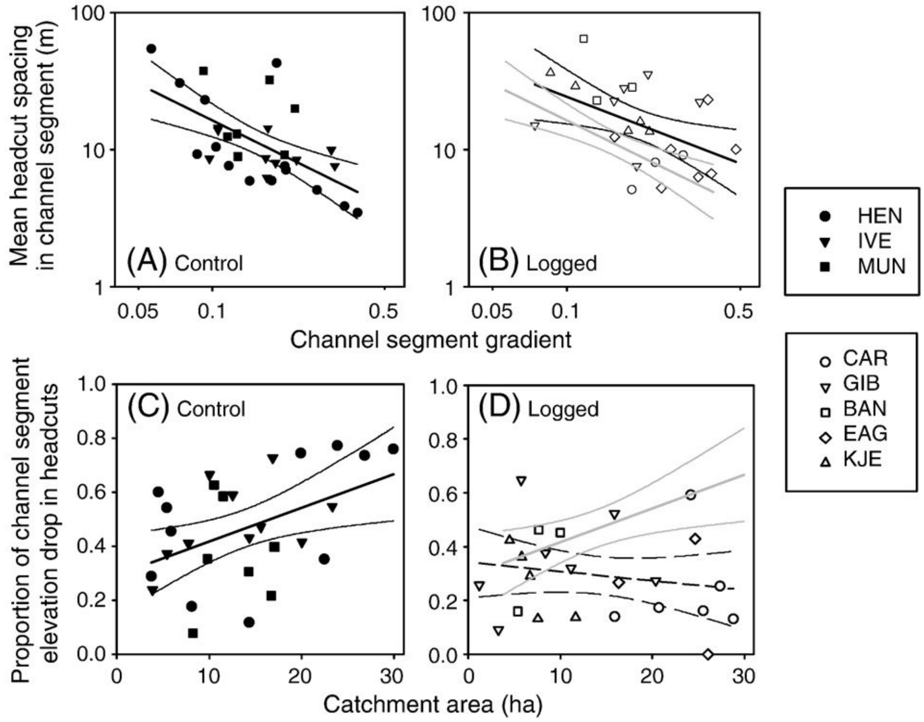

Fig. 4. Headcut spacing as a function of channel gradient in 50-m reaches of North Fork (A) control ($r^2 = 0.36$) and (B) logged ($r^2 = 0.23$) tributaries; and relative importance of elevation drop from headcuts in (C) control ($r^2 = 0.20$) and (D) logged ($r^2 = 0.03$) tributaries of North Fork Caspar Creek. The gray lines in Fig. 4B and D depict the regression and 95% confidence band for controls. Dashed lines indicate relations not significant at the 0.05 level.

area with stream power index expressed in control tributaries are not evident in recently logged tributaries (Fig. 5A). Relations between these attributes and watershed area, however, are strongly expressed in the pervasively gullied KJE tributary (Fig. 5B), in which gullies appear to have coalesced before monitoring began and which exhibits no significant relations to stream power index.

Results for South Fork tributaries, which were selectively logged in the early 1970s, are generally intermediate to those in North Fork logged and control watersheds. Headcut spacing in 50-m reaches and the proportional elevation drop accounted for by headcuts are most similar to those in logged North Fork watersheds (Fig. 6). Morphological data from South Fork tributaries generally show higher variance than found for either logged or control North Fork channels, which may in part result from the use of a l0-m DEM for the analysis rather than the 2-m DEM used for North Fork tributaries.

Mapping of upstream channel limits in North Fork control watersheds shows a mean stream power index of 0.69 ± 0.12 ha at the channel head (Fig. 7, Table 2). Results for the South Fork watershed (0.61 ± 0.09 ha) are not significantly different from those for North Fork control watersheds, while values for clearcut North Fork watersheds (0.34 ± 0.10 ha) are significantly lower than those in either control watersheds or South Fork tributaries. The catchment area at the head of forested channels averages 1.9 ± 0.3 ha, compared to 1.2 ± 0.5 ha for logged tributaries, representing a 28% increase in drainage density within 30-ha watersheds.

4.2. Rates of erosion at headcuts and streambanks

Measured rates of headcut retreat are highly skewed in part because of differences between activity levels at headcuts held in place by roots or woody debris and those not impeded by such features. Of the 52 headcuts observed over a four-year period in MUN, YOC, and ZIE tributaries, three migrated rapidly (>30 cm y^{-1}) during at least one year, for a total of 4.8 m during four headcut-years. The other 204 headcut-years produced no rapid retreat. The resulting mean rate of rapid retreat for all 52 monitored headcuts is 4.8 m for 208 headcut-years, or 2.3 cm y^{-1} per headcut.

For the subset of 11 headcuts that were surveyed in more detail between 2000 and 2002 or 2008, displacement totaled 158 cm over 64 headcut-years that exhibited no rapid retreat, producing a mean retreat of 2.5 cm y^{-1}. Individual retreat rates ranged from 0 to 30 cm y^{-1}, with two headcuts showing no retreat over the 8-year period. Recession monitored around painted roots over an 8-year period indicated an average retreat of 0.7 cm y^{-1} for 32 headcut-years. The combined estimate for gradual retreat is thus 1.9 cm y^{-1}, and the estimated total retreat rate is 4.2 cm y^{-1} with an expected accuracy of about ±50%. Data are too few and variance too high to isolate the effect of the 25-year recurrence interval storm of December 2005. Field observations suggest that many of the changes in headcut location noted between 2002 and 2006 may have resulted from the storm, but that the storm did not cause widespread or severe channel disruption.

As is the case for headcuts, banks show a skewed distribution of rates. Two monitored banks eroded rapidly (6 and 8 cm y^{-1}), while eight showed rates of <1 cm/y; retreat averaged 1.6 ± 1.2 cm y^{-1}. Erosion generally occurred either by block failure–often associated with undercutting–or by more widespread but gradual backwasting and spalling.

4.3. Suspended sediment yields

Lewis (1998) demonstrated that, with the exception of KJE, storm-based sediment yields increased significantly above expected levels after logging; these patterns are evaluated in more detail by Lewis et al. (2001). For the present study, we evaluate annual yields to assess the relative importance of individual watersheds as sediment sources. Mean annual suspended sediment yields at North Fork tributary gaging stations range between 2 Mg km^{-2} y^{-1} at watershed BAN before logging and 68 Mg km^{-2} y^{-1} at KJE after logging (Table 3).

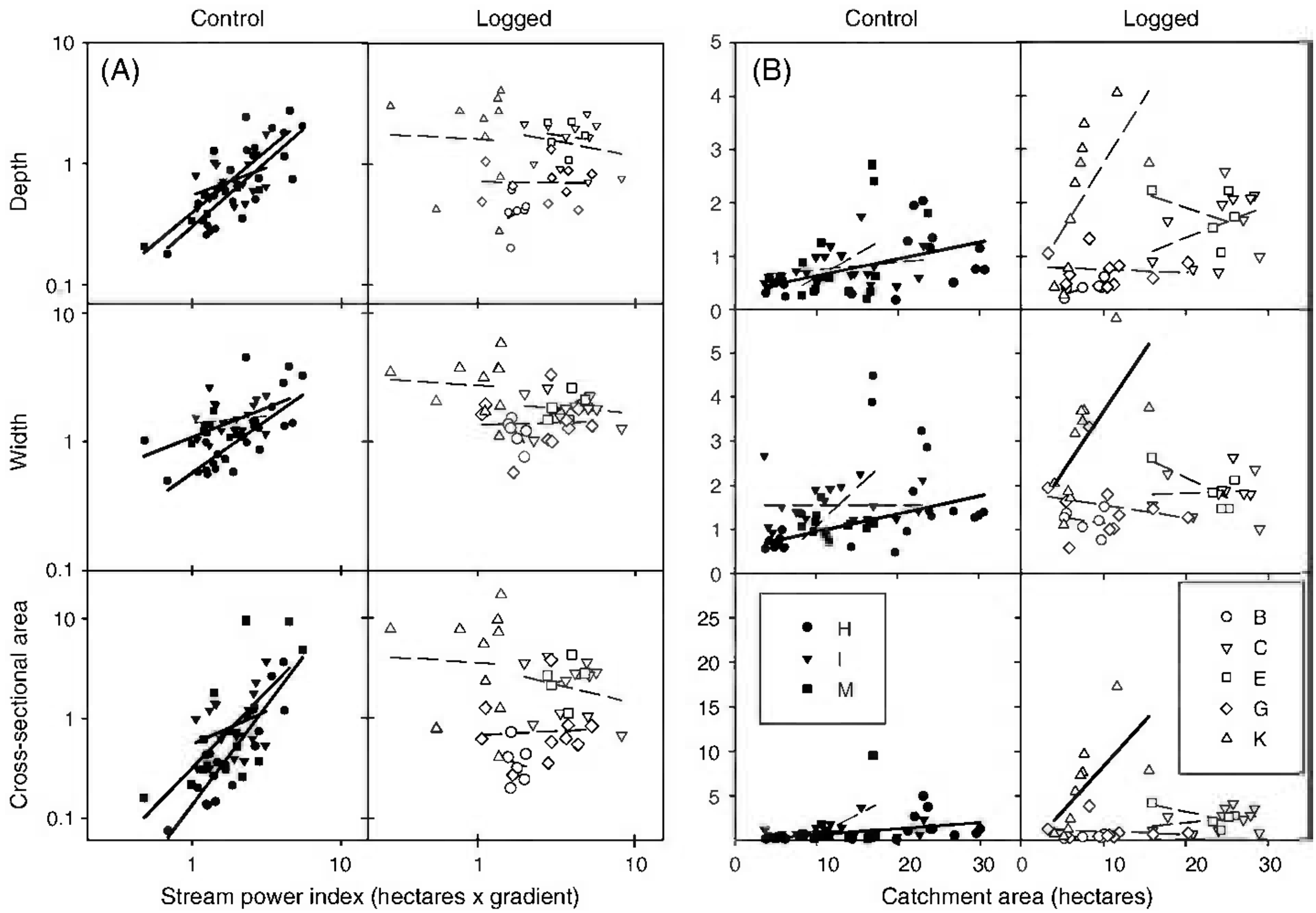

Fig. 5. Variation in mean segment depth (m), width (m), and cross-sectional area (m^2) with (A) stream power index and (B) catchment area in control and logged tributaries of North Fork Caspar Creek. Relations significant at the 0.05 level are depicted by solid lines.

4.4. Rates of erosion from other sources

Plot erosion data reported by Rice (1996) and here recalculated to reflect the post-calibration period reveal strong differences between logged and control watersheds. Logged watersheds show road and hillslope soil displacement rates averaging 2600 ± 1170 Mg km^{-2} y^{-1}, with 6 to 33% of the individual watershed totals associated with road erosion. In control watersheds, displacement averages about 8% of that calculated for the logged watersheds over the analogous period (Table 3), and 0% to 76% is associated with road erosion. Erosion on hillslope plots in control watersheds averages 67 ± 61 Mg km^{-2} y^{-1}. These calculations assume that all erosion voids are visible for 5 years in control watersheds. If the period of visibility is actually longer, as assumed by Rice (1996), the difference between displacements in logged and control watersheds would be accordingly greater. Hillslope and road erosion rates for the calibration period are highly uncertain. Differences in estimated displacement from roads during the calibration and post-calibration periods reflect the increase in road area in the treated watersheds and the expected influence of heavy road use in both treated and control watersheds. Overall, expected road sediment after the onset of road construction increased on average by factors of 16 in treated watersheds and 6 in controls.

Large-event surveys in the gaged watersheds disclosed 8 events capable of contributing sediment to streams during the calibration period and 25 in the post-calibration period. Mapped events were associated with major storms in 1986 (5 events), 1990 (4 events in two storms), 1993 (7 events), and 1995 (17 events in two storms). Field-based estimates for each of these events indicate that an average of about 25% of the sediment displaced was transported off-site. Annualized average volumes of off-site displacement from large events during the post-calibration period (Table 3) are not significantly different at the 0.05 level between treated watersheds (19 ± 14 Mg km^{-2} y^{-1}) and controls (26 ± 2 Mg km^{-2} y^{-1}).

5. Interpretation and discussion

Results of the field study are here interpreted to address three questions: (i) What is the relative importance of gully erosion between logged and forested tributary watersheds at Caspar Creek? (ii) How important is gully erosion relative to other sediment sources in the area? (iii) What mechanisms might contribute to differences in the extent and importance of gully erosion between logged and unlogged watersheds?

5.1. Sediment production from gully erosion in logged and forested tributaries

Once the distribution of channel characteristics and their associations with land-management categories are identified, it is possible to extrapolate information from the sampled portions of a watershed to the watershed as a whole. Measured headcut and bank erosion rates can then be applied to the channel network expected in each setting to allow comparison of average sediment production rates from gully sources typical of forested North Fork watersheds, recently clearcut

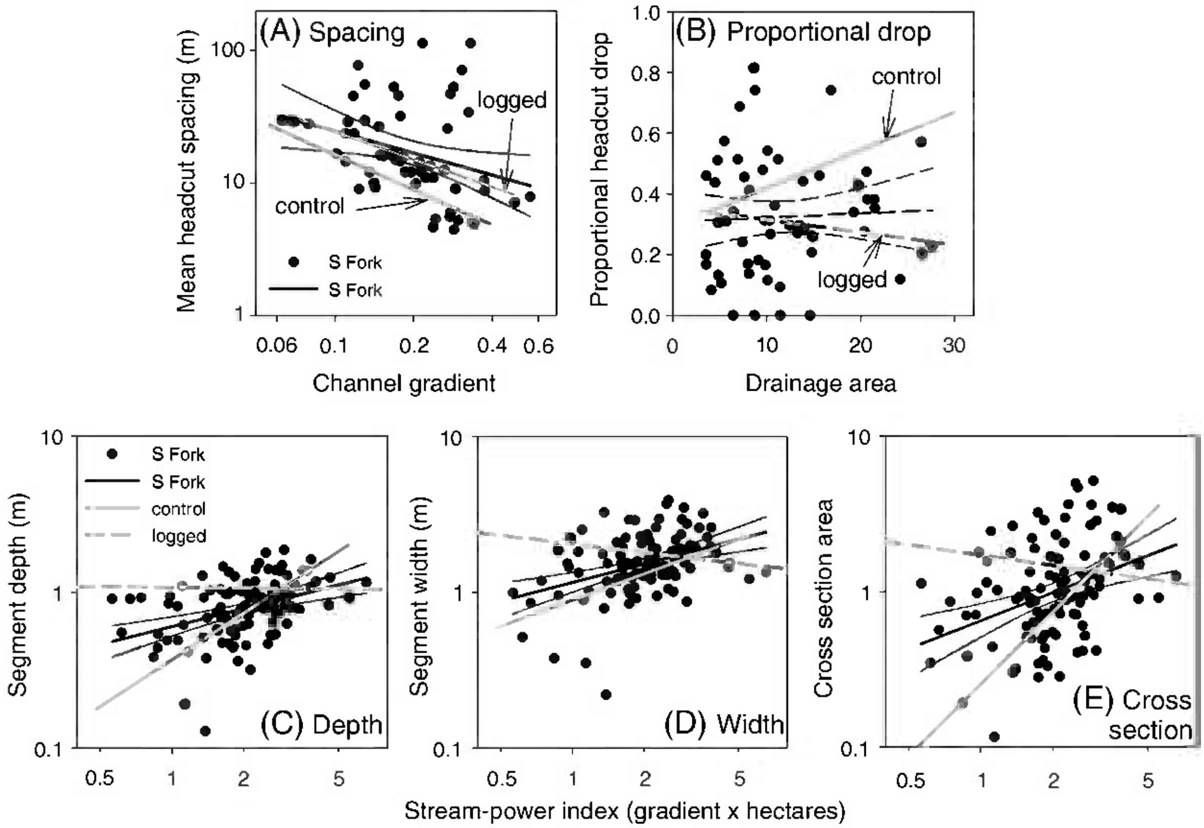

Fig. 6. Patterns of headcut distribution in South Fork tributary channels, and morphological characteristics of South Fork channels. Relations shown in gray depict analogous relations for North Fork tributaries (pooled regressions from Figs. 4 and 5), and relations significant at the 0.05 level are depicted by solid lines.

North Fork watersheds, and selectively logged South Fork watersheds. Calculations of sediment production from bank and headcut erosion are based on sparse data from a limited period, so estimates are not reliable. However, estimated sediment input rates are useful for indicating the potential magnitude of sediment contribution from the gullies and for assessing relative contributions among the three watershed treatment categories.

Sediment production by retreat of headcuts higher than 0.44 m in control watersheds can be estimated if we assume (i) the area of each eroding headcut face is equivalent to the product of the headcut height (relative to the base of the plungepool) and the channel width measured immediately downstream; (ii) the calculated average retreat rate of 4.2 cm y^{-1} for monitored headcuts applies to all headcuts higher than 0.44 m; and (iii) headcut frequency and dimensions in unmapped channels are similar to those in mapped reaches of similar size. Estimated sediment input from this source is then 28 Mg km^{-2} y^{-1} in North Fork control tributaries smaller than 30 ha (Table 2), with 19% of that sediment originating in channels draining areas of 1.9 to 3 ha, which were not characterized in control watersheds during the gully survey. Because headcut face areas are calculated as the product of channel width and the face height above the base of the plungepool, plungepool erosion is implicitly included as a component of headcut erosion.

A portion of the sediment eroded from headcuts and plungepools is deposited immediately downstream on the depositional downstream lip of the plungepool. Lip aggradation is estimated to account

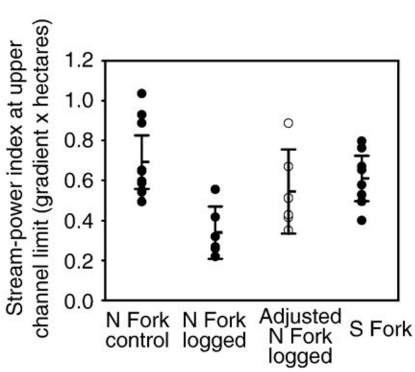

Fig. 7. Distribution of stream power indices at the upper limits of tributary channels. Error bars indicate the 95% confidence interval around the means (horizontal lines). "Adjusted" values account for hydrologic change after North Fork logging (see text Section 5.3.1).

Table 2
Comparison of channel characteristics for three watershed categories; dimensions represent means for 25-m channel segments; 95% confidence intervals are indicated for means.

Attribute	North Fork		South Fork
	Control	Clearcut[a]	
Catchment area at channel head (ha)	1.9 ± 0.3	1.2 ± 0.5	2.3 ± 1.0
Stream power index at channel head (ha)	0.69 ± 0.12	0.34 ± 0.10	0.61 ± 0.09
Channel width: mean at 3 to 7 ha (m)	0.97 ± 0.34	1.31 ± 0.38	1.37 ± 0.26
Channel width: mean at 15 to 25 ha (m)	1.83 ± 0.48	1.76 ± 0.25	2.16 ± 0.38
Channel depth: mean at 3 to 7 ha (m)	0.49 ± 0.07	0.53 ± 0.23	0.72 ± 0.11
Channel depth: mean at 15 to 25 ha (m)	1.16 ± 0.35	1.35 ± 0.40	1.02 ± 0.18
Estimated headcut erosion (Mg km^{-2} y^{-1})	28	50	37
Estimated plungepool deposition (Mg km^{-2} y^{-1})	7	12	9
Estimated bank erosion (Mg km^{-2} y^{-1})	54	73	66
1992–95 average suspended load (Mg km^{-2} y^{-1})	25 ± 17	39 ± 20	-

[a] Exclusive of watershed KJE.

Table 3
Estimated rates of sediment displacement and suspended sediment yields in North Fork sub-watersheds.

Site	Area (ha)	Years in category[a]	Suspended sediment yield	Road plot erosion[b]	Hillslope plot erosion[b]	Landslides and treethrows[c]	Gully erosion[d]
			($Mg\ km^{-2}\ y^{-1}$)				
Treatment watersheds, calibration period[a]							
BAN	10	6.1	2	0	67	0	52
CAR	26	6.1	6	44	67	0	90
EAG	27	4.7	8	6	67	3	68
GIB	20	4.9	9	58	67	27	82
KJE	15	3.8	67	5	67	0	136
Treatment watersheds, post-calibration period[a]							
BAN	10	3.9	9	330	1490	0	78
CAR	26	3.9	29	440	690	37	111
EAG	27	5.3	54	190	4290	31	90
GIB	20	5.1	29	510	2280	6	100
KJE	15	6.2	68	460	1680	20	142
Control watersheds, calibration period[a]							
HEN	39	5	14	0	67	11	82
IVE	21	5	9	44	67	1	74
MUN	16	5	21	6	67	0	77
Control watersheds, post-calibration period[a]							
HEN	39	5	30	0	33	24	82
IVE	21	5	7	1290	130	26	74
MUN	16	5	23	35	39	27	77

[a] "Calibration period" ends with the onset of new road construction at each logged watershed and after 5 years at each control watershed.
[b] Based on hillslope and road erosion plot measurements (Rice, 1996); values represent total amounts displaced.
[c] Based on landslides and treethrows mapped during channel surveys; values represent amounts transported off-site.
[d] Based on application of average bank and headcut erosion rate to the areas of bank and headcut face expected to be present in the watershed (see text); values represent amounts transported beyond the plungepool lip.

for approximately 25% of the combined headcut and plungepool erosion, as plungepool depths below the residual pool surface (i.e., below the depositional lip) average about 25% of the total headcut height on 23 headcuts mapped on the YOC-ZIE tributary in 2006.

We expect retreat rates for 0.26- to 0.44-m headcuts to be lower than 4.2 $cm\ y^{-1}$ and rates to be negligible for faces shorter than 0.26 m. But even at a retreat rate of 4.2 $cm\ y^{-1}$, the expected population of 0.26- to 0.44-m headcuts in control tributaries would produce no more than 4 $Mg\ km^{-2}\ y^{-1}$ of sediment if the assumptions listed above apply also to the small headcuts.

The monitoring results indicate an average bank erosion rate of 1.6 $cm\ y^{-1}$, a value similar to rates measured on vertical faces in a grassland gully in alluvial clay soils near Berkeley, CA (2 to 4 $cm\ y^{-1}$; Reid, 1989) and on near-vertical road-cut faces in a western Washington forest (1.6 $cm\ y^{-1}$; Reid, 1981), suggesting that the measured values are reasonable for the observed conditions. To estimate total sediment input from bank erosion in control watersheds, we assume (i) the proportion of banks "susceptible to erosion" is represented as the proportion mapped during a 1995 channel survey as either undercut or vertical and composed of either alluvium or colluvium; (ii) the estimated average gully-bank erosion rate of 1.6 $cm\ y^{-1}$ applies to the entire area of bank susceptible to erosion; and (iii) the measured relations between channel depth, stream power index, and drainage area defined for 3- to 30-ha watersheds apply to the full estimated drainage density of channels draining catchments of 1.9 to 30 ha.

The 1995 maps of North Fork tributaries indicate that at that time approximately half of the channel length was susceptible to erosion, with no significant difference in values between control and logged tributaries. The relation for channel depth as a function of drainage area obtained by pooling the data shown in Fig. 5 for control watersheds was then used to estimate mean depth for each channel increment throughout a hypothetical 30-ha control watershed. The above assumptions produce an estimated annual input from bank erosion of about 54 $Mg\ km^{-2}\ y^{-1}$ in the hypothetical watershed (Table 2).

About 9 $Mg\ km^{-2}\ y^{-1}$ of the calculated amount, or 17%, is expected to be from unmapped channels with drainage areas of 1.9 to 3 ha and so is based on extrapolation of the defined relations to smaller watersheds. A second estimate for the unmapped low-order reaches may be made by assuming that channel form near the head of a clearcut tributary is similar to that near the head of a control tributary, despite the difference in drainage areas at the channel head. If the mean bank face height of 0.5 m calculated for a mapped 40-m reach draining 1.2 to 1.3 ha of clearcut is assumed also to represent that near forested channel heads, the input by bank erosion from 1.9- to 3-ha drainage areas in control watersheds is estimated to be 12 $Mg\ km^{-2}\ y^{-1}$, a value similar to that estimated using morphological extrapolations.

Similar calculations employing the same estimate of the proportion of erodible bank for channels characteristic of South Fork watersheds provide an estimated input of 37 $Mg\ km^{-2}\ y^{-1}$ from headcut erosion and 66 $Mg\ km^{-2}\ y^{-1}$ from bank erosion in a 30-ha watershed, values 32% and 22% higher than corresponding estimates for North Fork control conditions (Table 2).

If headcut and bank retreat rates measured in North Fork control channels and South Fork channels are assumed to apply also to North Fork logged channels (exclusive of KJE), such channels would supply expected inputs of 50 $Mg\ km^{-2}\ y^{-1}$ from headcuts and 73 $Mg\ km^{-2}\ y^{-1}$ from banks (Table 2), values 79% and 35% higher than corresponding estimates for control conditions. The high values for headcut inputs relative to those in controls largely reflect the increased headcut face area expected in the extended low-order channel network and are offset slightly by the increase in headcut spacing downstream. Higher input rates from bank erosion reflect increases in both channel length and depth. Possible increases in headcut and bank retreat rates after logging and in percentage of the bank susceptible to erosion were not considered, so these calculations may underestimate erosion from these sources in logged watersheds.

5.2. Contribution of gully erosion to the sediment yield

To assess the importance of gully erosion relative to other sediment sources, we first compare estimates of sediment production from gullies to those from other erosion processes. Such comparisons, though instructive, do not account for differences in sediment delivery to streams and through stream channels, so we then evaluate correlations between tributary sediment yields and potential controlling influences.

5.2.1. Comparison of gully erosion rates to those of other erosion processes

The importance of gullying to sediment production can be evaluated relative to rates and patterns of erosion from other sources if total gully erosion is estimated for the length of channel present in each gaged watershed of the North Fork catchment. This calculation differs from the previous one in that channel characteristics within individual watersheds are taken into account instead of using regressed relations to construct typical conditions in a hypothetical watershed.

Sediment input from channel-bank erosion is calculated as the product of eroding bank area and average bank erosion rate (1.6 ± 1.2 $cm\ y^{-1}$) and is converted to units of mass by applying an average bulk density for the characteristic soil (1.5 $Mg\ m^{-3}$). The area of eroding banks in unmapped portions of each tributary channel network is estimated by assuming (i) channel heads are located at a drainage area of 1.9 ha for pretreatment conditions and 1.2 for logged conditions; (ii) regressions between channel depth and drainage area (Fig. 5) apply also to unmapped channels within each category of watershed; and (iii) 50% of the bank length is susceptible to erosion. Unmapped reaches (primarily first-order channels) account for 48% to 94% of the estimated channel length in individual North Fork watersheds (Table 1) and for an average of about half the estimated eroding bank area. Inputs from headcut erosion are calculated by applying an average retreat rate (4.2 ± 2.1 cm y^{-1}) to the estimated area of headcut face present in each tributary, using assumptions analogous to those employed for estimating bank erosion. Deposition on plungepool lips is then estimated as 25% of the volume eroded by headcut retreat and is subtracted from the total for each watershed.

Comparison of the estimated sediment inputs from gully erosion after logging to displacement rates estimated from road erosion plot data shows displacements from road erosion to average about 5 times greater than those accounted for by gully erosion (Table 3), and those calculated from hillslope erosion plot data to be an order of magnitude greater. Displacement from gullying, in turn, exceeds displacement from treethrows and landslides by a factor of 6.

Comparison of displacement rates to measured suspended sediment yields requires an estimate of the proportion of the total sediment load that is transported in suspension at the tributary gages. Bedload was not measured, but sediment volumes and particle sizes accumulated in the North Fork weir pond suggest that coarse sand and gravel transported as bedload contribute about 20 to 30% of the total clastic load at the North Fork weir. Comparison of the proportion of the sediment load accounted for by gravel at the weir (13%) to the proportion of gravel present in the upper 1 m of the watershed's dominant soils (25 to 50%) indicates that a high proportion of the gravel has broken down to smaller sizes before reaching the weir, suggesting that the proportion of sediment transported as bedload is likely to be substantially higher in tributaries. We here assume that 40% of the tributary sediment load is transported as bedload, so we expect total sediment loads to be about 67% higher than the suspended sediment yields noted in Table 3.

Comparison of rates of plot erosion with estimated total sediment yields demonstrates that sediment delivery rates for surface erosion sources are quite low, necessarily averaging <3% for logged watersheds. Field observations indicate that much of the sediment displaced on hillslopes is quickly redeposited on hillslopes, in low-order swales, or on valley flats and so does not contribute directly to sediment yield over the short term. Estimated sediment displacement from gully erosion is also more than sufficient to account for the total sediment loads, suggesting that deposition within channels and on valley flats may significantly influence the amount and timing of sediment exports from the tributary watersheds.

About half of the sediment from treethrows and landslides in the post-calibration period was generated during storms in HY 1995 and so would influence only a single year of the suspended sediment record for the period; most of the rest originated in 1993. Mean suspended sediment load for HY1995 in the logged watersheds was 83 Mg km^{-2}, so estimated total load averaged about 140 Mg km^{-2}. In comparison, sediment displaced by large events amounted to 63 Mg km^{-2} that year, suggesting that large events would not have provided the principal influence on sediment yield even during a year with a high incidence of landsliding.

In addition, examination of annual yields at tributary gaging stations indicates that ranks in suspended sediment yield were relatively consistent from year to year between 1991 and 1995 (Fig. 8), suggesting that yields during this period were largely controlled by chronic sediment sources rather than by discrete, sediment-producing events such as landslides. The highest rankings for tributaries were not associated with years in which significant landslides were observed in those tributaries.

The relative importance of landslides is likely to be greater over the long term because of the rare occurrence of very large slides. Between

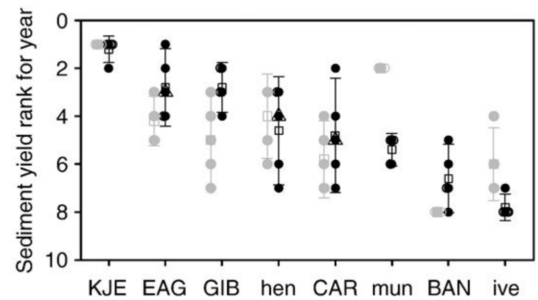

Fig. 8. Relative rankings in suspended sediment yield at tributary gages for HY1986–1990 (gray symbols) and HY1991–1995 (black). Open squares indicate mean rank for the period, and data points for years with significant landslides within individual watersheds are enclosed in triangles. Upper-case gage names indicate logged tributaries.

