Thermal Processes and Headwater Stream Temperature

- An understanding of thermal processes is required as a basis for understanding stream temperature dynamics, in particular for interpreting and generalizing from experimental studies of forestry influences.

- As a parcel of water flows through a stream reach, its temperature will change as a function of energy and water exchanges across the water surface and the streambed and banks.

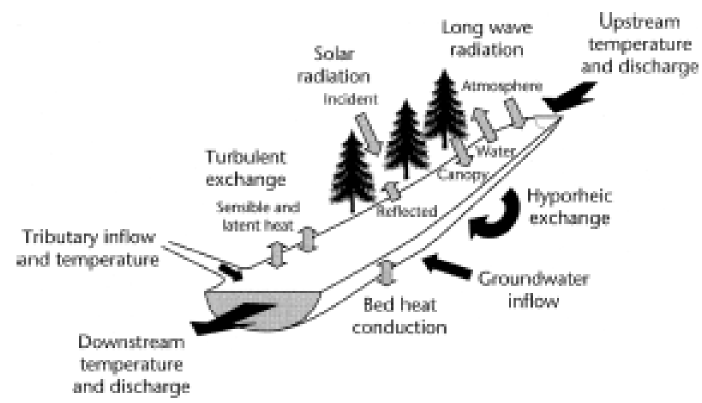


Figure 1. Factors Controlling Stream Temperature. Energy fluxes associated with water exchanges are shown as black arrows.

- Can be defined as a heat balance with expression of the radiation and advective exchange components.

- A form of the energy balance equation

Radiative Exchanges

- Radiation inputs to stream surface include incoming solar radiation (direct and diffuse) and long-wave radiation emitted by the atmosphere, forest canopy and topography.

- Canopy will reduce the direct component of solar radiation and will redistribute some of the diffuse component.

- Channel morphology (wide, narrow, and topographically shaded) will influence how much energy exchange occurs. Orientation can also affect how long the stream "sees" the direct solar during the day.

- When direct radiation comes from +30 degrees above the horizon, most of it can be absorbed within the water column and by the bed, and thus is effective at stream heating.

- Low solar angles at dawn and dusk, and during much of the annual solar cycle are not effective at stream heating because direct radiation comes in at too low an angle to be absorbed effectively.

- Incoming longwave radiation will be a weighted sum of the emitted radiation from the atmosphere, surrounding terrain, and the canopy, with the weights being their respective view factors.

Sensible and Latent Heat Exchanges

- Transfers of sensible and latent heat occur by conduction or diffusion and turbulent exchange in the overlying air.

- Sensible heat exchange depends on the temperature difference between the water surface and overlying air and on the wind speed.

- Where the stream is warmer than the air, heat transfer away from the stream is promoted by the unstable temperature stratification. Where the air is warmer than the stream, the heat transfer from the air to the stream is dampened by the stable air temperature stratification.

- Latent heat exchange also depends on atmospheric stability over the stream.

- Under intact forest cover, especially over small streams, lack of ventilation appears to limit the absolute magnitude of sensible and latent heat exchanges.

Bed Heat Exchanges and Thermal Regime of the Streambed

- Radiative energy absorbed at the streambed may be transferred to the water column by conduction and turbulent exchange and into the bed sediments directly by conduction and indirectly by advection where water infiltrates into the bed. Given that turbulent exchange is more effective at transferring heat than conduction, much of the energy absorbed at the bed is transferred into the water column, and the temperature at the surface of the bed will generally be close to the temperature of the water column, except where there may be local advection.

- Bed heat conduction depends on the temperature gradients within the bed and its thermal conductivity.

- The bed will normally act as a cooling influence on summer days and a warming influence at night, thus tending to reduce diurnal temperature range.

- Bed temperatures may be important biologically.

- The degree to which post-logging bed temperatures reflect changes in surface temperature depends on the local hydrologic environment.

Groundwater Inflow

- Groundwater is typically cooler than the streamwater during daytime, and warmer during winter and thus tends to moderate seasonal and diurnal stream temperature variations.
- Forest harvesting can increase soil moisture and ground water levels
- Increases in gw volume could act to promote cooling, or at least ameliorate warming.
- Some have argued cutting could increase groundwater temperature.
- There are no published research that has examined ground water discharge and temperature both before and after harvest as a direct test of the hypothesis of ground water warming.

Hyporheic Exchange

- Hyporheic exchange is a two-way transfer of water between a stream and its saturated sediments in the bed and riparian zone.
- Stream water typically flows into the bed at the top of a riffle and re-emerges at the bottom of a riffle.
- Hyporheic exchange can create local thermal heterogeneity and it can be important in relation to both local and reach scale temperature patterns in headwater streams.
- There are significant methodological problems associated with quantifying rates of hyporheic exchange and its influence on stream temperature.

Tributary Inflow

- Effects of tributary inflow depend on the temperature difference between inflow and stream temperatures and on the relative contribution to discharge and can be characterized by a simple mixing equation.

Longitudinal Dispersion and Effects of Pools

- Longitudinal dispersion results from variation in velocity through the cross-section of a stream. Not well studied, but could smooth and damp effects downstream.
- Deeper pools may have incomplete mixing creating thermal stratification.

Equilibrium Temperature and Adjustment to Changes in Thermal Environment

- For a given set of boundary conditions (e.g., solar radiation, air temperature, humidity, wind speed) there will be an "equilibrium" water temperature that will produce a net energy exchange of zero and thus no further change in temperature as water flows downstream.

- There is a maximum possible temperature a parcel of water can achieve as it flows through a reach at a given time, assuming that boundary conditions remain constant in time and space.

- Equilibrium conditions may not be achieved because the boundary conditions may changes in time and space before the water parcel can adjust fully to the thermal environment.

- Equilibrium temperature will be lower where there is substantial groundwater inflow, and will be higher for unshaded reaches.

- The rate at which a parcel of water adjusts to a change in the thermal environment depends on stream depth because for deeper streams, heat would be added to or drawn from a greater volume of water.

- Shallow streams adjust relatively quickly to a change in thermal environment.

- Flow velocity influences the length of time the parcel of water is exposed to energy exchanges across the water surface and the bed, and thus the extent to which the parcel can adjust fully to its thermal environment.

- Given that the depth and velocity of a stream tend to increase with discharge, the sensitivity of stream temperature to a given set of energy inputs should increase as discharge increases.

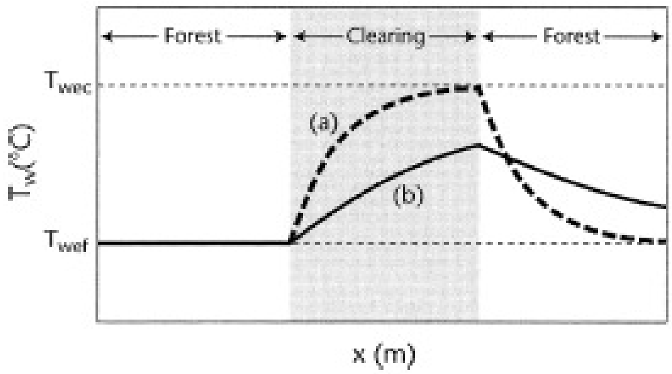


Figure 2. Schematic Temperature Patterns Along a Stream Flowing From Intact Forest, Through a Clear-Cut, and Back Under Intact Forest for (a) Shallow, Low Velocity, and (b) Deep, High Velocity Conditions (T_{wef} = equilibrium temperature in forest; T_{wec} = equilibrium temperature in clearing).

Thermals Trends and Heterogeniety Within Stream Networks

- o Small streams tend to be colder and exhibit less diurnal variability than larger downstream reaches
- o Small streams are more heavily shaded, will have a higher ratio of groundwater inflow, and are located at higher elevations (cooler air).
- o Local deviations from a dominant downstream warming tend may occur as a results of ground water inflow, hyporheic exchange, advection of water from other sources, or even changes in dominant variables such as air temperature.
- o Thermal heterogeneity has been documented at a range of spatial scales: with a pool, within a reach, within a river system.

Stream Temperature Response to Forest Management

- o Studies have occurred.
- o Some BACI, some not
- o Most studies in PNW in rain-dominated climates

Influences of Forest Harvesting Without Riparian Buffers

- Almost all streams that have buffers removed increase in summertime temperature.
- Harsh treatment yields high temperature response.
- Results appear to be more mixed in more recent years.
- Response in snowmelt not well studied. Still get increases.
- Winter temperatures have also not been well studied.

Influences of Forest Harvesting With Riparian Buffers

- Studies in rain-dominated catchments suggest that buffers may reduce, but not entirely protect against increases in summer stream temperature.
- A few studies in snow-dominated in Canada showed increase in temperatures.
- The protective effect of buffers can be compromised by blow-down.

Thermal Recovery Through Time

- Post-harvest temperatures should decrease through time as riparian vegetation recovers.
- Effects seem to last 5-10 years if riparian vegetation is allowed to recover.

Comparison With Studies Outside The Pacific Northwest

- Studies conducted elsewhere in the world are in many ways consistent with results from the PNW.
- However, difference in important environmental variables limit the comparability of results.

Effects of Forest Roads

- Some evidence for very small streams that even a road-right-of-way cut can be of sufficient length to cause local heating.

Downstream and Cumulative Effects

- You can get watershed level response—upstream to downstream translation

- o Downstream transmission of heated water would increase the spatial extent of thermal impacts.
- o Debate about whether down-stream cooling (how much, how fast) can have a significant effect.
- o Streams can cool in the downstream direction by dissipation of heat out of the water column or via dilution by cool inflows. Dissipation to the atmosphere can occur via sensible and latent heat exchange and long wave radiation from the water surface and evaporation.
- o Reported downstream temperature changes below forest clearings are highly variable. Some reports streams cooled, some report streams continued to warm in the downstream direction.
- o Whether cooling occurs may depend on ambient temperatures (only occurs when temperature is at a maximum)
- o Little process work to understand the mechanisms that allow cooling to occur.
- o Three factors may mitigate against cumulative effects of stream warming. 1) dilution could mitigate temps to be biologically suitable, 2) the effects of energy inputs are not linearly additive throughout a stream network due to systematic changes in balance of energy transfer mechanisms. 3) Intercepting environments (lakes, reservoirs)
- o May be secondary impacts like widening and shallowing from sedimentation

Monitoring and Predicting Stream Temperature and its Causal Factors

Monitoring Stream Temperature

- o Most recent studies have used submersible temperature loggers
- o Forward-looking infrared radiometery from helicopters has been used for investigating stream temperature patterns in medium to large streams. The application of this technology to small streams limited. Method can identify cool water areas.

Measuring Shade

- o Many different ways to measure shade (view-factor).

Predicting the Influences of Forest Harvesting on Stream Temperature

- There are empirical models (a few environmental variables can usually predict maximum temperature within a degree or two with about r^2 of 0.60 to 0.70)

- There are physically-based models. There are a variety of them with different assumptions, formulations, variables to inform, complexity. Most, including the simplest, predict temperature accurately.

Discussion and Conclusions

Summary of Forest Harvesting Effects on Microclimate and Stream Temperature

Biological Consequences and Implication for Forest Practices

- Briefly discusses non-fish potential effects

- A better understanding is required of how changes in the physical conditions in small streams and their interactions with chemical and biological processes influence their downstream exports.

- One tree height should cover it.

Issues For Future Research (Moore et al. 2005)

- Riparian microclimates have been relatively little studies, both in general and specifically in relation to the effects of forest practices.

- Shade is the dominant control on forestry-related stream warning in small streams.

- Determining shade in small streams is difficult and refined and consistent methods are needed.

- Hemispherical photography might be the way to go to solve subjectivity and methods problems.

- The effects of low and deciduous vegetation in controlling temperature in very small streams is not well understood.

- Further research should address the thermal implications of surface/subsurface hydrologic interactions, considering both local and reach scale effects of heat exchange associated with hyporheic flow paths.

- Bed temperature patterns in small streams and their relation to stream temperature should be researched in relation to stream the effects on benthic invertebrates and nonfish species.

- The hypothesis that warming of shallow ground water in clearcuts can contribute to stream warming should be addressed, ideally by a combination of experimental and process/modeling studies.

- The physical basis for temperature changes downstream of clearings needs to be clarified. Are there diagnostic site factors that can predict reaches where cooling will occur. Such information could assist in the identification of thermal recovery reaches to limit the downstream propagation of stream warming. It could also help identify areas within a cut block where shade from a retention patch would have the greatest influence.

The Physiological Basis for Salmonid Temperature Response

- Water temperature governs the basic physiological functions of salmonids and is an important habitat factor.

- Fish have ranges of temperature wherein all of these functions operate normally contributing to their health and reproductive success. Outside of the range, these functions may be partially or fully impaired, manifesting in a variety of internal and externally visible symptoms. Salmon have a number of physiologic and behavioral mechanisms that enable them to resist adverse effects of temporary excursions into temperatures that are outside of their preferred or optimal range. However, high or low temperatures of sufficient magnitude, if exceeded for sufficient duration, can exceed their ability to adapt physiologically or behaviorally.

- Salmon are adapted over some evolutionary time frame to the prevailing water temperatures in their natural range of occurrence, and climatic gradient are among the primary factors that determine the extent of a species' geographic distribution on the continent.

- Salmon are considered a "cold water" species, and generally function best within the range of ambient temperatures in water bodies within their natural range of occurrence. This range is 0-30^{o}C for salmonids, where end temperatures are lethal and mid range temperatures are optimal. The southern limit of the natural range of salmonids coincides with the occurrence of summer water temperatures of 30^{o}C.

- The effects of temperature are a function of magnitude and duration of exposure. Exposure to temperatures above 24^{o}C of sufficient continuous duration can cause mortality.

- Salmon can tolerate each successively lower temperature for exponentially increasing intervals of time. Temperatures above 22^{o}C are stressful. Lengthy exposure to higher temperatures include loss of appetite and failure to gain weight, competitive pressure and displacement by other species better adapted to prevailing temperatures, or disease.

- Growth occurs best when temperatures are moderate and food supplies are adequate. High and low temperatures limit growth. Optimal temperatures for growth are in the range of 14 to 17^{o}C, depending on species.