1986 and 2006, for example, total sediment displacement from landslides and near-stream treethrows throughout the North Fork watershed averaged over 200 Mg km^{-2} y^{-1}, and the two largest slides were responsible for about 50% of the displacement. Had the period of extensive suspended sediment monitoring (1986–1995) included one of these major landslides, landsliding would have undoubtedly influenced short-term sediment yields; but results here suggest that influences from the landslides and treethrows mapped during the study period are not expressed as acute short-term increases in yield. Both of the largest slides occurred on logged slopes and triggered debris flows, resulting in considerable volumes of deposition within and adjacent to downstream channels. Such deposits are likely to be remobilized later by channel incision, thus eventually contributing additional suspended sediment through gully erosion.

5.2.2. Correlation analysis

The relative importance of various sediment sources can be further evaluated by analyzing correlations between sediment displacement rates and suspended sediment loads in the gaged watersheds to identify the combination of source inputs that best explains the observed distribution of yields. Irrespective of sediment delivery ratios and grain size distributions, the relative distribution of suspended sediment yields among watersheds is expected to reflect the relative distributions of sediment displacements from the processes that most strongly influence sediment yields. For this analysis, accuracy of the estimated sediment inputs is not as important as the relative values between watersheds within a source type.

We carried out a step-wise multiple regression of mean suspended sediment yields (*SSY*, Mg km^{-2} y^{-1}) from calibration and treatment periods in each tributary watershed against estimated erosion from gullies (G, Mg km^{-2} y^{-1}), large events, road plot erosion, hillslope plot erosion (P_h, Mg km^{-2} y^{-1}), and watershed area to evaluate the relative influence of these factors on the distribution of suspended sediment yields. The resulting model (Fig. 9) employs only two variables:

$$SSY = -41.5 + 0.702\,G + 0.0050\,P_h \quad \text{adjusted}\,R^2 = 0.77 \tag{1}$$

The relation is highly significant at the 0.05 level, with gully erosion by itself explaining 73% of the variance; the hillslope term is only marginally significant ($p = 0.05$). Results suggest that gullying and associated processes in logged watersheds contribute about three to seven times as much sediment to the sediment yield as processes assessed by hillslope erosion plots. In forested watersheds the difference is greater, with gullying responsible for about 50 to 200 times as much as hillslope erosion. Delivery of hillslope sediment to streams depends in part on the extent of the channel network, so the

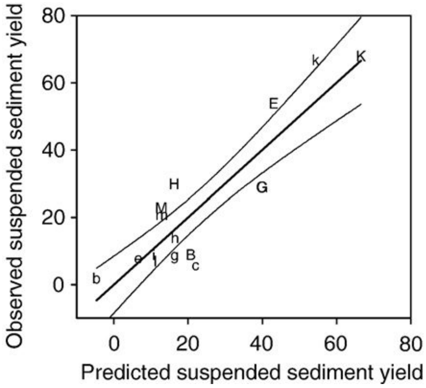

Fig. 9. Observed suspended sediment yields (Mg km^{-2} y^{-1}) versus those predicted using Eq. (1). Lower case initials represent data for the calibration period in each watershed. Points C and G are superimposed.

apparently overwhelming importance of gully erosion may actually reflect both in-channel erosion and the increased efficiency of hillslope sediment delivery due to network expansion after logging.

If bedload is assumed to account for 40% of the total sediment yields at the tributary gages, about half the estimated sediment eroded from banks and headcuts is not delivered to the downstream gage, suggesting that either the rate of gully erosion is overestimated or that significant deposition is occurring within channels and on floodplains. An average channel aggradation rate of 2 mm y^{-1} would account for the undelivered sediment, and if aggradation is indeed occurring at this rate it should become evident from cross-section measurements in the near future. However, given the uncertainties associated with the rate calculations, we expect that imprecision in estimated rates is likely to account for most of the discrepancy.

5.3. Mechanisms of influence from logging

Results of the study suggest that clearcut logging at Caspar Creek caused a significant increase in gully-related sediment inputs (Table 2). Such increases might reflect either increased runoff associated with logging or direct disturbance of low-order channels during cable-yarding operations. Several kinds of information suggest that hydrologic change is likely to be the dominant influence in the present case.

5.3.1. Evidence from network extent

Field measurements suggest that channel heads migrated upslope after logging (Fig. 7, Table 2) and that the stream power index associated with the channel head locations is significantly lower in logged watersheds than in controls. However, the utility of the stream power index (calculated as drainage area × gradient) as a measure of actual stream power relies on the assumption that drainage area is a valid index for relevant discharges. Once a site is logged, the relation between drainage area and characteristic discharges changes.

In North Fork watersheds, the mean peakflow for flows with recurrence intervals longer than 0.15 y increased by about 60% in the 2 years following logging (Reid and Lewis, 2007), with increases approaching an asymptote of 34% for flows occurring fewer than 3 times a year. Effective maximum stream power per unit drainage area during storms thus averaged 60% greater after logging, so a post-logging stream power index of 0.34 ha is equivalent to a pre-logging index of 0.54 ha. If this hydrologic shift is accounted for, the recalculated index at the channel head is not significantly different at the 0.05 level between logged and control watersheds (Fig. 7).

This finding may indicate either that the hydrologic change is sufficient to explain the shift in channel-head location or that the hydrologic change prevented healing of mechanically disrupted sites over the period following yarding. Low-order tributaries are most susceptible to influences from direct disturbance, and the prevalence of subsurface soil pipes in headwater channels would make these sites particularly sensitive to mechanical disruption. However, changes in channel characteristics in logged watersheds along the 65% of the mapped channel length protected by buffer strips could not be explained by direct disruption because disturbance within the buffers is minimal.

5.3.2. Evidence from nested gages

If hydrologic change is an important influence on in-channel erosion after logging, channels downstream of logged watersheds should show increases in sediment yield that cannot be explained simply by changes in sediment input from upstream.

A pair of nested stream gages provides the data needed to assess downstream variations in sediment yield in a gullied tributary. Gage DOL (with a 77-ha catchment) lies downstream of gage EAG (27 ha) in the North Fork watershed. The catchment above the EAG gage was logged in 1991, and hillslopes abut most of the channel length in the EAG catchment. In contrast, the catchment between the EAG and DOL gages has not been logged since 1904, and valley-fill terraces buffer much of the channel from hillslope inputs. Several small tributaries enter this reach, but alluvial fans at most tributary mouths appear to trap much of their coarse sediment load and some fine sediment. Visible sources of sediment within the reach include incised stream banks, headcuts, and near-stream treethrows.

Suspended sediment loads measured during storms at the EAG gage were subtracted from corresponding loads at the DOL gage to estimate the load derived from the unlogged portion of the watershed. Loads for the pre-logging period in both portions of the watershed were then regressed against the mean of loads at control watersheds HEN and IVE to allow prediction of expected loads. The ratios of observed to expected loads from the unlogged portion of the watershed show a response similar in initial timing and magnitude to that from the logged portion of the watershed (Fig. 10), though the downstream portion reattained pre-logging values more quickly than the logged portion. Sediment input increased again in the DOL reach after regrowth in the EAG watershed was thinned in 2001. The sediment record thus indicates that upstream logging influenced sediment production in unlogged portions of the downstream watershed where there was no direct disturbance, suggesting that hydrologic change may indeed have been influential.

However, it is also possible that a portion of the sediment appearing to have originated within the DOL reach may represent

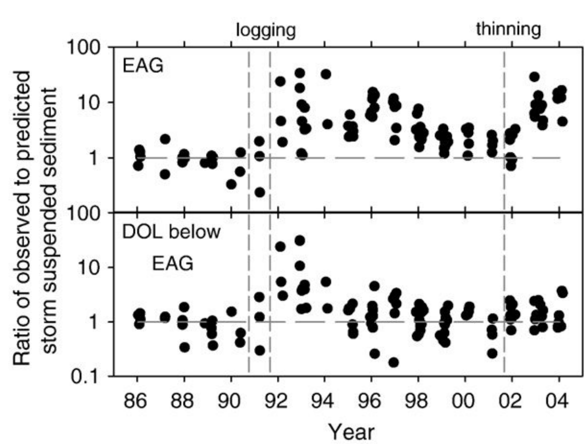

Fig. 10. Deviations from expected suspended sediment load in the clearcut EAG watershed and in the unlogged portion of the DOL watershed downstream of the EAG gage. Values are corrected for bias from back-transformation from logarithms (Baskerville, 1972).

breakdown of bedload into suspendible sizes: bedload originating in EAG would not be included as part of the EAG suspended load, but if it then becomes suspendible downstream it would be recorded as suspended load at the DOL gage. The potential magnitude of this effect can be estimated if bedload is assumed to represent 40% of the total sediment load at EAG. DOL produced 197 Mg of excess suspended sediment between 1991 and 1995, compared to 38 Mg from EAG, indicating that breakdown could explain no more than 13% of the post-logging sediment increase from DOL even if all EAG bedload were transformed to suspended load within the DOL channel. In actuality, much of the EAG bedload is trapped on a minimally channeled fan immediately downstream of the gaging station, so breakdown is likely to explain substantially less than 13% of the increase.

At a larger scale, Lewis et al. (2001) evaluated post-logging changes in suspended sediment yield across the full suite of nested North Fork stream gages and found that observed changes in suspended sediment yield after logging were more closely correlated with changes in flow than with other indices of management activity levels.

5.3.3. Morphological changes associated with headcut migration

Results of morphological comparisons for logged and control tributaries also are consistent with a situation in which channel morphology has been influenced by increased flows after logging. Contrasts in morphology between logged and control watersheds are revealed not so much by changes in average channel dimensions as by changes in the relations between channel dimensions and controlling variables such as stream power and drainage area (Fig. 5), and by increased spacing between headcuts after logging (Fig. 4).

The contrasting patterns may reflect an increase in the frequency with which headcuts are destabilized after logging. Newly mobilized headcuts can retreat rapidly until they encounter an upstream plungepool or a hardened lip supporting another headcut. Two discontinuous gullies will then have coalesced, thereby decreasing the number of headcuts present and increasing mean headcut spacing while not strongly affecting headcut height. Mean channel cross-sectional area would increase in areas where gullies coalesce. Reaches experiencing large changes in any particular year are likely to be distributed randomly, as the ability for a headcut to retreat rapidly is strongly influenced by the condition of elements armoring its lip. Overall patterns of distribution for morphological characteristics are thus likely to become less regular as the system continues to adjust to the sequence of changing hydrologic conditions.

Increased coalescence of headcuts in logged tributaries is also consistent with observed differences in the relative importance of stream power and watershed area as predictors of morphological characteristics in control watersheds and in the pervasively gullied watershed KJE (Fig. 5). In general, where gullies are present as a series of headcuts, headcut location–and the resulting distribution of incised reaches–might strongly reflect local conditions. For a channel segment with uniform discharge, incision is expected to persist along the steeper portions (high stream power), while low-gradient portions (low stream power) may aggrade. In contrast, where gullies have coalesced, incised reaches also span low-gradient reaches between initial headcut locations. Because the fine-scale patterns of incision and deposition in discontinuous gullies are superimposed on the broader pattern of a general increase in channel size downstream, which persists when gullies coalesce, stream power would then become a less efficient predictor of attributes such as width, depth, and cross-sectional area than would drainage area alone.

Morphological characteristics associated with reaches having low headcut frequencies are expected to diverge between logged and control watersheds as logged channels adjust to altered conditions. If low frequencies in logged reaches indicate coalescence of headcuts rather than absence of incision, mean channel depths in reaches with few headcuts should be greater in logged watersheds than in control watersheds, and this is indeed the case. In channel segments where headcut frequency is less than about 10/100 m of channel, mean channel depth is significantly greater in logged watersheds (1.6 ± 0.4 m) than in control watersheds (0.7 ± 0.2 m, Fig. 11). In the South Fork watersheds, the distribution of channel depths as a function of headcut spacing is most similar to that in North Fork control watersheds.

5.4. Contrasts between North Fork and South Fork channels

The intermediate position of South Fork channel morphological relations relative to those in North Fork control and clearcut watersheds (Fig. 6) may in part reflect ongoing recovery from second-cycle logging effects in the South Fork. The South Fork was logged 28 to 30 years before the gully survey, while North Fork watersheds had been logged only 10 to 12 years earlier. Differences in silvicultural strategy and yarding techniques would also contribute to differing results. The selective logging employed in the South Fork resulted in less hydrologic change from altered vegetation than in the clearcut North Fork watersheds, but the use of tractors for yarding produced widespread compaction that continues to generate overland flow at some sites. In addition, South Fork logs were skidded along some low-order channels, directly disrupting the channels and likely producing a very different distribution of channel forms immediately after logging than was present after North Fork logging. Also in contrast to the North Fork, South Fork roads and landings were located mid-slope and in riparian zones, strongly

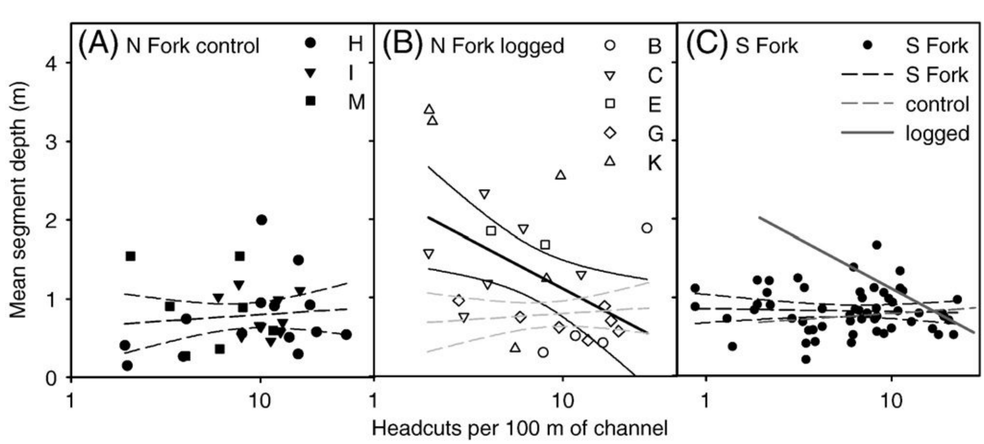

Fig. 11. Mean depth and headcut frequency in 50-m channel segments of (A) North Fork control, (B) North Fork logged, and (C) South Fork tributaries. Relations significant at the 0.05 level are depicted by solid lines.

increasing the potential for channel disruption by road drainage and mechanical disturbance.

5.5. Implications for forest management

Efforts to prevent and mitigate logging-related erosion in rain-dominated watersheds of the Pacific Northwest have generally relied on the use of buffer strips and on reduction of road-related erosion. Road erosion was found not to be of major consequence at North Fork Caspar Creek (Lewis et al., 2001), possibly reflecting drainage control efforts, road closures following logging, and pervasive water-barring of abandoned roads. Most North Fork roads and log landings are located near ridge tops, further reducing their potential for contributing sediment directly to channels. Robust buffer strips were incorporated into the logging plan, providing extensive filter strips below upland sediment sources and preventing direct disturbance to a significant portion of the stream network. Despite these measures, suspended sediment yields increased significantly after logging, and much of the increase appears to originate from gully-related processes that are not amenable to mitigation either through road improvements or buffer strips. If increased runoff after logging generates sediment from within downstream channels, control of excess sediment from this source would be possible only through management of the level of hydrologic change induced by logging, and this would require either management of the rate of logging within a watershed or modification of the silvicultural strategy used.

Logging prescriptions generally consider only the distribution of channel types present at the time that plans are developed and so do not reflect the possibility that the channel network may expand after logging. The apparent 28% increase in drainage density after logging at Caspar Creek would strongly increase the connectivity between hillslope sediment sources and the downstream channel network. Plans to maintain a prescribed distance between ground-disturbing activities and stream channels are defeated if channel networks expand into the disturbed sites after logging.

Logging plans on lands administered by US Federal agencies or regulated by California State agencies are required to include an evaluation of potential cumulative and indirect impacts of the planned logging, but considerable uncertainty and controversy have at times surrounded the definition of what constitutes an adequate impact analysis. Observation of the Caspar Creek gullies suggests that future analyses might usefully consider the possible influence of logging-related hydrologic changes on downstream channel morphology and sediment inputs.

Interest is growing in the use of indirect methods for inferring long-term erosion rates to allow comparison to management-related sediment inputs. Several studies have evaluated concentrations of cosmogenic ^{10}Be in soils and sediment to estimate long-term input rates (e.g., Kirchner et al., 2001). In the case of Caspar Creek, Ferrier et al. (2005) used results of such a study to conclude that recent erosion rates evaluated from monitoring data at Caspar Creek are lower than rates characteristic of the pre-logging period. Such conclusions rest heavily on the assumption that the distribution of sediment sources that produced the sampled sediment is typical of the distribution present over the period for which long-term rates are to be inferred. However, examination of the Caspar Creek tributaries indicates that gullying is now pervasive, that it probably initiated with or was greatly accelerated by first-cycle logging, and that many of the gullies excavate cosmogenically “pristine” sediment sources such as buried saprolite and bedrock. Under these conditions, samples obtained from in-channel sediments will contain lower ^{10}Be concentrations than would be expected from sediment exported before gully initiation, and estimated “long-term” erosion rates may instead disproportionately reflect accelerated erosion resulting from first-cycle logging.

6. Conclusions

Results of the Caspar Creek study suggest that erosion along incised channels is an important source of sediment in tributary watersheds. Gullying in the area appears to have expanded with first-cycle logging of the late 1800s. Channels had not yet recovered from the earlier impacts at the onset of second-cycle logging nearly 100 years later, and increased runoff resulting from second-cycle logging accelerated erosion within the still-incised channels. Recently logged watersheds show a higher drainage density than controls, with the extent of the increase similar to that expected on the basis of the observed change in runoff.

Because an appreciable portion of the increased sediment input at Caspar Creek is associated with hydrologic changes caused by logging and because a significant portion of the excess sediment is generated along channels in and downstream of the logged areas, the strategies most often used in the region to reduce sediment inputs from logging–control of road-related erosion and establishment of riparian buffer strips–are not effective for reducing an important component of the logging-related sediment input at Caspar Creek. In addition, efforts to reduce impacts from surface erosion by ensuring that soil-disturbing activities are not carried out near streams would need to take into account the potential for upslope expansion of the channel network after logging.

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Chapter **10**

Evaluating Forest Management Effects on Erosion, Sediment, and Runoff: Caspar Creek and Northwestern California

Raymond M. Rice, Robert R. Ziemer, and Jack Lewis

ABSTRACT: The effects of multiple logging disturbances on peak flows and suspended sediment loads from second-growth redwood watersheds were approximately additive. Downstream increases were no greater than would be expected from the proportion of the area disturbed. Annual sediment load increases of from 123 percent to 269 percent were measured in tributary watersheds but were not detected at the main channel gages, implying that sediment was being temporarily stored in the intervening channels.

KEYWORDS: automatic sampling, California Forest Practices Act of 1973, clearcut, discriminant analysis, erosion, erosion hazard rating (EHR), evapotranspiration, fish, floods, interception lag time, landslides, peak flows, roads, sedimentation, selective harvest, slope stability, stage-based sampling, storm volume, streamflow, suspended sediment, turbidity-based sampling, transient snowpack, water yield

Introduction

The debate about the impact of commercial forest management on fish and the identification of acceptable management practices make robust, well-designed watershed research essential. The climate and geology of northwestern California make it an ideal place to study the erosion, sedimentation, and streamflow effects of forest management. Northwestern California is well-watered with average annual precipitation ranging from a little under 40 inches near the coast to more than 120 inches on some of the higher coastal peaks. In this Mediterranean climate, about 90 percent of the annual precipitation in the commercial forest zone falls as rain between October and April. However, some of the largest floods have been enhanced by the melting of a transient snow pack. Five large floods have struck the area in the last 50 years. Those floods and the geologic setting produce high erosion and sedimentation rates. The area is characterized by high rates of tectonic uplift, resulting in slopes averaging 45 percent. The western two thirds of the area is dominated by the highly erodible Cretaceous and Jurassic rocks of the Franciscan Assemblage (Bailey et al. 1964). In the eastern portion, there are many deeply weathered granitic plutons, which are the source of landslides and severe surface erosion. As a consequence of the foregoing factors, rivers of northwestern California have the highest sediment loads in the United States and some are comparable to the sediment-laden rivers of Asia.

A redwood-dominated (*Sequoia sempervirens*) forest begins at the coast and extends inland approximately 30 miles. Associated species include Douglas fir (*Pseudotsuga menziesii*), Sitka spruce (*Picea sitchensis*), grand fir (*Abies grandis*), and western hemlock (*Tsuga heterophylla*). As the environment becomes dryer and warmer inland, the forest changes to one dominated by

Douglas fir, with lesser amounts of white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), and sugar pine (*Pinus lambertiana*). Logging in northwestern California began in the 1850s in response to the demands of the gold rush. In the coastal redwood zone, forests were clearcut to provide lumber for the cities of the San Francisco Bay area and to clear the land for pasture. Often the latter objective was not met because the sprouting redwoods persisted in spite of repeated burning. Logging the coastal redwood forests has continued since the 1850s; with some areas now having been logged for a third time. Inland, the logging history is quite different. With the exception of isolated areas near mining operations during the gold rush, large-scale commercial logging of the inland softwood conifers did not begin until after World War II. The coastal redwood forests were mostly privately owned, while the inland commercial timber stands were mainly on national forests.

Both inland and on the coast, past logging and mining practices left a legacy that is affecting current management. Logging has impacted the coastal watersheds most heavily. Corduroy roads were built in tributary drainages for skidding logs to main channels. Splash dams were used during the earliest logging to facilitate log drives to the mills. Stream channels were straightened and riparian vegetation removed to reduce the chance of log jams. The remnants of these activities exist today. The shift from log drives to rail transport in many cases only moved the disturbance from the streams to the stream banks since gentle right-of-way grades were needed. These railroad rights of way became main truck haul roads following World War II, when yarding and hauling shifted almost exclusively to tractors and trucks. Main skid trails and landings were commonly located in or adjacent to tributary stream channels.

Although there had been forest practice legislation in California since 1945, early regulations dealt mainly with regeneration and fire control (Arvola 1976). The California Forest Practice Act of 1973 marked a dramatic change in the level and focus of forest practice regulation by the State of California. The 1973 regulations addressed a broader range of environmental concerns, and have subsequently been further expanded in response to the California Environmental Quality Act and Section 208 of Public Law 92-500 (1972 amendments to the Federal Water Pollution Control Act). These laws, together with the Endangered Species Act, have become the vehicles by which the general public and various special interests influence forest practices.

Caspar Creek Experimental Watersheds

The North Fork and South Fork Caspar Creek watersheds are between three and seven miles from the coast and seven miles south-southeast of Fort Bragg, California. They are part of the Jackson Demonstration State Forest. Between 1860 and 1906, the watersheds were clearcut and then burned to clear the ground for yarding. Splash dams were constructed and there were extensive modifications to the main stream channels to accommodate log drives.

Watershed experiments, conducted at Caspar Creek by the USDA Forest Service and California Department of Forestry and Fire Protection since 1963, offer an insight into the hydrology of northwestern California and effects that changing forest practices have had on erosion, sediment yield, and streamflow (Figure 10-1). Approximately 35 percent of the two watersheds have slopes <30 percent. Approximately 7 percent of the North Fork is steeper than 70 percent, but <1 percent of the South Fork is that steep.

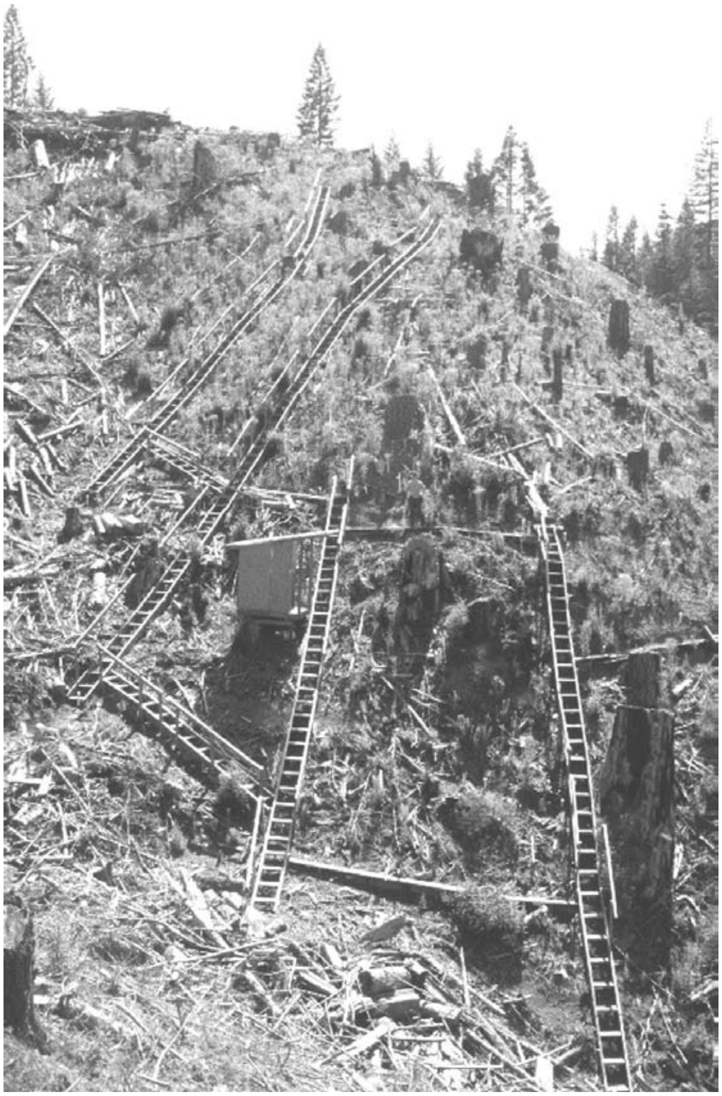

Figure 10-1. Process studies of soil moisture, subsurface flow, and pipeflow at Caspar Creek assist in understanding effects of logging on streamflow and erosion.

South Fork Study

The first phase of research at Caspar Creek was a traditional paired watershed experiment initiated in 1963 when concrete broad-crested weirs with a 120-degree V-notch were installed in the 1,047-acre South Fork and the 1,179-acre North Fork (Rice, Tilley, and Datzman 1979).

The calibration phase lasted until hydrologic year 1967 (October 1966 - September 1967; HY-67). A main haul road was constructed in the South Fork in 1967; approximately three-fourths of its length within 200 ft of the stream channel. Logging was deferred until the summer of HY-71 to allow an estimate of the road's effect independent of logging disturbances. Selective logging began at the weir and proceeded up the watershed during two additional seasons. A total of 38.3 million board feet of timber was harvested from the watershed -- 64 percent of the stand volume. The study was fortunate in that the calibration and postlogging periods each included 30-year peak flows. Consequently, inferences made from the study are not weakened by statistically questionable extrapolations of data in either period.

Peak Streamflow

The effects of road building and logging on peak flows were evaluated by Ziemer (1981) and Wright et al. (1990). The results agreed with the preponderance of paired watershed studies on the west coast of the United States. Ziemer (1981) found statistical differences associated with logging and road building only in peak flows <0.01 cfs ac^{-1}. Wright et al. (1990) found changes only in peaks <0.02 cfs ac^{-1}.

Storm Runoff

Wright et al. (1990) also investigated possible logging effects on the hydrograph lag time and volume of stormflow. As with peak flow, significant differences in quick flow volume and total stormflow volume were only found for the class of storms having peaks of <0.02 cfs ac^{-1} and volumes <42,700 ft^3. Lag time was decreased by approximately 1.5 hours for all three segments of the hydrograph. Therefore, because the total hydrograph was merely shifted forward in time but remained unchanged in shape, it seems unlikely that this response to logging would have any effect on channel stability or sediment transport.

Water Yield and Summer Low Flows

Keppeler and Ziemer (1990) tracked several aspects of postlogging water yield, especially during the low-flow season, from HY-71 to HY-83. Average annual water yield increased 15 percent or 0.3 ac-ft (AF) ac^{-1}. Most of that increase occurred during the winter rainy season. Summer low-flow volume increases averaged 29 percent (0.04 AF ac^{-1}) and the minimum streamflow rate averaged 38 percent (0.000036 ft^3 sec^{-1} ac^{-1}) higher than that predicted by the prelogging relationship with the North Fork. The average length of the part of the low flow period when flow in the South Fork was <0.2 cfs (0.0002 ft^3 sec^{-1} ac^{-1}) was shortened by 43 days from 1972 to 1978 -- a 40 percent reduction. Beginning with 1979, the durations of these low flows returned to the prelogging pattern and the summer minimum flow dropped below prelogging levels (Keppeler 1998). In summary, the authors concluded that in spite of the benefits measured in their study, a dependence on selective logging for water yield increases was impractical and would be of minimal importance compared to other forest management and production goals.

Sediment Loads

Although, as mentioned earlier, streamflows before and after logging the South Fork were comparable, the sediment record was clouded by the occurrence of two streamside landslides in 1974. The landslide in the North Fork control watershed was 10 times larger than the landslide in the logged South Fork. As a consequence, the suspended sediment load for the North Fork that year was 4.5 times larger than that predicted from its relationship with flow during all the other years. In view of this anomaly, Rice, Tilley, and Datzman (1979) decided to use the predicted rather than the observed sediment load for 1974 to estimate the effect of logging. Had they not done so, the estimated effect of logging in that year would have been a reduction of 6.75 ft^3 ac^{-1}. Using the adjustment, the authors estimated that the logging and road construction resulted in an additional 257 ft^3 ac^{-1} of sediment from HY-68 to HY-76-- almost a threefold increase above that expected from the watershed in an undisturbed condition. In a subsequent analysis using the suspended loads of the North Fork as a control and making adjustments for 1974 landslides, Lewis (1998) estimated 181 ft^3 ac^{-1} of excess sediment in the first six years after logging was initiated (Figure 10-2a).