- Salmon have been shown to increase growth in streams where riparian canopy rwas removed due to increased light and food availability, despite the occurrence of warmer temperatures.

- Larger size generally increases survival and reproductive success.

- Growth rates are important for anadromous salmonids, who must reach minimum sizes before they are able to migrate to the ocean. Missing normal migration windows by being too small or too large may have negative effects on success in reaching the ocean.

- The temperature of rivers and streams ranges over the full range of temperatures within the range utilized by salmonids during the course of the year. The summer maximum temperatures are generally those of most concern.

- The most thermally tolerant salmonid species occur in California (steelhead, chinook and coho). Of these species, coho are the most thermally sensitive.

Temperature Exposure in Natural Streams and Potential Effects of Forest Practices

- Water temperature generally tends to increase in the downstream direction with stream size as a result of systematic changes in the important environmental variables that control water temperature. As streams widen, riparian canopy provides less and shade until some point in a river system where it provides no significant blocking effect. Cooler groundwater inflow also diminishes in proportion to the volume of flow in larger streams.

- The lowest order streams have the coolest water temperatures near groundwater temperature (11-14^{o}C). Higher order streams are near ambient air temperatures (20-26^{o}C). The range of water temperature from lower to higher orders in California rivers and streams during the warmest period in the summer spans much of the tolerable temperature range for salmonids. Water temperature typical of higher order streams are within stressful levels for salmonids.

- Removal of riparian vegetation may increase stream temperatures up to the ambient air temperature, depending on the natural extent of shading and the proportion of canopy removed. Thus, temperatures typically observed only in downstream reaches may occur in tributary streams.

- Salmonid distribution within stream systems and within the region reflects temperature tolerance. Coho are found in the cooler waters associated with headwater streams and within the coastal zone where climate is strongly influenced by the Pacific Ocean. Steelhead have somewhat higher thermal tolerance, and are more widely distributed.

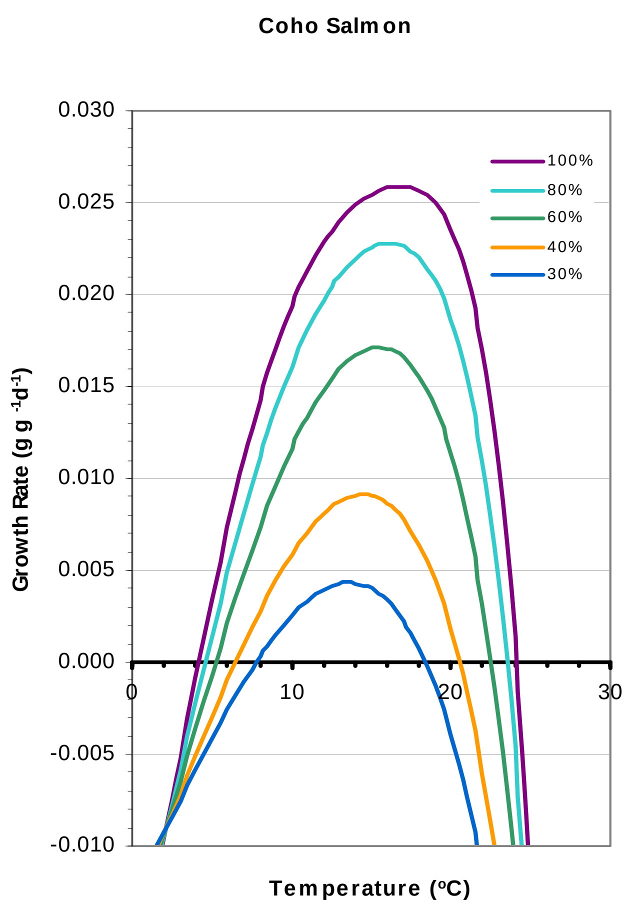
3) TAC Primer on Temperature and Salmon and Watershed Patterns (The Physiological Basis for Salmonid Temperatures)

The Physiological Basis for Salmonid Temperature Response

Water temperature is a dominant factor affecting aquatic life within the stream environment (Hynes 1970). Water temperature affects important stream functions such as processing rates of organic matter, chemical reactions, metabolic rates of macro-invertebrates, an cues for life-cycle events (Sweeney and Vannote 1986). Water temperature plays a role in virtually every aspect of fish life, and adverse levels of temperature can affect behavior (e.g. feeding patterns or the timing of migration), growth, and vitality.

Water temperature governs the rate of biochemical reactions in fish, influencing all activities by pacing metabolic rate (Frye 1971). Fish are poikilothermic or "cold-blooded". This means that fish do not respond to environmental temperature by feeling hot or cold. Rather, they respond to temperature by increasing or decreasing the rate of metabolism and activity. Water temperature is the thermostat that controls energy intake and expenditure.

Figure 1. Coho salmon daily growth rate as a function of temperature and daily food ration.



The role of temperature in governing physiologic functions of salmonids has been studied extensively (Brett 1971; Elliott 1981; reviewed in Adams and Breck 1990; Brett 1995, McCullough 1999). The relationship between energetic processes and temperature have been quantified for many fish species with laboratory study. Energetic processes are expressed as functions of activity rate in relation to temperature. The relationships between energy-related functions and temperature follow two general patterns: either the rate increases

continuously with rise in temperature (e.g., standard metabolic rate, active heart rate, gastric evacuation), or the response increases with temperature to maximum values at optimum temperatures and then decreases as temperature rises (e.g., growth rate, swimming speed, feeding rate) (Brett 1971, Elliott 1981). Each function operates at an optimal rate at some temperature and less efficiently at other temperatures.
For example, daily growth as a function of temperature is shown in Figure 1. Beginning with the coolest temperatures (0^{o} C), growth increases with temperature up to the optimal due to increasing consumption and food conversion efficiency. At temperatures above the optimal, growth rates decline as consumption declines in response to temperature and metabolic energy costs increase (Brett 1971, Elliott 1981, Weatherly and Gill 1995). Because the shape of growth curves is relatively broad at the maximum, there is little or no negative effect of temperature several degrees above optimum. Some investigators define the optimal temperature as the temperature at which maximum growth occurs, and refer to the range of temperature where growth occurs as "preferred" temperatures (Elliott 1981).

The general form of this relationship is similar for all salmonid species, varying somewhat in the details of growth rates and optimal temperatures. All salmonids have a similar biokinetic range of tolerance, performance, and activity. They are classified as temperate stenotherms (Hokanson 1977) and are grouped in the cold water guild (Magnuson et al. 1979). Significant differences in growth rate and temperature range exist among families of fish (Christie and Regier 1988). Some families grow best in colder temperatures (e.g. char), and many grow better in warmer temperatures (e.g. bass). Differences in the specific growth/temperature relationships among species in large measure explain competitive success of species in various temperature environments.

The range of environmental temperature where salmonid life is viable ranges from 0-30 ^{o}C, with critical temperatures varying somewhat by species. Salmonid physiologic functions operate most effectively in the mid regions of the range where growth is also optimized. Physiological functions are impaired on either end of the temperature range so that the geographic distribution of prevailing high or low temperatures ultimately limits the distribution of the species in the Salmonidae family (Eaton 1995).

The effects of temperature are a function of magnitude and duration of exposure. Figure 2 from Sullivan et al. 2000 summarizes the general relationship of salmonid response to temperature exposure. Salmon species are similar in this pattern, but vary somewhat in the temperatures zones of response.

Exposure to temperatures above 24^{o}C can elicit mortality with sufficient length of exposure. The temperature where death occurs within minutes is termed the ultimate upper incipient lethal limit (UICL). This temperature is between 28- 30^{o}C, varying by salmon species. Clearly, salmon populations are not likely to persist where this temperature occurs for even a few hours on a very few days each year (Eaton 1995).

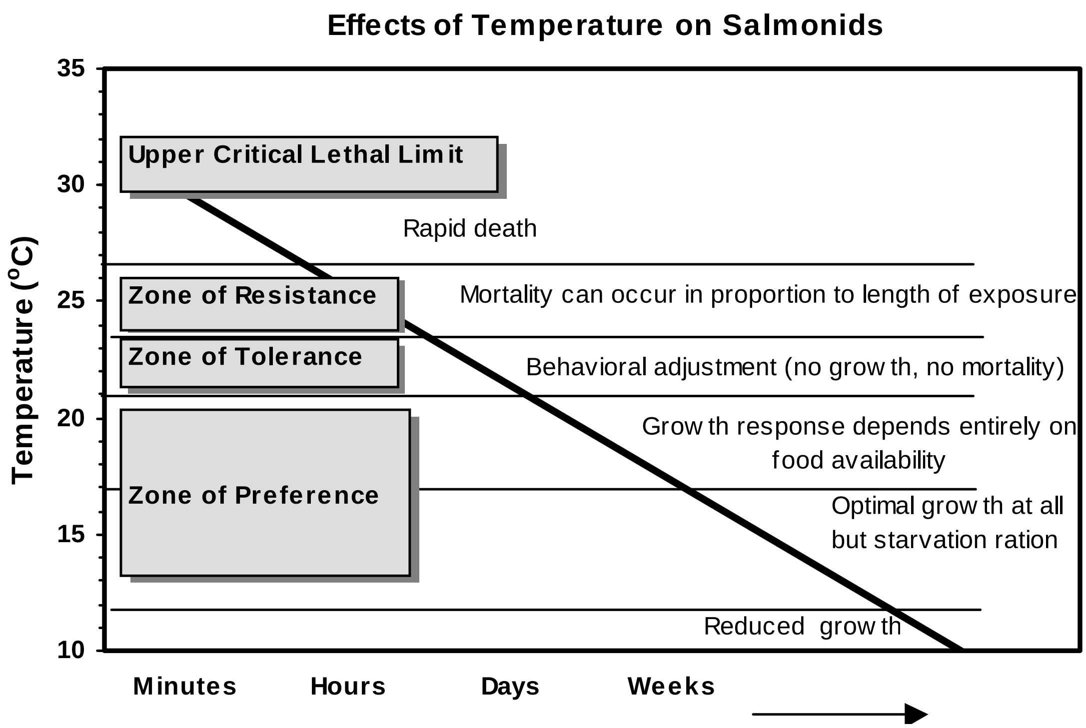
Lethal exposure is defined as up to 96 hours of continuous exposure to a given temperature.

Salmon can tolerate each successively lower temperature for exponentially increasing intervals of time. They do so by altering food consumption and limiting the metabolic rate and scope of activity (Brett 1971, Elliott 1981, Weatherly and Gill 1995). This resistance to the lethal effects of thermal stress enables fish to make excursions for limited times into temperatures that would eventually be lethal (Brett 1956; Elliott 1981). The period of tolerance prior to death is referred to as the “resistance time” (Figure 2) (Hokanson 1977, Jobling 1981). Salmon can extend their temperature tolerance through acclimation. Brett (1956) reported that the rate of increase in ability to tolerate higher temperatures among fish is relatively rapid, requiring less than 24 hours at temperatures above 20^{o}C. Acclimation to low temperatures (less than 5^{o}C) is considerably slower.

Laboratory and field studies have repeatedly found that salmon can spend very lengthy periods in temperatures between 22 and 24^{o}C without suffering mortality (Brett 1995, Bisson et al. 1988; Martin 1988). Temperatures within this range may be stressful, but are not typically a direct cause of mortality (Brett 1956). Temperatures that cause thermal stress after longer exposures, ranging from weeks to months, are termed chronic temperature effects. Endpoints of lengthy exposure to temperature that are not physiologically optimum may include loss of appetite and failure to gain weight, competitive pressure and displacement by other species better adapted to prevailing temperatures (Reeves et al. 1987), change in behavior, or susceptibility to disease. Werner et al. (2001) documented correlations between stream temperature, size of juvenile steelhead and heat shock protein expression.

Fish may be able to avoid thermal stress by adjusting behavior, such as moving to cooler refugia. Numerous observers have observed behavioral adjustment by seeking cool water refugia when temperature in normal foraging locations reaches 22°C (Donaldson and Foster 1941; Griffiths and Alderdice 1972; Wurtsbaugh and Davis 1977; Lee and Rinne 1980; Bisson et. al. 1988; Nielsen et al. 1994, Tang and Boisclair 1995; Linton et al. 1997; Biro 1998). Fish resume feeding positions when temperatures

Figure 2. General biological effects of temperature on salmonids in relation to duration and magnitude of temperature (from Sullivan et al. 2000).



decline below this threshold. At very low temperatures, salmonids cease feeding and seek cover under banks or within stream gravels (Everest and Chapman 1972).

Less quantifiable in a dose-response context are relationships involving temperature and disease resistance, and temperature effects on sensitivity to toxic chemicals and other stressors. (Cairns et al. 1978). For temperature to affect the occurrence of disease, disease-causing organisms must be present, and either those organisms must be affected by temperature or fish must be in a weakened state due to the effect of temperature. Some disease-causing organisms may be more prevalent at high temperature, others are more prevalent at low temperature, and some are not temperature-related. Thus, the interaction of temperature and disease is best evaluated on a location-specific basis.

If energy intake is adequate to fuel the physiological energy consumption, mediated in large part by the environmental temperature, then the organism can live in a healthy state and grow. Growth is a very important requirement for anadromous salmon living in fresh water. Salmon emerge from gravels in their natal streams measuring approximately 30 mm in length and weighing approximately 0.5 gram. Adults returning to spawn 3 to 5 years later typically measure 500 to 1000 mm in length and weigh from 5 to 20 kg depending on species. This enormous increase in body mass (greater than 5000 times) must be accomplished within a very limited lifespan. Salmon have evolved from a fresh water origin to spend a major portion of life in a marine habitat where there is far greater productivity and where the majority of growth occurs (Brett 1995).

Juvenile salmon must achieve the first six times increase in weight in their natal stream before they can smolt and migrate to the ocean (Weatherly and Gill 1995). Coho and steelhead generally smolt within 1 year, but can require as long as 3 years to achieve sufficient size to begin the transition to salt water. The long-term exposure of salmonids to temperature during their freshwater rearing phase has an important influence on the timing of smoltification and the ultimate size fish achieve (Warren 1971, Brett 1982, Weatherly and Gill 1995, Sullivan et al. 2000).