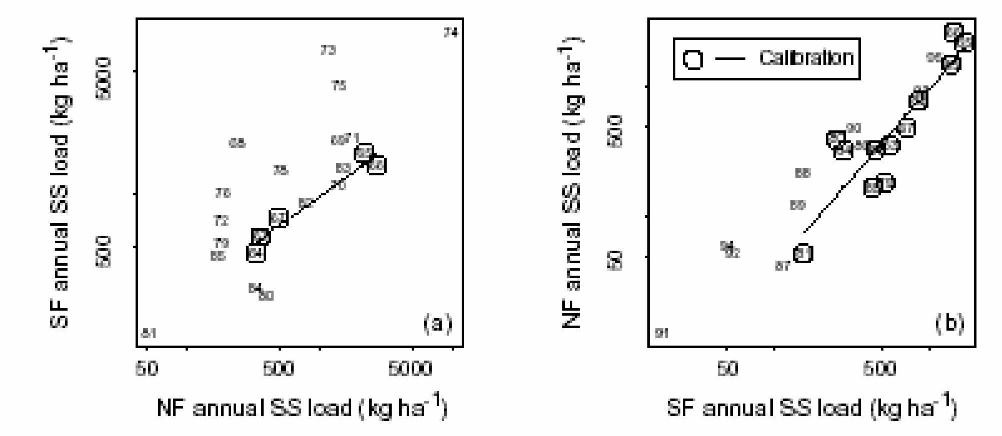

Figure 10-2. Comparison of suspended sediment loads after logging (a) South Fork of Caspar Creek and (b) North Fork of Caspar Creek.

NOTE: Numbers represent water year. South Fork was roaded in 1968,and approximately 65 percent of the timber volume was logged from 1971 to 1973. In the North Fork, roading and logging proceeded concurrently, with 13 percent of the timber volume removed from 1985 to 1986 and 37 percent removed from 1989 to 1992. Calibration years for (b) include both pretreatment and postrecovery years from (a).

Erosion

Road-related erosion was estimated to be 11,500 ft^3 mi^{-1} (Krammes and Burns 1973). However, Krammes and Burns acknowledge being unable to measure all sources of erosion, and their estimate is considerably below that of a 1976 study by McCashion and Rice (1983). Rice, Tilley, and Datzman (1979) estimated that sediment related to the road construction exceeded estimated erosion from the road by approximately 35 percent because of the presumed underestimation of road-related erosion. The discrepancy was attributed mainly to a 25,000 ft^3 failure of the old South Fork splash dam in 1968. The sediment was considered a result of the road because a stream crossing immediately upstream may have caused the failure. Logging-related erosion was estimated on seven plots totaling 94 ac. The estimated rate was 1,150 ft^3 ac^{-1}. That rate was almost five times greater than that on 18 similar plots measured elsewhere in northwestern California by the same field crew during the same season.

North Fork Study

Cumulative watershed effects had become an important management concern when the second Caspar Creek watershed study was designed. One objective of the study in the North Fork of Caspar Creek was to test for synergistic cumulative effects. To accomplish that goal, 13 additional stream gages were installed on the mainstem and tributary channels of the North Fork. The cumulative effects hypothesis tested was that unit area peak flows or sediment loads increased as a function of watershed area in addition to the proportion of watershed area treated.

The study also became an evaluation of the effect of forest practice rules that had been revised since the South Fork study. Other than the total proportion of the timber volume removed, the logging of the North Fork had little in common with the logging of the South Fork. The North Fork timber was harvested by clearcutting and 92 percent of it was cable yarded to landings high on the slopes. All roads were located well away from stream channels. Tractor yarding was limited to the gentle, upper slopes (Figure 10-3). Streamside buffers, from which 34 percent of the timber volume was thinned, protected streams that supported aquatic life. Logging began in the upper watershed and proceeded downstream to enhance the chance of detecting cumulative effects.

Figure 10-3. *Tractor logging was restricted to the gentle upper slopes in North Fork of Caspar Creek*

Peak Streamflow

As with the South Fork study, when peak flows at the North Fork weir were regressed against those at the South Fork weir, the postlogging regression was found to not be statistically significantly different from the regression before logging. However, when the test was repeated using two control tributary watersheds within the North Fork, postlogging peak increases were statistically significant (Ziemer 1998). This finding suggests that the earlier conclusions of no significant change (Ziemer 1981; Wright et al. 1990) may have been the result of a large variance requiring a large change before being detected.

For the cumulative effects analysis, an aggregated regression model was fit simultaneously to all the subwatershed peaks (Lewis et al. 2001). The peak flow equation was a function of peaks in the control tributary watersheds, proportion of area logged, the interaction between logged area and antecedent wetness (Figure 10-4), and time since logging. No variables related to roads, skid trails, fire lines, burning, or herbicide application improved the equation. The estimated average peak flow increases for the two-year return period storm was 27 percent for the 100 percent clearcut tributary watersheds and 9 percent for the 50 percent clearcut North Fork. The analysis found an approximately linear recovery rate of 8 percent per year for the first seven years after logging. The coefficient of the variable testing synergistic cumulative effects was not significant ($p = 0.21$).

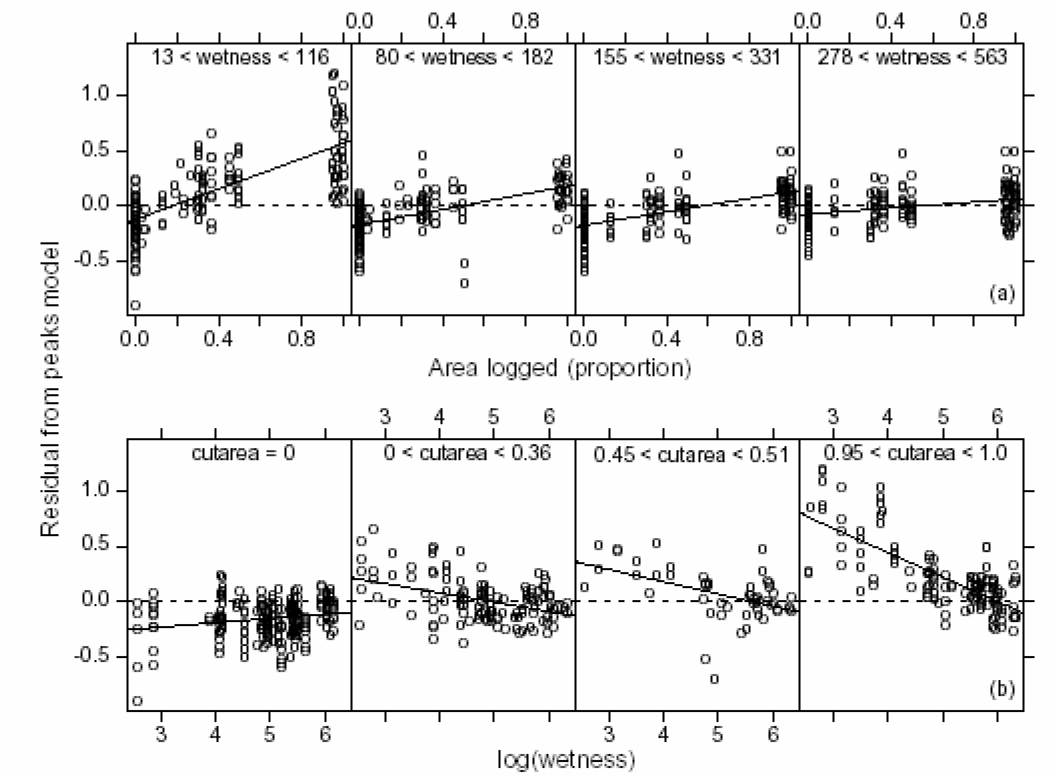

Figure 10-4. Residuals from simple linear regressions of logarithm of peak flow in treatment watersheds versus controls: (a) Effect of cutting decreases with increasing wetness; (b) effect of wetness becomes increasingly negative with more cutting.

Storm Runoff

The results of the runoff volume analysis were very similar to those of the peak flow analysis. The largest relative increases were found in the smallest storms and driest antecedent conditions. However, the average increase for runoff volumes greater than 21 ft^3 ac^{-1} declined with storm size and then leveled off at an average increase of 30 percent for clearcuts and 13 percent in partially clearcut watersheds. The mean percentage increases were still positive even under the wettest conditions of the study -- 27 percent for clearcuts and 16 percent for partially clearcut watersheds (Ziemer 1998).

The annual storm runoff volume (the sum of the storms measured) increased an average of 60 percent (2.26 AF ac^{-1}) in clearcut watersheds and 23 percent (0.87 AF ac^{-1}) in partially clearcut watersheds. Based on the complete discharge records at the North Fork weir, the storm runoff volume included in this analysis represents approximately 45 percent of the total annual runoff volume.

Water Yield and Summer Low Flows

AS with the South Fork study, an 8 percent increase (0.2 AF ac^{-1}) in annual water yield was found following logging in the North Fork (Keppeler 1998). The minimum summer low flow rate was increased 148 percent. Unlike in the South Fork, no recovery trend was detected, suggesting that water yield effects will persist longer after clearcutting than when a similar timber volume is removed from a watershed in a selection cut. These differences in water yield recovery are probably related to changes in rainfall interception and evapotranspiration.

Sediment Loads

Much more so than in the peak flow analysis, sediment load estimates in the North Fork study benefited from improved technology. The North Fork sediment estimates were made from samples taken by automatic pumping samplers using variable probability sampling (Thomas 1985).

Using the South Fork as a control, following logging there was a 28 percent increase in suspended sediment discharge above that predicted by pretreatment regression (Figure 10-2b) and an 8 percent decrease in total sediment load (suspended plus pond accumulation). Neither of these estimates was statistically significant (Lewis 1998). However, when three unlogged North Fork tributaries were used as controls, the effect of logging was a statistically significant 89 percent increase in suspended sediment load. Only storm flows were sampled for that estimate, but these storm flows carried approximately 90 percent of the total suspended sediment load in the North Fork.

In the aggregate analysis, logging was associated with a mean annual increase in suspended sediment load of 212 percent (3.5 ft^3 ac^{-1} yr^{-1}) in clearcut watersheds and 73 percent (3.5 ft^3 ac^{-1} yr^{-1}) in partially clearcut watersheds. The most important explanatory variable in the aggregate analysis was the increased volume of stormflow. With one exception, downstream increases in sediment load were no greater than would be expected by the proportion of the watershed disturbed. However, that may be because the excess sediment produced in tributary watersheds had not yet reached downstream gages.

Erosion

Road-related erosion in the North Fork was approximately 5,000 ft^3 mi^{-1}, but, because these roads were located away from streams, this erosion did not contribute significantly to sediment loads. Logging-related erosion in the cable-yarded North Fork was approximately 650 ft^3 ac^{-1} -- substantially less than that measured in the tractor-yarded South Fork.

Erosion Studies

The study of erosion has some advantages over the study of sediment resulting from forest management activities. Estimating sediment effects usually involves a combined calibration period and treatment period spanning a decade or more. Because of the time and expense, sediment studies are anecdotal -- the watersheds are chosen for study rather than being a random sample of all watersheds. Often even the choice of a control watershed is not left to chance. Erosion studies, in contrast, can be based on a random sample of the population of interest. They also take less time, even if researchers must wait several years to allow erosion processes to respond to the disturbance before collecting data. The weakness of erosion studies is that usually sediment, not erosion, is the parameter of interest, and estimating the delivery of eroded material to a watercourse and its subsequent transport is a very uncertain exercise.

Because of the geology, steep terrain, and climate of the redwood region, mass movement is the dominant erosion process. Sheet erosion was estimated in only one of the studies discussed below. The lack of such estimates stems as much from the difficulty and uncertainty of measuring sheet erosion as from the presumed minor role that sheet erosion plays in total sediment yield. Estimates of rills, gullies, and mass movements are usually based on the volume of the void left by the erosion. Occasionally void volumes are reduced by estimates of local deposition, in which case the estimates are almost as uncertain as those of sheet erosion.

Erosion hazard ratings (EHR) are often used as a guide for designing appropriate forest practices. California's forest practice rules currently contain an EHR patterned after one used by the USDA Forest Service. As far as we know, its efficacy has never been tested. An earlier version was tested by Datzman (1978) and rejected when it was found to be significantly correlated ($r^2 = 0.63$) only with tractor-yarded old-growth logging. Rice and Datzman (1981) used the same data in a regression analysis in an attempt to predict erosion volume from site and operational variables. Their equation -- which was based on slope, aspect, geology, and yarding method -- was only moderately successful ($r^2 = 0.43$). Mass movements accounted for 55 percent of the measured erosion, gullies 37 percent, and rills 8 percent. The study led to two insights: that most (68 percent) of the erosion was on 4 of the 102 plots and that operator behavior was responsible for a large part of the variability in erosion. These findings influenced the focus of several subsequent studies.

Concurrently, an inventory of road-related erosion was made of 344 one-mile segments of Forest Service roads (McCashion and Rice 1983; Rice and McCashion 1985; Rice and Lewis 1986). McCashion and Rice (1983) concluded that the road caused approximately 60 percent of the measured erosion and that conventional engineering methods or minor relocation of the right-of-way could have avoided only 24 percent of the total erosion. They found that 95 percent of the road-related erosion was by mass movements, including bank sloughing. Furthermore, erosion from seasonal roads was approximately 50 percent greater than that from all-weather secondary roads, even though the rights-of-ways of the latter were 50 percent wider. Like Rice and Datzman (1981), McCashion and Rice (1983) found their data to be highly skewed. Three mass movement events accounted for 85 percent of the natural erosion and 33 percent of all the erosion measured. Two other studies used linear discriminant analysis to identify high-risk sites on forest roads. Rice and McCashion (1985) randomly divided the data into two groups. One of these groups was used to develop an equation separating the stable sites ($<1{,}420$ ft^3 mi^{-1}) from the unstable group ($>1{,}420$ ft^3 mi^{-1}). It correctly classified 75 percent of the sites. When tested against the other half of the data, the equation correctly classified 74 percent of those sites. The sites classified as unstable were responsible for 82 percent of the total erosion. In a second discriminant analysis, Rice and Lewis (1986) contrasted conditions at 0.2 mi. road segments containing large erosion features with a random sample from all 0.2 mi. segments. It had a similar classification accuracy based on only two variables -- slope and geology.

Even before these studies quantified the importance of mass movements in erosion associated with forest management, problems on national forests fostered research aimed at estimating landslide risk. Discriminant analysis is well suited for such studies because it can produce probabilistic estimates of risk; the task is to identify the portion of the landscape that is at risk. Rice and Pillsbury (1982) studied the English Peak batholith in the Klamath Mountains and produced an equation based on slope, prelogging crown density, drainage area, and distance to a stream. The equation had a classification accuracy of almost 90 percent with the data used in its development. When that equation was tested on two granitic batholiths in Oregon, its classification accuracy was 74 percent on one and 51 percent on the other (Rice, Pillsbury, and Schmidt 1985) -- an example of the danger of overfitting developmental data. When the least significant variable was dropped from the equation, the prediction accuracies became 86 percent, 76 percent, and 75 percent -- suggesting that the deleted variable, drainage area, was related to some process unique to the English Peak batholith.

As a result of rapid tectonic uplift, topography that is typical of these coastal mountains consists of a steep “inner gorge” adjacent to streams with gentler terrain above. Slides in the inner gorge are of particular concern because they tend to have high sediment delivery to the streams. Furbish and Rice (1983) sampled 85 clearcut patches on the Six Rivers National Forest.

They found that inner gorges, which accounted for approximately 30 percent of the study area, produced 88 percent of the landslide volume. Outside the inner gorge, 85 percent of the slide volume was associated with roads. Slides occupied 2.6 percent of the inner gorge area and 0.14 percent of the remainder of the study area (Furbish 1981).

In 1985 the findings that a small proportion of large landslides typically yielded a disproportionately large fraction of the total measured erosion led the state of California to provide most of the funding for the Critical Sites Erosion Study (CSES). The CSES sample population consisted of all areas on private land where harvesting had been completed between November 1978 and October 1979. Access was granted to 75 percent of the area. It was presumed that the six-year delay after logging would have allowed enough time for weaknesses in the site or the execution of the harvest to be expressed as excess erosion. The objective was to learn how the sites with large erosion events differed from stable sites. Data were collected on all erosion features displacing more than 350 ft^3 from 2 ac, square plots. Critical plots (>2,700 ft^3 ac^{-1}) were found during a reconnaissance of the available harvest areas. Control plots were selected at random. The overall study was divided into two substudies. In one, an interdisciplinary team attempted to learn why the erosion had occurred, based on their appraisal of site conditions and how the harvest or roadwork had been executed (Durgin, Johnston, and Parsons 1989). The researchers developed a numerical index to quantify their evaluations. The other substudy used discriminant analysis to distinguish between critical and control sites (Lewis and Rice 1989). Both substudies analyzed roads and harvest areas separately.

Durgin, Johnston, and Parsons (1989) found that, similar to earlier studies, almost all of the erosion measured was produced by 12 percent of the area studied. They concluded, "Inherent physical factors had the major influence on susceptibility of a road or harvest area to erosion events. On road plots, management factors played an almost equal role to site factors, but on harvest plots, management factors played a decidedly secondary role." They found that roads, which covered 4 percent of the area studied, produced 76 percent of the erosion. Lewis and Rice (1989) sifted through 56 variables before deciding that the most robust equations for both road and harvest sites were based on only three variables. Extensive testing persuaded the authors that the addition of other variables that were "known" to affect slope stability would risk overfitting the developmental data. The chosen equations included slope, horizontal curvature, and rock strength for harvest areas and slope, horizontal curvature, and soil hue for road plots. These variables seem to be good surrogates for the forces affecting slope stability. Slope indexes the balance between the downslope component of gravitational force promoting failure and friction at the failure plane resisting it; horizontal curvature represents sites of convergent subsurface flow promoting failure and accumulations of quasi-stable colluvium susceptible to failure; rock strength indexes the resistance to failure; and hue was apparently indexing subsurface water because soils associated with high risk were frequently gleyed soils of low chroma, indicating waterlogged conditions. The road equation had an estimated classification accuracy of 78 percent, and the harvest area equation had an estimated classification accuracy of 69 percent. Lewis and Rice (1990) concluded that these accuracies were high enough to make the functions useful tools for estimating risk and presented a method by which economic and environmental objectives could be balanced objectively.

Rice and Lewis (1991) found that with no site-specific information, the probability of a slope-failure critical harvest site was 0.007 on the northern California coast and 0.001 inland. The probability of a critical road site was 0.022 on the coast and 0.006 inland. As a consequence of scarcity of critical sites, any acceptable risk threshold that correctly identifies a large proportion of critical sites will also misclassify as critical a substantial number of sites that will not fail as the result of management activities (Figure 10-5). For example, a threshold that correctly identified 90 percent of critical road sites on the coast would also misclassify 40 percent of the stable sites. Conversely, a threshold that correctly classified 90 percent of the stable sites would misclassify 55 percent of the critical sites. Any site with at least a 1.2 percent probability of slope failure on the coast or 0.3 percent probability inland would have to be classified as critical, in order to identify 90 percent of the critical sites. These low risk thresholds seem counterintuitive, but they reflect the fact that a wide net must be thrown in order to catch such rare fish. Probably the most effective use of any discriminant function is for screening areas that are steep enough to contain potential mass wasting sites. The sites classified as critical could then be subjected to geotechnical investigation if the expected proportion of "false positives" is unacceptable.

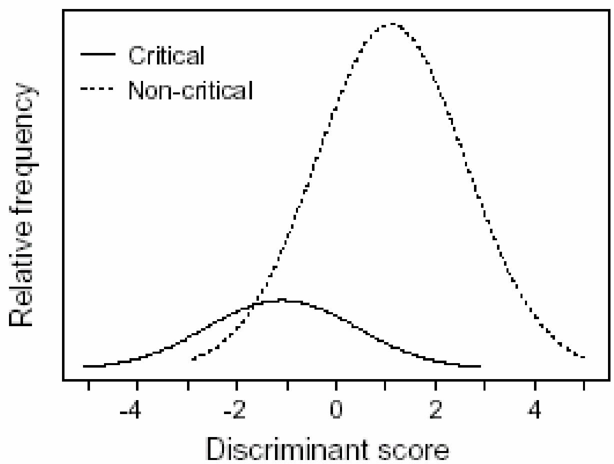

Figure 10-5. Relative frequency of critical and noncritical sites for various risk ratings (discriminant scores) on harvest areas. (Frequency of noncritical sites has been scaled to be underrepresented by a factor of 10.)

Measurement and Analytical Capabilities

Streamflow

With the nearly universal application of digital computers, traditional methods of recording have largely given way to electronic recording of stage via data loggers. This has eliminated the need to store charts and tape and to digitize or otherwise transfer them to electronic media. Data loggers normally record the gage height sensed by pressure transducers, rather than traditional float or bubble-gage sensors. With these changes, reliability has been improved significantly by eliminating failures associated with mechanical clocks, pens, tapes, pulleys, float switches, and so forth. Of course that advance does not eliminate problems associated with sediment in flumes, debris in weirs, and plugging of stilling well intakes or failures related to transducers, batteries, and wiring. Therefore, it is still important to make frequent site visits to check functioning of equipment.

Suspended Sediment Sampling and Estimation

Manual Sampling

In most small and medium-sized watersheds, including those that are seasonally dominated by snow, most of the annual suspended sediment usually is transported during a few large streamflow events. Automated data collection is essential to effectively capture such events. Although it is possible to rely solely on manual measurements, important storm flows are infrequent and difficult to predict; when they do occur, trained personnel may not be available to collect the required information. Infrequent, systematic manual sampling will not provide adequate information to make credible suspended sediment load estimates under these conditions. Although manual sampling -- for example, with DH-48 (FIASP 1958) or D-49 (FIASP 1965) samplers -- is necessary to measure the mean and variability of sediment concentration within a stream cross-section, it is generally inadequate for defining the temporal variability. The result is that concentrations for periods of interest must be estimated by relying on discharge and sediment rating curves. However, discharge is a very imperfect predictor of sediment concentration (Walling and Webb 1988). Although rating curves may be adequate for some rivers and some purposes, their inaccuracy seriously limits their usefulness in detecting changes in sediment transport.

Automated Sampling

The use of data loggers at gaging stations facilitates innovative approaches to suspended sediment sampling because of their programmability and capability of controlling multiple sensors and devices. Several manufacturers now offer turbidity probes that can be deployed on a continuous basis in streams. Therefore, information about discharge and/or turbidity can be used to control an automatic pumping sampler.

Automatic pumping samplers have some fundamental limitations that are important under some conditions. They collect point samples that are not guaranteed to represent the cross-sectional mean concentration. Their representativeness must be verified by examining the relation between simultaneous pumped and depth-integrated samples. At Caspar Creek, because the channels are small and the suspended material is generally fine, we have found the pumped sample concentrations need little or no correction. In larger channels, especially those with coarser sediments, larger differences can be expected between the pumped and depth-integrated sample concentrations. A more significant problem in streams with very coarse sediment loads is pumping sampler inefficiency with coarse sand. The problem is exacerbated if the pump must be situated at a much higher elevation than the sampler intake and when water velocity exceeds the sampler intake velocity. At one location where the pump was approximately 13 ft above the intake and where point velocities exceeded 15 ft sec^{-1}, we found greater than 90 percent of the sand that was coarser than 0.5 mm was escaping the sampler. It is unlikely that correcting to depth-integrated samples can salvage the data in such situations.

Automated sampling programs reduce the cost per sample. However, they usually are designed to sample more frequently than manual programs; hence, they will typically involve increased total costs for processing samples, in addition to the extra purchase and maintenance costs of automated field equipment. Automated sampling programs also require more technical skills for installation, maintenance, troubleshooting, data processing, and analysis. The superior coverage of storms and the potential for much more accurate load estimates justifies the extra costs for many applications.

Stage-Based Sampling

Automated sampling programs often have used a variety of stage-based algorithms for sampling. One stage-based algorithm uses variable probability sampling to increase sampling frequency as a continuous function of stage (Thomas 1985, 1989). This approach was used at Caspar Creek for 10 years, but was discontinued because it tends to collect too many samples if estimates are needed for each storm. Time-stratified sampling (Thomas and Lewis 1993) is an approach that gives less variable sample sizes for storms. This approach breaks hydrographs into time periods (strata) of varying lengths that depend on the stage height and direction at the start of each stratum. Two or three samples are collected randomly in each stratum. Another approach, flow-stratified sampling, chooses from a set of sampling frequencies corresponding to specific rising and falling flow classes (Thomas and Lewis 1995). Flow-stratified sampling is best suited for estimating seasonal or annual loads. All of these methods utilize probability sampling; therefore, they have the desirable feature that sediment load and its variance can be computed without bias.

Turbidity Threshold Sampling

Turbidity-threshold sampling (TTS) bases real-time sampling decisions on turbidity, measured by an instream probe. TTS attempts to collect enough samples to develop a regression from each storm event, so that concentration can be reliably estimated from turbidity for any period with significant sediment transport. The algorithm employed in TTS collects a sample each time a specified turbidity condition, defined by its magnitude and direction, is satisfied. Discharge information is used only to disable sampling when either the turbidity probe or pumping sampler intake are not adequately submerged. Regressions of concentration on turbidity are often linear, with small residual variance. Sediment loads from all but the smallest storm events can be accurately estimated from TTS samples. Variance estimation is also possible but is less reliable than variance estimation from probability sampling designs (Lewis 1996).

KEY LESSONS LEARNED IN NORTHWEST CALIFORNIA

- Proportional increases in peak flows resulting from both clearcutting and selective logging are greatest for smaller events.
- Increases in peak flows from clearcutting are related to proportion of area logged, watershed wetness, time after logging, and storm size.
- Inability of some studies to detect changes in peak flows after logging may be due to high variance rather than an absence of effect.
- Increases in water yield and summer low flows diminish over time and will probably be of minimal importance compared to other forest management and production goals.
- Suspended sediment loads increased almost three-fold from selective logging and road construction prior to implementation of the 1973 Forest Practice Act.
- Smaller, but statistically significant, increases in sediment were associated with clearcutting and roads under forest practice rules in effect in 1990.
- A few sites contribute most of the erosion and sediment from forest operations.
- Slope angle, horizontal curvature (topographic concavity), rock strength, and soil color can be used to identify sites with a high risk for landsliding.
- Improved monitoring and data analysis technology is available and can greatly benefit watershed studies.

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Sediment Production on Forest Road Surfaces in California's Redwood Region: Results for Hydrologic Year 2005-2006

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Abstract: This paper describes results of the first year (hydrologic year 2005-2006) of a study of surface erosion on forest roads on the Jackson Demonstration State Forest, Mendocino County, California. Estimated annual sediment production varied greatly among sites, from 0.05 kg/m^2/yr to more than 4 kg/m^2/yr. The estimated share of suspended sediment in total sediment production ranged from 33% to 86%. In this paper, we describe the study method, present basic results, and discuss practical issues in data collection and interpretation.

Keywords: Forest roads, surface erosion, sediment production

Introduction

This paper describes instrument and study design, data collection, and preliminary results from a pilot study of surface erosion associated with forest roads on the Jackson Demonstration State Forest in California's coast redwood region. The study is a joint effort by the California Department of Forestry and Fire Protection and NOAA Fisheries. Our goals were to test the technology and to collect data over a representative range of conditions. These data are useful in that they were previously unavailable for the redwood region and because they provide a basis for assessing the predictive power of erosion simulation models.

Annual sediment production for hydrologic year 2005-2006 varied significantly across road segments, from much less than 1 $kg/m^2/yr$ to more than 4 $kg/m^2/yr$. Rocked segments produced much less sediment than most unsurfaced segments, though two unsurfaced segments produced amounts comparable to the rocked segments. The estimated share of suspended solids in total sediment production ranged from 33% to 86%, with no clear relation to road characteristics. Annual rainfall measured in the Caspar Creek Watershed, where four of the ten sites are located, was 147% of normal, the third wettest year since 1962. A more definitive assessment of sediment production on our study sites will emerge as we develop a time series of data in subsequent years and overcome some of the technical difficulties encountered in this first year of the study, as described below.

Methods

Study Area and Site Characteristics

The ten road segments in this road surface erosion study are located on the Jackson Demonstration State Forest in western Mendocino County, California. The study area has a Mediterranean climate, with most of the precipitation occurring during the months of November through April. The area is mountainous, with elevations ranging from sea level to 880 meters (2,850 feet) in the east. Topography is generally steep and dissected as a result of rapid uplift rates. Underlying geologic materials are dominated by coastal belt Franciscan sandstone, and soils range from gravelly loam to fine-grained with a high clay content.

Study sites were chosen to represent a range of grade, surface, and traffic conditions typical of forest roads in the redwood region. Site selection was also influenced by operational considerations such as placement on the hillslope, travel distances, and reducing the risk of vandalism. Road segments were generally crowned with an inside ditch. Individual site conditions varied in topography, cut-bank height, ditch vegetation, and overhead canopy. Road surface conditions were also variable. Some unsurfaced roads contained a fraction of native rock, while rocked roads had variations in the condition of applied rock. Sporadic surface flow was present from cut banks at four sites and varied with storm intensity. The road segments selected are representative of many roads on the forest and in the region, but differ from contemporary new construction standards that require outsloping and rolling dips to reduce concentration of runoff. Basic information on the ten study sites is presented in Table 1.

Surface runoff on each road segment is directed to an inside ditch, from which a culvert directs it to devices that measure runoff and sediment production, as described below. Thus, the inside ditch is part of the road segment profile, although its relative contribution to sediment production on the road segment is unknown. The catchment area for runoff on each site was estimated from the base of the cut bank to the crown in the road that serves as a "water divide." Because the actual catchment area for each segment cannot be known precisely, and will likely vary to some extent with rainfall intensity, we are exploring the sensitivity of our results to potential measurement error. However, in this paper we report results based on the single catchment area value deemed most probable for each segment.

Instrument Design and Calibration

Our method of estimating runoff and sediment production is based on a design by Black and Luce (2007). Each site has a settling basin that captures coarse sediment generated on the road segment, a tipping bucket with event logger that enables estimation of total runoff, and a splash device that collects a subsample of the runoff for analysis of suspended solids. A 5 ml subsample (c. 0.05% of tipping volume) is collected at each tip of the approximately 10-liter bucket and flows through a flexible tube into a closed 19-liter (5-gallon) bucket, which acts as a reservoir for composite post-storm sampling.

Fig. 1: The settling basin and tipping bucket during initial calibration. The settling basin, made of 122 cm (48") diameter corrugated metal pipe, catches runoff from the road segment, allowing coarse sediment to settle. The runoff then passes through a rectangular arm and into the tipping bucket, shown here just after having tipped to the right. A subsample intake opening, obscured by the water in this photo, catches a 5-ml flow sample, which is routed to a bucket from which a composite sample is later collected for laboratory analysis of suspended solids.

Each tipping bucket is calibrated by providing flow at a known rate, using a flow meter, and recording the resulting duration between tips. Repeating this procedure with different flow rates enables estimation of a calibration curve relating flow rate to duration between tips, as shown in Figure 2. Durations recorded by the data logger can then be used in conjunction with the calibration curves to estimate runoff on each segment, generating a fine-scale hydrograph for each segment throughout the winter season.