The size of salmonids during juvenile and adult life stages influences survival and reproductive success (Brett 1995). Larger size generally conveys competitive advantage for feeding (Puckett and Dill 1985, Nielsen 1994) for both resident and anadromous species. Smaller fish tend to be those lost as mortality from rearing populations (Mason 1976; Keith et al. 1998). Larger juveniles entering the winter period have greater over-wintering success (Holtby and Scrivener 1989; and Quinn and Peterson1996). Growth rates can also influence the timing when salmon juveniles reach readiness for smolting. Missing normal migration windows by being too small or too large, or meeting a temperature barrier, may have a negative effect on success in reaching the ocean (Holtby and Scriverner 1989).

How large a salmon can grow in a natural environment is fundamentally determined by environmental and population factors that determine the availability of food. Water temperature regulates how much growth can occur with the available food. Brett et al. (1971) described the freshwater rearing phase of juvenile salmon as one of restricted environmental conditions and generally retarded growth. Many studies have observed an increase in the growth and productivity of fish populations in streams when temperature (and correspondingly) food is increased. This tends to occur even in the cases where temperatures exceed preferred and sometimes lethal levels (Murphy et al. 1981, Hawkins et. al., 1983, Martin 1985, Wilzbach 1985, Filbert and Hawkins 1995).

Table 1 summarizes results from laboratory and field studies of coho and steelhead temperature response (from Sullivan et al 2000). Steelhead and coho are similar, though not identical, in the temperatures at which various functions or behaviors occur. Importantly, Sullivan et al (2000) showed that even though the laboratory optimal growth temperatures for steelhead are within a narrower and cooler range than those of coho (e.g. their "growth curves"), steelhead grow better than coho when exposed to higher temperatures in natural streams. These authors suggest that this disparity results from a greater efficiency in obtaining food in natural environments by steelhead, thus allowing them to generally obtain a higher ration of food. Bisson et al (1988b) showed that the body form of these two fish differ, enabling steelhead to feed efficiently in riffle habitats where food supply is more abundant. Thus, steelhead have a higher "net temperature tolerance" than coho.

Table 1. The spectrum of coho salmon and steelhead response at temperature thresholds synthesized for field and laboratory studies in Sullivan et al (2000). Threshold values are approximations, due to lack of consistency in reporting temperature averaging methods among studies. Temperature thresholds are standardized to the average 7-day maximum to the extent possible to allow comparison of field and laboratory study observations.

Biologic Response	**COHO Approximate Temperature ^{o}C**	**STEELHEAD Approximate Temperature ^{o}C**
Upper Critical Lethal Limit (death within minutes)-Lab	29.5	30.5
Geographic limit of species—Stream annual maximum temperature (Eaton 1995)	30	31.0
Geographic limit of species—Warmest 7-Day Average Daily Max Temperature (Eaton 1995)	23.4	24.0
Acute threshold U.S. EPA 1977—Annual Maximum	25	26
Acute threshold U.S. EPA 1977— 7-day average of daily maximum	18	19
Complete cessation of feeding (laboratory studies)	24	24
Growth loss of 20% (simulated at average food supply)	22.5	24.0
Increase incidence of disease (under specific situations)	22	22
Temporary movements to thermal refuges	22	22
Growth loss of 10% (simulated at average food supply) (7-day average of daily maximum)	16.5	20.5
Optimal growth at range of food satiation (laboratory)	12.5-18	10-16.5
Growth loss of 20% (simulated at average food supply) 7-day average of daily maximum	9	10
Cessation of feeding and movement to refuge	4	4

Optimal temperatures for both Chinook salmon fry and fingerlings range from 12 C to 14 C, with maximum growth rates at 12.8 C (Boles 1988). [These numbers seem much to low compared to other studies. Need reference.] With the exception of some spring-run Chinook salmon, most Chinook juveniles do not rear in streams through the summer and are therefore not typically exposed to late-summer conditions. A significant portion of spring-run Chinook salmon, however, reside in streams throughout the summer. These salmon are also the only salmonid that must cope with summer water temperatures as adults. They typically enter the Sacramento River from March to July and continue upstream to tributary streams where they over-summer before spawning in the fall (Myers et al. 1998). Adult spring-run Chinook salmon require deep, cold pools to hold over in during the summer months prior to their fall spawning period. When these pools exceed 21 C adult Chinook salmon can experience decreased reproductive success, retarded growth rate, decreased fecundity, increased metabolic rate, migratory barriers, and other behavioral or physiological stresses (McCullough 1999).

There has been some suggestion that there may be genetic adaptations by local populations that confer greater tolerance to temperatures. However, literature on temperature thresholds for salmonids, as summarized in Table 1 is remarkably consistent despite differences in locations of subject fish (Sullivan et al. 2000, Hines and Ambrose 2000, Welsh et al. 2001).

One problem encountered in synthesizing laboratory and field studies is how to characterize the widely variable stream temperature characteristics of a stream in either a physically or biologically meaningful way is lack of standardization on reporting summary statistics. The measures of 7-day maximum values have been shown to have biological meaning (e.g. Brungs and Jones 1977). These types of metrics also provide useful indices for comparing temperature among streams. Sullivan et al (2000) showed that all of the short-term high temperature criteria relate closely to one another when calculated from the same stream temperature record (7-day mean and maximum, annual maximum temperature, and long-term seasonal average). However, longer-term measures are better indicators of general ecologic metabolism. For example, degree-summation techniques sum duration of time (days, hours) above a selected threshold temperature.

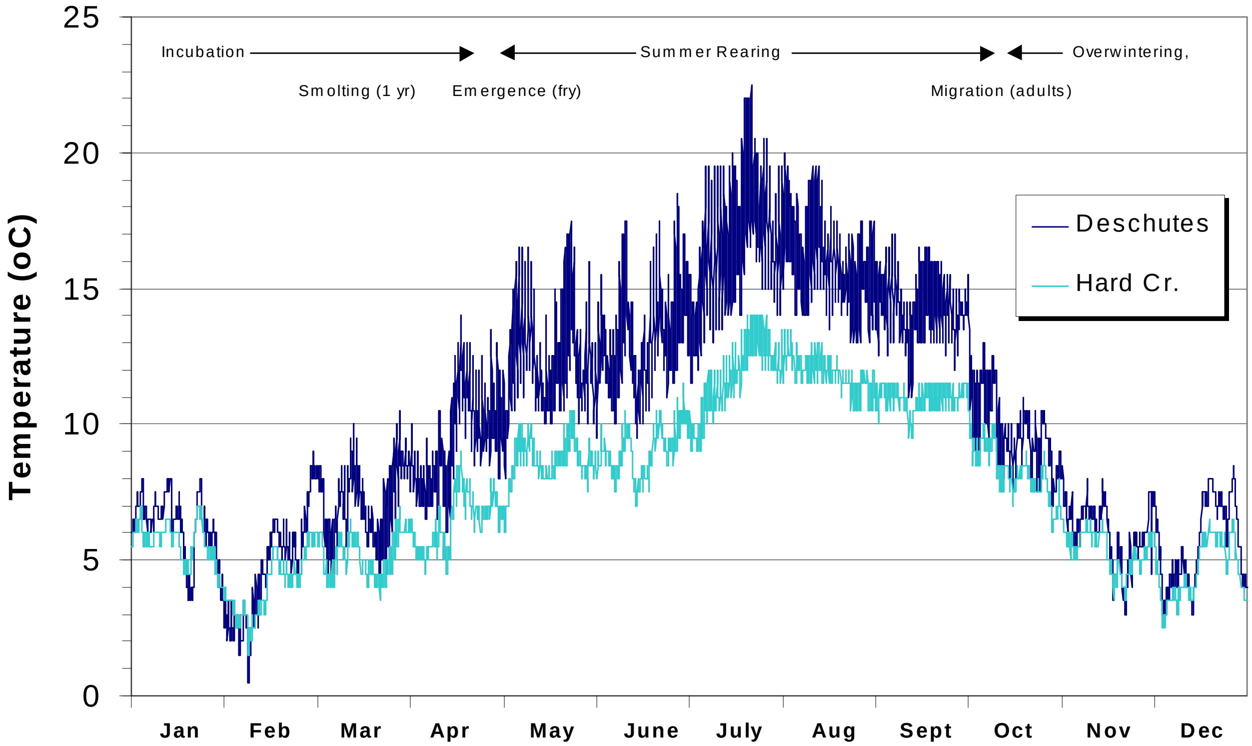
Temperature Patterns and Salmonid Species Distribution Within Watersheds

Temperatures supporting the physiologic functions of fish species reflect the ambient temperatures likely to be found in streams in each species' natural range of occurrence (Hokanson 1977). For salmonids, this range is from 0 to less than 30^oC (see Table 1). Within the range of distribution of salmonids in the Pacific Northwest, there is a west to east climatic gradient reflecting the marine influence at the coast and the orographic effects of interior mountain ranges. Coastal zones are characterized by maritime climates with high rainfall that occurs during the winter and dry warm summers. Interior zones are dryer, and rainfall may occur as rain or snow. Summers are very dry, and temperatures often hotter than coastal zones, although elevation can have a significant cooling effect. Comparison of river temperatures associated with forested regions throughout Washington, Oregon and Idaho show generally consistent occurrence of temperatures within the temperature tolerance of salmonids (Sullivan et al. 2000).

The temperature of streams and rivers within the range of distribution of salmonids in the Pacific Northwest and California typically vary widely on both temporal and spatial scales. For example, the range of hourly temperature over a year period for a smaller headwaters stream and larger mainstem river located within a forested watershed in Washington are shown in Figure 3. (The figure also shows the typical phase and migration timing for coho and steelhead salmon.) Similar patterns are observed in forested regions of California.

Active feeding and positive growth can occur at any time during the year when temperature is within the positive growth range illustrated in Figure 1. Juvenile salmon experience preferred temperatures for much of the year, and may experience stressful temperature conditions for relatively little time during the year. Water temperatures between 8 and 22°C tend to be the most prevalent temperatures observed in natal rivers and streams in the Pacific Northwest (Sullivan et al. 2000). Temperatures high enough to directly cause mortality are rare within the region where salmon occur. Temperatures high enough to cause stress (>22°C) may be common, especially in

Figure 3. Water temperature of the Deschutes River (148 km^2) and Hard Creek (2.3 km^2), a headwater tributary, near Von. Data are hourly measurements.

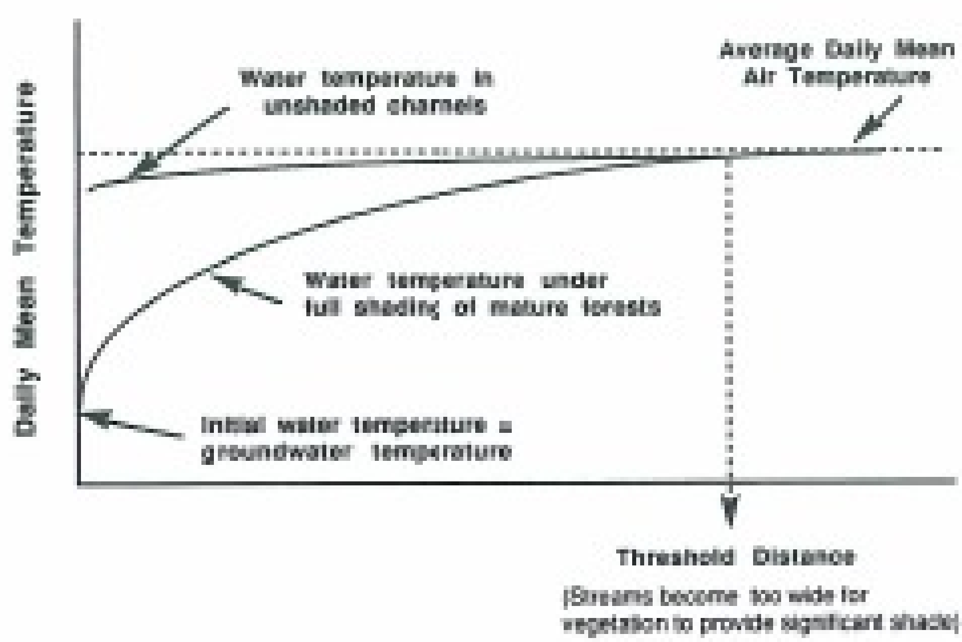


higher order streams.

Watershed Temperature Patterns

Stream temperature tends to increase in the downstream direction from headwaters to lowlands. (Hynes 1970, Theurer et al 1984). The dominant environmental variables that regulate heat energy exchange for a given solar loading, and determine water temperature are stream depth, proportional view-to-the-sky, rate and temperature of groundwater inflow, and air temperature (Moore et al, 2005). Increasing temperature in the downstream direction reflects systematic tendencies in these critical environmental factors. Air temperature increases with decreasing elevation (Lewis et al. 2000). Riparian vegetation and topography shade a progressively smaller proportion of the water surface as streams widen (Spence et al. 1996), until at some location there is no effective shade at all (Beschta et al. 1987, Gregory et al. 1991). Streams gain greater thermal inertia as stream flow volume increases (Beschta et al. 1987), thus adjusting more slowly to daily fluctuations in energy input. The typical watershed temperature pattern is illustrated in Figure 4.

Figure 4. General pattern of temperature at the watershed scale and potential range of response to forest removal. (from Sullivan et al. 1990).



Low order streams tend to be the coolest within the stream system. Low order streams are close to source areas and emerge near groundwater temperatures. They are typically shallow, steep and narrow, and are well-shaded, depending on overstory vegetation. Mid-order streams have wider channels and therefore less shade, greater flow volume, and moderate gradient. Tributary inflow is the main source of external flow contribution (as opposed to groundwater inputs). Higher order streams characteristically have low gradients, wide channels, and large volumes of water. Riparian vegetation and topography provide little insulation. The thermal inertia of the large volume of flow, and rapid mixing by turbulent flow generally overwhelms any lateral inputs (tributaries or phreatic groundwater) relatively quickly, allowing only isolated pockets of colder water. These streams may have large alluvial aquifers that may create significantly cooler zones from hyporheic flow; particularly in streams with complex channel features.