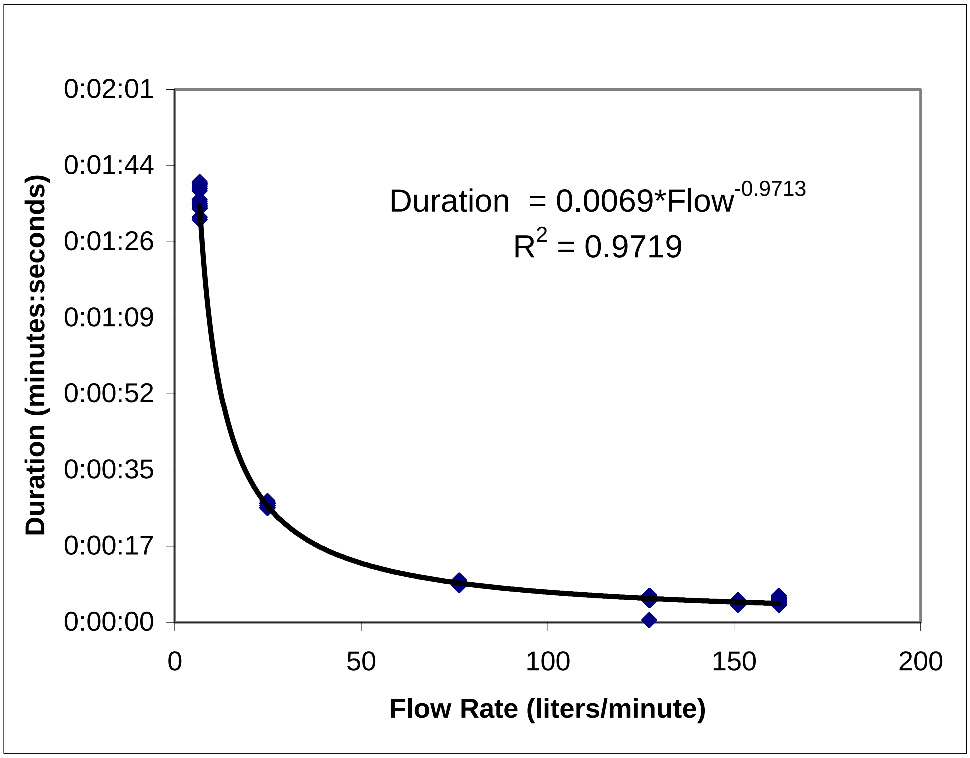

Fig. 2: Pre-season calibration curve for site 10. Fitting a power curve to tipping time data at known flow rates provides a relation between tipping times, downloaded from the data logger, and flow rates. After this relation is inverted to express flow as a function of duration, it is used in equation 1 below to estimate total runoff.

Data Collection and Analysis

Runoff and suspended solids data were collected throughout the rainy season. Regular visits, sometimes several per week, to the study sites were necessary to download tipping times from the event loggers. When the 19-liter buckets containing splash samples appeared to be nearing capacity, or when a sample had not been taken for several weeks, a mechanically agitated composite sample was collected from the buckets for analysis of suspended solids at a commercial laboratory using the EPA 160.2 protocol. The buckets were then cleaned and emptied. We measured total suspended solids (TSS) rather than suspended sediment concentration (SSC) due to budget considerations and the finding in Gray *et al.* (2000) that these measures differ little when the proportion of coarse sediment is small. In our study, the settling basin captured most of the coarse sediment before TSS samples were taken, minimizing the likely divergence between TSS and SSC.

To estimate runoff on each segment i during an interval j between consecutive tips of the tipping bucket, the flow rate associated with interval j's duration (derived from the calibration curve) is multiplied by that duration. That is,

$$1) \quad RUNOFF_{ij} = DURATION_{ij} * FLOW_{ij} = DURATION_{ij} * (\frac{DURATION_{ij}}{\alpha_{ij}})^{1/\beta_{ij}}$$

where $RUNOFF_{ij}$ is the runoff volume from site i during interval j, $DURATION_{ij}$ is the time elapsed during interval j, $FLOW_{ij}$ is the runoff rate derived from the calibration curve for intervals of j's duration, and α_{ij} and β_{ij} are the calibration coefficients for instrument i in interval j (in the example given in Figure 2, these are 0.0069 and –0.9713, respectively). Note that the calibration coefficients are time-varying, due to interpolation between pre- and post-season calibration values, a point we address in the discussion.

Letting k be an index of the period between two consecutive samples of total suspended solids, production on site i for the period k was estimated as the product of estimated runoff (L) and the associated TSS concentration (mg/L):

$$2) \quad TSS_{ik} = SAMPLE_{ik} * RUNOFF_{ik} = SAMPLE_{ik} * \sum_{j \in k} RUNOFF_{ij}$$

Note that $RUNOFF_{ik}$ is calculated as the sum of runoff over the between-tip intervals j, of which there are generally thousands per interval k between successive TSS samples. Thus, this measure of TSS_{ik} assumes the estimated TSS concentration $SAMPLE_{ik}$ applies to all runoff during period k. The results reported below reflect this assumption, as we have not yet tested the sensitivity of our results to other possibilities (e.g., interpolation between consecutive estimates of TSS, since discrete sampling of TSS leads to estimates that may take large steps from interval to interval, whereas they are almost certainly varying more smoothly over time).

Finally, annual values for runoff and TSS on site i are simply the sums of periodic values over j and k, respectively.

The coarse sediment captured by each settling basin was weighed at the end of the winter season with a 2,000 kg (+/- 500g) electronic dynamometer/crane scale. Each settling basin was topped off with water and the tank containing sediment and water was weighed and compared to previous weights of the tank and water without sediment. Several tanks filled mid-season, requiring weighing by hand due to restricted winter access on unsurfaced roads. The tanks were emptied by hand into 19-liter buckets that were transferred to a 38-liter (10-gal) bucket prior to weighing with the 2,000 kg crane scale mounted on a surveying tripod. The transfer to the 38-liter bucket was intended to minimize the number of measurements and effects of scale error. The mass estimates of the tank with the sediment-water mixture and with water alone were used to estimate the mass of dry sediment as in Black and Luce (2007):

3) $$M_S = \frac{\rho_S (M_{TSW} - M_{TW})}{\rho_S - \rho_W}$$

where M_S is the dry mass of sediment, M_{TSW} the mass of the tank with sediment and water, M_{TW} the mass of the tank filled with water alone, ρ_S is the particle density of sediment, and ρ_W is the density of water. We assume a sediment particle density of 2750 kg/ m^3 (see Wosika 1981, Appendix B) and a water density of 1000 kg/m^3.

Results

Table 1 shows summary results from the first year and information on characteristics of the ten road segments in the study. Figure 3 presents the same sediment information in a different form to allow easier comparison. The limited number of replications and variety of topographic conditions do not support a meaningful statistical analysis of the relation between sediment production and segment characteristics. Nevertheless, these results suggest that the rocked roads in this study produce less sediment than the native-surface roads, as expected. The proportions of coarse and suspended sediments vary considerably among sites, but not in any obvious relation to road characteristics: there are both rocked and unrocked roads that produce high relative proportions of both fine and coarse sediment.

Table 1. Site Characteristics and Summary Results for HY2005-2006. Winter traffic was not measured because it was limited to occasional light-duty vehicles. The final column is the ratio of suspended sediment to total sediment.

	Surface	Winter Traffic	Ditch (percent vegetated)	Grade	Area (m2)	Total Sediment (kg/m2)	Suspended Sediment Share
Site 1 (Rd 1000-1)	Unrocked	Light	10%	6%	1031	3.76	59%
Site 2 (Rd 240-1a)	Unrocked	None	0%	4%	716	4.15	41%
Site 3 (Rd 90-1)	Unrocked	None	10%	6%	634	1.34	73%
Site 4 (Rd 210-2)	Unrocked	None	10%	6%	778	0.07	49%
Site 5 (Rd 210-1)	Unrocked	None	10%	7%	560	0.12	33%
Site 6 (Rd 240-1)	Unrocked	None	0%	9%	399	4.57	54%
Site 7 (Rd 600-4)	Rocked	Light	75%	4%	757	0.05	35%
Site 8 (Rd 620-4)	Rocked	Light	30%	7%	452	0.23	86%
Site 9 (Rd 640-7)	Rocked	Light	30%	7%	723	0.10	58%
Site 10 (Rd 640-1)	Rocked	Light	20%	4%	573	0.23	40%

Using an assumed road surface bulk density of 1,600 kg/m^3 (Coe 2006), the sediment production rates given in Figure 3 correspond to surface depth loss rates of 0.03 mm/yr to 2.85 mm/yr.

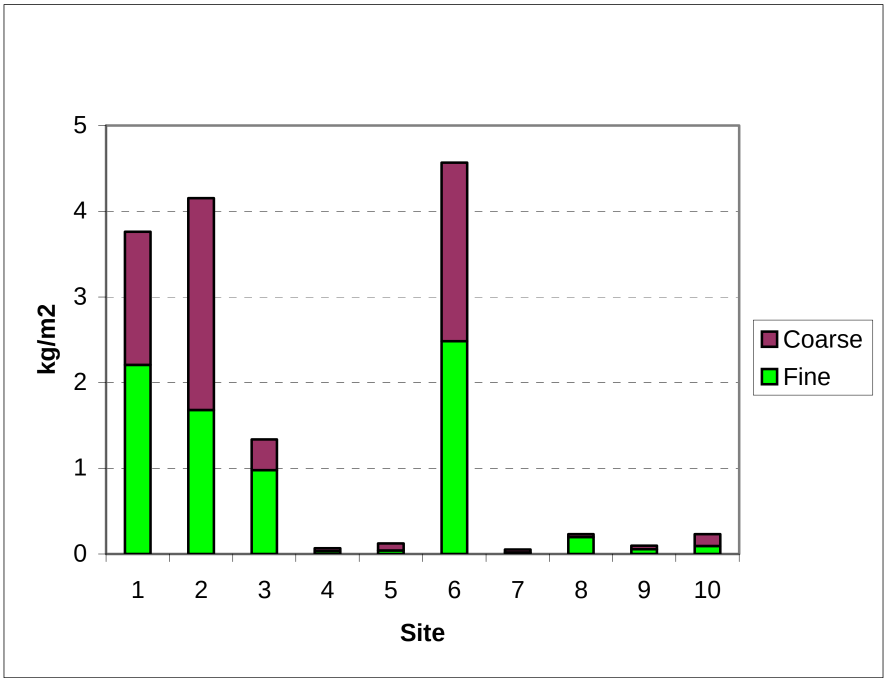

Fig. 3: Coarse and Fine Sediment Production, HY2005-2006. Data are normalized by area. Sites 1-6 are unsurfaced while sites 7–10 are rocked. While the highest-producing road segments were all unsurfaced, two unsurfaced segments (4 and 5) produced less sediment than the higher-producing rocked segments (8 and 10).

Discussion

Study results to date indicate that both total sediment production rates and the share of suspended sediment in total sediment differ greatly among sites. All four of the highest-producing segments in our study are unsurfaced, but two unsurfaced segments (4 and 5) produced less sediment than at least two rocked segments (8 and 10). Some of this variation can be attributed to known site variability, e.g., site 2 was the only one graded in the summer of 2005. Other factors, such as geology and ditch function, no doubt played a role as well, but we did not have sufficient data to support a statistical analysis of factors influencing variability in sediment production. Two points are particularly worth noting in interpreting our results. First, HY2005-2006 was a particularly wet year, with the North Fork of Caspar Creek reaching its highest peak flow since 1974. Second, we have not attempted to control for the influence of traffic, since all our road segments are either closed in winter or are believed to be used by 10 or fewer light-duty vehicles (pickups and sedans) per week.

Our results were generally consistent with other recent studies. An application of the WEPP simulation model (Ish and Tomberlin 2007) generated a mean long-term surface erosion rate estimate of 4.14 kg/m^2 on native surface roads in our study area, which is similar to the higher sediment production rates shown in Fig. 3. In studies from the interior portions of California, Coe (2006) reported a 16-fold difference in median sediment production rates between rocked and un-rocked road segments in the central

Sierra Nevada, while Korte and MacDonald (2007) found that native and mixed surface roads produced approximately three times the sediment as gravel surfaced roads in the southern Sierra Nevada.

The results presented here are preliminary, as they represent data from a single year with unusually heavy precipitation, and we have not yet explored the sensitivity of the results to uncertainties about sediment concentrations, catchment areas, and equipment function. A particularly important example of the latter was marked differences in the calibration coefficients for some tanks before and after the rainy season. Because we cannot know the rate at which the calibration coefficients changed during the season, the results reported here are based on a simple linear interpolation over time between the pre- and post-season calibration coefficients. Examining the sensitivity of our results to other possible patterns of change in the calibration coefficients—for example, such that the initial coefficients were operative until the last day of the season, or that the final coefficients were operative after the first day of the season—will enable us to bound the range of results consistent with our pre- and post-season calibration measures.

Additionally, several known technical problems add to the uncertainty in these HY2005-2006 results. Because it was a heavy rain year and the data loggers could only record 8150 tips, 47 of 338 total data downloads (14%) indicated that the data logger had filled, resulting in some lost data. There was also some minor equipment damage due to site visitors, e.g., on two occasions the tubing directing runoff subsamples to a collection reservoirs was removed. On site 7, an old buried culvert was found to be directing a significant amount of water from the study segment under the road and away from our instruments.

There are important questions related to road surface runoff and erosion that are beyond the scope of our study. We have not attempted to develop a statistical analysis of the factors contributing to road sediment production, to assess delivery of sediment to the stream network, nor to investigate the share of organic material in sediment production (although we have begun to examine the organic/inorganic breakdown in the second season of data collection).

The goals of this study are more limited: 1) to examine the feasibility of a particular approach to estimating road surface erosion in the redwood region, and 2) to generate estimates of sediment production on a representative range of road segments. The results presented here suggest that the method generates useful information, though at a significant cost: instrumentation at each site cost approximately $1800, while project initiation and data collection during the first year required approximately one staff person-year.

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Even-Aged Management and Landslide Inventory, Jackson Demonstration State Forest, Mendocino County, California[1]

Julie A. Bawcom[2]

Abstract

Tree removal associated with clearcutting in a coastal redwood forest does not alone initiate numerous shallow landslides that deliver large quantities of sediment to watercourses. This landslide inventory focused on the relationship between vegetation removal in a predominantly second-growth redwood forest and shallow landslides. Deep-seated dormant landslide features were mapped to record if reactivation had occurred within clearcuts. This field-based inventory included mapping all fifty clearcut units and characterizing landslides by type, cause, age, sediment delivery and stream classification. Thirty-two active landslides were found; and all except two were associated with older roads. The percent of the total landslide volume to the volume of landslide sediment reaching a watercourse for each watershed varied from 34 to 75 percent delivery. Sediment delivery depended on the proximity of the road to higher order watercourses. Many deep-seated dormant rockslides were mapped within clearcut units with no reactivation except within old road fills. Mapping nearby uncut or partially cut control units with similar slope characteristics as the clearcuts have yielded similar road failures.[3] Results of this study support the current focus on road rehabilitation and decommissioning for watershed restoration.

Key words: forestry, Jackson State Forest, landslides, redwoods

Introduction

Clearcutting in North Coast redwood forests is suggested as being a significant factor contributing to an increase in landsliding. Available scientific data are not consistent in documenting an association between the two. To better understand the relationship between vegetation removal and slope stability in a second-growth redwood dominated conifer forest, a landslide inventory was prepared for all units that have been clearcut on Jackson Demonstration State Forest (JDSF) in coastal Mendocino County *(fig. 1)*. JDSF is within the redwood forest region in western Mendocino County between Fort Bragg and Willits, California *(fig. 1)*. It is the largest of eight California Demonstration State Forests, having a total area of 48,652 acres, including over 100 miles of fish bearing streams and over 300 miles of actively maintained forest roads.

The four watersheds where clearcutting occurred are in the west half of JDSF. These are the South Fork Noyo River (17,348 acres), a tributary to the Noyo River;

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[3] Unpublished mapping by author.

Berry Gulch (7,993 acres), tributary to the Little North Fork of Big River; Hare Creek (6,179 acres), a small coastal stream, and the North Fork of Caspar Creek (1,168 acres),[4] a portion of a small coastal stream *(fig. 1)*. All four of these watersheds are dominated by second-growth redwood with varying percentages of Douglas-fir and hardwoods, primarily tanoak (Henry 1998).

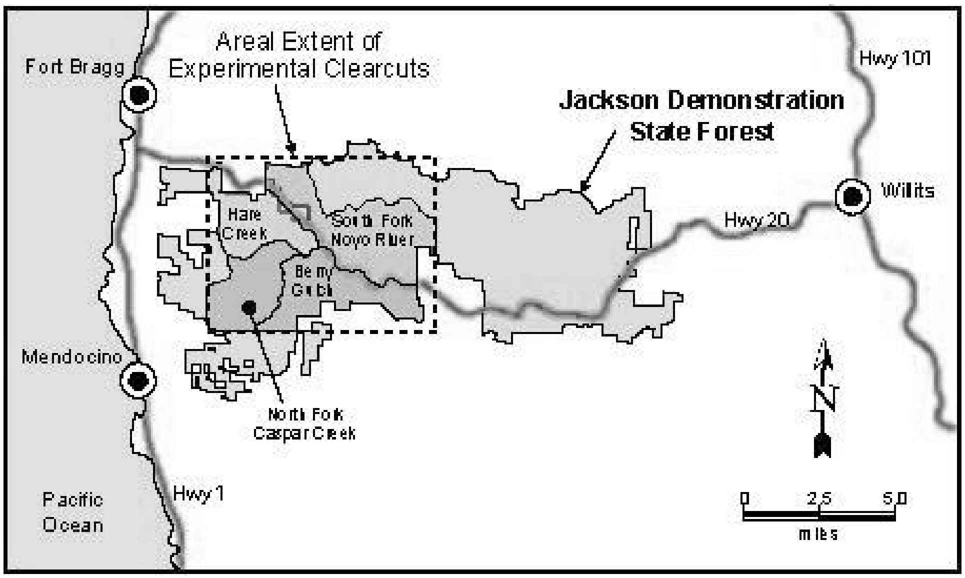

Figure 1—Map of Jackson Demonstration State Forest and the study watersheds. (Unpublished mapping by the author.)

The bedrock geology of the four watersheds is fairly uniform Coastal Belt of the Franciscan Complex (Kilbourne 1982, Kilbourne and Mata-Sol 1983, Manson and others 2001, Short and Spittler 2001, Spittler and McKittrick 1995). Marine sandstones, siltstones, mudstones and conglomerates are dominant, with minor belts of volcanic rocks locally exposed. Weathering and mechanical fracturing is highly variable, and can change dramatically over short distances. The sedimentary rocks are generally massive to poorly bedded sandstone with local outcrops of thin-bedded friable siltstone and shale. A blocky fracture overprints all of the bedrock units. Local areas of deeply weathered and sheared materials with low shear strengths occur in the area studied. The slopes of the clearcut areas are highly variable.

Historic land use pre-dating the establishment of JDSF includes logging of old-growth forest between 1860 and 1947. Splash dams and animal teams were utilized along portions of Big River and Caspar Creek (1860 to 1890). Logging methods changed to steam donkey and railroad in the late 1800s and continued until the beginning of World War II. Many remnant historic logging features (railroad grades, trestles, steam donkey cables and blocks and portions of splash dams) are evident throughout the State Forest. Tractors were first used in the woods prior to World War II, gradually becoming the predominant method of yarding timber by the 1940s. Road construction and use of logging trucks began in the 1930s and replaced the

[4] The North Fork of Caspar Creek watershed is defined as the portion of the basin above the weir constructed in 1962 for the Caspar Creek Watershed Study (Ziemer 1998).

railroad by the mid-1940s. The majority of the main roads on JDSF were constructed between 1950 and 1980.

An association between clearcut harvesting of coniferous forests and landsliding is clearly demonstrated in many parts of the world (Bishop and Stevens 1964, Gray 1970, Montgomery and others 2000, O'Loughlin 1974, Swanston and Swanson 1976). Analyses of this association typically identify the loss of root strength as a controlling mechanism (Abe and Ziemer 1991, Wu and Swanston 1980, Ziemer 1981). Most of these studies involve non-sprouting species, such as pine, varieties of fir, and Douglas fir.

Methods

Between 1980 and 1995, fifty forested blocks were clearcut within nineteen timber sales on JDSF. Approximately 1800 acres were clearcut in four separate watersheds. Most of the clearcut blocks in this study consisted of 80- to 100-year old second-growth stands that naturally regenerated following clearcut logging of the old-growth forest between 1860 and 1947. This majority of second entry clearcuts on JDSF represent a unique data set. Since 1994, JDSF has limited the amount of even-aged management conducted and has transitioned from clearcutting to a system that includes retention of structure trees for habitat purposes.

For this study all of the landslides within modern clearcut units were mapped on the ground using field-mapping methods. Aerial photo interpretation was completed for gathering background data using several sets of photos (1947, 1964, 1981, 1984, 1988, 1996, 2000). Logging history, road construction, date and type of logging method and site preparation were recorded. Within one of the four watersheds, the North Fork of Caspar Creek is part of a paired watershed study of the effects of logging and road building on stream flow, sedimentation, anadromous fish and fish habitat (Lewis and others 2001, Ziemer 1998). Detailed information on the sub-watersheds of the North Fork (14 gauging stations, rain-gauges, subsurface drainage and soil piping sites, and the solar radiometer site) is documented in Ziemer (1998). In addition to geomorphic mapping (Spittler and McKittrick 1995) an inventory of sediment sources, including landslides, are compiled yearly in the North Fork Caspar Creek database (Elizabeth Keppeler, USFS-PSW, personal communication, 2004). Cafferata and Spittler (1998) identified that storm sequences meeting the criteria for triggering landsides have occurred in all phases of the Caspar Creek study, with the greatest number in 1998. Precipitation amounts of at least two inches in one day combined with five inches in three days or eight inches in ten days are thought to trigger landslide events. Three record water years of precipitation occurred with 79.03 inches in 1983, 61.38 inches in 1995, and 80.50 inches in 1998. Because of the close proximity of the other watersheds to Caspar Creek *(fig. 1)*, all of the clearcuts in this study were exposed to rainfall conditions capable of triggering landslides. The landslides mapped for this study were compared to a shallow landslide potential map developed for JDSF by Vestra Inc. in 1997 using the SHALSTAB model. Just over half of the landslide failures occurred within a potential instability rating of moderate or high.

Results

South Fork Noyo River

Seventeen clearcut units totaling 557 acres were mapped for landslides along the South Fork Noyo River *(fig. 2)*. Six of the clearcut units had been broadcast burned after logging. Most of the clearcut units have a Northwest orientation except for three units along the Bear Gulch tributary. Clearcut logging of the original old growth occurred in this basin between 1900 and 1930. Modern clearcutting occurred between 1985 and 1990. Only three westernmost clearcut units had been selectively harvested using tractors in 1968 and 1969. A total of 67 miles of roads are present in the watershed. Slopes are variable, ranging from 10 percent to over 70 percent. About one-half of the units are on dormant landslides (deep-seated rockslides) mapped for this study. None of these deep-seated landslides reactivated after clearcutting. Fifteen landslides (not including the dormant landslides) were mapped in and near the modern clearcut units, with six of these exhibiting evidence of displacement since the most recent logging entry. All of the 15 recently active landslides in clearcut units are shallow debris slides and fill slumps (rotational fill-slides) related to midslope roads constructed in the 1980's. No in-unit landslides occurred in South Fork Noyo clearcut units following the recent harvest. Three slides represent a total of 117 cubic yards of sediment measured in the field delivered into first order watercourses with a delivery ratio for South Fork Noyo River of about 70 percent *(table 1, fig. 2)*.

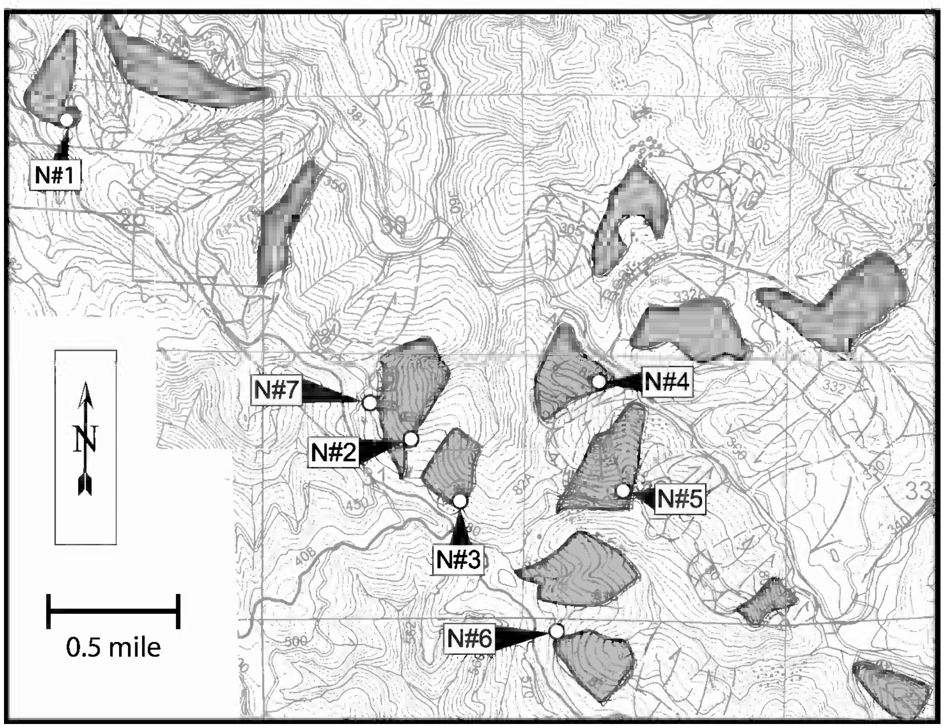

Figure 2—South Fork Noyo River clearcuts and mapped landslides.

Table 1—*South Fork Noyo River landslides summary (refer to map in* figure 2*).*

Slide no.	Volume (yd^3)	Age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
N1	90	1988	# 259 1979	90	Class III	100
N2		1988	#91 1987	None	None	N.A.
N3	15	1997	#91 1987	15	Class III first order	100
N4	213	1988-1989	#80 1987	None	N.A.	N.A.
N5	415	1997	#80 1987	None	N.A.	N.A.
N6	670	1987	# 81 1987	None	N.A.	N.A.
N7	60	1997	# 91 1987	12	Class III first order	20

Berry Gulch

Nine clearcut units totaling 228 acres in Berry Gulch are located along an unnamed northern tributary of Berry Gulch on south-southwest facing slopes *(fig. 3)*. Most of the units are situated in the upper reaches of the watershed. Five of the nine units were broadcast burned after logging. In addition to the clearcut units, a total of forty miles of roads are present in the Berry Gulch watershed. The old-growth forest was clearcut between 1900 to 1920 using steam donkeys to yard the trees to a railroad spur within the stream channel. The second-growth stands were harvested between 1964-1968 using tractors and 1960s roads constructed along mid and lower slopes. Recent clearcuts occurred between 1987 and 1994 using the 1960s road system. Hill slopes range between 20 and 80 percent. About half of the clearcuts are within dormant deep-seated landslides. A total of eight landslides were mapped (not including the dormant landslides) in the area of the clearcut units occurring between the 1960s logging and the present. Only three of the slides occurred in the 1980s and 1990s, following modern clearcutting. All eight landslides are shallow debris slides and shallow rockslides related to roads or landings. Three landslides delivered sediment to watercourses *(table 2, fig. 3)*. No in-unit landslides occurred in the Berry Creek clearcuts following the recent harvesting. A total of 675 cubic yards of sediment measured in the field was delivered to first order watercourses, with a total delivery ratio for Berry Gulch at about 75 percent.

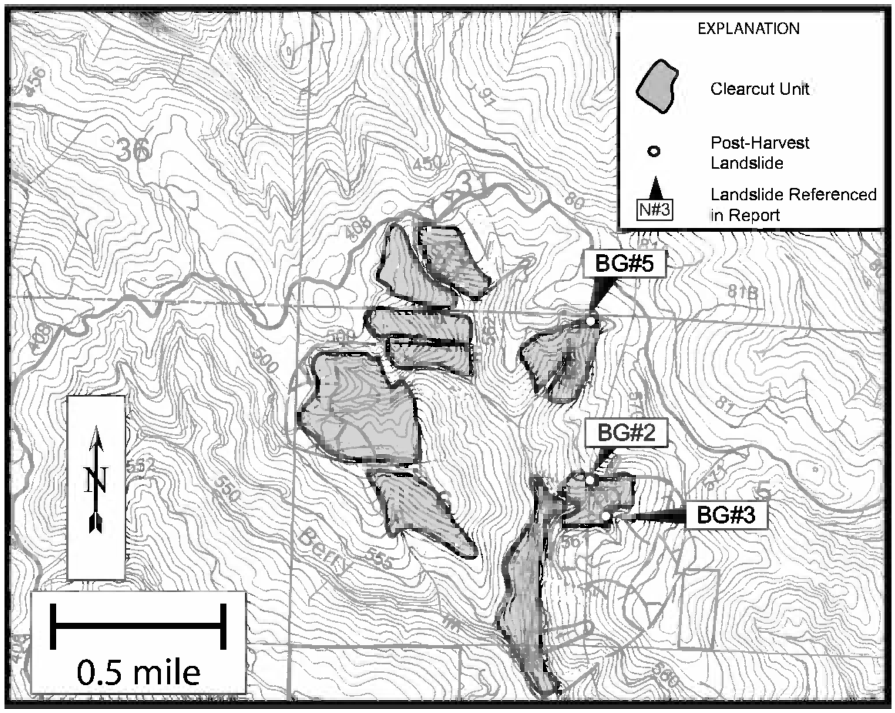

Figure 3—Berry Gulch clearcuts and mapped landslides.

Table 2—*Berry Gulch landslide summary(refer to map,* figure 3).

Slide no.	Volume (yd^3)	Failure age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
#BG2	129	1975?	# 560 1965	90	Intermit. class III	70
#BG3	100	1995?	# 561 1965	100	Intermit. class III	100
#BG5	670	1980's	# 560 1965	485	class III	72

Hare Creek

Hare Creek is a northwest trending coastal stream with two main tributaries: Bunker Gulch to the north and South Fork Hare Creek to the south. Sixteen clearcut units totaling 353 acres were harvested between 1982 and 1990 *(fig. 4)*. Seven of the sixteen units were broadcast burned after logging. Forty-six miles of roads are located within the watershed built in the 1950s to1980s. The original old growth was clearcut between 1880 and 1900. Topography is highly variable, with slopes under 50 percent along the eastern portion of Hare Creek, and steeper 70 percent slopes along portions of the South Fork and western section of Hare Creek. Only one dormant landslide was mapped in the area of modern clearcutting. A total of twelve landslides occurred, many around the time of road construction before modern clearcutting. All these historically active features are shallow debris slides and fill slumps (rotational fill/slides) related to landings, roads and skid trails constructed in the 1950s to 1980s on lower mid-slopes and along steep stream channel banks. No in-unit landslides occurred in Hare Creek clearcut units following the recent harvesting. Six of the landslides delivered sediment to a watercourse. A total of 4,530 cubic yards of sediment measured in the field was delivered to a first, second or third order watercourse, with a total delivery ratio for Hare Creek of about 34 percent *(table 3, fig. 4)*.

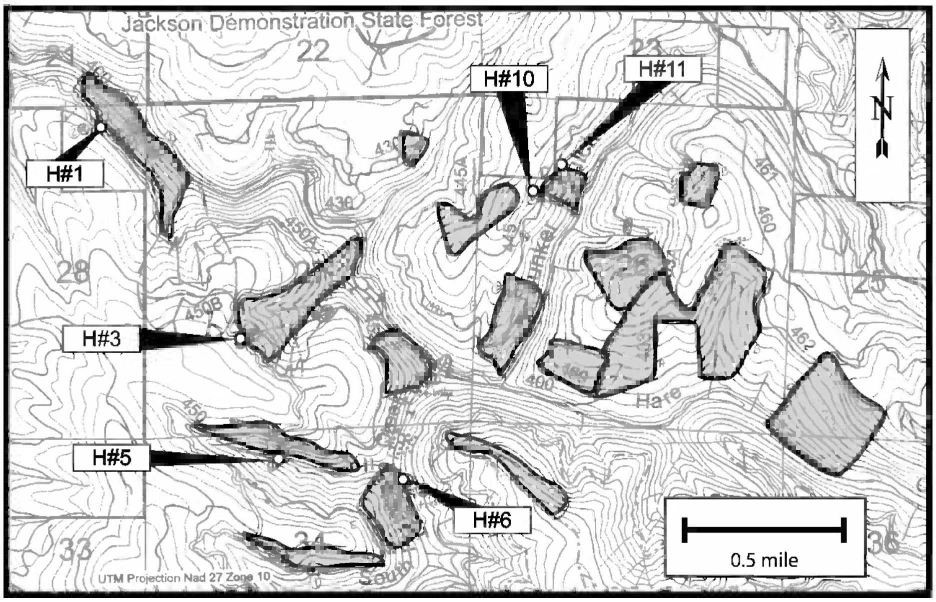

Figure 4—Hare Creek clearcuts and mapped landslides.

Table 3—*Hare Creek landslides summary (refer to map,* figure 4*).*

Slide no.	Volume (yd^3)	Failure age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
#H1	4800	1995-96	#450 1950's+	300	Hare Creek Class I	6
#H3	930	1997	#450 1950'+	400	Class III	43
#H5	4000	1992-93	#453 1973-74	3500	Class II watercourse	87
#6	3000	1970's	Railroad 1920s	50	South Fork Hare Creek	2
#10	267	1984 +1997	#445 1983-84	200 +50 1997	Bunker Gulch,	94
#11	500	1984-85	#445 1983-84	20-30	Class III	6

North Fork Caspar Creek*:* The North Fork of Caspar Creek is a north-northeast trending tributary of a small coastal stream. Caspar Creek was initially logged between 1860 and 1904 using splash dams to transport logs (Napolitano and others 1989). Ten modern clearcut units totaling 681 acres (about half the watershed) were harvested between 1985 and 1992 in the North Fork of Caspar Creek *(fig.5)*.

These clearcuts are part of a forty-year ongoing study of the watershed effects of harvesting and road building. Four of the ten clearcut-units were broadcast burned for site preparation. The recent logging used 7.1 miles of existing roads and 5.2 miles of new roads located near ridges (Cafferata and Spittler, 1998). The main road providing access to the units (Road 500) was constructed in the 1950s. After the modern clearcut logging in the North Fork Caspar Creek, both the logged and unlogged areas were examined, and a total of six slope failures were found in the clearcut units: four in-unit and two related to roads. Four of the slope failures contributed sediment to a first order watercourse. Two of the largest in-unit delivering landslides (C98, C207) were partly triggered by concentrated road drainage into the clearcut slope. The two slope failures were gully formation of a collapsed soil pipe and within the watercourse and lake protection zone (WLPZ) buffer where channel slumping occurred (C129, C160). As of the spring of 1998, the size and number of landslides in the North Fork Caspar Creek basin were similar in logged and unlogged units and the volume of sediment discharged by landslides from areas cut or uncut was also about the same (Cafferata and Spittler 1998). The two landslides and two slope failures in the North Fork Caspar Fork had a total volume of 9,464 cubic yards delivering about 4000 cubic yards and a total delivery ratio of 42 percent *(tables 4* and *5, fig.6)*.

Table 4—*Caspar Creek landslides summary (refer to map,* figure 5*).*

Slide no.	Volume (yd^3)	Failure age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
#C98	5574	1995	1985	2200	Class III	39
#C129	110	Jan 1996	N.A.	110	Class III	100
#C160	100	1998	N.A.	50	Class I	50
#C207	3680	Dec 2002	1950	1600	Class III	44

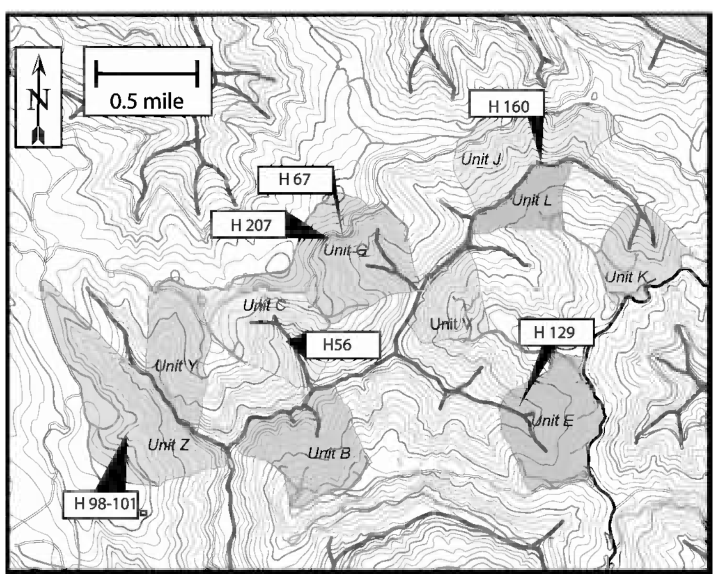

Figure 5—North Fork Caspar Creek clearcuts and mapped landslides.

Discussion and Conclusions

Jackson Demonstration State Forest is unique due to its availability for all types of research in the coastal redwood region, the previously un-entered second growth redwood stands and the uniformity of the underlying geologic bedrock. This research project was able to study the removal of trees in 80- to 100-year old second-growth stands that naturally regenerated following clearcut logging of the old-growth forest between 1860 and 1947. All other studies related to harvesting and slope stability has been completed in areas with predominantly non-sprouting tree species. The inventory on JDSF represents a unique data set.

This study attempts to document the relationship between clearcutting and shallow landslides that deliver sediment to watercourses in a redwood dominated conifer forest. All fifty units, totaling about 1800 acres, within four watersheds were clearcut, and many were broadcast burned for site preparation following logging. All were subjected to storms capable of triggering landslides (Cafferata and Spittler 1998) with thirteen storm sequences above thresholds for initiating landsliding. Following the most recent clearcut logging (1980-1995), a total of thirty-two landslides failed within clearcut units and no deep-seated dormant landslides showed evidence of reactivation except road fill failures.

Of the thirty-two landslide failures, all but two have an association with old roads and landings. Mapping by the author of nearby uncut or partial cut control units revealed similar road related failures. The four in-unit landslides occurred in the Caspar Creek watershed. The first one in the uncut watercourse and lake protection zone (WLPZ), another as gully formation from a collapsed soil pipe and the last two in-unit debris slides associated with poor road drainage. Only the North Fork Caspar Creek watershed with 50 percent of the watershed clearcut as part of a paired watershed study had in-unit landsliding associated with upslope road drainage. The in-unit landsliding may be attributed to the large unit area of vegetation removal along steep slopes and use of other treatment methods such as pre-commercial thinning.

The total amount of sediment delivered from landslides after clearcutting in all four watersheds was about 8,800 cubic yards *(table 5),* with 98 percent of that total associated with old roads and landings. In comparison, one natural debris slide that failed in 1975 and 1998 in another portion of the State Forest (North Fork of the South Fork Noyo River) in uncut fifty-year old second growth delivered about 10,000 cubic yards.

Table 5—*A summary of the number of sediment delivering landslides in each watershed, amount delivered and the delivery ratio (in percent) for each watershed.*

Watershed	Percent acres clearcut	# of slides	Road-related	In-unit slides	# of slides delivering sediment	Delivery (yd^3)	Delivery ratio percent
SF Noyo	4	6	6	0	3	115	70
Berry Gulch	5	8	8	0	3	675	75
Hare Creek	9	12	12	0	6	4530	34
NF Caspar	50	6	4	2	4	3960	42
Total		**32**	**28**	**4**	**16**	**8,740**	

The results of this inventory suggests that vegetation removal associated with clearcutting alone has not been a significant contributor to slope instability or delivery of sediment, and older road, skidtrail and landing fills are the predominant source of shallow landsliding and stream aggradation. Currently there are no other published studies similar to this inventory within coastal redwood forests. The results of this study are supported by other unpublished studies conducted in several nearby watersheds on private industrial timberlands.[5] The results of this inventory also find no increase in the rate of landsliding or initiation of movements of older dormant landslides within clearcuts on JSDF. Results show a clear relationship between roads, particularly older roads as the main source of sediment delivery to streams from shallow landslides and erosion.

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MONITORING STUDY GROUP
CALIFORNIA STATE BOARD OF FORESTRY AND FIRE PROTECTION

Modified Completion Report MONITORING PROGRAM

Implementation and Effectiveness of Forest Practice Rules related to Water Quality Protection

MONITORING RESULTS FROM 2001 THROUGH 2004

Ruben Grijalva
Director
Department of Forestry and Fire Protection

Mike Chrisman
Secretary for Resources
The Resources Agency

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Governor
State of California

July 2006
SACRAMENTO, CALIFORNIA

ABSTRACT

The California Forest Practice Rules (FPRs) (Title 14, California Code of Regulations) are designed in large part to protect water quality and aquatic habitat in forested watersheds during and after silviculture activities (Figure 1). The critical questions then become: 1) At what rate are the water quality related FPRs being properly implemented?, and 2) When properly implemented, how effective are these FPRS in protecting water quality by retaining canopy and groundcover in watercourse and lake protection zones (WLPZs), by preventing erosion, by preventing sediment transport, and/or by preventing sediment transport to stream channels? The Modified Completion Report (MCR) program focused on answering these two basic questions using forensic monitoring data collected on a random selection of 281 Timber Harvesting Plans (THPs) and randomly selected sites within those THPs. The data were collected in the field primarily by the California Department of Forestry and Fire Protection's (CDF's) Forest Practice Inspectors and were analyzed by CDF's watershed staff in Sacramento, California. Overall, the MCR monitoring study found that: 1) The rate of compliance with FPRs designed to protect water quality and aquatic habitat is generally high, and 2) FPRs are highly effective in preventing erosion, sedimentation and sediment transport to channels when properly implemented. There are specific areas where improvements in implementation and/or effectiveness could be made, and these are enumerated with specific recommendations at the end of this report. The findings of the MCR monitoring project are comparable to the findings of the earlier Hillslope Monitoring Program (HMP) project (Cafferata and Munn 2002).

KEY TERMS: water quality, aquatic habitat, forestry, monitoring, streams, California Forest Practice Rules (FPRs) (Title 14, California Code of Regulations), Timber Harvesting Plans (THPs) watercourse and lake protection zones (WLPZs), roads, watercourse crossings, WLPZ canopy, groundcover, erosion, sediment transport, and sediment transport to channels.

Figure 1. A small watercourse or stream in a forest in California.

CALIFORNIA DEPARTMENT OF FORESTRY AND FIRE PROTECTION

Modified Completion Report
MONITORING PROGRAM: MONITORING RESULTS FROM 2001 THROUGH 2004

July 2006

by Clay A. Brandow, Peter H. Cafferata and John R. Munn
California Department of Forestry and Fire Protection

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The Monitoring Study Group (MSG) is a standing committee of the BOF made up of members of the public, resource agencies (both state and federal), three universities, and the timber industry. The agencies listed above make up the MSG; the names listed above are the current primary representatives for these agencies at MSG meetings. The MSG chair is appointed by the Board of Forestry and Fire Protection (BOF) and the group is staffed by CDF. Each agency and organization is responsible for determining the appropriate person(s) to serve as a representative on the MSG (i.e., the BOF does not make formal appointments to the MSG).

Modified Completion Report

Executive Summary

A key objective of California's Forest Practice Rules (FPRs) is to protect the beneficial uses of water (Figure 2). To determine whether this is being accomplished, the Board of Forestry and Fire Protection and the California Department of Forestry and Fire Protection (BOF/CDF) have established a long-term monitoring program, which includes a number of monitoring projects that are briefly described at the end of this Executive Summary. The Modified Completion Report (MCR) project is a major component of this long-term program. This report:

- Describes MCR monitoring conducted from 2001 through 2004,
- Summarizes and analyzes the MCR monitoring results, and
- Makes findings and recommendations based on those results.

The purpose of the MCR project has been to determine the adequacy of both implementation and effectiveness of the Forest Practice Rules (FPRs) that are used to protect water quality and riparian/aquatic habitat.

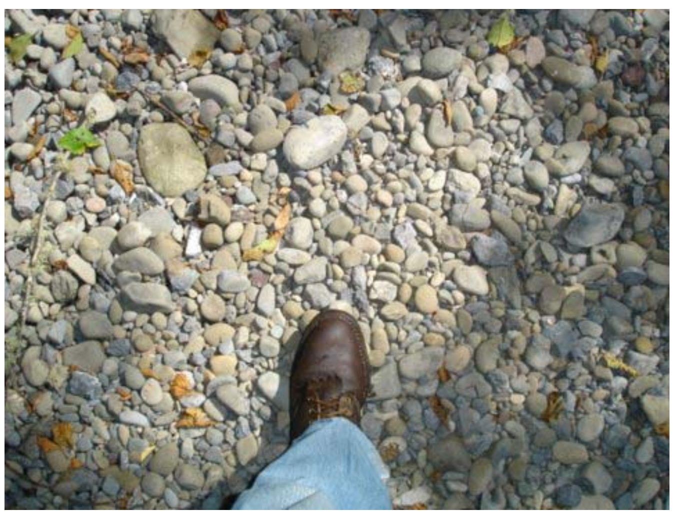

Figure 2. Substrate of a watercourse or stream in a forested watershed on the California coast. Reaches with clean gravel are an important habitat component of many forested streams. A key objective of the water quality related FPRs is to prevent transport of excessive fine sediment (e.g., sand and silt) to watercourse channels.

MCR monitoring is an extension of the normal timber harvest inspections and Completion Reports that CDF is required to conduct on timber harvesting plans (THPs) by the California Forest Practice Act and the FPRs. MCR data was collected by CDF Forest Practice Inspectors on a random sample of THPs at the time of plan completion and/or during the erosion control maintenance period.

Based on the findings of CDF's earlier Hillslope Monitoring Program (HMP) project (Cafferata and Munn 2002), the MCR project has focused on the following landscape features:

1) Watercourse and Lake Protection, including:
    - WLPZ Percent Total Canopy
    - WLPZ Groundcover and Erosion Features
2) Roads, and
3) Watercourse Crossings

Although the MCR project used a different random sample of THPs than the HMP (1996-2001) and was performed by CDF Inspectors instead of a third-party contractor, the results of these two studies are comparable. Furthermore, the MCR and HMP watercourse crossing effectiveness results compare well with findings of other California studies, such as the USDA Forest Service's Best Management Practices Effectiveness Program (BMPEP) (USFS 2004).

The *MCR Monitoring Procedures and Methods* are included in Appendix A of this report and are found on-line at:

http://www.bof.fire.ca.gov/board/msg_archives.asp

In both the MCR and the HMP studies, effectiveness of erosion control measures is based on the assumption that if soil is kept on site and out of stream systems, then water quality and riparian and aquatic habitat are protected from the effects of increased sedimentation.

Like HMP monitoring, MCR monitoring found that: 1) The rate of compliance with the FPRs designed to protect water quality and aquatic habitat is generally high, and 2) the FPRs are highly effective in preventing erosion, sedimentation and sediment transport to channels when properly implemented.

In most cases, Watercourse and Lake Protection Zone (WLPZ) canopy and groundcover exceeded Forest Practice Rule (FPR) standards. For Class I and Class II WLPZs, average total percent canopy was 84% for the Coast area (Region 1), 68% for the Inland North area (Region 2) and 73% for the Inland South area (Region 4). With rare exceptions, WLPZ groundcover exceeds 70%, patches of bare soil in WLPZs exceeding the FPR standards are rare, and erosion features within WLPZs related to current operations are uncommon. Moreover, in most cases, actual WLPZ widths were found to meet or exceed FPR standards and/or widths prescribed in the applicable THP.

There are rare instance were WLPZ canopy and groundcover do not meet FPR standards, either naturally or as a result of harvesting operations. Detection, and where possible, prevention or abatement of these rare occurrences is an important key to water quality protection. Because these occurrences are rare, rapid ocular inspection of as many high-risk WLPZs as possible is the recommended method of detection for enforcement purposes, saving the more rigorous and time consuming measurement method and procedures to follow up on observed problems and document possible WLPZ violations.

When properly implemented, road-related FPRs were found to be highly effective in preventing erosion, sedimentation and sediment transport to channels. Overall implementation of road-related rules was found to meet or exceed required standards 82% of the time, was marginally acceptable 14% of the time, and departed from the FPRs 4% of the time. Road-related rules most frequently cited for poor implementation were waterbreak spacing and the size, number and location of drainage structures.

This low rate of non-compliance is important because erosion and sedimentation was found to be much more likely at road-related features where the FPRs are not properly implemented. Additionally, erosion, sedimentation and sediment transport is much more likely at road-related features where there was a departure from the applicable FPRs. For example, when there is a departure from the rule, the chance of erosion is about 1 in 2, the chance of sediment transport is about 1 in 3, and the chance of sediment transport to a channel 1 in 10. But where the FPR implementation is acceptable or better, the chance of erosion is about 1 in 20, and the chance of sediment transport or sediment transport to a channel is equal to or less than 1 in 100. In addition, more than half of the departures from the FPRs are concentrated in the worst six percent of all road segments. Finding and fixing the drainage and discharge problems on these few bad segments would have the greatest impact on improving road-related water quality problems for the least cost.

Watercourse crossings present a higher risk of discharge into streams than roads, because while some roads are close to streams, all watercourse crossings straddle watercourses. Overall, 64% of watercourse crossings had acceptable implementation of all applicable FPRs, while 19% had at least one feature with marginally acceptable implementation and 17% had at least one departure from the FPRs. Common deficiencies included diversion potential, fill slope erosion, culvert plugging, and scour at the outlet.

All these topics and more are covered in detail in the full report. Findings and recommendations can be found at the end of the report.

MCR Project Context:
Brief Synopsis of BOF/CDF Long Term Monitoring Program

The BOF/CDF *Long Term Monitoring Program* (LTMP) has had three main components from 1996 through 2004. These are: 1) Modified Completion Report (MCR) Monitoring, 2) the Hillslope Monitoring Program (HMP), and 3) Cooperative Instream Monitoring Projects (CIMPs). An additional component, the *Interagency Mitigation Monitoring Program* (IMMP), will build on the HMP and the MCR projects and is currently being designed by an interagency team.

HMP monitoring was conducted from 1996 through 2002. MCR monitoring was conducted from 2001 through 2004. CDF plans to revise and re-start MCR monitoring in 2006. CIMPs began in 1997 and are ongoing. IMMP monitoring will begin as soon as the monitoring study design is completed.

MCR monitoring is an extension of the normal timber harvest inspections and Completion Reports that CDF is required to do on THPs under the California Forest Practice Act and the Forest Practice Rules (FPRs). MCRs are done by CDF Forest Practice Inspectors on a random sample of THPs at the time of THP completion and/or during the erosion control maintenance period. MCR used a different random sample of THPs than the HMP, but the results are comparable. The MCR random sample analyzed in this report included 281 plans, all THPs. The HMP random sample analyzed in Cafferata and Munn (2002) included 300 plans, of which 295 were THPs and five were Non-Industrial Timber Management Plan – Notices of Timber Operations (NTMP-NTOs). Plan submission dates in the two random samples ranged from 1993 to 2002 for the MCR random sample analyzed in this report and from 1991 to 2000 for the HMP random sample analyzed in Cafferata and Munn (2002).

HMP monitoring assessed a random sample of completed THPs that had over-wintered from one to four years, using an outside contractor. The objective of the HMP was to evaluate the implementation and effectiveness of Forest Practice Rules and special THP provisions specifically designed to protect water quality and riparian and aquatic habitat.

The CIMPs measure water quality and aquatic habitat parameters in selected basins. The objectives are two-fold: 1) to establish baselines and trends, and 2) to gage the effects of all activities in a watershed on the beneficial uses of water. It is often difficult to establish cause and effect (i.e., link current management practices to instream conditions), and instream monitoring is not specific to the impacts of timber management alone. Instream monitoring is important in establishing whether overall efforts to protect the beneficial uses of water are succeeding or failing, and can address cumulative watershed impacts.

The IMMP is being developed to provide information regarding forestry-related practices at high-risk sites where practices have been designed to protect water quality. The IMMP will use multi-agency teams composed of representatives from CDF, California Department of Fish and Game (CDFG), California Geological Survey (CGS), and the Regional Water Quality Control Boards (RWQCBs). It is anticipated that this team approach will provide a balance of interests for all the Review Team agencies and provide greater public confidence in the monitoring results.

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*Indicates individuals who assisted in the collection of MCR data on one or more randomly selected THPs.

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List of Abbreviations

BMPs	Best Management Practices
BOF	California State Board of Forestry and Fire Protection
CDF	California Department of Forestry and Fire Protection
CDFG	California Department of Fish and Game
CDPR	California Department of Parks and Recreation
CFA	California Forestry Association
CGS	California Geological Survey
CIMP	Cooperative Instream Monitoring Project
CLFA	California Licensed Foresters Association
CPSS	Certified Professional Soil Scientist
CSES	Critical Sites Erosion Study
EEZ	Equipment Exclusion Zone
EHR	Erosion Hazard Rating
ELZ	Equipment Limitation Zone
ESU	Evolutionarily Significant Unit
PA	Forest Practice Act
FPRs	Forest Practice Rules (Rules)
HMP	Hillslope Monitoring Program
LTMP	Long-Term Monitoring Program
LTO	Licensed Timber Operator
LWD	Large Woody Debris
MAA	Management Agency Agreement
MCR	Modified Completion Report
MSG	Monitoring Study Group
NMFS	National Marine Fisheries Service
NPS	Non-point Source
NTMP	Non-industrial Timber Management Plan
NCRWQCB	North Coast Regional Water Quality Control Board
NTO	NTMP Notice of Timber Operations
PE	Professional Engineer
PG	Professional Geologist
PH	Professional Hydrologist
PHI	Preharvest Inspection
PMP	Pilot Monitoring Program
QA/QC	Quality Assurance/ Quality Control
RCD	Resource Conservation District
RPF	Registered Professional Forester
Rules	Forest Practice Rules (FPRs)
RWQCB	California Regional Water Quality Control Board
SMZ	Streamside Management Zone
SWRCB	State Water Resources Control Board
TMDL	Total Maximum Daily Load
THP	Timber Harvesting Plan
UCCE	University of California Cooperative Extension
USEPA	U.S. Environmental Protection Agency
USFS	U.S. Department of Agriculture, Forest Service
WLPZ	Watercourse and Lake Protection Zone

Modified Completion Report—Final Report

Introduction

The purpose of the Modified Completion Report (MCR) project has been to determine the adequacy of the implementation and effectiveness of California's Forest Practice Rules (FPRs) used to protect water quality and riparian/aquatic habitat. This has been done using information collected by CDF Forest Practice Inspectors during Timber Harvesting Plan (THP) completion report inspections and erosion control maintenance inspections. The MCR data was collected from January 2001 to July 2004. Based on the findings of CDF's earlier Hillslope Monitoring Program (Cafferata and Munn 2002), the MCR project has focused on the following landscape features:

1) Watercourse and Lake Protection Zones, including:
- **WLPZ Percent Total Canopy**
- **WLPZ Groundcover and Erosion Features**

2) Roads, and
3) Watercourse Crossings

Background Information

California's modern Z'berg-Nejedly Forest Practice Act (FPA) was adopted in 1973, with full field implementation occurring in 1975. During the subsequent three decades, a variety of monitoring projects have examined the implementation and effectiveness of California's Forest Practice Rules in protecting water quality. These monitoring efforts are in addition to the California Department of Forestry and Fire Protection (CDF) Forest Practice compliance inspection program that has been in place for over 30 years. Under the FPA, Timber Harvesting Plans (THPs) must be submitted to CDF for review and approval prior to conducting commercial timber harvesting on non-federal timberlands. The THPs are then reviewed for compliance with the FPA and the Forest Practice Rules adopted by the Board of Forestry and Fire Protection (BOF), and for conformity with other state and federal regulations protecting watersheds and wildlife. Multi-disciplinary teams composed of representatives of CDF, the Department of Fish and Game (CDFG), Regional Water Quality Control Boards (RWQCBs), and the California Geological Survey (CGS), conduct Preharvest Inspections (PHIs) of THP areas to determine whether the proposed timber operations comply with requirements of the FPA and the FPRs. During PHIs, additional mitigation measures beyond the standard rules are often recommended based upon site-specific conditions. This report focuses on water quality issues, but the added THP mitigation also relates to habitat protection, public safety, and the protection of other public trust resources. Additional inspections during active timber operations and the post-harvest period when logging is completed ensure compliance with the Act, the FPRs, and specific provisions of the THP.

The State Water Resources Control Board (SWRCB) certified the Forest Practice Rules and review process as Best Management Practices (BMPs) under Section 208 of the Federal Clean Water Act in 1984, with a condition that a monitoring and assessment program be implemented. Initially, a one-year qualitative assessment of forest practices was undertaken in 1986 by a team of four resource professionals (Johnson 1993). The team audited 100 THPs distributed across the state and produced the final "208 Report" (California SWRCB 1987). This report indicated that the Rules were generally were effective when properly implemented on terrain that was not overly sensitive and that poor FPR implementation was the most common cause of observed water quality impacts. The team recommended several changes to the FPRs based on their observations.

The Critical Sites Erosion Study (CSES) was an additional water quality monitoring project in the 1980's related to timber operations conducted within watersheds throughout northern California. The CSES project determined site characteristics on THPs that can be used to identify area susceptible to large erosion events and identified management factors that have contributed to erosion events. This project collected data during 1985 and 1986 on management and site factors associated with existing large erosion events on a random sample of 314 THPs covering over 60,000 acres (Durgin and others 1989, Lewis and Rice 1989, Rice and Lewis 1991).

In 1988, the BOF, CDF, and the SWRCB entered into a Management Agency Agreement (MAA) that required improvements in the FPRs for protection of water quality based on needs described in the "208 Report." At this point, the SWRCB approved final certification of the FPRs as Best Management Practices. The U.S. EPA, however, withheld certification until the conditions of the MAA were satisfied, one of which was to develop a long-term monitoring program (LTMP).

In response to the MAA conditions, the BOF formed an interagency task force in 1989, later known as the Monitoring Study Group (MSG). The primary purpose of the MSG was to develop a long-term monitoring program that could test the implementation and effectiveness of the FPRs in protecting water quality. From 1989 to 1999, the MSG was an "ad hoc" committee of the BOF that met periodically to: 1) develop the long-term monitoring program, and 2) provide guidance to CDF in implementing monitoring programs. With public input, the MSG developed a LTMP with both implementation and effectiveness monitoring components, and conducted a pilot project to develop appropriate techniques for both hillslope and instream monitoring that was conducted from 1993 to 1995 (Rae 1995, Tuttle 1995, Spittler 1995, Lee 1997).

The primary goal of the MSG's LTMP has been to provide timely information on the implementation and effectiveness of forest practices related to water quality for use by forest managers, agencies, and the public. Both CDF and the BOF placed initial emphasis on hillslope monitoring because it can provide a more immediate, cost effective and direct feedback on impacts from current timber operations when compared to instream monitoring (particularly channel monitoring which involves coarse sediment parameters) (Reid and Furniss 1999). As stated in Robben and Dent (2002), it is usually easier to identify a sediment source and quantify the volume of sediment it produced, compared to measuring sediment in the watercourse and tracing it to the source.

Two state-sponsored hillslope monitoring programs have been conducted from 1996 through 2004: first the Hillslope Monitoring Program (HMP) and then the Modified Completion Report (MCR) Monitoring Program. The HMP ran from 1996 to 2002, with data collection by highly qualified independent contractors. Interim and final reports were prepared by CDF (BOF 1999, Cafferata and Munn 2002). The first phase of the Modified Completion Report (MCR) monitoring program, which is the subject of this report, was implemented from 2001 to 2004 as a more cost-effective approach than the HMP, utilizing CDF Forest Practice Inspectors to collect onsite monitoring data as part of required Work Completion Reports.

Complementing these hillslope (onsite) monitoring efforts are several cooperative instream monitoring projects located throughout California. These include:

- Caspar Creek (CDF and USFS-Pacific Southwest Research Station)
- Garcia River (CDF, NCRWQCB, MCRCD, MRC, Maillard Ranch, The Conservation Fund)
- Wages Creek (CDF, Hawthorne Timber Company/Campbell Timberland Management)
- Judd Creek (CDF, Sierra Pacific Industries)
- Little Creek (CDF, Cal Poly San Luis Obispo, Sierra Pacific Industries)

The Caspar Creek project is a paired watershed study that has measured hydrologic changes, erosion impacts, sediment production, cumulative effects, and biological impacts from logging and road construction in second-growth redwood/Douglas-fir forests since 1962.[1] The Judd Creek and Wages Creek studies were developed to test the effectiveness of the FPRs and the THP review process in protecting water quality at the THP scale in Tehama and Mendocino Counties, respectively. The Garcia River project is designed to determine if sediment and turbidity conditions are improving for anadromous salmonids at five tributary stations (Barber and Birkas 2005). The Little Creek project is evaluating the effects of selective timber harvesting and will determine if current highly regulated practices in the Santa Cruz Mountains are adequately protecting the beneficial uses of water from adverse sediment-related impacts.