Water temperature in larger rivers without riparian shading is in equilibrium with, and close to, air temperature. In smaller streams, water temperature is depressed below air temperature due to the cooling effects of groundwater inflow and the shading effects of the forest canopy (Sullivan et al. 1990; Moore 2005). The minimum temperature profile in Figure 4 indicates the general pattern of water temperature in streams in a fully forested watershed. The coolest temperatures will be observed in the smallest streams and will be near prevailing groundwater temperature. As the effects of these insulating variables lessens in the downstream direction, water temperature moves closer to air temperature until the threshold distance where riparian canopy no longer provides effective shade and the water temperature is closely correlated with air temperature alone (Kothandaraman 1972). It is likely that the shape of the minimum line varies both with basin air temperature and with differences in natural vegetation.

Various authors have reported the likely summertime temperatures that mark the highest and lowest temperatures on this curve for streams and rivers of the Pacific Northwest and California used by salmonids. Minimum groundwater temperatures are approximately 10-13^{o}C (Sullivan et al. 1990, Lewis et al. 2000). Maximum temperatures typically range from 20 to 26^{o}C (Sullivan et al. 2000, Lewis et al. 2000) depending on location.

Removal of vegetation in headwater streams may allow temperature to increase up to (but not exceed) the basin air temperature maxima. Thus, the potential response of water temperature to forest harvest may be large in small streams, but only small, and difficult to detect in mid to large size watersheds.

Fish Species Distribution Within Watersheds

Salmonid species found in California include Chinook (O. tshawytscha), coho (O. kisutch), and steelhead (O. salmo). These species are the most temperature tolerant of the anadromous species in the salmonidae family. The southern-most extent of the natural range of salmon is found at latitude approximately equal to San Francisco, dipping further south along the coast. Eaton (1995) showed a strong relationship between prevailing summertime maximum temperatures and the end of the range of occurrence.

Salmon species throughout their range have evolved to use different parts of the river system during their freshwater rearing phase. Systematic changes in the occurrence or dominance of species within river systems in part reflects the temperature patterns as one important component of habitat. Differences among species can confer competitive advantages in relation to environmental variables that influence the species' distribution (Brett 1971, Baltz et. al. 1982, Reeves et al. 1987, DeStaso and Rahel 1994).

Steelhead have higher net temperature tolerance, are widely distributed within the northern region of California and occupy a broader range of habitats including larger rivers and smaller streams. Coho have the lowest net temperature tolerance of the salmonids found in California, and are found primarily where temperatures are coolest for most of the year. They primarily occur in the low to mid-order tributaries within the coastal zone. (reference for distribution).

Chinook salmon are perhaps the most temperature tolerant of all salmon species. They have the highest optimal temperatures for growth and fastest growth rates of all the salmonids. Fall run chinook emerge from gravels in spring and move to the larger (warmer) rivers where their growth rate allows them to migrate to the ocean with weeks to a few months. They migrate out of the river before the warmest summer temperatures occur. An exception are spring-run Chinook salmon. Some juveniles reside in streams throughout the summer. These salmon are also the only salmonid that must cope with summer water temperatures as adults. They typically enter the Sacramento River from March to July and continue upstream to tributary streams where they over-summer before spawning in the fall (Myers et al. 1998). Adult spring-run Chinook salmon require deep, cold pools to hold over in during the summer months prior to their fall spawning period. When these pools exceed 21 C adult Chinook salmon can experience decreased reproductive success, retarded growth rate, decreased fecundity, increased metabolic rate, migratory barriers, and other behavioral or physiological stresses (McCullough 1999).

California Regional Temperatures

To date, there has been no California-wide water temperature study or synthesis of available information. A regional stream temperature study was conducted within the Coho ESU by the Forest Science Project at Humboldt State University (Lewis et al. 2000). The area where coho occur within California is delineated by the Coho ESU includes the northern coast zone and portions of the interior Klamath region. Water temperature was measured at hundreds of sites in a variety of streams and rivers well distributed within the area from approximately San Francisco northward to the Oregon border, and from the coast to approximately 300 km inland. Stream size varied from watershed areas as small as 20 to a maximum of over 2,000,000 hectares. The assessment included new data and historical analysis of historic temperature assessments, augmented with recently measured temperature at the same locations as earlier measurements.

Results of the study provide some general insight into maximum summer stream temperatures within this region of California.

- The regional study confirmed the general increasing trends in temperature from watershed divide to lowlands.

- The annual maximum temperature ranged from 12-25^{o}C in the coastal zone and 14-32^{o}C inland beyond the coastal influence. Temperature as high as 32^{o}C occurs, but is rare.

- The cooling influence of the coastal fog belt on air temperature extends as far inland as 50 km in some rivers, and is significant enough to affect water temperature within a distance 20 km from the coast in some locations. The effect of the cool air is sufficient to reduce some river temperatures by as much as 5-7^{o}C degrees by the time water reaches the ocean. These help prevent prolonged exposure to stressful temperatures. The coast fog zone is the dominant zone for coho productivity in the state.

- Maximum temperature in rivers in the coastal fog belt can exceed 20^{o}C

- No one geographic, riparian, or climatic factor explains water temperature with high precision. Multiple regression models developed from the data explain about 65% of the variability, similar to finding in other parts of the Pacific Northwest (Sullivan et al. 1990).

- The coolest maximum temperatures (<18^{o}C) are most likely to occur where:

    - Distance from divide is less than 10 km.

    - Canopy cover is >75%

- The probability of achieving temperature of <20^{o}C decreases at 1) lower canopy closure, 2) distance from divide as an indicator of stream size, and 3) with distance from the coast.

- There is relatively small difference in maximum water temperatures between interior and coastal streams of similar watershed areas in basins less than 100,000 hectares in size.

What needs to be understood better for California:

- the availability of cool water at the watershed and population scale

- the overall cumulative effect of temperature on the annual basis.

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Primer on Sediment Riparian Exchanges Related to Forest Management in the Western U.S.

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PRIMER: SEDIMENT RIPARIAN EXCHANGE FUNCTION:
Erosion and Erosion and Sediment Processes in California's Forested Watersheds

Erosion is a natural process that is well described for California in several college textbooks (Norris and Webb 1990, Mount 1995). California's evolving landscape reflects the "competing processes of mountain building and mountain destruction", with landslides, floods, and earthquakes working as episodic forces which often create major changes (Mount 1995). In general, the land surface is sculpted by the forces of erosion: water, wind, and ice. The physical and chemical composition of the rock determines how it weathers by these forces. The role of running water in shaping the earth's surface is considered the most important of all the geologic processes and has received the greatest attention by researchers (Leopold et al. 1964; Morisawa 1968).

The rates of natural erosion are very high in the State's regions having greater amounts of rain and snow, such as the geologically young mountains of the Northern Coast Ranges, Klamath Mountains, and Sierra Nevada (Norris and Webb 1990). Mean annual precipitation was shown to be a relatively precise indicator of climatic stress on sedimentation in Northern California (Anderson et al. 1976).

Soil erosion processes on upland watersheds include: a) surface erosion (e.g., dry ravel, sheet and rill), b) gullying, and c) mass movement or wasting (e.g., soil creep and landslides, such as slumps, earthflows, debris slides, large rotational slides). These can occur singly or in combination. Falling raindrops can be a primary cause of surface erosion, especially where soils have little vegetative cover (Brooks et al. 1991). Erosion products deposited by water become "sediment", brought to a channel by gravity and erosive forces. The water-related, or "fluvial", processes active within the stream channel and floodplain are: 1) the transport of sediment; 2) the erosion of stream channel and land surface; and 3) the deposition or storage of sediment.

Sediment Sizes, Transport & Measurement

Sediment is any material deposited by water, but research usually describes sediment according to its size, means of transport, and method of measurement (MacDonald et al. 1991, Leopold 1994). Inorganic sediment ranges in size from very fine clay to very large boulders. Particle size classes tend to be split into a different number of size categories by physical scientists (AGI 2006) and by biologists (Cummins 1962). The Modified Wentworth Scale is commonly used by biologists (Waters 1995) and includes 11 particle sizes and names: clay, silt, sand (five classes), gravel, pebbles, cobbles, and boulders. In addition, sediment includes particulate organic matter, composed of organic silts and clays and decomposed material. Grain size terminology can also vary:

- *Fine-grained sediment* ("fines") includes the smaller particles, such as silt and clay (usually <0.83 mm in diameter). The largest size class for this category varies, sometimes including sand and small gravel (1-9 mm) (Everest et al. 1987).

- *Coarse-grained sediment* represents the larger particles, such as gravels and cobbles. It makes up the bed and bars of many, if not most, rivers. The smallest size class for this category varies, and sometimes includes sand and small gravel (1-9 mm).

Whatever the term used, it is important to understand the sediment definition and particle size that each research article is using before extrapolating the results.

Sediment is transported by streams as either *suspended load* of the finest particle sizes (from clay to fine sand <2.0 mm) that are carried within the water column, or as *bedload* of the larger particles (from coarse sand to boulders) that never rise off the bed more than a few grain diameters. Higher velocity and steeper streambed slope can transport larger grain size, for example.

Since the measurement of sediment transport levels can be problematic, it is done in several ways. (For detailed descriptions of common methods, including the strengths and limitations of each, see MacDonald et al. 1991, Gordon et al. 1992, and Waters 1995.)

Suspended sediment samplers measure direct suspended sediment concentration (SSC) in milligrams of sediment per liter of water (mg/l). Since most sediment transport takes place during high flows, samples must be taken during these periods to develop long-term averages. Many samples are needed near peak discharges to determine the error margin. Two types of samplers can be used: depth-integrating and point-integrating.

Turbidity is a measure of the ability of light to be transmitted through the water column (e.g., the relative cloudiness). Turbidity sampling and meters are often used as a substitute for the direct measurement of the suspended sediment load of a selected stream reach, but the relationship may vary and requires a careful study design to make accurate correlations Turbidity is frequently higher during early season runoff and on the rising limb of a storm's runoff; automated data collection is now being used to more accurately capture such infrequent events (Eads and Lewis 2003). Turbid water may also be due to organic acids, particulates, plankton, and microorganisms (which can be ecologically beneficial); interpretation must therefore be carefully done. In redwood-dominated watersheds of north coastal California, Madej (2005) found the organic content of suspended sediment samples ranged from 10 to 80 weight percent for individual flood events. Turbidity is not a good indicator for movement of coarse-grained sediments, such as sand in granitic watersheds, since these larger grain sizes move at the bottom of the water column or as bedload (Morisawa 1968; Sommarstrom et al. 1990; Gordon et al. 1992).

Bedload measurement can be a difficult method since this larger-sized sediment must be collected manually during high flows when bedload is in transport. While there are different types of methods and equipment, the Helley-Smith bedload sampler has become the standard for bedload measurement, especially for coarse sand

and gravel beds. Multiple samples must be taken per cross-section of stream. Bedload cannot be collected automatically as readily as suspended sediment can. Bedload as a percentage of suspended load can range from 2-150 percent; 10 percent bedload would be a conservative estimate for a storm event with muddy-looking water in a gravel-bed stream.

Sediment that is deposited within stream channels can be measured by changes in channel characteristics. The most common methods include: a) channel cross-sections, b) channel width / width-depth ratios; b) pool parameters (e.g., fines stored in pools (V*)), c) bed material (particle-size distribution, embeddedness, surface vs. subsurface particle size); d) longitudinal profiles in upstream-downstream directions (e.g., using the "thalweg", the deepest part of the stream channel).

Fluvial Processes and Sediment

Stream reaches can be defined by the dominant fluvial processes: erosion /transport / storage (Schumm 1977; Montgomery and Buffington, 1997; Bisson, *et al,* 2006). The steep headwaters tend to be the source of erosion, the middle elevation streams are the transfer zone, and the low elevation streams are the depositional zone. However, any given stream reach demonstrates all three processes over a period of time; the relative importance varies by location in the watershed.

Natural Sources of Sediment

Within the riparian zone, natural sediment sources and the effects of the riparian zone tend to vary by the type of channel reach (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). The uppermost parts of many source reaches are characterized by exposed bedrock, glacial deposits, or colluvial valleys or swales. Stream reaches in bedrock valleys are usually strongly confined and the dominant sediment sources are fluvial erosion, hillslope processes, and mass wasting. The colluvial headwater basins have floors filled with colluvium which has accumulated over very long periods of time. Such channels as may exist are directly coupled with the hillslopes, and their beds and banks are composed of poorly graded colluvium. Stream flow is shallow and ephemeral or intermittent. The colluvial fill is periodically excavated by debris flows which scour out the stream channels and delivery large quantities of sediment and large woody debris to downstream reaches (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). There is often is no distinctively riparian vegetation bordering the channels.

A bit further downstream, transport reaches commonly still have steep gradients, are strongly confined and subject to scouring by debris flows. Stream beds are consequently characterized either by frequent irregularly arranged boulders or by channel-spanning accumulations of boulders and large cobbles that separate pools. The boulders move only in the largest flood flows and may have been emplaced by other processes (e.g., glacial till, landslides). Streams generally have a sediment transport capacity far in excess of the sediment supply (except following mass wasting events). Dominant sediment sources are fluvial and hillslope processes and mass wasting (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). The transition between transport and response reaches is especially likely to have persistent and pronounced impacts from increased sediment supply (Montgomery and Buffington, 1997).

In the higher response reaches, stream gradients and channel confinement become more moderate. Incipient floodplains or floodprone areas may begin to border the channels, so they are not so coupled to hillslope processes. The typical channel bed is mostly straight and featureless with gravel and cobble distributed quite evenly across the channel width; there are few pools. Where the bed surface is armored by cobble, sediment transport capacity exceeds sediment supply, but unarmored beds indicate a balance between transport capacity and supply. Dominant sediment sources are fluvial processes, including bank erosion, and debris flows are more likely to cause deposition than scouring (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). There is usually distinctively riparian vegetation along the channel.