In addition to hillslope and instream monitoring efforts, numerous monitoring projects have been supported, or are currently being supported, by CDF that provide critical information related to monitoring techniques and/or answer key questions regarding forest practice implementation and effectiveness.[2] Examples of these projects include:

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[1] Caspar Creek published papers are found at:
http://www.fs.fed.us/psw/topics/water/caspar/caspubs.shtml

[2] MSG reports and supported reports are found at:
http://www.bof.fire.ca.gov/board/msg_supportedreports.asp

- Testing Indices of Cold Water Fish Habitat (Knoop 1993)
- V-Star Tests in Varying Geology (Lisle 1993, Lisle and Hilton 1999)
- Erodible Watershed Index (McKittrick 1994)
- Evaluation of Road Stream Crossings (Flanagan and others 1998)
- Sediment Storage and Transport in the South Fork Noyo River Watershed, Jackson Demonstration State Forest (Koehler and others 2001)
- Central Sierra Nevada Sediment Study (MacDonald and others 2004, Coe 2006)
- Sediment Composition as an Indicator of Stream Health (Madej 2005, Madej and others, in press)

Summary of Other Related Studies

Several monitoring-related studies have been completed in California over the past decade that are related to the monitoring work described in this report. A brief description of these related projects is given below, and a comparison of the results of these study results to those of MCR results is presented in the appropriate section of this report -- WLPZ and Groundcover Monitoring, Road Monitoring or Watercourse Crossing Monitoring.

<u>BOF/CDF Hillslope Monitoring Program (HMP)</u>
The HMP conducted a statewide evaluation of the implementation and effectiveness of California's Forest Practice Rules (FPRs) from 1996 through 2002 using an annual, random sample of 50 completed THPs and NTMPs that had over-wintered from one to four years. Detailed information was collected from sampled plans in the summer months. This included data on: (1) randomly located road, skid trail, and watercourse and lake protection zone (WLPZ) segments, as well as randomly located landings and watercourse crossings; and (2) large erosion events (e.g., mass wasting features) where they were encountered. Winter documentation of fine sediment delivery to streams was not undertaken by this program. The monitoring work was done by highly qualified independent contractors who acted as third party auditors (Ice and others 2004). A report of interim findings was prepared (California State BOF 1999), and a final report based on 300 plans was completed in 2002 (Cafferata and Munn 2002). Data revealed that implementation rates of the FPRs related to water quality were high, averaging 94%, and that individual practices required by the rules were effective in preventing hillslope erosion when properly implemented. WLPZs were found to retain high levels of post-harvest canopy and surface cover as required by the FPRs, and these high levels were found to be effective in preventing harvesting related erosion. In those instances where erosion sites were identified, they were nearly always associated with inadequate implementation of the appropriate rule required by the FPRs. Roads and associated watercourse crossings were found to have the highest frequency of problems. These conclusions were generally similar to those reached in an earlier audit of 100 THPs (California SWRCB 1987).

USFS Best Management Practices Evaluation Program (BMPEP)

Water quality monitoring data collected from 1992 through 2002 on National Forest lands located in California was reported in 2004, fulfilling monitoring commitments to the SWRCB (USFS 2004). Twenty-nine different on-site monitoring protocols were used to evaluate BMP implementation and effectiveness. Altogether, there were approximately 3,900 random evaluations made for the 18 National Forests, with the most occurring on the Klamath and the least on the Los Padres. Most of the observations were for engineering and timber-related BMPs. Both implementation and effectiveness for a BMP were rated at the same time following 1-2 overwintering periods. If impacts to water quality were found, the observer estimated the magnitude, duration, and extent of impacts. A statistically significant relationship between BMP implementation and effectiveness was found for 16 of the 29 BMP protocols. In general, the results show that while some improvements are necessary, the program performed reasonably well in protecting water quality on National Forest lands in California. BMP implementation and effectiveness were relatively high for most activities and elevated effects on water quality were relatively infrequent, particularly in recent years. For all activities combined, BMPs were implemented 85% of the time, and were effective at 92% of the sites at which they were implemented. Effects classified as elevated were typically caused by lack of or inadequate BMP implementation and most elevated effects were related to engineering practices. Roads, and in particular stream crossings, were found to be the most problematic.

Colorado State University (CSU) Sierra Nevada Sediment Study

Dr. Lee MacDonald and graduate student Drew Coe measured sediment production rates on the Eldorado National Forest and on Sierra Pacific Industries timberlands in the Central Sierra Nevada (Coe and MacDonald 2001, 2002; MacDonald and others 2004; Coe 2006). Approximately 150 sediment fences were installed in the summers of 1999 and 2000. Field investigations focused on (1) quantifying sediment production and sediment delivery from timber harvest, roads, wild and prescribed fires, off-road vehicles, and undisturbed areas; (2) quantifying the year-to-year variability in sediment production; and (3) determining the effect of key site variables (MacDonald and others 2004). MacDonald and others (2004) found that roads, high-severity wildfires, OHV trails, and certain skid trails on granitic soils were the dominant sediment sources. The mean road sediment production rate was 0.9 kg/m^2, 0.1 kg/m^2 from skid trails, 0.4 kg/m^2 from ORV trails, 1.1 kg/m^2 from high severity burn sites, and 0.001 kg/m^2 from minimally disturbed sites. Native surface roads produced 10-50 times more sediment than rocked roads and most sediment delivery related to roads occurred at or near stream crossings. Additionally, they found that sediment production rates were highly variable between sites within a year as well as between years. Multivariate analyses indicated that the dominant controls on road sediment production included road contributing area (A), road gradient (S), annual erosivity (E_A), and road surfacing (rock vs. native surface; T). An empirical model containing these variables explained 54% of the variability in annual road sediment production.

USFS-PSW Research Station and CDF—Caspar Creek Watershed Study

Suspended sediment and bedload have been measured at the North and South Forks of Caspar Creek for more than 40 years (Ziemer 1998, Lewis and others 2001, Keppeler and others 2003). Caspar Creek is a small coastal watershed situated between the Noyo and Big River drainages in western Mendocino County. The Caspar Creek data set is unique in California, since it is the only forested experimental watershed currently in operation with a continuous, long-term flow and sediment record (Ziemer and Ryan 2000). Results show that improved forestry practices after 1974 have significantly reduced sediment yields. Selection logging conducted prior to the implementation of the modern FPRs in the South Fork of Caspar Creek produced from 2.4 to 3.7 times more suspended sediment than clearcutting in the North Fork under the modern FPRs (Lewis 1998). In the North Fork of Caspar Creek following clearcut harvesting of almost half the watershed in three years under the modern FPRs, suspended sediment monitoring showed that annual sediment loads increased 123-269% in the tributaries. At main-stem stations, however, increased loads were detected only in small storms and there was little effect on annual sediment loads. Most of the suspended sediment measured at the North Fork weir resulted from one large landslide that occurred in January 1995. Road rehabilitation work was conducted during the summer of 1998 on three miles of road that had had been constructed along the South Fork in 1967. A total of 33 watercourse crossings were abandoned, removing a total of approximately 28,500 cubic yards of fill material. Surveys of the abandoned crossings have shown that downcutting following large winter storm events resulted in 854 cubic yards of sediment production, or three percent of the total amount of sediment removed, with an average loss of approximately 26 cubic yards per crossing. Over 70% of this material came from three crossings, or 9% of the abandoned crossings surveyed (Cafferata and Munn 2002).

<u>Klein—Sanctuary Forest Stream Crossing Excavations in the Upper Mattole River Basin, 2002-2003</u>

The Sanctuary Forest, Inc. is implementing an erosion control and prevention program to reduce long-term sediment yield in the upper Mattole River watershed, with the focus on decommissioning unneeded forest roads that pose sedimentation risks. Klein (2003) conducted a monitoring project to determine volumes of erosion following road removal at excavated crossings and impacts to water quality. Erosional void dimensions were measured at 18 excavated crossings. Both channel scour and bank slumps were documented for each crossing. Survey work was not conducted prior to the onset of winter rains, so channel scour was estimated by making field measurements of scarp heights and top widths at geometric transition points within the excavation. Most of the erosion was found in the excavated channel areas, but erosion was also documented above crossings where culverts had been located. The total sediment delivery for the first winter was 279 yds^3, with an average of 15.5 yds^3 per crossing. Sediment yield for individual crossings ranged from over 50 yds^3 to less than 2 yds^3. Four crossings (approximately 22% of the excavated crossings) produced roughly half the total sediment volume. In general, channel scour strongly dominated sediment yield. Bank slumps were relatively minor except at one removed crossing.

Modified Completion Report (MCR) Study Design

Overview

Under the FPA, Public Resources Code (PRC) Section 4586 requires that within six months of the receipt of the Work Completion Report specified in PRC Section 4585, the director shall determine, by inspection, whether the work described in the report has been properly completed in conformity with the rules and regulations. If so, a report of satisfactory completion is issued. If not, the director shall take such corrective action as he or she determines appropriate. MCR is a slight modification to this process. MCR adds a monitoring step, which is designed to collect data on the implementation and effectiveness of the FPRs designed to protect water quality.

The initial MCR monitoring design was a simple check list used in the late 1990's by CDF inspectors during the Work Completion Report inspection that is required on all THPs. This approach had several deficiencies. First, even though the check list forms were to be turned-in for all THPs undergoing Work Completion Report inspections, in practice forms were turned-in for only a small, non-random fraction of the completed THPs. Since the sample was not random, it was not possible to tell whether this was a representative sample of all THPs. Second, the check list only included categories for deficient implementation or effectiveness of listed FPRs. This implied that absence of a check mark always meant no deficiency, which was not always true. And third, because the check list instructions did not include criteria for site selection, it was not possible to determine what bias might have been introduced by the choice of sampling locations.

To solve these problems the MCR protocols were revised to include:

1) Random selection of THPs for monitoring to ensure a representative sample,
2) Forms that required a mark or an entry for each question to indicate whether it had been answered or deemed not applicable, and
3) Criteria for random selection of monitoring sites within each THP.

Random Selection of THPs

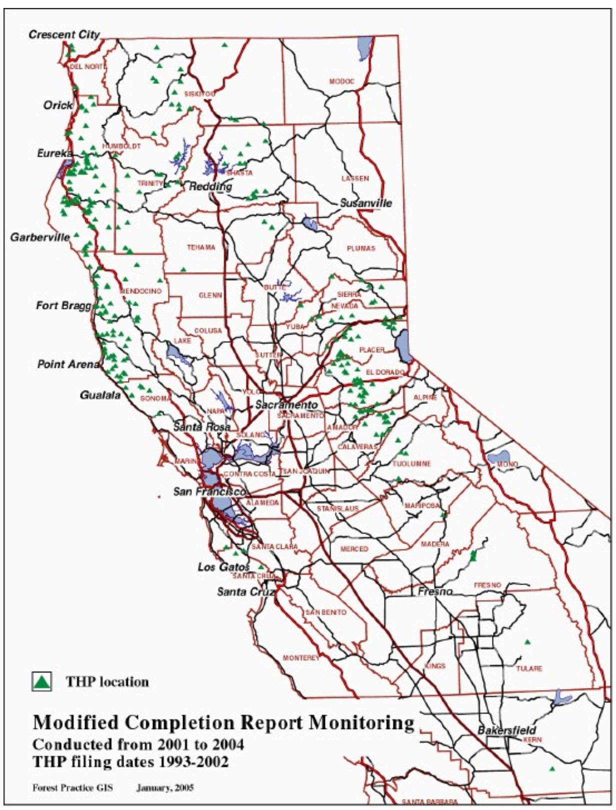
The MCR monitoring was performed on a random sample of completed THPs. The initial target sample size was 25% of all THPs undergoing Work Completion Report inspections. This percentage was subject to change based on staffing levels and workload, and the sample size was revised downward from 25% to 12.5% on February 25, 2002. A 12.5% sample represented about 125 THPs in 2002.

To obtain a random sample, pick-lists of randomly selected THP numbers were generated and distributed to Forest Practice Inspectors. One list was generated for THPs dated 1990 through 1999; and separate, annual lists were generated for THPs approved in 2000, 2001, 2002, and 2003. There were no THPs with a filing date of 2004 or later in this sample, because no plans filed in 2004 were completed by July 1, 2004. To avoid confusion, the same list of numbers was used for all three CDF

Regions. This does not affect the randomness of the sample because each region assigns its own, consecutive THP numbers, starting with 001, annually. If the THP number for a completed plan matched one of the numbers on the random list for a given year, then that THP was selected for monitoring.

A program used to produce lists of random THP numbers was written by State Forests Research Coordinator Tim Robards of CDF in collaboration with CDF watershed scientist Clay Brandow. In this approach, each number from 1 to 1000 is individually compared to a randomly generated number that gives a one in "X" chance of selection. For example, to get a 12.5% sample, "X" equals 8, and each THP number has an independently determined one-in-eight chance of being selected. This provides a random, 12.5% sample of completed THPs regardless of the number of THPs approved in any given year.

The MCR project has not yet included Non-Industrial Timber Management Plan (NTMP) Notices of Timber Operations (NTO), while the Hillslope Monitoring Program did include some NTMPs. Neither the MCR random sample nor the HMP random sample included harvesting operations conducted under Exemption or Emergency Notices.



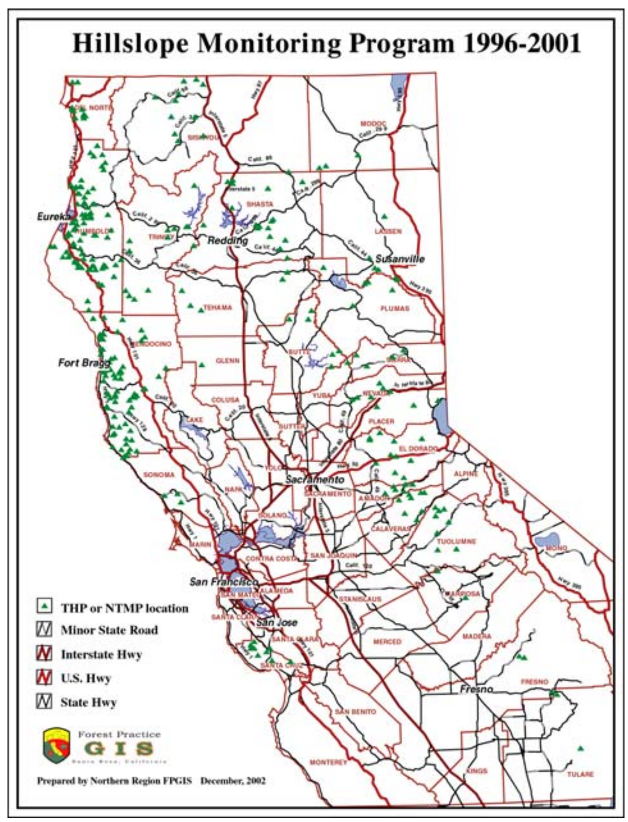


Figure 3. General locations of THPs randomly selected for MCR monitoring from 2001 to 2004 on the left, compared to the general locations of THPs randomly selected for HMP monitoring from 1996-2001 on the right.

Plotting the locations of THPs selected for MCR monitoring from 2001 to 2004 produces a statewide pattern of sampling sites that is remarkably similar to a plot of THP and NTMP sample sites selected for the HMP from 1996 through 2001 (See Figures 3 & 4).

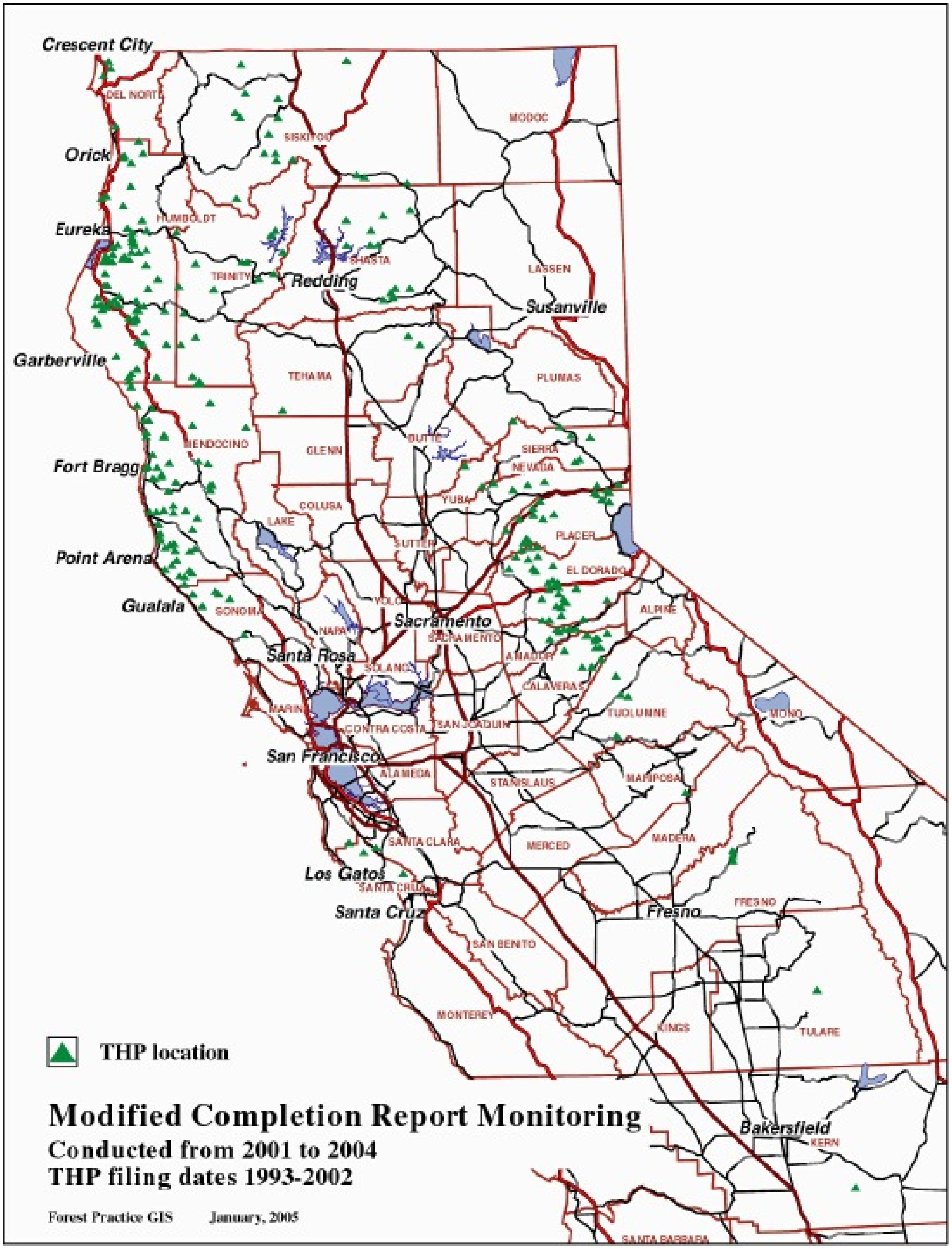


Figure 4. General locations of THPs randomly selected for MCR Monitoring from 2001 to 2004. This is simply an enlargement of the map of MCR THP distribution shown on the left in Figure 3.

The similarity of geographic patterns is the expected outcome, since MCR and HMP monitoring used independent, random samples of roughly equal size of THPs completed California. This similarity of geographic patterns is further evidence that both random samples are representative of the whole population.

Data Collection

Most of the MCR monitoring data was collected by CDF Forest Practice Inspectors, with some assistance from other CDF staff. On a small number of the THPs, monitoring assistance was provided by Regional Water Quality Control Board staff, California Department of Fish and Game staff, or landowner representatives (generally the Registered Professional Foresters (RPFs) who prepared and/or administered the THP).

Data was collected on paper forms. To avoid ambiguities from blanks in the data, responses such as "N/A" (for "not applicable") were required for all entries that might otherwise be left empty Despite training on filling out the data collection forms, blanks were still a problem. This has required some interpretation of the meaning of items left blank for subsequent data analyses. For future monitoring efforts, a solution to this problem is to use electronic data loggers that will not allow field observers to complete the form without all of the required entries.

The methods and procedures used in data collection for this report are documented in Modified Completion Report Monitoring Procedures and Methods (rev.4/9/03), which is listed in this report as Appendix A. An electronic copy of the *Modified Completion Report Monitoring Procedures and Methods (rev.4/9/03)* is available on line at:

http://www.bof.fire.ca.gov/board/msg_archives.asp

Implementation and Effectiveness Evaluations

All four sites (WLPZ segment, road segment, and two watercourse crossings) were evaluated for implementation at the time of the final Work Completion Report inspection(s). The sample road segment and watercourse crossings drainage structures were to be evaluated a second time for effectiveness during the post-completion erosion control maintenance inspection(s), after at least one over-wintering period. In some cases, the implementation evaluation was done after one or more over-wintering period(s) and the effectiveness evaluation was done on the same visit. In other cases, the effectiveness inspections were not done for lack of a second visit. Consequently, the subset of THPs with roads and crossings rated for effectiveness is smaller than the sub-set of the THPs with roads and crossings rated for implementation.

Effectiveness information recorded included erosion features present (if any), source and cause of erosion features, impact to water quality, and adequacy of road and crossing design and construction.

Between November 2000 and June 2003, field training sessions on MCR data collection were conducted on THPs located in several CDF units located around the state. Seventy-five individuals took part in the training. Most of these were CDF inspectors, but some RWQCB staff were also present.

Quality Assurance/Quality Control (QA/QC)

Quality assurance consists of actions to ensure adherence to data collection and analysis procedures, while quality control is associated with actions to maintain data collection and analysis consistent with study goals through checks of accuracy and precision. The quality assurance program was composed of three components: 1) qualifications and practical experience of CDF Forest Practice Inspectors, 2) a detailed field training program, and 3) protocols provided in the *Modified Completion Methods and Procedures* document (See Appendix A).

The quality control program consisted of self-evaluation of the data collection forms for completeness in the field and a second evaluation of the forms by watershed staff at CDF Headquarters. Questions were resolved through direct communication between the Forest Practice Inspectors and watershed staff.

To ensure completeness of THP samples, lists of recently completed THPs subject to MCR Monitoring were generated quarterly using the Forest Practice System (FPS) data base and the MCR random pick-lists. These lists of THP numbers were checked against lists of MCR monitoring reports received in Sacramento, and responsible Forest Practice Inspectors were contacted about missing reports.

Regional Distribution of Monitored THPs

CDF has four Administrative Regions, three of which are included in this monitoring and will be referred in this report by short, descriptive names:

1) North Coast Region 1 is referred to as **"Coast"**,
2) Cascade Region 2 is referred to as **"Inland North"**
3) Central Sierra Region 4 is referred to as **"Inland South"**

Southern Region 3, which includes southern California and the eastern slope of the Sierra Nevada south of the Carson River, is arid, except at the highest elevations, which are for the most part federal lands. The region contains very little private or state forest lands and generates very few THPs. Consequently, Southern Region 3 was not included in this study. Also, in some portions of the of the report, notably the section on roads, the combined areas of Inland North and Inland South are referred in the aggregate as simply **"Inland."**

All of the 281 plans selected for MCR monitoring were THPs, while the 300 plans selected and analyzed for the HMP included 295 THPs and 5 NTMPs.

The distribution of plans by CDF Administrative Region was somewhat different for the MCR project than in the HMP. For MCR Monitoring, percentages of Coast (R-1), Inland North (R-2) and Inland South (R-4) plans were 52%, 27% and 21%, respectively (see Figure 5). For the HMP, the percentages of Coast (R-1), Inland North (R-2) and Inland South (R-4) plans were 62%, 26% and 13%, respectively (see Figure 6). Simplifying the comparison by combining the inland categories gives a Coast vs. Inland ratio of about 50/50 for the MCR sample of THPs and about 60/40 for Hillslope Monitoring Program sample.

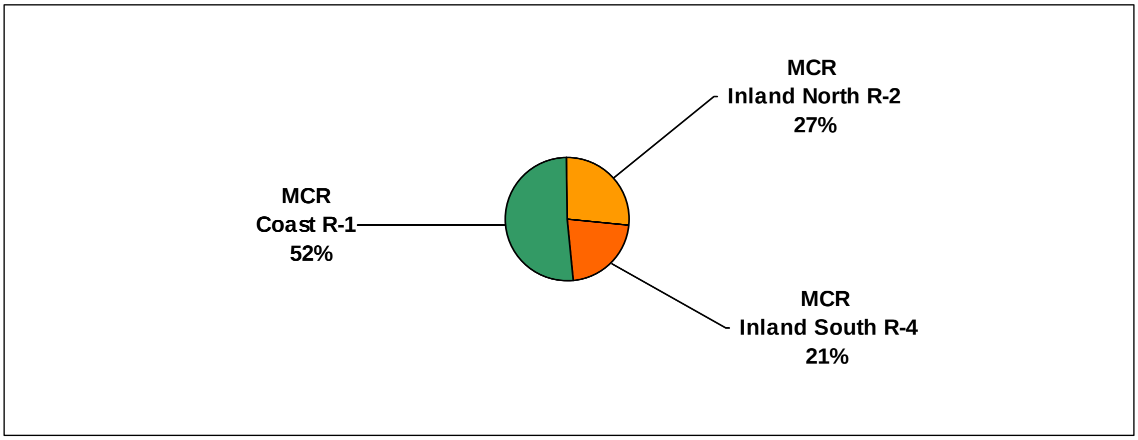


Figure 5. Distribution of MCR Monitoring Randomly Sampled THPs by Region.

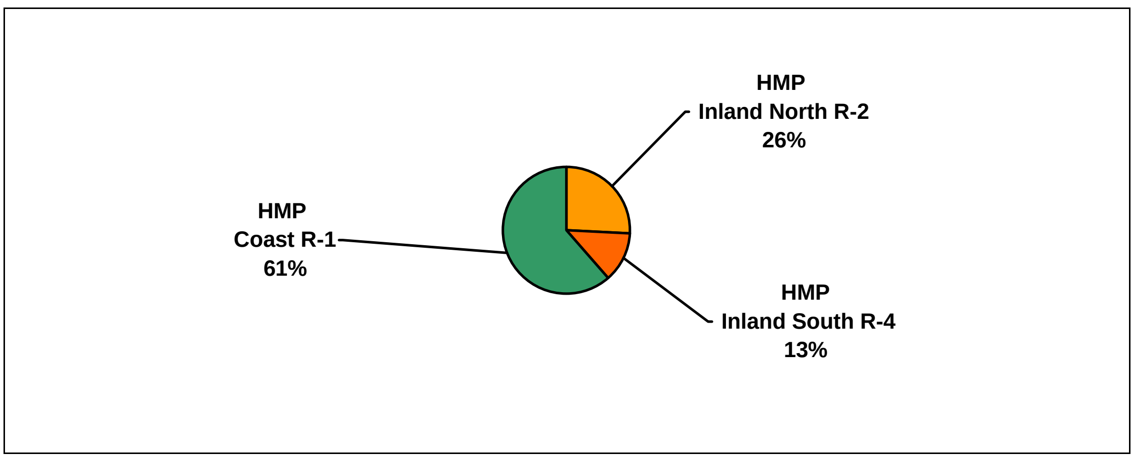


Figure 6. Distribution of HMP Randomly Sampled THPs by Region.

General locations of THPs randomly selected for MCR monitoring are shown plotted on the map of CDF Administration Regions below in Figure 7. Note the clustering; this clustering is representative of the clustering in the population of all THPs completed from 2001 through 2004. A similar pattern of clustering was observed in the HMP random sample (1999-2001).

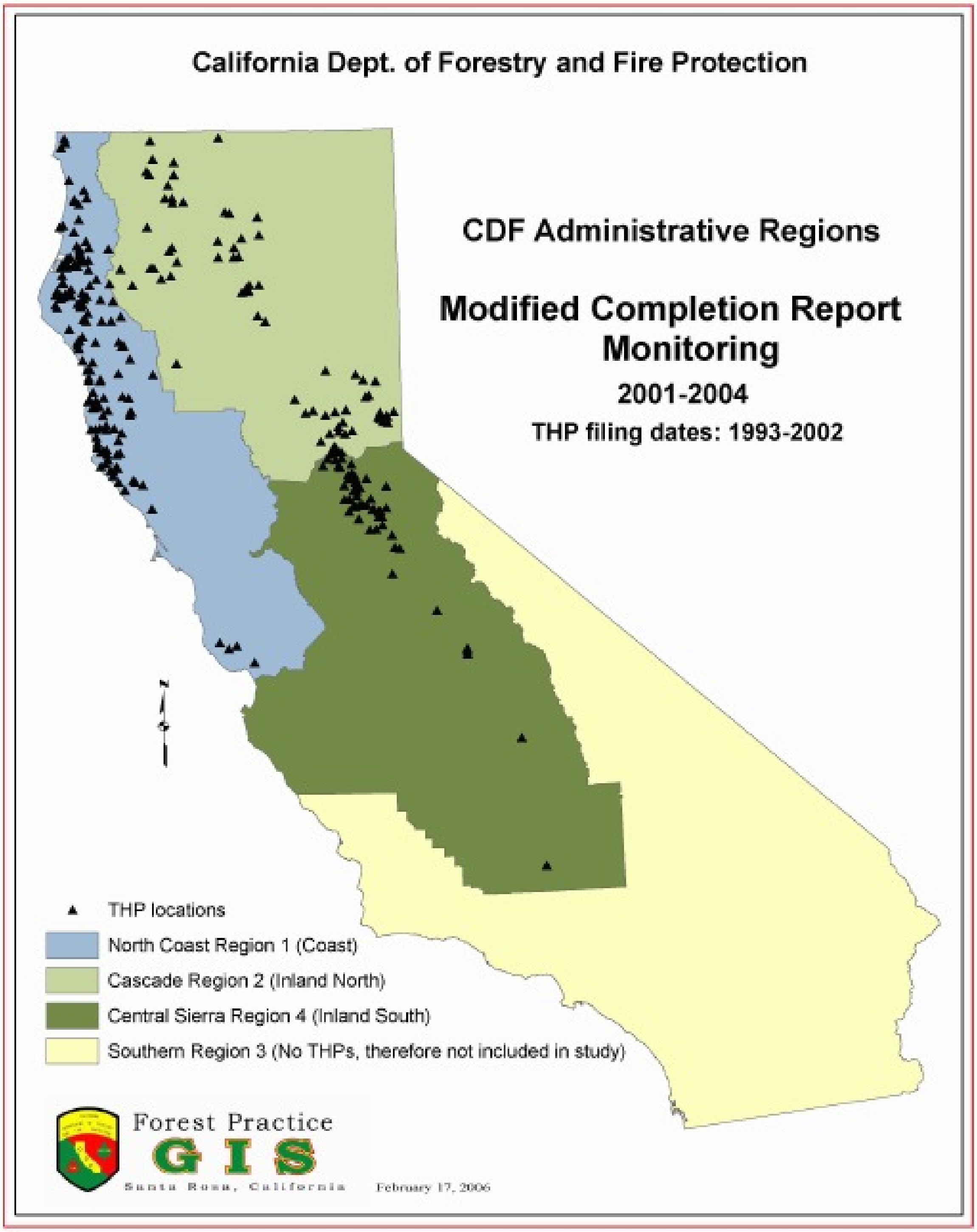


Figure 7. General locations of THPs randomly selected for MCR Monitoring from 2001 to 2004 by CDF Administrative Region.