Also in low to moderate gradients, braided reaches may form where the sediment supply is far in excess of transport capacity (e.g., glacial outwash, mass wasting) and/or stream banks are weak or erodible (Buffington, *et al*, 2003). Channels are multi-threaded with numerous bars. The bars and channels can shift frequently and dramatically, and channel widening is common. The size of bed particles varies widely. Banks are typically composed of alluvium. Bank erosion, other fluvial processes, debris flows, and glaciers are the dominant sediment sources. Distinctively riparian vegetation is common, and is especially important in providing root strength to weak alluvial deposits (Bisson, *et al*, 2006).

In lower-elevation, lower-gradient response reaches, channels are generally sinuous, unconfined by valley walls, and bordered by floodplains. Beds are composed of gravel or sand arranged into ripples or dunes with intervening pools. Sediment supply exceeds sediment transport capacity, so much of the finer sediment is deposited outside the channel onto the floodplain. The dominant sediment sources are fluvial processes, bank erosion, inactive channels, and debris flows. Distinctively riparian vegetation typically grows on the floodplain where it plays important roles in: i) reinforcing weak alluvial banks and floodplains, and ii) providing hydraulic roughness to reduce erosion during overbank flooding (Montgomery and Buffington, 1997; Bisson, *et al,* 2006).

Natural sediment production in undisturbed watersheds can vary significantly, depending upon soil erodibility, geology, climate, landform, and vegetation. Delivery of sediment to channels by surface erosion is generally low in undisturbed forested watersheds, but can vary greatly by year (Swanston 1991). Annual differences are caused by weather patterns, availability of materials, and changes in exposed surface area. Sediment yields for surface erosion tend to be naturally higher in rain-dominated than in snow-dominated areas. Soil mass movement is the predominant erosional process in steep, high rainfall forest lands of the Pacific Coast. The role of natural disturbances in maintaining and restoring the aquatic ecosystem is becoming more recognized by scientists using interdisciplinary approaches (Reeves et al. 1995).

California Examples

Landslides are an important sediment source in northern coastal ranges of California, particularly where they were active in the wet period of the late Pleistocene and have remained dormant for long periods. If reactivated by undercutting at the toe, these slides can deliver immense amounts of sediment to channels (Leopold 1994). Kelsey (1980) found in the Van Duzen River basin that avalanche debris slides accounted for headwater erosion storage, but that natural fluvial hillslope erosion rates were quite low. In the North Coast range, small headwater streams tend to aggrade their beds during small storms and degrade during large, peak flow events. However, in larger streams, sediment aggrades during large events and gradually erodes during smaller ones (Janda et al.1978).

Sediment budgets offer a quantitative accounting of the rates of sediment production, transport, storage, and discharge (Swanson et al. 1982; Reid & Dunne 1996). They are performed in California by academic researchers (Kelsey 1980; Raines 1991), consultants (e.g., Benda 2003), and agencies. In a review of sediment source analyses completed for agency-prepared Total Maximum Daily Load (TMDL) allocations in nine north coast California watersheds, the amount of the “natural” sediment source contribution ranged from a low of 12% to a high of 72% over the past 20-50 year period (Kramer et al. 2001). An evaluation of sediment sources in a granitic watershed of the Klamath Mountains found 24% of the erosion and 40% of the sediment yield to be natural background levels in 1989 (Sommarstrom et al. 1990). Post-fire erosion can be a major component of sediment budgets in semi-arid regions of California (Benda 2003).

Role of Riparian Vegetation

Forested riparian ecosystems influence sediment regimes in many ways. First, riparian plant species are adapted to flooding, erosion, sediment deposition, seasonally saturated soil environments, physical abrasion, and stem breakage (Dwire et al. 2006). Sediment transported downslope from overland flow passes by riparian vegetation, where it can accumulate or be transported through the riparian area (USEPA 1975; Swanson et al. 1982b). The significance of vegetation’s role in providing bank stability and improving fish habitat was first recognized as early as 1885 (Van Cleef 1885). Riparian plant roots help provide streambank, floodplain, and slope stability (Thorne 1990; Abernathy and Rutherford 2000; NRC 2002) and can bind bank sediment, reducing sediment inputs to streams (Dunaway et al. 1994). Bank material is much more susceptible to erosion below the rooting zone, but vegetated banks are typically more stable than unvegetated ones (Hickin 1984). Soil, hydrology, and vegetation are interconnected in bank stability, though the understanding has developed more slowly

(Sedell and Beschta 1991; NRC 2002). For example, the effect of riparian vegetation roots on the mass stability of stream banks may be overestimated in erosion models, according to recent research (Pollen and Simon 2005). In a study on the Upper Truckee River, California, a willow species provided an order of magnitude more root reinforcement than lodgepole pine and reduced the frequency of bank failures and sediment delivery (Simon, Pollen, and Langendoen 2006).
Riparian vegetation patterns appear to indicate specific landforms and local hydrogeomorphic conditions; the patterns differ by geographic location and climate, such as semi-arid versus humid regions (Hupp and Ostercamp 1996). Since streamside areas tend to have high moisture and low soil strength, they are vulnerable to compaction and physical disturbance (Dwire et al. 2006). For some sediment processes originating from upslope of the riparian zone, vegetation may have little influence. Large, deep-seated landslides are probably not affected by streamside plants and downed wood, for example (Swanson et al. 1982b). Current conditions of riparian plant communities need to be viewed in the context of the historical alterations to the landscape, including land management (NCASI 2005).

Effects of Sediment on Aquatic Life of Streams

While erosion processes can provide sources of gravels for fish spawning, excessive sediment deposition can be harmful to aquatic life. Habitat needs for anadromous salmonid fish of the Pacific Coast are well described by Bjornn and Reiser (1991), with a review of the effects of fine sediment on fish habitats and fish production compiled by Everest et al. (1987), Furniss (1991), Walters (1995), Spence et al. (1996), and CDFG (2004). A brief summary of the effects of sediment on critical life stages of salmon and trout is as follows:

- Spawning: Fine sediment can become embedded in spawning gravels, reducing the abundance and quality available for spawning and possibly preventing the female from excavating her nest (redd); excessive sediment loading can cause channel aggradation, braiding, widening, and increased subsurface flows, all reducing spawning gravel abundance; excess sediment can fill pools that are needed for rest and escapement of adults migrating upstream to spawn.
- Egg Incubation: Excessive fine sediments can suffocate or impede egg development or developing alevins by reducing or blocking intragravel water flow, oxygenation, and gas exchange. Organic sediment, however, can provide valuable food (e.g., bugs) for fish (Madej 2005).
- Juvenile Rearing: Coarse and fine sediment can fill pools, which reduces the volume of habitat available for critical rearing space and the population that can be sustained; fine sediment can cover the streambed and suffocate benthic macroinvertebrates, reducing availability of important food source (Suttle et al. 2004). Chronic turbidity from suspended fine sediment interferes with feeding effectiveness of fry and smolts, reducing their growth rate or forcing them to emigrate (Sigler et al. 1984; Newcombe and Jensen 1996; Rosetta 2004).

The review by Everest et al. (1987) demonstrated that the effects of fine sediment on salmonids are complex and depend on many interacting factors: species and race of fish, duration of freshwater rearing, spawning escapement within a stream system, presence of other fish species, availability of spawning and rearing habitats, stream gradient, channel morphology, sequence of flow events, basin lithology, and history of land use (Furniss et al. 1991). It also should be noted that research on the effect of "fine sediment" on salmonid reproduction (e.g., percent survival of fry emergence from eggs) varies in the definition of sediment size, ranging from 0.85mm to 9.5 mm, but tends to focus on 2.0 millimeters or less (Everest et al. 1987). One needs to be careful in interpretation of the literature when comparing the effects of differently defined "fines" (Sommarstrom et al. 1990.)

The first major literature review on the aquatic effects of human-caused sediment was published in 1961 by California Dept. of Fish and Game biologists Cordone and Kelley, who concluded that sediment was harmful to trout and salmon streams. Productive streams, at every trophic level, contain stored sediment and large organic debris and are more productive than channels with too little or too much sediment (Everest et al. 1987). An early California study of streams with increased sedimentation found that fish biomass decreased in some streams and increased in others (Burns 1972). Stream macroinvertebrate diversity was significantly decreased in stream reaches below failed logging road crossings, implying the effect of higher sediment levels (Erman et al. 1977). In a review of stream characteristics in old-growth forests, the authors noted that many streams in California have naturally high sediment loads, including an abundance of fines less than 1 mm, but historically these streams supported healthy populations of salmonids (Sedell and Swanson 1984).

Forest Management & Sediment Effects

The literature on the erosion and sediment impacts of forest operations is quite extensive, though much of it comes out of the Pacific Northwest. Most of the California research on private forestland has focused on the north coastal redwood region, particularly in the Caspar Creek Experimental Watershed of the Jackson Demonstration State Forest in Mendocino County (e.g., Zeimer 1998; Rice et al. 2004) and in the Redwood Creek watershed as part of Redwood National Park related research (e.g., Best et al. 1995; Madej 2005).

Historic Logging Practices

Certain mid-20th century logging practices were clearly identified as harming water quality. Clearcut logging, of large portions of a watershed down to the edge of streams, and the logging road system, were noted as a major source of sediment in earlier studies in Oregon (Brown and Krygier 1971; Swanson and Dyrness 1975) and California (Cordone and Kelly 1961; Burns 1972). Cordone and Kelley in 1961 perceived that the bulk of stream damage was caused by carelessness and could be prevented "with little additional expense", they thought at the time. Over thirty years ago, Burns (1972) examined logging and road effects on juvenile anadromous salmonids in northern California streams, with all streams showing sediment increases following logging. Evidence was also gathered to show that good logging practices could reduce sedimentation problems in the western region (Haupt and Kidd 1965; Brown 1983).

Sediment and other impacts led to a series of increasingly protective measures for forestry operations on public and private lands in the U.S. In 1973, California's State Water Resources Control Board recommended improved timber harvest and road construction methods at the time of the passage of the State Forest Practice Act but prior to the adoption of the Forest Practice Rules in 1975 by the Board of Forestry (SWRCB 1973). Tighter stream protection rules were later required by the State, as described under Riparian Buffers below. Berbach (2001) describes the evolution of such measures for private forestland in California.

Roads as a Major Source of Sediment

Logging roads have historically been the largest, or one of the largest, sources of forest management-related sediment (Trimple and Sartz 1957; Megahan and Kidd 1972; Burns 1972; Anderson et al. 1976; Adams & Ringer 1994). One study found that roads can contribute more sediment per unit area than that from all other forestry activities, including log skidding and yarding (Gibbons and Salo 1973). Roads can affect streams directly through the acceleration of erosion and sediment loadings, the alteration of channel morphology, and changes in the runoff characteristics of watersheds. Sedimentation was often greatest when major storm events occurred immediately after construction, while surface erosion usually declined over time with revegetation of roadsides and natural stabilization (Beschta 1978). A long-term study in Caspar Creek in Mendocino County found similar results, but also a lag of sediment transport as material only moved during periods of high runoff and streamflow (Krammes and Burns 1973). In landslide prone terrain, road-related erosion could continue unless certain design, construction and maintenance practices were carried out, or high erosion hazard areas were avoided. Much of the research of logging road effects was on roads that had been constructed in the 1950's, 60's and 70's, before improved road location and design to minimize potential slope stability and erosion problems were applied. By the early 1990s, steps were being taken to minimize the negative effects of roads on streams through both construction and maintenance practices (Furniss et al. 1991; Weaver and Hagans 1994).

Channel crossings, within the riparian area, are often the primary cause of water quality problems associated with roads and the resultant ecological impacts (USFS 1976; Erman et al. 1977; Forman and Alexander 1998). Debris blockages of undersized culverts and flood flows can cause the failure of the logging road stream crossing, delivering large volumes of crossing-fill sediment directly into the channel. In a long-term erosion evaluation of the Redwood Creek watershed, researchers found significant gullying problems due to logging roads, particularly due to diversions at plugged stream culverts or ditch relief culverts (Hagans et al. 1986). These diversions created complex channel networks and increased downslope drainage density, yet 80% of all gully erosion was avoidable, the authors stated, through minor changes in road construction techniques.

Heavily used, unsurfaced logging roads also can produce significantly more sediment and turbidity than abandoned roads, with one study in Washington State showing a 130 fold increase (Reid and Dunne 1984). Road surface sediment can drain into roadside ditches and then into streams, delivering fine sediment detectable by turbidity sampling below the road (Bilby et al. 1989). The problem can be effectively minimized, the authors noted, by draining the ditch onto the forest floor in small quantities to infiltrate, by using better road construction and surfacing material, and by leaving woody debris within the stream. Ketcheson and Megahan (1996) evaluated the potential sediment filtration effectiveness of the riparian zone below road fills and culverts in granitic terrain, finding that road sediment travel distance increased with increasing volume of eroded material.

In some locations, road placement within the stream riparian zone can encroach on the floodplain and channel and force streamflows to the opposite bank, potentially destabilizing the hillslope and causing increased landsliding. Roads located within the landslide-prone inner valley gorge, where very steep slopes are adjacent to streams, are at high risk of frequent or iterative failure (Furniss et al. 1991). A study in the Klamath Mountains of northwestern California noted this relationship (Wolfe 1982). If roads must be located in a valley bottom, a buffer strip of natural vegetation between the road and the stream is recommended (Furniss et al. 1991).

High quality roads and better maintenance are likely to reduce the amount of material supplied to channels from hillslopes, reduce the amount of sediment mobilized along low order streams, and reduce the sediment delivery rate to high order streams (Furniss et al. 1991; Slaymaker 2000). In the past decade, methods to inventory logging road drainages for their potential to deliver sediment have become more standardized (Flanagan et al. 1998; CDFG 2006). Road erosion studies need to be examined in the context of geology and soil types, such as the highly erosive granitics (e.g., Megahan and Kidd 1972).