Random Site Selection within Randomly Selected THPs

Up to four monitoring sites were located on each THP. These included:

1) A 200 foot WLPZ segment along a Class I or Class II watercourse,
2) A 1000 foot road segment, and
3) Two crossings of Class I, Class II or Class III watercourses.

For THPs that lacked one or more of these sites, forms were turned-in with the notation: "Not applicable to this THP."

Methods of random site selection for WLPZ segments, road segments, and watercourse crossings within a selected THP are described elsewhere in this report under the methods section for each of these features.

The use of randomly selected sampling sites within the THP allowed inspectors to focus in detail on whether the FPRs applicable to that site were: 1) properly implemented, and 2) effective in protecting water quality by preventing erosion, sediment transport, and discharge into channels.

MCR Monitoring:
WLPZ Canopy and Groundcover

I. Methods

Monitoring Timelines and WPLZ Selection

A 200-foot long WLPZ segment was randomly selected for MCR monitoring from each of the randomly selected THPs with one or more WLPZs. This was not possible in some cases, because Class I or Class II watercourses were not present on all of the randomly selected THPs. Within the WLPZ, sample segment zone width and percent total canopy were measured (Figure 8), and groundcover conditions were observed. Also, where they existed within the WLPZ segment, three additional items were observed and recorded: 1) erosion features, 2) untreated patches of bare mineral soil, and 3) timber harvesting that occurred on this entry.

Figure 8. Pete Cafferata, CDF, making canopy cover measurements using a sighting tube.

Selecting the 200-foot WLPZ segment began with the inspector delineating all of the Class I and Class II WLPZs on the THP map(s). Then a scale was used to mark 200 foot segments along all of the delineated WLPZs. Each of these segments was given a number. Then a random number between 1 and the maximum number of segments was identified using a random number table or a pocket calculator random number generator, and the segment number corresponding to the identified random number was selected for sampling. Where both sides of the creek were harvested, a coin flip was used to determine which side of the stream to monitor. Random selection of WLPZ reaches was used to capture a representative sample of WLPZ conditions. This is different than the objective of WPLZ enforcement inspections. For enforcement purposes, segments are selected for canopy measurement based on apparent violations. Therefore, enforcement data represents worst-case post-harvest WLPZ conditions, while MCR measurements represent average WLPZ conditions for the study period.

The MCR procedures used for WLPZ canopy measurement were modified from Preharvest Inspection (PHI) and enforcement action procedures developed by Robards (1999). In both procedures, canopy is determined using a sighting tube, but the number of observations for the MCR procedure is 50, as compared to 100 for the enforcement procedure. Average WLPZ width for the MCR was determined by pacing within the segment sampled for canopy cover, and groundcover was estimated by ocular observation. Additionally, fresh erosion features in the MCR sample segment (i.e. gullies, rills, or areas of sediment deposition) were noted. The advantages to using similar WLPZ canopy/surface cover sampling methods for PHIs, enforcement, and MCR sampling included continuity of techniques, reduced training needs, and data comparability.

Sampling Procedures

The following sampling procedures apply to both Class I and Class II WLPZs. The target sample size for canopy measurements was 50 sighting tube points, regardless of the size of the sampled area. The distance (D) between points was calculated using the following formula, where width and length refer to the width and length of the sampled WLPZ segment:

$$D = \sqrt{\frac{width \times length}{50}}$$

Since the standard MCR sample length is 200 feet, this equation can be simplified to:

$$D = 2\sqrt{width}$$

When applied to standard widths of 50, 75, 100 and 150 feet, D is 14, 17, 20 and 28 feet, respectively. For convenience, the WLPZ width stated in the THP was used to determine D for field measurements, even if the actual WLPZ width flagged on the ground was found to be different during subsequent field work.

WLPZ transects were started at the watercourse transition line at one end of the WLPZ segment. From there, the first sample point was located on a line perpendicular to the watercourse at a distance that was calculated using a random number between zero and one times the measurement interval distance D. From the first sample point, the distance D was paced perpendicular to the stream to reach the next sample point, and so on until the next point would exit the flagged WLPZ. The WLPZ transect was then turned 90º for distance D to start of a new line perpendicular to the stream. This procedure was repeated until 50 sample points were measured, whether this completed the final line or not. The resulting measurement pattern is similar to what is shown in Figure 9.

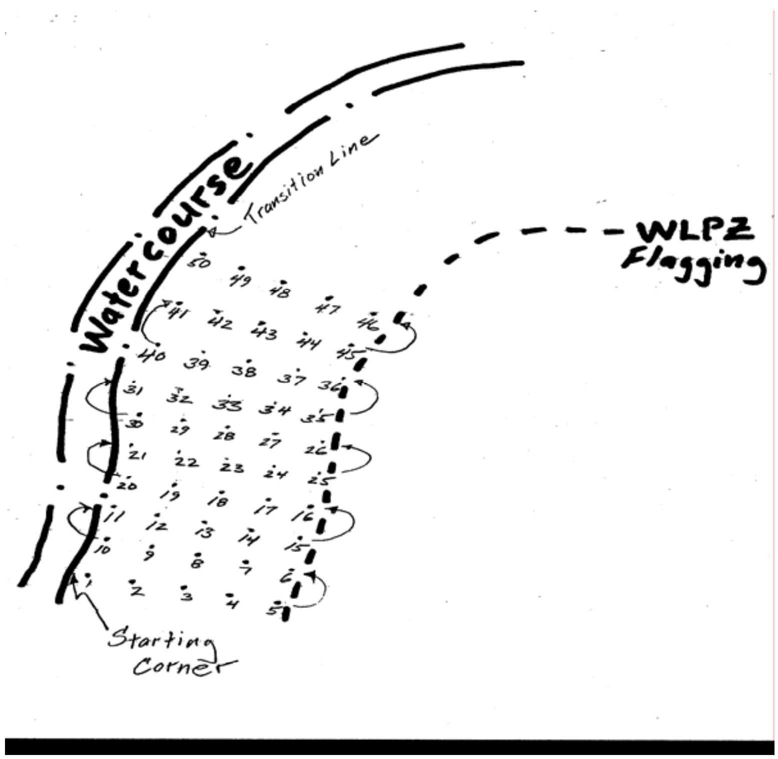


Figure 9. Typical pattern of canopy sighting and groundcover observation points within a typical randomly sampled WLPZ segment.

At each sample point, the inspector recorded total canopy as either a hit or miss, using a sighting tube (shown in Figure 10) as follows: (1) the sighting tube was leveled in front of one eye using the horizontal and vertical bubbles, (2) the dot in the center of the tube was lined up with circle in the center of the tube, and (3) the dot was evaluated as to whether it intercepted an object above the observer, such as needles, a leaf or a tree branch. Hits were recorded as “+” in the hit column and misses were recorded as “-” in the miss column on the WLPZ data form. When deciduous trees were encountered without leaves in the winter, it was assumed that leaf cover would be present in the summer months.

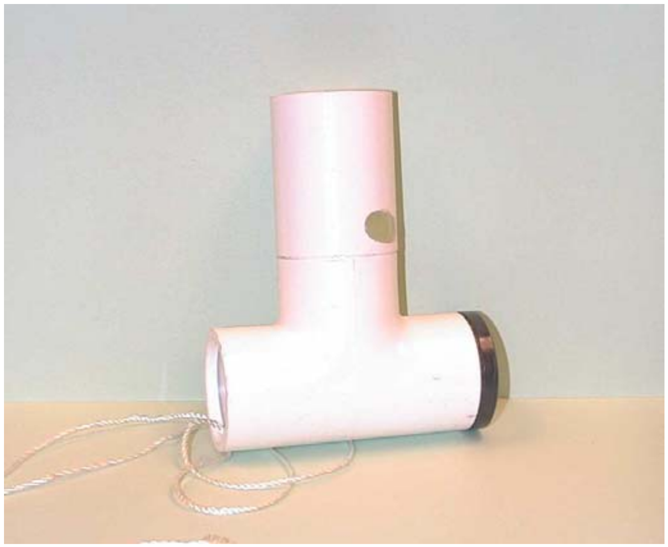


Figure 10. Example of a sighting tube used for making WLPZ canopy measurements.

The proportion of the ground surface covered with duff, litter, gravel larger than ¾ inch, and other protective material was also estimated and recorded at each sample point. In addition, the presence of erosion features or sediment deposition encountered during the transect was documented in association with the nearest sample point, along with information about feature type (i.e., gully, rilling, or areas of sediment deposition) and the feature's approximate size (width, depth, and length) in feet. Each erosion feature was recorded only one time, even if it was observed at more than one location, and a check box for "No erosion features observed in the sample WLPZ segment" was included on the data form to ensure that absence of recorded erosion features was not an oversight.

Following completion of the WLPZ transect, an overall assessment of conditions in the WLPZ segment was made, including whether or not there had been harvesting (yes or no), and if there had been harvesting how much canopy was removed, using three categories: <10%, 10-30%, and 30-50%.

An example of a completed form is included in the Modified Completion Report Methods and Procedures (see Appendix A).

II. Results

WLPZ segments were located in 187 of the 281 THPs included in the MCR sample. The regional distribution was 110 WLPZ segments on the Coast (CDF Region 1), 49 in the Inland North area (Region 2) and 28 WLPZ segments in the Inland South area (CDF Region 4.)

WLPZ Percent Total Canopy

Average percent total canopy cover in WLPZs was higher in the Coast than in the Inland areas. Looking at Class I and II watercourses together, average percentages for the Coast are in the mid to low eighties, and are around seventy for both Inland North and Inland South. In Table 1, below, the column for overall average includes all WLPZ results within each Region. The next two columns to the right split the overall sample into WLPZ segments with no harvest in this entry (the current THP) and WLPZ segments with harvest as part of this entry.

Class I & II WLPZs	Overall	No Harvest	Harvest
Coast (Region 1)	84% n = 110	86% n = 55	82% n = 55
Inland North (Region 2)	68% n = 49	72% n = 12	67% n = 37
Inland South (Region 4)	73% n = 28	69% n = 15	77% n = 13

Table 1. Average percent total canopy in WLPZs by Region for Class I and Class II watercourses combined. The number of segments included in each average equals "n."

Results for Class I watercourses alone are similar (Table 2). Note that the number of WLPZ segments (n) represented in some of these averages is very small. Consequently, the 10 percent difference between average percent canopy for harvested and unharvested WLPZs in the Inland South area is probably not meaningful.

Class I WLPZs	Overall	No Harvest	Harvest
Coast (Region 1)	84% n = 29	83% n = 14	84% n = 15
Inland North (Region 2)	69% n = 18	74% n = 3	68% n = 15
Inland South (Region 3)	71% n = 5	65% n = 2	75% n = 3

Table 2. Average percent total canopy in WLPZs by Region for Class I watercourses. The number of segments included in each average equals "n."

The percent total canopy results for WLPZs along Class II watercourses are also similar to both the combined and Class I results (Table 3).

Class II WLPZs	Overall	No Harvest	Harvest
Coast (Region 1)	84% n = 81	87% n = 41	81% n = 40
Inland North (Region 2)	67% n = 31	70% n = 9	65% n = 22
Inland South (Region 3)	73% n = 23	70% n = 13	78% n = 10

Table 3. Average percent total canopy in WLPZs by Region for Class II watercourses. The number of segments included in each average equals "n."

The MCR percent total canopy results for WLPZs are strikingly similar to the findings of the Hillslope Monitoring Program, which used similar canopy measurement techniques, but was based on a completely different random sample of THPs. The importance of this will be covered in more depth in the WLPZ discussion section.

WLPZ Erosion Features

Of the 187 WLPZs sampled, 19 (~10 percent) had one or more erosion features. Of the 19 WLPZs with erosion features, only 2 (or about one percent) had erosion features related to current timber operations. Of the two WLPZ segments with erosion features related to current timber operations, one involved sediment deposition from erosion on a landing upslope, and the other was a gully that resulted from soil with less than 70% groundcover. In the first case, the WLPZ functioned as it should to intercept sediment originating from upslope erosion. In the second case, removal of groundcover as part of the timber operation led to erosion and sediment production, based on field observation.

The causes of the 17 WLPZ erosion features not related to current timber operations were described as follows:

- 6 inner gorge erosion sites,
- 2 streambank failures,
- 1 sediment deposition from a scarp,
- 4 originated from old skid trails/roads,
- 1 gully from a county road,
- 1 eroding cow trail, and
- 1 breached irrigation ditch.

Inner gorge erosion, streambank failures and scarps are natural features of the California landscape, and are common on California's north coast. County roads, cow trails, and irrigation ditches are land management features related to uses other than timber harvesting. Skid trails and skid roads from past timber operations reflect past practices that are not generally permitted under current FPRs.

Other WLPZ Results

Other WLPZ information collected as part of the MCR inspections included WLPZ length, width, canopy removal, understory canopy, and groundcover. Blanks have been interpreted as missing data and were not included in the calculation of average values. In some cases, however, data points with a value of zero may have been left blank.

The average total length of Class I WLPZ in the sampled THPs was 1,309 feet on the Coast (Region 1) and 1,770 feet in the Inland areas (Regions 2&4). The average total length of Class II WLPZ in the sampled THPs was 3,369 feet on the Coast and 3,396 feet Inland.

For all Regions, actual WLPZ widths as paced were equal (within ±5 feet) to the width prescribed in the THP 58% of the time, greater than prescribed 35% of the time, and less than prescribed 7% of time.

The average prescribed WLPZ widths for Class I streams were 129 feet, 92 feet and 75 feet for the Coast, Inland North and Inland South, respectively. WLPZ widths measured on the ground were generally wider than prescribed widths. The average actual widths for Class I streams were 145 feet, 94 feet and 94 feet for the Coast, Inland North and Inland South, respectively. On Class II watercourses, the average prescribed WLPZ widths were 85 feet, 64 feet and 63 feet for the Coast, Inland North and Inland South, respectively. Again, the actual widths were wider than the prescribed widths on average. The average measured widths were 93 feet, 69 feet and 67 feet for the Coast, Inland North and Inland South, respectively.

Canopy removal by current timber operations within sampled WLPZ segments was extremely variable. For Class I watercourses in all Regions, 18 WLPZ segments had no canopy removal, 19 had less than 10% of the canopy removed, 12 had 10% to 30% of the canopy removed, and none had more than 30% canopy removal. For Class II watercourses in all Regions, 64 WLPZ segments had no canopy removal, 44 had less than 10% removed, 25 had 10% to 30% removed, and none had more than 30% canopy removal.

Total canopy has two components: understory canopy and overstory canopy. Based on ocular estimates, the remaining understory canopy in Class I WLPZs was 50% or greater 92% of the time, and the remaining overstory canopy was 50% or greater 96% of the time. Likewise for Class II WLPZs, remaining understory canopy was 50% or greater 91% of the time, and remaining overstory was 50% or greater 92% of the time.

The “Threatened and Impaired Watershed Rule Package Requirements (T&I Standards)” for overstory canopy came into effect on July 1, 2000. They only apply to Class I watercourses in specific watersheds in THPs filed after mid-year 2000. To the question “Does this Class I watercourse meet the T&I standards?” inspectors answered 25 WLPZs did meet the standards, 6 did not, and in 10 the standards were not applicable. There were 11 instances of apparent missing data were the question was not answered.

Regarding WLPZ groundcover, both live and dead, 70% groundcover is a threshold at which surface erosion is normally prevented. Class I WLPZ percent groundcover was equal to or greater than 70% on average 93%, 81%, and 60% of the time for the Coast, Inland North and Inland South, respectively. Similarly, Class II WLPZ percent groundcover was equal to or greater than 70% on average, 93%, 90%, and 71% of the time for the Coast, Inland North and Inland South, respectively. Untreated patches of bare mineral soil equal to or greater than 800 square-feet, or greater than a threshold specified in the THP, were reported in only one Class I WLPZ, which was located on the Coast, and in three Class II WLPZs, one of which was on the Coast and two of which were in the Inland South.

III. Discussion

The MCR results for percent WLPZ total canopy are strikingly similar to the earlier findings of the Hillslope Monitoring Program (Cafferata and Munn 2002), which used similar canopy measurement techniques but was based on a completely different random sample of THPs. Comparisons of these results for Class I watercourses are shown in Table 4 and Figure 11, and Class II watercourse comparisons are shown in Table 5 and Figure 12. Such similarity of results from two independent studies indicates that these averages are a true representation of the current status of WLPZ total canopy cover on recently completed THPs in California.

Table 4. Comparison of MCR (2001-2004) and Hillslope Monitoring Program (1999-2001) results for average percent WLPZ total canopy by Region for Class I watercourses. The number of segments represented in each average equals "n."

Class I WLPZ Comparison	MCR Monitoring (2001-2004) Class I WLPZ percent total canopy	HMP (1999-2001) Class I WLPZ percent total canopy
Coast (Region 1)	84% n = 29	83% n = 27
Inland North (Region 2)	69% n = 18	61% n = 17
Inland South (Region 4)	71% n = 5	67% n = 13
Inland (Regions 2&4 combined)	69% n = 23	64% n = 30

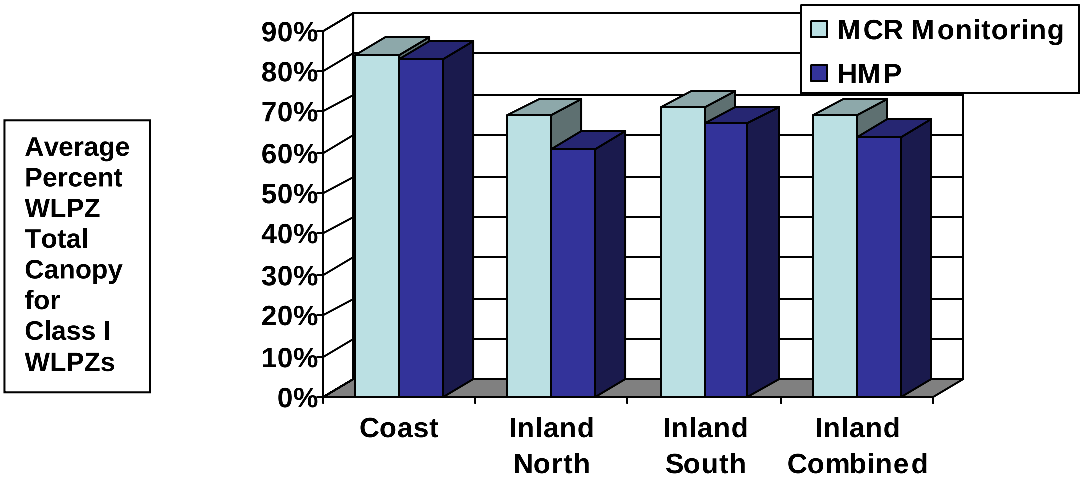


Figure 11. Graphic comparison of MCR (2001-2004) and Hillslope Monitoring Program (1999-2001) results for average percent WLPZ total canopy by Region for Class I watercourses.

Table 5. Comparison of MCR (2001-2004) and Hillslope Monitoring Program (1999-2001) results for average percent WLPZ canopy by Region for Class II watercourses. Number of segments represented in each average equals "n."

Class II WLPZ Comparison	MCR Monitoring (2001-2004) Class II WLPZ percent total canopy	HMP (1999-2001) Class II WLPZ percent total canopy
Coast (Region 1)	84% n = 81	80% n = 109
Inland North (Region 2)	67% n = 31	62% n = 46
Inland South (Region 4)	73% n = 23	74% n = 19
Inland (Regions 2&4 combined)	70% n = 54	66% n = 65

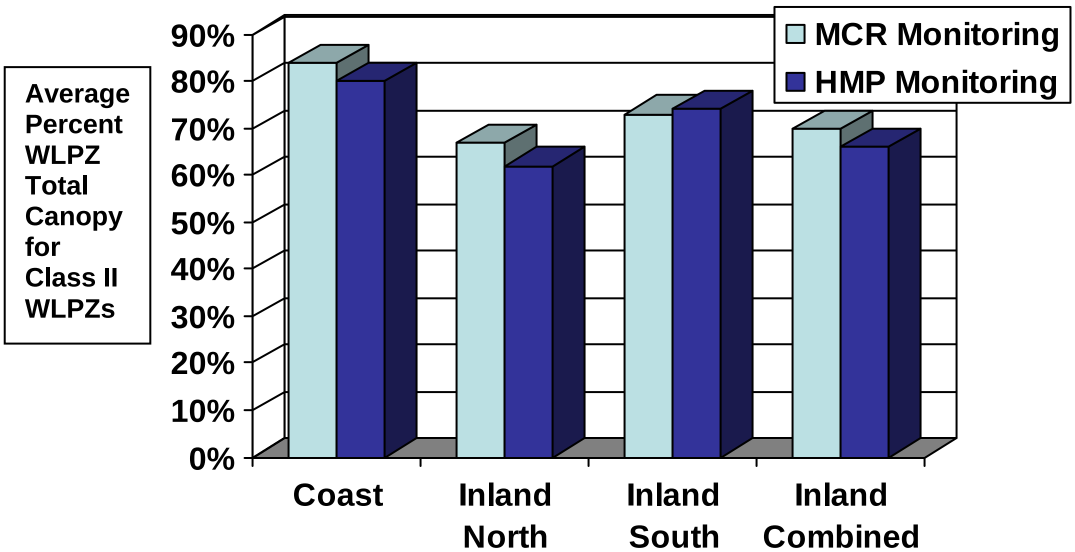


Figure 12. Graphic comparison of MCR (2001-2004) and Hillslope Monitoring Program (1999-2001) results for average percent WLPZ total canopy by Region for Class II watercourses.

Both the MCR and HMP results for percent WLPZ canopy indicate that the FPR standards are generally being met; however, there are rare instances of WLPZs with harvesting done under a current THP that do not meet FPR standards, which are potentially citable violations. Consequently for enforcement purposes, the best strategy to detect such infrequent violations is do quick ocular assessments of as many WLPZs as possible, and reserve more accurate but time-consuming canopy measuring techniques for WLPZs that appear to be probable violations. This observation will be reflected in the recommendations at the conclusion on this report.

Also, as in the HMP, MCR observations of WLPZ groundcover and erosion indicate that WLPZs function well to prevent erosion and sediment transport from current timber operations, assuming they have adequate groundcover and are free of significant patches of bare soil, which was generally found to be the case.

MCR Monitoring:
Roads

I. Methods

Road Segment Selection and Monitoring Timelines

The procedure for randomly selecting a road segment on a THP is described in detail in the *Modified Completion Report Monitoring Procedures and Methods* (see Appendix A). Briefly, a single 1,000-foot long road segment was selected for monitoring on each THP selected for MCR Monitoring (Figure 13). The basic concept is that results from randomly selected segments when aggregated provide unbiased estimates of hillslope erosion, sediment transport off the road prism, and sediment transport to channels.

Figure 13. Pete Cafferata, CDF, recording road observations at a rolling dip. Orange box on his right hip is a hip-chain which meters-out string for tracking distances of specific road-related features along a 1000-foot sample segment.

The initial study design included visiting each road segment twice: first during the Work Completion Report inspection to evaluate implementation, and then during the erosion control maintenance period to evaluate effectiveness after at least one overwintering period. In practice, most of the randomly selected road segments had been through at least one overwintering period prior to the Work Completion Report inspection, therefore most of the evaluations of implementation and effectiveness were done on the first visit.

Segments of roughly equal length (approximately 500 to 1,000 feet) were marked along all of the roads shown on the 1:24,000 scale THP road map. Each segment was then assigned a number. Using either a random number table or function on a calculator, a random number was generated between 1 and the highest numbered segment. The mid-point of the road segment matching the random number was used as the starting point for the 1,000-foot road segment. Direction from the starting point was decided by a coin flip, assuming a 1,000- foot sample road segment could be obtained in either direction.

Not all of the randomly sampled THPs had a single, 1,000-foot long road segment that was suitable for sampling. In these cases, where possible, a sample segment shorter than 1000-feet was monitored. On randomly selected THPs without roads suitable for monitoring (e.g., all of the roads used in the THP were either public roads or residential driveways), no road monitoring was done.

The location of the starting point was marked in the field, often by writing a message such as “Begin MCR Road Sample Segment” and noting the date on flagging attached to a nearby permanent object or vegetation. The *hip-chain* string would then be attached to the starting point and the counter set to zero. While walking the sample road segment, each road-related feature was evaluated and its distance from the start point recorded using the *hip-chain*, until reaching approximately 1,000 feet from the starting point or the end of the road, whichever came first.

Both the procedure and the form used for evaluating road segments were similar to those used in the HMP. Specific methods and the road form are available in the *Modified Completion Report Monitoring Procedures and Methods* (Appendix A). In short, the beginning and ending distances from the segment starting point of all road-related features (e.g., inside ditches, cut banks, waterbreaks, cross drains, etc.) were recorded, regardless of whether or not they presented a water quality problem. Consecutive numbers were assigned to each recorded feature, which, in combination with the THP and segment number, became a unique identifier for that feature. Then codes were recorded to indicate the type of feature and any associated drainage problems, erosion causes, erosion source areas, and sediment production. The dimensions of erosion features were also to be recorded, but this was not done consistently.

The rule numbers used in MCR monitoring were based on the California Forest Practice Rules (CDF 2000) (see Table 6). Unfortunately, the numbering of the FPRs tends to change from year to year with each new version of the rule book. Also, because the road-related rules are located in several sections of the book and because there is often more than one FPR from more one section of the book that covers a road-related feature or issue, the road-related rules tend to be complex. The roads discussion section describes what is being done to remedy this situation.

The California Forest Practice Rules for 2006, with the complete wording of each rule, is available in hardcopy from CDF Headquarters in Sacramento and on-line at http://www.fire.ca.gov/php/rsrc-mgt_forestpractice.php.

Table 6. Summary of road-related Forest Practice Rules that were available for selection for the implementation and effectiveness evaluations for each sample road segment.

Modified Completion Report **Road FPR Pick List (Column C)** Revised 8-11-00		
Type	Rule No.	Description
Waterbreaks	914.6(c) 934.6(c) 954.6(c)	Waterbreak spacing according to standards.
	914.6 (f) 934.6 (f) 954.6 (f)	Where waterbreaks don't work--other erosion controls.
	914.6(g) 914.6(g) 954.6(g)	Waterbreaks constructed with a depth of at least 6 inches cut into firm roadbed.
Roads	923.1(a) 943.1(a) 963.1(a)	Road shown on THP map correctly.
	923.1(a) 943.1(a) 963.1(a)	If landing on road >1/4 ac or required substantial excavation-shown on map.
	923.1(c) 943.1(c) 963.1(c)	Logging roads and landings shall be planned and located, when feasible, to avoid unstable areas.
	923.1(d) 943.1(d) 963.1(d)	For slopes >65% or 50% within 100 feet of WLPZ, soil treated to minimize erosion.
	923.1(e) 943.1(e) 963.1(e)	New logging roads shall not exceed a grade of 15%, except that for 500-foot pitches with max. 20% grades.
	923.1(f) 943.1(f) 963.1(f)	Adequate numbers of drainage facilities provided to minimize erosion.

Type	Rule No.	Description
Roads (continued)	923.1(g) 943.1(g) 963.1(g)	New roads shall be single lane with turnouts, and constructed with balanced cut and fills where feasible.
	923.1(h) 943.1(h) 963.1(h)	Road construction shall be planned to stay out of WLPZs.
	923.1(h) 943.1(h) 963.1(h)	If logging roads will be used from the period of October 15 to May 1, hauling shall not occur when saturated soil conditions exist on the road.
	923.2(b) 943.2(b) 963.2(b)	Sidecast minimized for slopes >65% for distances >100 feet.
	923.2(c) 943.2(c) 963.2(c)	Compacted fill on roads with >50% sideslopes.
	923.2(d) 943.2(d) 963.2(d)	Fills constructed with insloping approaches, etc.
	923.2(e) 943.2(e) 963.2(e)	Breaks in grade above/below throughfill.
	923.2(f) 943.2(f) 963.2(f)	On 35% sideslopes remove organic layer of soil prior to placing fill.
	923.2(g) 943.2(g) 963.2(g)	Proper placement of excess material to avoid polluting streams.
	923.2(h) 943.2(h) 963.2(h)	Drainage structures of sufficient size, number and location to carry runoff water.
	923.2(h) 943.2(h) 963.2(h)	Drainage structures of sufficient size, number and location to minimize erosion.
	923.2(i) 943.2(i) 963.2(i)	Trash racks, etc. installed where appropriate.
	923.2(j) 943.2(j) 963.2(j)	No wood debris in road fills.