Some studies have compared the effects of old to new forest practices. Cafferata and Spittler (1998) compared the effects of logging in the 1970s to the 1990s in the Caspar Creek watershed in Mendocino County found that “legacy” roads continue to be significant sources of sediment deca des after construction.   Recent Total Maximum Daily Load (TMDL) studies in north coastal California watersheds assessed sediment sources over multiple decades, but the analyses did not distinguish whether logging road-related sediment originated from roads constructed before or after the Forest Practice Act in 1973 (Kramer et al. 2001). However, timber operations under the “modern” Forest Practice Rules produced an estimated erosion rate one-tenth that of pre-1976 practices on a tributary of Redwood Creek (Best et al. 1995). Rice (1999) cautioned about direct comparisons of different studies with different objectives, but concluded that road-related erosion in Redwood Creek was significantly reduced due to improved road standards (e.g., better sizing and placement of culverts). In 1999, the Scientific Review Panel on California Forest Practice Rules and Salmonid Habitat made nine recommendations on road construction and maintenance, including the removal of legacy roads within the riparian zone (Ligon et al. 1999).

Riparian Buffers in Forest Management

The concept of using vegetation and/or obstructions to form buffer strips to minimize or retard downslope sediment movement has been applied to agricultural and forestry operations for many years (Broderson 1973; USEPA 1975). Buffer strips are defined as riparian lands maintained immediately adjacent to streams or lakes to protect water quality, fish habitat, and other resources (Belt et al. 1992). Limiting mechanical harvesting activities within streamside zones is appropriate to protect their vulnerability to compaction and physical disturbance, due to high moisture and low soil strength factors (Dwire et al. 2006).

The U.S. Forest Service adopted the Streamside Management Zone (SMZ) in the 1970s as a Best Management Practice (BMP), for closely managed harvesting, to act as an effective filter and absorptive zone for sediment, to protect channel and streambanks, and other benefits (USFS 1979). Each National Forest's Forest Plan also has Standards and Guidelines for the protection of riparian areas, including specific BMPs (Belt et al. 1992). In 1975, the California Board of Forestry first adopted the Stream and Lake Protection Zone (SLPZs) as part of the state's Forest Practice Rules (FPRs); these riparian zone protections were later expanded by the Watercourse and Lake Protection Zone (WLPZ) in 1983, 1991 and 2000 (Berbach 2001). While the benefits of such riparian protections are not challenged, the extent of the buffer strips (i.e., upslope and upstream) to balance ecological, water quality, and management needs continues to be debated (Dwire et al. 2006).

Direct physical disturbance of stream channels and soils within the riparian area by timber harvest activities can increase sediment discharge (Everest et al. 1987). In a 1975 California field study, physical damage to streambanks during logging was caused by equipment operating through streams, by yarding and skidding timber through channels, and by removal of streamside vegetation. Failed road crossings deposited sediment into the streams, reducing the diversity of the aquatic invertebrate community (Erman et al. 1977). Grant (1988) identified a method, primarily through aerial photograph analysis, to detect possible downstream changes in riparian areas due to upstream forest management activities.

More recent studies have looked at the design of forest riparian buffer strips to protect water quality. The authors of one literature summary stated, "we cannot overemphasize the importance of maintaining the integrity of the riparian zone during harvest operations" in relation to erosion and sedimentation processes (Chamberlin et al. 1991). The use of riparian buffers and BMPs has generally decreased the negative effects of forest harvest activities on surface water quality (Belt et al. 1992; Norris 1993). However, even an intact riparian buffer strip cannot prevent significant amounts of hillslope sediment from entering a stream via overland flow (due to infiltration and saturation excess in severely disturbed soil) or from debris slides originating outside the riparian zone (Belt and O'Laughlin 1994; O'Laughlin & Belt 1995).

One area of research receiving more attention is the riparian zone within headwater and low order streams (e.g., first and second). Sediment deposited in low order streams (which tend to be Class III under FPR rules) may be delivered to high order streams (e.g., third and fourth) that are usually Class I and II. Moore (2005) summarizes the latest results of this headwater research in the Pacific Northwest. MacDonald and Coe (2007) have recently investigated the influence of headwater streams on downstream reaches in forested areas, including the connectivity and effects of sediment. These recent research papers and others on this topic need to be thoroughly examined before consensus can be reached on the conclusions.

In recent years, the use of riparian buffer zones as a management tool has increased. For public lands in the Pacific Northwest, Riparian Reserves (RR) were set aside under the Northwest Forest Plan in 1994, where silvicultural activities were not allowed for multiple reasons, including water quality (Thomas 2004). For private forest lands, stream protection zones have increased in importance and restrictions in the past decade due to the federal and state listings of anadromous salmonid species as threatened or endangered (Blinn and Kilgore 2001; Lee et al. 2004). The current WLPZ rules for California were tightened from the 1991 Rules to protect listed fish species under the "Threatened or Impaired" (T/I) Rules, adopted as Interim Rule Requirements by the BOF in 2000, based in part on the recommendations of the Scientific Review Panel (Ligon et al. 1999; Berbach 2001). Research is now needed on the effects of these newer riparian protection zones, with comparisons made to previously designated zones.

Recent Sediment Evaluations of Forest Practices

Evaluations of forest practices producing and delivering sediment, as a nonpoint pollution source, revealed that Best Management Practice (BMP) implementation was generally good across the U.S., but cases of noncompliance persisted (especially for road and skid trail BMPs (SWRCB 1987; Binkley and Brown 1993). The authors recommended compliance and effectiveness monitoring must therefore be an ongoing activity.

The Board of Forestry's Monitoring Study Group (MSG) has overseen two recent evaluations of the effectiveness of the Board's Forest Practice Rules (FPRs). The Hillslope Monitoring Program (Cafferata and Munn 2002) evaluated monitoring results from 1996 through 2001, while the Modified Completion Report (Brandow et al. 2006) continued analysis of data from 2001 through 2004. Both studies 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. The 2006 report concluded the following:

> In most cases, Watercourse and Lake Protection Zone (WLPZ) canopy and groundcover exceeded Forest Practice Rule (FPR) standards. 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…
>
> 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…
>
> 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…

Attention has recently focused on riparian management of low order streams by management agencies, the public, and scientists. Gaps in knowledge are still being identified for the Pacific region and the diversity of riparian management standards continue to be debated (Young 2000; Moore 2005).

What We Do Not Know or Do Not Yet Agree Upon:

- The need for buffer strips along low order (e.g., 1st, 2nd) streams to prevent or minimize the delivery of sediment to higher order streams during forestry operations.
- The amount of forest management that can be performed within a designated riparian buffer zone without accelerating sediment production and delivery.
- The sediment effects of the newer, riparian protection zones for forest management, with comparisons made to previously designated zones.
- The relevance of forest management research on sediment relationships in riparian zones in other western states to California, and the relevance of such research in California's north coastal redwood region to other region's of the state.

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Primer on Water Riparian Exchanges Related to Forest Management in the Western U.S.

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of the
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PRIMER : WATER RIPARIAN EXCHANGE FUNCTION

Salmonid Life-Cycle Needs Related to Water

Important habitat characteristics for salmonids in streams include minimum streamflow, obstructions to flow that create debris dams and have other effects on stream shape, and gravel necessary for spawning (Botkin and others 1994). The riparian zone along streams influences all of these factors. Streamflow, and the sediment this flow transports, interact with large wood, boulders, and bedrock outcrops to produce physical characteristics of streams required by fish, including side channels in floodplains, and pools and riffles in small main-stream channels.

The amount, velocity, and depth of water required by salmonids varies depending on the life stage. Bjornn and Reiser (1991) present a comprehensive review of this topic for North American salmonids. Migrating fish require water depths that allow upstream passage [e.g., minimum water depths of 0.09 m to 0.12 m for chum salmon, depending on substrate particle size (Sautner and others 1984)]. Streamflow affects the amount of spawning habitat available by regulating the area covered by water and the velocities and depths of water over gravel beds [e.g., velocities ranging from 0.3 to 3.0 m/s and a minimum depth of 0.18 m (Thompson 1972)]. Stream discharge, followed by water velocity, are the most important factors in determining the amount of suitable living space for rearing salmonids [e.g., velocities < 10 cm/s for newly emerged salmon and trout fry (Everest and Chapman 1972); depths ranging from water barely deep enough to cover juveniles to > 1 m (Bjornn and Reiser 1991)].[1] In general, salmonid carrying capacity increases as streamflow increases up to a point, and then levels off or declines if velocity becomes excessive (Bjornn and Reiser 1991, Murphy 1995).

Minimum streamflows in both summer and late fall are critical for juvenile rearing and successful spawning for salmonids, respectively. Murphy (1995) reported that minimum streamflow in summer limits salmonid carrying capacity on a broad scale. For example, total commercial catch of coho salmon off of Washington and Oregon was found to be directly related to the amount of summer streamflow when the juveniles were in streams two years before (Smoker 1955, Mathews and Olson 1980). Botkin and others (1994) found that streamflow, especially the minimum flow in November three and four years prior to adult returns, accounted for most of the variation in adult spring Chinook adult salmon returning to spawn in the Rogue River in Oregon.

Effects of Forest Management on Peak Flows, Low Flows, and Water Yield

The effects of forest management activities on streamflow have been studied since the early 1900’s and are summarized in Ziemer and Lisle (1998) and Moore and Wondzell (2005). Changes in peak flows, low flows, and water yield resulting from forest removal are very complex. The magnitude of change to both water yield and peak flows depends on the amount and location of the harvest, the stand age and composition of the vegetation removed, soil and lithologic characteristics, topography, and climatic conditions. The persistence of the effect is largely determined by the rate and composition of vegetation re-occupying the disturbed site.

[1] Note that in an area with numerous deep pools and cool groundwater contribution, discharge and velocity can be very low, compared to an area without pools.

In terms of aquatic habitat, key hydrologic concerns relate to changes in summer low flows, and in peak flows and their effects on channel stability and sediment transport (Moore and Wondzell 2005). In a comprehensive review of forestry impacts on aquatic habitats, Botkin and others (1994) concluded that there is no evidence or reason to believe that changes in flow due to forest harvest would be deleterious to fish. They state that increases in flood peaks would be expected to cause a slight increase in channel mobility and an increase in the transport of bed sediment (factors that relate to spawning and rearing habitat), but there do not appear to be field studies relating changes in flooding to degradation of fish habitat.

Peak Flow Changes

Ziemer and Lisle (1998) provide a comprehensive description of how changes in peak flows associated with forest management vary with watershed size, type of precipitation, season, and flood magnitude. In general, the effects of forest practices are more pronounced and easier to detect in small watersheds, greater in areas where rain-on-snow events occur, greater in the fall months, and greater for frequent runoff events. More detailed information on these principles and specific examples and are provided in the paragraphs that follow.

Substantial (e.g., $\geq$30-50% clearcut) harvesting in small to medium-sized watersheds[2] over short time periods is required to noticeably increase small to medium recurrence-interval peak flows associated with timber harvesting. Limited harvesting in riparian areas alone cannot affect flood frequency or magnitude.

Ziemer (1998) reported a 9 percent increase in 2-year peak flows following clearcutting approximately 50 percent of the North Fork Caspar Creek watershed (5 km^2), located near Fort Bragg, California.[3] Ziemer and Lisle (1998) state that: "There is little evidence that forest practices significantly affect large floods produced by rain. However, it is possible that clearcutting exacerbates some rain-on-snow floods, although the magnitude of such an effect is highly variable and difficult to measure or detect."[4] They also explain that the greater the size of the flood or basin being investigated, the less likely that there will be any detectable changes caused by forest practices.

Specific peak flow studies in the Pacific Northwest confirm these conclusions. Thomas and Megahan (1998) found that treatment effects decreased as flow event size increased and were

[2] Ziemer and Lisle (1998) define small basins as having drainage areas $\leq$ 1 km^2 (~250 ac) and large basins as >100 km^2 (~25,000 ac). Medium-sized basins can be considered be on the order of 10 km^2 (~2,500 ac).
[3] The WLPZ Forest Practice Rules tested in the North Fork Caspar Creek watershed were those in effect from 1983 to 1991 (e.g., Class I buffer strips of 200 ft for slopes >70%). In 1991, maximum Class I WLPZs were reduced to 150 feet for slopes >50%.
[4] Snow accumulation tends to be higher in openings than under forest canopies, with cut blocks typically accumulating about 30 percent to 50 percent more snow. Removal of the forest canopy exposes the snow surface to greater incident solar radiation as well as to higher wind speeds, which can increase sensible and latent heat inputs. During mid-winter rain-on-snow events, melt rates are typically governed by sensible heat transfer from the relatively warm air, condensation of water vapor onto the snowpack, and in some cases by the sensible heat of rainfall. Under these conditions, snowmelt may significantly augment rainfall, increasing the magnitude of flood peaks (Moore and Wondzell 2005).

not detectable for flows with 2-year return intervals or greater for small treated watersheds that were either clearcut or patchcut with roads in the H.J. Andrews Experimental Forest, located in the western Cascade Mountains of Oregon in the rain-on-snow zone. Beschta and others (2000) analyzed the same data and concluded that treatment effects were unlikely for peak flows with recurrence intervals of approximately 5 years or greater, and that a relationship could not be found between forest harvesting and peak discharge in the large basins.

In a broad summary of the literature, Moore and Wondzell (2005) reported that peak flows increased following forest harvesting in most studies in coastal catchments, with increases ranging from 13 percent to over 40 percent based on the original analyses. They also found that in coastal watersheds, the magnitude of forest practice-related peak-flow increases declined with increasing event magnitude in most cases, with the greatest increases typically associated with autumn rain events on relatively dry catchments. Moore and Wondzell (2005) state that peak flow change does not appear to be related in any simple way to the percentage of basin area cut or basal area removed, and that estimates of post-treatment recovery rates varied among studies.

Timber harvesting affects the amount of interception loss that takes place in forested watersheds. This, in turn, may influence changes in winter peak flows. Interception loss has been reported as approximately 20% in coastal California forests (Reid and Lewis, in press), and more generally as about 10 to 30 percent of total rainfall, depending on canopy characteristics and climatic conditions (Moore and Wondzell 2005). Differences in interception loss between logged and unlogged areas are likely to explain the majority of the observed increases in larger winter peak flows, when transpiration is at its annual minimum (Ziemer 1998, Lewis and others 2001).

Small increases in peak flows (≤ 10%) for 2-5 yr return interval events have been found to be relatively benign and have not been judged to be capable of substantially modifying the morphology of the stream channels (Ziemer 1998). This is due to the fact that the magnitude of peak flow changes is substantially less than the within-a-year and year-to-year variability in streamflows. The changes are within the normal range of variability of streamflows (Grant and others 1999).

In addition to harvesting effects, roads can have significant hydrologic impacts (Coe 2004). Several studies have shown that logging roads can intercept shallow subsurface flow and rapidly route it to the stream network, potentially leading to increased peak flows in headwater basins (Moore and Wondzell 2005), or possibly delayed peaks in larger watersheds due to desynchronization of peak flows from tributary basins. Pathways linking the road network to stream channels include roadside ditches draining directly to streams, and roadside ditches draining to culverts that feed water into incised gullies (Wemple and others 1996). Accelerated runoff at the road segment scale also results since haul roads have compacted surfaces with low permeability that generate overland flow in even moderate rainstorms (Coe 2004, Moore and Wondzell 2005).

At the basin scale, paired-watershed studies have not shown strong evidence to support road-induced increases in peak flows. Studies may have been hampered by insufficient pre-treatment calibration data, lack of treatment replication, and poor experimental control (i.e., road building and timber harvesting have often occurred simultaneously or in quick succession) (Thomas and Megahan 1998, Coe 2004). Modeling studies have shown that increases in peak flows due to roads were approximately equal to the effects from timber harvesting (i.e., canopy removal) in an experimental watershed in western Washington (Bowling and Lettenmaier 2001). The effect of both activities declined as the flow recurrence interval increased. Additionally, modeling studies suggest that roads can decrease baseflow during the critical summer months (Tague and Band 2001). However, much uncertainly still exists regarding the hydrologic effects of roads at the watershed scale (Coe 2004, Royer 2006). If there are impacts from road building on peak flows, these effects will be more pronounced and easier to detect in smaller basins (Ziemer and Lisle 1997).

Channel aggradation, or filling of the channel bed with sediment, can have a significant effect on flood height or flooding. Where aggradation is severe, it is more important for overbank flooding than changes in runoff due to logging operations (Lisle and others 2000). Widespread channel aggradation can occur in low gradient reaches of watersheds if the sediment production rate has been significantly accelerated above background rates by mass wasting and surface erosion and delivery processes. If this happens, similar magnitude peak flows to those which would have occurred earlier can cause more extensive over-bank flooding downstream because of reduced channel capacity. These flood events would be the consequence of rainfall/runoff/channel aggradation interactions, rather than rainfall/runoff interactions. The area flooded would be changed by the altered channel configuration, even if the amount of water remained the same.

Low Flow Changes

Forest removal in mountainous watersheds will increase low summer and early fall streamflows, as well as total water yield. Botkin and others (1994) reported that while total water flow in a stream is important to salmon, flow increases during summer and early fall that can augment streamflow at a critical season for juvenile rearing are more important than the changes in magnitude of total annual flow. Nearly all published reports on timber harvesting and resulting changes in summer low flows have shown that streamflow will either increase or remain unchanged in proportion to the amount of vegetation removed in the watershed. Harvested areas contain wetter soils than unlogged areas during periods of evapotranspiration, and hence higher groundwater levels and greater late-summer streamflow (Chamberlin and others 1991).

Studies have documented that the post-treatment recovery rates are highly variable depending on the severity of the treatment and the vegetation reoccupying the site, along with physiographic and climatic characteristics. Often increases are fairly short-lived, as regeneration begins to utilize surplus soil moisture and intercepts precipitation. After approximately 10-30 years, baseflow (and peak flow rates) have returned to normal or decreased below pre-harvest levels due to rapidly growing hardwoods that transpire more water than mature conifer trees (Murphy 1995, Moore and Wondzell, 2005). Long-term effects of logging on summer low flows likely depends primarily on species composition before and after harvest (Spence and others 1996, Moore and Wondzell 2005). In general, summer low flows are more sensitive to transpiration from riparian vegetation than from vegetation in the rest of the catchment (Moore and Wondzell 2005).

One example in California of documented water yield changes with both selective harvesting and clearcutting has taken place in the Caspar Creek watershed. The effects of selective logging on low flows were examined in the South Fork Caspar Creek watershed, where 64 percent of the second-growth stand volume of coast redwood and Douglas-fir was tractor logged from 1971 to 1973. Statistically significant summer low flow enhancements were evident for 7 years after logging. Minimum discharge increases averaged 38 percent after the selective harvesting and summer low flow volumes increases averaged 29% between 1972 and 1978 (Keppeler and Ziemer 1990, Rice and others 2004). The average length of the part of the low flow period when flow in the South Fork was less than 0.2 cfs was shortened by 43 days form 1972 to 1978, a 40% reduction. As in previous studies, most of the enhanced streamflow (average annual water yield) increase (approximately 90 percent) was realized during the rainy season while greater relative increases were witnessed during the summer low flow period (Keppeler 1986).

In the North Fork Caspar Creek watershed, approximately 50 percent of the watershed was clearcut harvested over about 7 years (1985 to January 1992).[5] Minimum discharge increases averaged 148 percent at the North Fork weir and flow enhancement persisted through hydrologic year 1997 with no recovery trend observed. The larger increases in the North Fork were probably due to wetter soils in the clearcut units, where little vegetation was present to use the additional moisture (Keppeler 1998). This data suggests that water yield effects will persist longer after clearcutting than when a similar timber volume is removed from a watershed with selective cutting. These differences in water yield recovery are probably related to changes in rainfall interception and evapotranspiration (Rice and others 2004). Enhanced summer low flows improve aquatic habitat in stream channels. In the Caspar Creek study, higher discharge levels increased habitat volumes and lengthened the flowing channel network along logged reaches during the summer and early fall months (Keppeler 1998).

The amount of increased water flow caused by forest management activities on summer low flows of large rivers is unknown, but Botkin and others (1994) state that based on studies extrapolated elsewhere, it is reasonable to assume that there would be a small positive effect. Given the importance of low flow increases to salmonid production, however, this change may be significant.

Annual Water Yield Changes

For total annual water-yield changes with forest management, most small-watershed studies have shown that in areas with significant precipitation (>100 cm/yr or ~40 in/yr), increases in streamflow are proportional to the reduction in forest cover. This is due to reduced losses from evapotranspiration by the trees in rain-dominated systems. Moore and Wondzell (2005) reported that in rain-dominated small catchments, clearcutting and patch-cutting increased yields by up to 6 mm for each percentage of basin harvested, while selective cutting increased yields by up to about 3 mm for each percentage of basal area removed. Increased water yield, however, is not uniformly distributed seasonally or throughout the rotation in the Pacific Northwest and California. Most of the annual increase occurs in the winter high-runoff season and during the wetter years, rather than during the summer season and drought years, when the

[5] Most of the clearcut harvesting (45.5%) took place from the spring of 1989 to January 1992 (Henry 1998).

additional water is needed (Ziemer 1987).[6] When vegetation reduction in a watershed is less than 20 percent, the expected water-yield increase is not measurable and the remaining trees will likely use as much water as the original stand (Bosch and Hewlett 1982).

Ziemer (1987) summarized the literature on this subject and reported that total water yield increases resulting from management in larger basins would be very small and not measurable. For example, Kattelmann and others (1983) estimated that for National Forest lands in Sierra Nevada watersheds, streamflow could only be increased one percent if multiple use/sustained yield guidelines were followed.

While there is some evidence in the arid southwestern United States that expansion of the phreatophytic riparian forests along rivers can contribute to streamflow declines (Thomas and Pool 2006), this does not appear to be a significant concern for most California watersheds with coniferous forests. For forest streams with narrow strips of riparian forest, riparian vegetation water use is usually a small portion of the overall water budget and probably has minor influence on annual water yield (Dr. Julie Stromberg, Arizona State University, Tempe, AZ, personal communication). As an example, complete felling of a strip of riparian vegetation in a small watershed at Coweeta Hydrologic Laboratory in North Carolina produced only very minor water yield increases (Hewlett and Hibbert 1961). With the limited harvesting in riparian zones that is allowed under the current forest practice rules in California, water-yield increases are not expected to be measurable.

Stormflow Generation

Water is transferred through riparian zones to channels by surface and subsurface flow. Shallow or lateral subsurface flow from hillslopes in steep forested watersheds in the western United States is widely recognized as a main contributor to stream flow generation; however, processes that control how and when hillslopes connect to streams are still being studied. Much of the difficulty in deciphering hillslope response in the stream is due to riparian zone modulation of these inputs (McGuire and McDonnell 2006).

A key concept for forested watersheds is that there is great temporal and spatial variability in how water is transferred to the channel. Streamflow in small forested headwater basins is usually generated from an expanding and contracting source area, often denoted as the variable source area, representing a fraction of the total basin area. The source of streamflow is usually that part of the basin nearest the perennial, intermittent, and ephemeral channels. Source areas (the hydrologically-active areas that contribute directly to stormflow) can vary from only one percent of the total basin area in small storms to 50 percent or more in very large storms. The percentage of saturated source area in a watershed is topographically sensitive (i.e., higher percentages occur with gentler slopes). The source areas within a watershed are very dynamic, expanding and contracting during events as the influx of precipitation progresses and then ends.

[6] This was observed in areas with rain-dominated winter periods, where summer storms are infrequent, as is found in California. In contrast, experimental studies on eastern U.S. watersheds (rain-dominated) have shown that peakflow and water yield increases dominate during the growing season months, since approximately half of the annual precipitation (in the form of higher-intensity convective storms) occurs from May through October.

Moisture redistribution continues following the rain event as slower lateral hillslope drainage supplies additional moisture to lower slope positions. Direct runoff and its source area increase due to channel expansion and slope water movement (Hewlett and Nutter 1970, Troendle 1985). Riparian areas associated with perennial and larger intermittent streams remain at or near saturation during the winter and hence are hydrologically active for transporting water by saturated overland flow and rapid subsurface flow via soil macropore and/or displacement flowpaths. Smaller intermittent and ephemeral streams are only active when the hydrologic network expands sufficiently to incorporate steeper-gradient channels. Ephemeral first order channels (typically Class III watercourses) flow only in response to direct rainfall, and, although they are part of the hydrologic network, they do not generally have riparian zones because hydrophilic (water-dependent or water-loving) plants are usually absent.

Water Exchange and Transfer within the Riparian/Floodplain Zone

Water is exchanged in riparian zones, and larger floodplains in several ways. Streams either gain water from inflow of groundwater (i.e., gaining stream—moving water from the riparian zone to the channel) or lose water by outflow to groundwater (i.e., losing stream—moving water from the channel into the riparian zone). Many streams do both, gaining in some reaches and losing in other reaches. Input of cold groundwater to the bottom of pools can be a key refugia feature for anadromous fishes in summer months (Osaki 1988).

The riparian zone has been conceptualized as a zone of transmission of ground water and hillslope water to the stream channel, as well as a direct router of precipitation and snowmelt when the riparian water table rises to the ground surface. Between storms, and even during small storms with dry antecedent conditions, subsurface inputs from adjacent hillslopes are often minimal. At these times, two-way exchanges of water between the stream and the riparian aquifer (hyporheic exchange) can become important (Moore and Wondzell 2005). The hyporheic zone is an area adjacent to the channel and below the floodplain (if present) where surface water and groundwater mix. Hyporheic zones link aquatic and terrestrial systems and serve as transition areas between surface water and groundwater systems. The hyporheic zone contains species common to both surface and subsurface systems, including a diverse community of macroinvertebrates. Few hyporheic studies have focused on unconstrained headwater streams in the Pacific Northwest. Consequently, the knowledge of hyporheic hydrology draws largely upon studies of larger, unconstrained streams.

Transpiration by vegetation in the riparian zone may extract groundwater from the riparian aquifer, producing a diurnal decrease in riparian water-table level and in streamflow, followed by recovery at night. Lundquist and Cayan (2002) report that diurnal cycles are evident in many western river records and that daily variation in streamflow is often 10-20% of the daily mean flow. Harvesting in the riparian zone can have a significant influence on riparian-zone hydrology through its effect on transpiration and water-table drawdown, potentially dampening or eliminating diurnal fluctuations in discharge and increasing low-flow discharges (Bren 1997). During extended periods of low flow, sections of small streams dry up wherever stream discharge is insufficient to both maintain continuous surface flow and satisfy water losses through the bed and banks. Stream drying may occur frequently in the headmost portions of the channel network, interrupting connectivity (Moore and Wondzell 2005). Also, forestry-related changes in channel morphology can substantially influence stream-aquifer interactions. Channel incision and simplification of channel morphology during large floods can substantially lower water tables and reduce exchange flows of water between the stream and the riparian aquifer (Wondzell and Swanson 1999).

Neither the effect of forest harvesting nor the effect of riparian buffer strips on hyporheic exchange flows has been directly examined in small headwater streams (Moore and Wondzell 2005). Moore and Wondzell (2005) hypothesize, however, that because channel morphology strongly controls hyporheic exchange, it is reasonable to assume that timber operations that lead to losses in channel complexity would reduce interactions between the stream and the riparian aquifer. In contrast, they state that efforts to minimize management impacts on channels, such as retention of riparian buffer strips, would help preserve stream-aquifer interactions. The ecological implications of decreased stream-aquifer interactions are stated as being difficult to predict with current knowledge. Moore and Wondzell (2005) report that Wondzell and Swanson's research (1996) suggests that such decreased interactions could lead to reduced nutrient cycling and reductions in stream productivity.