Type	Rule No.	Description
Roads (continued)	923.2(k) 943.2(k) 963.2(k)	No overhanging banks.
	923.2(l) 943.2(l) 963.2(l)	Fell trees >12” dbh with >25% of roots exposed by road.
	923.2(m) 943.2(m) 963.2(m)	Sidecast extending >20 ft treated to avoid erosion.
	923.2(o) 943.2(o) 963.2(o)	Discharge onto erodible fill prevented waterbreaks installed to discharge into cover.
	923.2(p) 943.2(p) 963.2(p)	Waterbreaks installed according to standards in FPR 914.6 [934.6, 954.6].
	923.2(q) 943.2(q) 963.2(q)	Drainage facilities in place and functional by October 15, except waterbreaks on roads in use until rains begin to produce overland flow.
	923.2(s) 943.2(s) 963.2(s)	Completed road construction shall be drained by outsloping, waterbreaks, and/or cross-draining by October15.
	923.2(t) 943.2(t) 963.2(t)	Winter roads surfaced where necessary.
	923.2(u) 943.2(u) 963.2(u)	Slash and other debris from road construction placed so as not to discharge into Class I and II streams.
	923.2(v) 943.2(v) 963.2(v)	Road construction activities in the WLPZ, except for stream crossings or specified in the THP, shall be prohibited.
	923.4(a) 943.4(a) 963.4(a)	Road maintenance completed during erosion control period.
	923.4(b) 943.4(b) 963.4(b)	Upon completion of timber operations, temporary roads and associated landing shall be abandoned properly FPR 923.8).
	923.4(c) 943.4(c) 963.4(c)	Waterbreaks maintained to minimize erosion. Erosion controls maintained during maintenance period.

Type	Rule No.	Description
Roads (continued)	923.4(d) 943.4(d) 963.4(d)	Watercourse crossings facilities and drainage structures shall be kept open.
	923.4(e) 943.4(e) 963.4(e)	Roadside berm removed or breached, except where needed for erosion control.
	923.4(f) 943.4(f) 963.4(f)	50-year flow design minimum for drainage structures.
	923.4(g) 943.4(g) 963.4(g)	Temporary roads blocked by start of winter.
	923.4(h) 943.4(h) 963.4(h)	Prevent excessive loss of road surface.
	923.4(i) 943.4(i) 963.4(i)	Soil stabilization where needed to prevent discharge.
	923.4(j) 943.4(j) 963.4(j)	Drainage ditches maintained to allow flow of water.
	923.4(k) 943.4(k) 963.4(k)	Prevent discharge from cuts, fills and sidecast. slopes.
	923.4(l) 943.4(l) 963.4(l)	Maintain trash racks.
	923.4(m) 943.4(m) 963.4(m)	Maintain drainage structures to prevent discharge.
	923.4(n) 943.4(n) 963.4(n)	Maintain drainage structures to prevent diversions.
	923.4(o) 943.4(o) 963.4(o)	Use heavy of equipment, road maintenance in WLPZ is prohibited during the wet season, except in emergencies.
	923.6 943.6 963.6	Wet spots rocked or otherwise treated.

II. Results

Two-hundred and forty-four (244) road segments were rated for implementation of FPRs related to water quality protection. Most of these segments were approximately 1,000 feet long. Some segments were shorter, commonly on plans without a single 1,000 foot long segment, and a few were longer. Using an average length of 1,000 feet, 244 segments equates to approximately 46 miles of road, which is about the distance from Sacramento to Stockton or from San Francisco to San Jose.

Implementation

In this random sample of road segments, a total of 1,991 road features were evaluated for implementation of the FPRs, which gives an average of 43 features per mile of road. Of these 1,991 features, there were 83 departures from the FPRs, or about 1.8 departures per mile of road. It is important to note that these departures tend to be clustered on short sections of bad road. For example, just five road segments out of the total of 244 segments account for 33 of the departures. In other words, the worst 2% of the road mileage accounted for 40% of the departures. This finding has important implications for both road managers and regulators that will be discussed more fully in roads discussion section.

As shown below in Figure 14, of the 1,991 implementation evaluations, 4% were rated as departures from the FPRs, 14% were rated as marginally acceptable, 76% were rated as acceptable, and 6% were rated as exceeding the FPR requirements (greater than acceptable implementation).

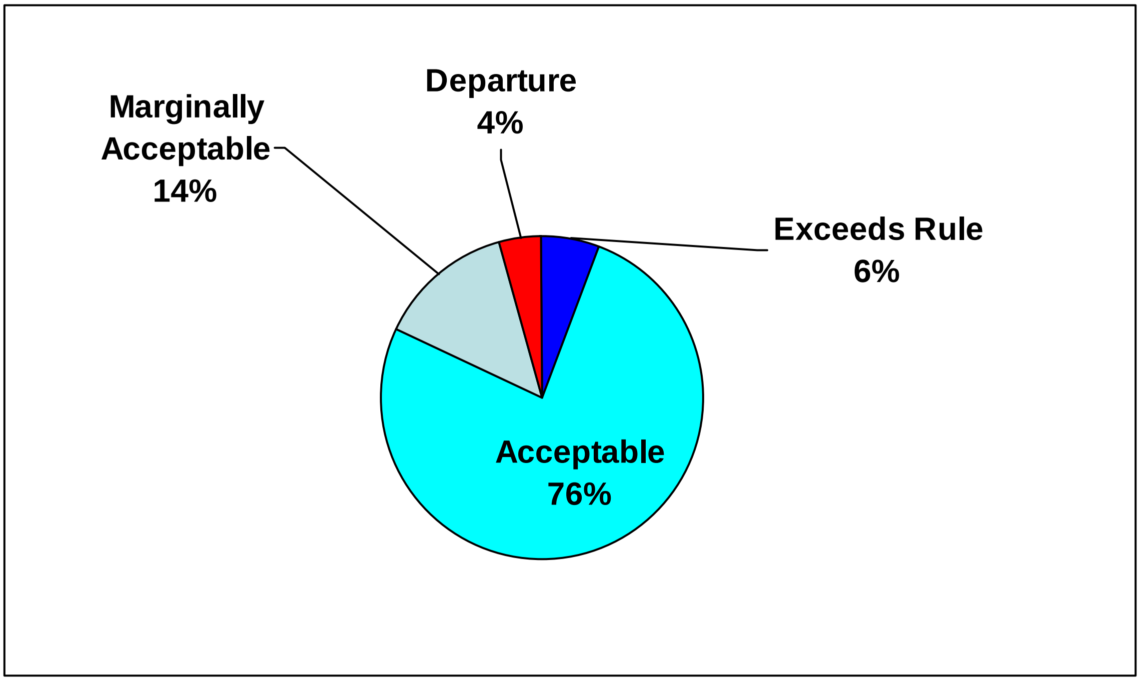


Figure 14. Overall road-related features rated for implementation (n = 1,991).

The Coast (CDF Region 1) accounted for 1,285 of the total 1,991 road features rated for implementation, and 706 were Inland (CDF Regions 2 &4). On the Coast, 2% of the evaluated road features were rated as departures from the FPRs, 15% were rated as marginally acceptable, 76% were rated as acceptable, and 7% were rated as exceeding the FPR requirements (Figure 15).

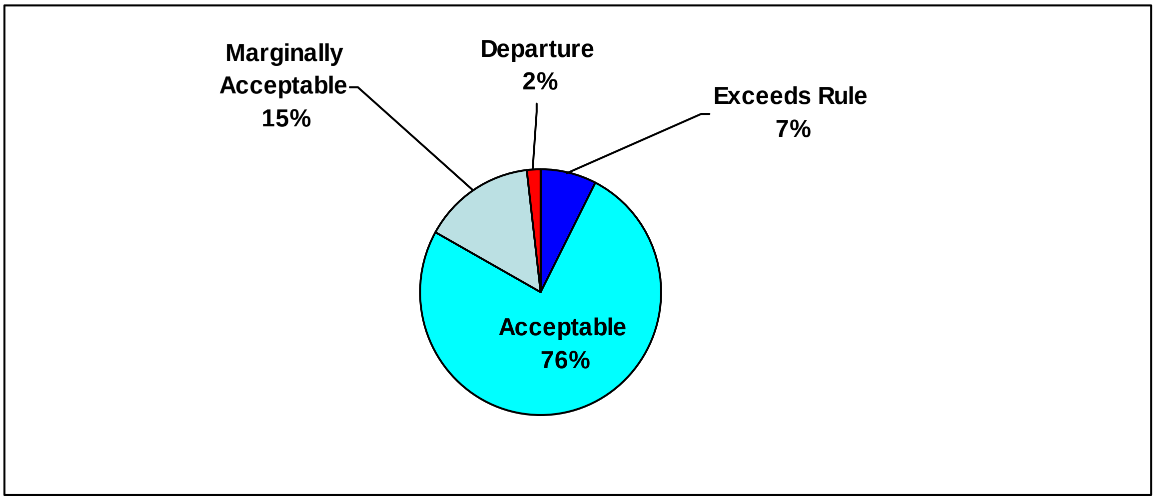


Figure 15. Coast (CDF Region 1) road-related features rated for implementation (n = 1,285).

Inland, 8% of the evaluated road features were rated as departures from the FPRs, 11% were rated as marginally acceptable, 78% were rated as acceptable, and 3% were rated as exceeding the rule (Figure 16).

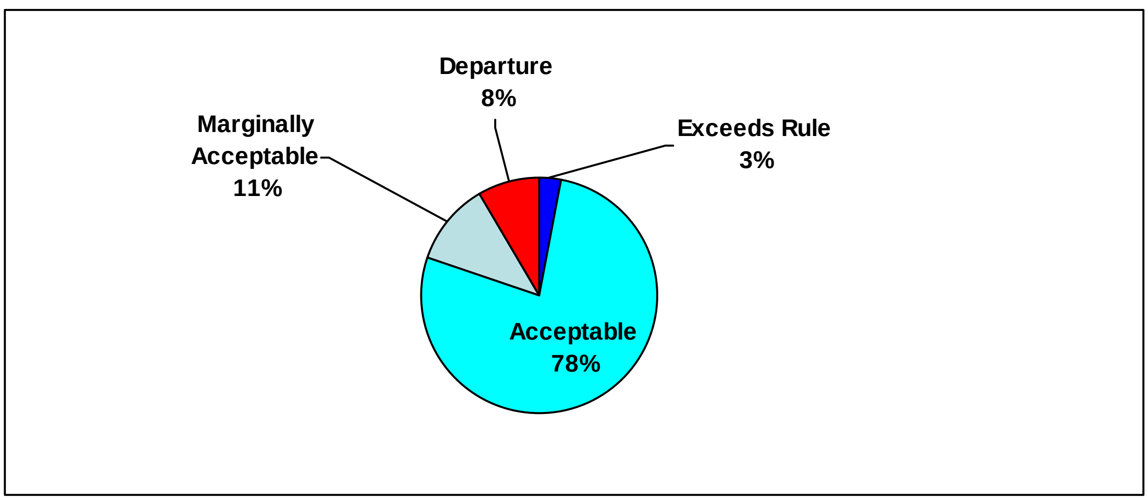


Figure 16. Inland (Regions 2 & 4) road-related features rated for implementation (n = 706).

There is a notable difference between the departure rates of 2% and 8% for coastal and inland regions, respectively. Combining the departure and marginally acceptable ratings for the coast region and also for the inland regions gives much closer results of 17% and 18%. Therefore, it is possible that the difference in departure rates could be an artifact of where inspectors conducting the MCR evaluations in the different regions choose to draw the line between departures vs. marginally acceptable implementations of FPRs. Determining whether this difference is real or not would require having personnel conducting the MCR inspections work and/or train across regions.

Assuming that departure rates for the Coast and Inland regions have been consistently evaluated, there are greater opportunities for improved implementation Inland, where the worst 6% of road segments account for three-quarters of the observed departures. Consequently, preventing departures on the worst 6% of the road mileage would hypothetically reduce the inland departure rate from 8% to a much more acceptable 2%, as shown in Figure 17, below.

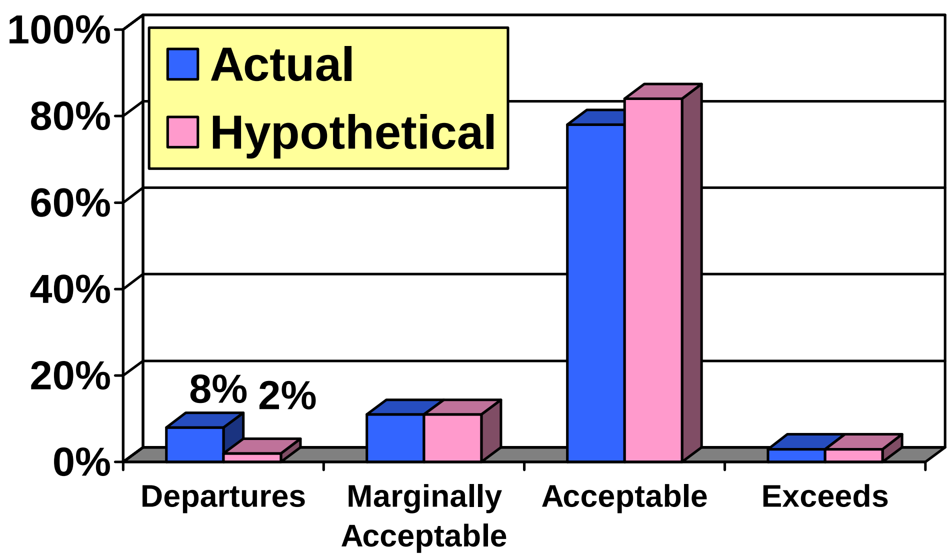


Figure 17. Inland (CDF Regions 2 & 4) hypothetical exercise: What would happen to the departure rate if we found and fixed the worst 6% of all road segments? Answer, the departure rate would hypothetically drop significantly from 8% to 2%.

On the Coast, the departure rate is already a relatively low 2%, and fixing the worst 6% of the road mileage brings the departure rate down to 1% (Figure 18).

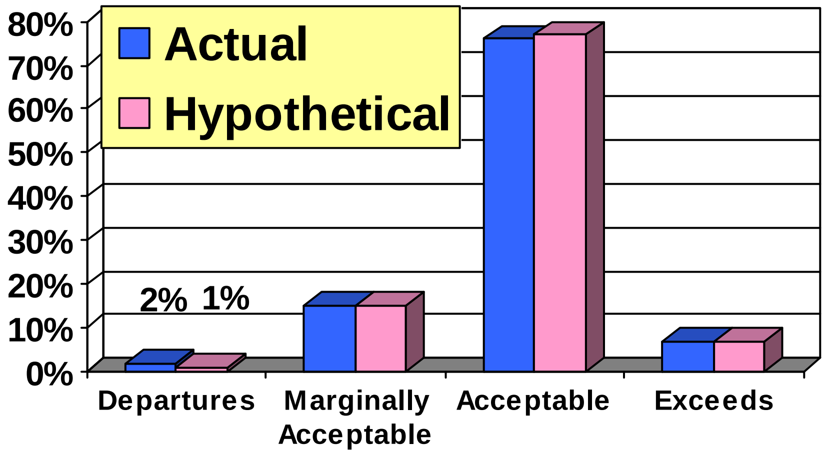


Figure 18. Coast (CDF Region 1) hypothetical exercise: What would happen to the departure rate if we found and fixed the worst 6% of all road segments? Answer, the departure rate would hypothetically drop slightly from 2% to 1%.

Figure 19. Example of road segment built to drain properly in wet weather. Note the two functional dips and their spacing.

The monitoring results demonstrate that most road features are implemented properly (figure 19), since 96% of the road features were rated marginally acceptable or above, as shown in Figure 14 presented earlier. However, there is still room for improvement, and these improvements can and should be focused on areas where it is possible to further reduce the impacts of roads on water quality.

When looking at specific types of features related to observed departures from the FPRs, there is very a definite pattern. Overall, 95% of the observed road-related departures involve FPRs directly related to providing proper drainage. Some of the remaining five percent of departures may also be directly or indirectly affected by drainage. Figure 20, shown below, groups the 95% of departures that are definitively related to drainage into five major categories, and a list of these departures by specific FPR is provided at the end of this section.

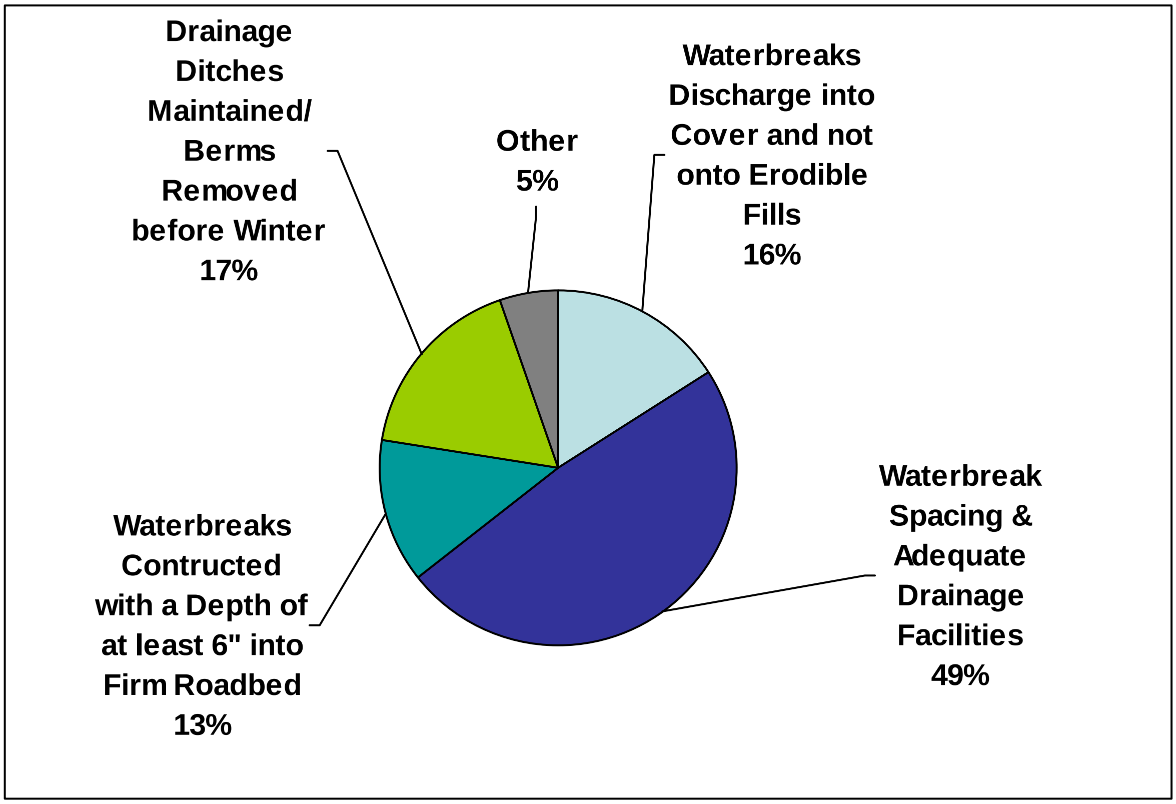


Figure 20. Departures from the road-related FPRs – percentages by category.

As demonstrated in Figure 20, the waterbreak spacing and adequate drainage category accounts for about half of the departures; drainage ditches maintained/ berms removed before winter category accounts for 17%. The waterbreaks discharge into cover and not onto erodible fills category accounts for 16%. The waterbreaks constructed with a depth of at least six inches into firm roadbed category accounts for 13%, and the catch-all category of "other" accounts for only 5% of the departures.

Effectiveness

A total of 130 out of the 244 sampled road segments were rated for FPR effectiveness, which (assuming an average segment length of 1,000 feet, as described above) equates to about 24 miles of sampled roads. These 130 road segments included 1,147 road-related features that were evaluated and rated for effectiveness and are subsets of the 244 road segments and 1,991 features rated for implementation, respectively.

All road segments rated for effectiveness had been through at least one wet season. An important caveat is that selection of road segments rated for effectiveness was not completely random, but neither was it systematic. At the time the monitoring study was designed, it was thought that all road segments in the sample would eventually be rated for effectiveness. This topic is discussed further in the discussion section.

As shown in Figure 21, below, evidence of erosion was found on 109 of the 1,147 road-related features rated for effectiveness. Sediment transport was found associated with 36 of the 109 erosion features, and 9 of those 36 features had evidence of sediment transport to a watercourse channel.

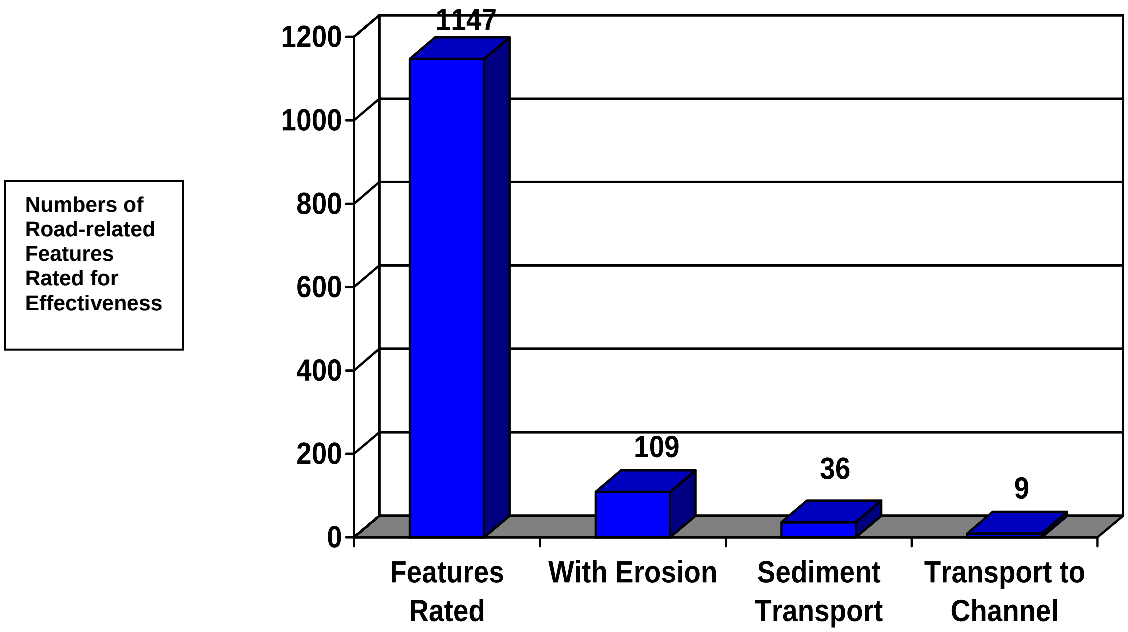


Figure 21. Road-related features rated for effectiveness, comparing the total features rated to the number with evidence of erosion, sediment transport and transport to channel.

When calculated as a percentage of the total features rated, 9.5% of the road features evaluated for effectiveness had erosion, 3.1% showed signs of sediment transport, and 0.8% showed evidence of sediment transport to a channel, as shown in Figure 22.

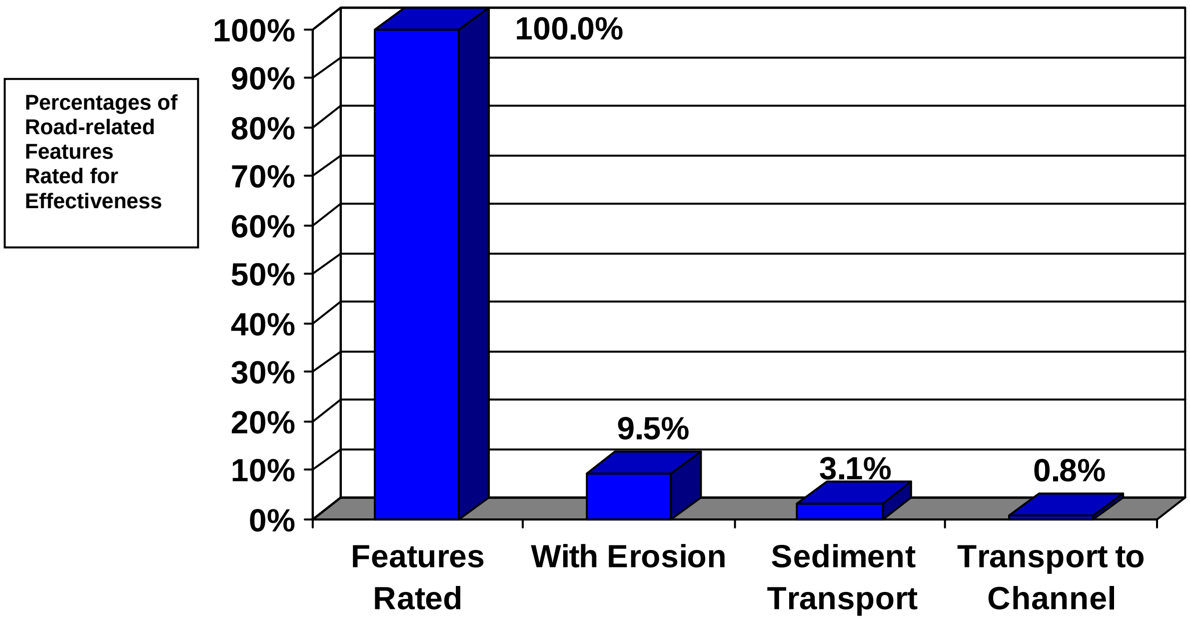


Figure 22. Road-related features rated for effectiveness as percentages, comparing the total features rated to the number with evidence of erosion, sediment transport and transport to channel.

Dividing the data into regions yields 639 road-related features rated for effectiveness on the Coast (CDF Region 1) and 508 Inland (CDF Regions 2 & 4). Of these, 35 and 74 had evidence of erosion, 9 and 27 showed evidence of sediment transport, and 4 and 5 had evidence of transport to a channel for the coast and inland regions, respectively, as shown in Figure 23.

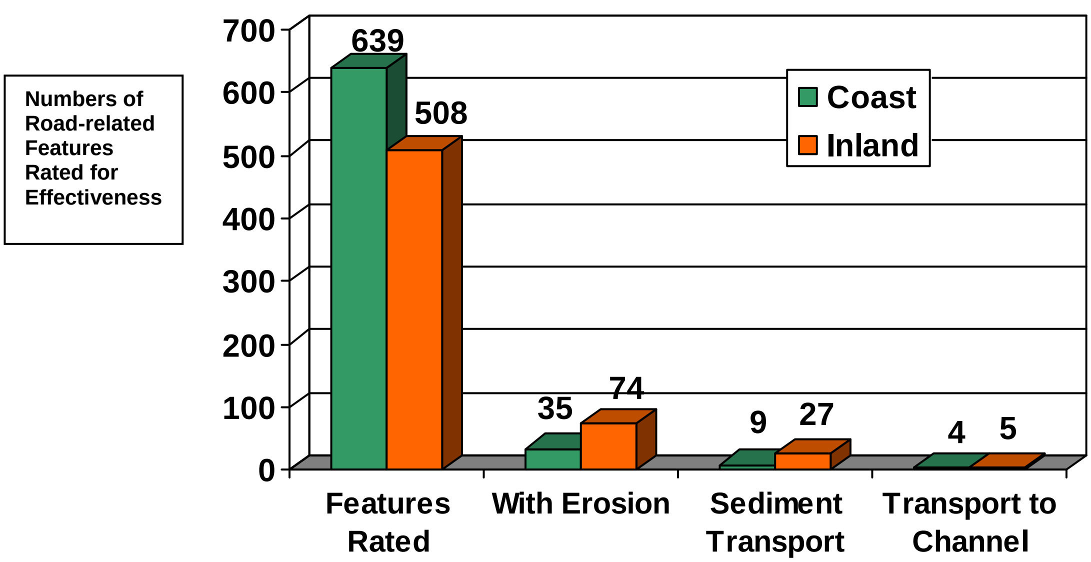


Figure 23. Coast vs. Inland road-related features rated for effectiveness, comparing the total features rated to the number of features with evidence of erosion, sediment transport and transport to channel.

Expressing these results as percentages, as shown in Figure 24, allows an easier comparison between regions. Erosion was found on 5.5% of the road-related features on the Coast versus a much higher 14.5% Inland. Evidence of sediment transport was observed on 1.4% of road-related features on the Coast and on 5.3% Inland. Evidence of sediment transport to channels was found on 0.6% of the road-related features on the Coast and 0.9% Inland.

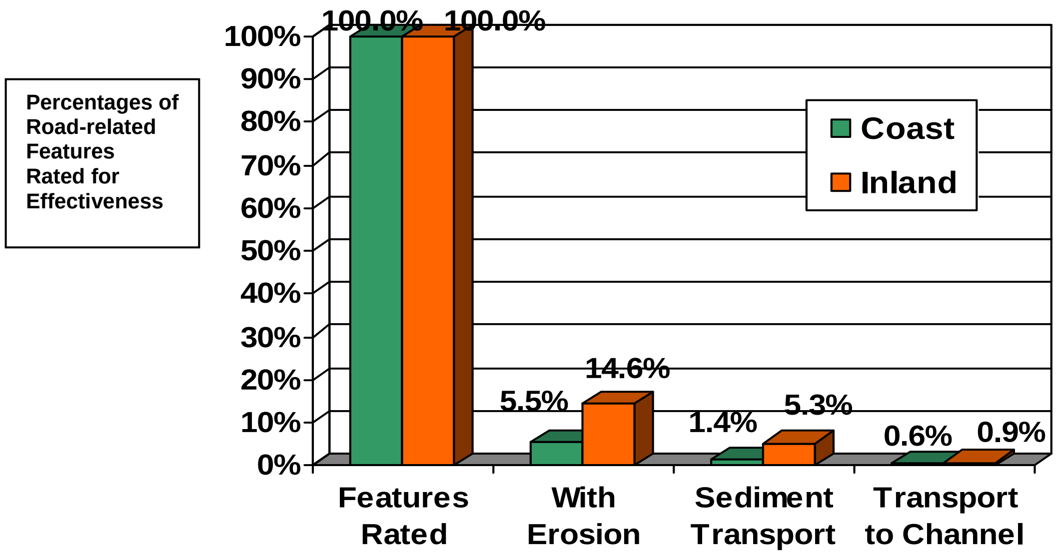


Figure 24. Coast vs. Inland road-related features rated for effectiveness as percentages, comparing the total features rated to the percentage of features with evidence of erosion, sediment transport and transport to channel.

Inland road-related features show signs of erosion and sediment transport more frequently than road-related features on the Coast; however, the percentage of road-related features showing evidence of sediment transport to channels is about the same on the Coast and Inland. One possible explanation for this is that timberlands on the Coast generally get more rainfall than timberlands in Inland and consequently develop denser networks of natural channels, which put road-related features closer to more channels.

Implementation vs. Effectiveness

Better implementation of the road-related FPRs resulted in greater effectiveness in preventing erosion, sediment transport, and sediment transport to channels. While properly implemented road FPRs occasionally failed to prevent erosion, sediment transport, and discharge, improperly implemented FPRs failed at a much higher rate.

Of the 1,147 road-related features that were evaluated for both implementation and effectiveness, 5% had implementation that exceeded the FPR, 78% had acceptable implementation, 12% had marginally acceptable implementation, and 5% were