Forest Management Impacts on Water Transfer/Exchange Processes

Forest management activities include timber falling, timber yarding, road and crossing construction and use, site-preparation activities, herbicide applications, forest thinning, etc. Forest operations on a watershed-basis can influence surface and subsurface runoff in several ways. For example, decreased interception loss increases the amount of water infiltrating the soil, leading to higher water-table levels during storms (Moore and Wondzell 2005). Limited timber falling and tree removal in riparian zones alone will reduce interception loss and evapotranspiration, but will likely have little impact on streamflow (low flows, peak flows, or annual water yield), as discussed previously. In contrast, ground-based yarding activities in riparian zones and floodplains of larger river systems can adversely impact important overflow channels used by salmonids during high winter storm discharges. Additionally, riparian areas are vulnerable to both compaction and physical disturbance during ground harvesting operations due to areas of high soil moisture and low soil strength that are common within streamside zones. These concerns, along with riparian and aquatic habitat protection, provide a basis for limiting mechanical harvesting activities within riparian zones (Dwire and others 2006).

Considerably less is known about forest management impacts associated with small headwater channels when compared to larger fish bearing watercourses. Even though streamflow is sporadic in ephemeral first order channels (typically Class III watercourses), it is capable of transporting fine sediment down to fish-bearing streams. Rashin and others (2006) found that at several study sites in Washington, delivery of sediment to unbuffered tributaries resulted in adverse impacts to fish-bearing streams that were otherwise adequately protected by riparian buffers.

Field evidence from the Caspar Creek watershed suggested that unbuffered, headwater stream channels, particularly in burned areas, contributed significantly to suspended sediment loads. Lewis and others (2001) state that sediment increases in the North Fork Caspar Creek tributaries probably could have been reduced by avoiding activities that denuded or reshaped the banks of the small headwater channels. Much of the post-harvest increases in sediment yield in the North Fork were attributed to harvest-induced storm flow volume increases (Lewis and others 2001), suggesting that the hydrologic changes can be practically and not just statistically significant (Moore and Wondzell 2005). Therefore, there is evidence that increased flows in small headwater channels, as well as disturbance of these channels, can produce increased downstream sediment transport. Further discussion of sediment delivery is provided in the California State Board of Forestry and Fire Protection's Technical Advisory Committee (TAC) Sediment Primer.

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PC 4/20/07

GARCIA RIVER TREND AND EFFECTIVENESS MONITORING: SPAWNING GRAVEL QUALITY AND WINTER WATER CLARITY IN WATER YEARS 2004 AND 2005, MENDOCINO COUNTY, CALIFORNIA

5-15-2006
Final Report to California Department of Forestry and Fire Protection

By Ridge to River
Teri Jo Barber
Anna Birkas

For Mendocino County Resource Conservation District

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EXECUTIVE SUMMARY

Have changes in land management resulted in changes in spawning gravel quality or winter water clarity in the Garcia River, the first on the northern California coast to receive a TMDL? To begin to answer this question, turbidity and spawning gravel quality (particle size distribution and permeability) were measured in four Garcia River watershed tributaries in water years 2004 and 2005.

Fine sediments filling spawning gravels are deleterious to salmon gravel redds because interstitial gravel spaces become filled with particles that clog the semi-permeable nest. As permeability is impaired, the flow of dissolved oxygen to incubating eggs is reduced. This can result in nest fouling, as metabolic waste products accumulate. In 1999 and 2004, four Garcia River tributaries were sampled for spawning gravel particle size distributions and permeability. Bulk spawnable gravels were collected in 6 gallon samples, repeated eight times per tributary, and sieved into particle size classes. From these samples, the relative accumulation or depletion of finer particles (0.85 mm, 6.5 mm, and 8.0 mm size classes, collectively referred to as fines) between 1999 and 2004 was determined. Permeability was measured directly in spawnable gravels with portable backpack water pumps. Permeability rates and particle size distributions were used to predict salmonids survival to emergence. Gravel redds were significantly more permeable at Pardaloe and Mill Creeks, and measurably (but not significantly) more permeable at Inman Creek and the South Fork Garcia River in 2004 verses 1999. Fewer fines in spawnable gravels were found at Pardaloe Creek and South Fork in 2004 than in 1999, but these improvements were not statistically significant. Further study with a larger sample size is necessary to determine whether these measurable differences are due to natural variation or to true improvements in spawning gravel quality.

Winter water clarity was tested with automated turbidity sensors every ten minutes in water years 2004 and 2005. Water clarity is occluded by sediments suspended in the water column during storms (peak turbidity) and to a lesser extent between storms (chronic turbidity). Chronic turbidity can impair sight and feeding of over-wintering salmonids and causes gill abrasion. Subwatershed South Fork experienced the lowest total number of hours with turbidities exceeding a threshold of 30 NTUs during WY 04 and 05. Mill Creek had the lowest total hours with turbidities exceeding 60 and 150 NTU thresholds in WY 2004 while Pardaloe and Mill shared these prizes in WY 2005. South Fork had the lowest storm-related peak turbidity, closely followed by Pardaloe. Pardaloe had the lowest timber harvest rate and road density of the five watersheds studied over a 17-year period. South Fork had relatively high road density but had recently experienced basinwide restoration. In general, the other streams had no restoration work and had moderate to high timber harvest intensities. Excepting Mill Creek, these basins exhibited greater peak and chronic turbidity. Road restoration work and past timber harvest intensity may explain these differences. But natural variation and the many factors in a watershed that affect stream and gravel quality may dominate. Further research must be conducted before conclusions can be definitively drawn. However in evaluating whether funding spent on watershed restoration erosion controls is effective in improving instream conditions, this news is encouraging.

ACKNOWLEDGEMENTS

The extension of recording turbidimeter and pumping water sampler technologies to an increasing list of watersheds, agencies, citizenry, and hydrologists have provided an exciting arena in which to observe water quality changes. This is possible through the technology transfer afforded by the staff of USFS Redwood Sciences Laboratory in Arcata, California. We benefited greatly from expertise provided by Rand Eads, Jack Lewis, and Liz Keppeler. Kevin Fauchet, employed by Campbell Timberland Management, supplied assistance in de-mystifying the ISCO pumping sampler. Ben Monmonier provided technical assistance maintaining our turbidity monitoring stations. Bill Trush, Geoff Hales, and Darren Mierrou provided expertise in the gravel monitoring component of the study as well as some guidance with our turbidity analysis. Hydrologist Randy Klein corrected our raw data and provided a finished data set on which to run our turbidity analysis. He also guided our analysis with his 2003 document *Duration of Turbidity and Suspended Sediment Transport in Salmon Bearing Streams in Northcoastal California*. Thorough editorial assistance on our first draft was provided from California Department of Forestry and Fire Protection's John Munn and Hydrologist Pete Cafferata, both with CDF's Monitoring Study Group who funded the purchase of our instrumentation. Funding for turbidity station maintenance and data analysis were provided by California's Water Resources Control Board through their Regional Water Quality Control Board whose helpful contributors include Dave Hope, Bob Tancredo, and Kathleen Daily – to name a few.

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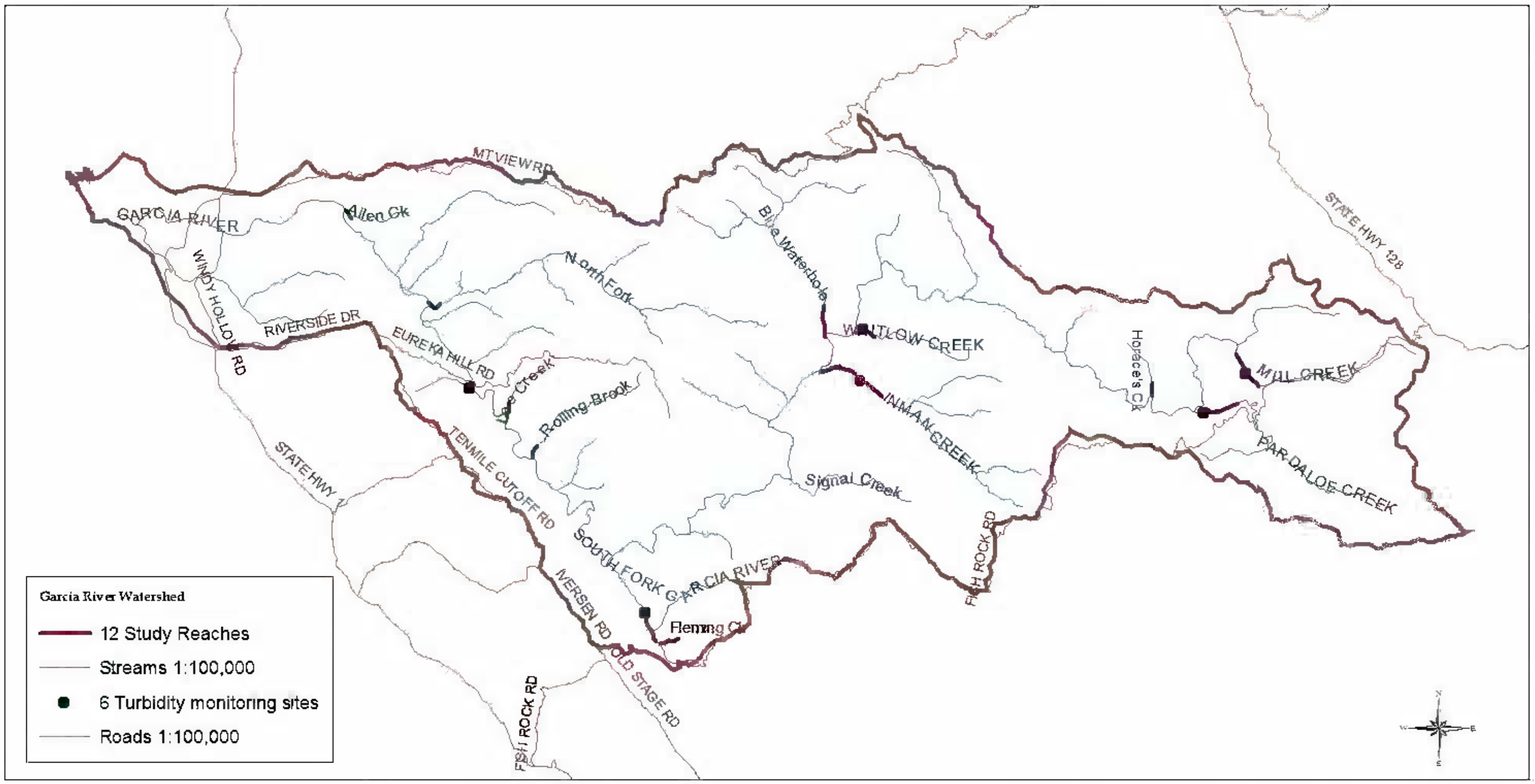


A project of:

The Mendocino County Resource Conservation District and Ridge to River, Westport, CA

December 13, 2005

Figure 1. Map of Garcia River Watershed with 6 MCRCD Turbidity Monitoring Sites and 12 MCRCD Study Reaches

0 1,650 3,300 6,600 9,900 13,200 Meters

0 4,850 9,700 19,400 29,100 38,800 Feet

1 inch equals 9,333 feet

This project funded by the State Water Resources Control Board and the CA Department of Forestry and Fire Protection

GARCIA RIVER TREND AND EFFECTIVENESS MONITORING: SPAWNING GRAVEL QUALITY AND WINTER WATER CLARITY IN WATER YEARS 2004 AND 2005, MENDOCINO COUNTY, CALIFORNIA *Ridge to River 5-15-2006*

INTRODUCTION

STATEMENT OF PURPOSE

Sediment delivery in the Garcia River has resulted in a loss of quantity and quality of salmonids habitat. Both California Department of Forestry and Fire Protection (CDF) and the North Coast Regional Water Quality Control Board (NCRWQCB) have responded to the degradation of North Coast watersheds. The NCRWQCB (1998) established a Total Maximum Daily Load (TMDL) for the Garcia River of 552 tons per square mile per year. This represents a 60% reduction in the estimated average annual sediment load of 1380 tons per square mile per year delivered to the Garcia River Watershed. The Garcia River was the first watershed in California to receive a TMDL allocation for excessive sedimentation. CDF responded by funding instream and upslope monitoring projects to determine whether implementation of current Forest Practice Rules are controlling observable erosion delivery to Class I fish-bearing streams. The primary goal of the Board of Forestry and Fire Protection's Monitoring Program is to assess the effectiveness of the Forest Practice Program in protecting water quality. The Monitoring Study Group, an advisory committee of the Board of Forestry and Fire Protection, concluded that instream monitoring should be used to assess current conditions and long- term channel trends. The Monitoring Study Group made the Garcia River watershed its first cooperative instream monitoring project in 1997.

Sediment delivery from roads to fish bearing streams is perhaps the single most important problem for salmon bearing watersheds for which there is a probable solution. In 1999, Trout Unlimited was awarded a California Department of Fish and Game (DFG) SB 271 grant to treat 8.75 miles of road for drainage improvements and erosion controls in the South Fork Garcia River watershed. Work was conducted on Mendocino Redwood Company (MRC) property with the goal of preventing over 28,000 cubic yards of sediment delivery to stream channels. Ten instream habitat sites were also funded for implementation in South Fork Garcia. This work was implemented in 2000 and 2001 (Craig Bell, Point Arena, personal communication, 2005). The purpose of upslope road-based erosion control is to minimize sediment delivery to fish-bearing streams. The time period between 2001 and 2003 provided two years following watershed restoration work to allow erosional adjustments at the sites undergoing drainage improvements and erosion control. Beginning in 2004, conditions in the stream channel were documented following the restorative treatments. This study will illuminate whether and to what extent channel conditions have changed following restorative treatments in the South Fork.

Both CDF and the NCRWQCB have contributed financially to the Garcia Monitoring Project. CDF's Hillslope Monitoring Program final report recommended that CDF support further instream monitoring studies to answer questions about the effectiveness of the Forest Practice Rules and to determine if current practices are complying with

Regional Water Quality Control Board Basin Plan standards designed to protect salmon spawning and rearing habitat (Cafferata and Munn, 2002).

This project will assist the NCRWQCB and CDF, as well as landowners in the Garcia River Watershed, to answer the following question: How and to what extent are sediment- related instream conditions improving for anadromous salmonids in the Garcia River? Gravel quality and instream water clarity are parameters known to be sensitive to watershed restoration. Other protocols previously measured in Garcia River tributaries in 1998-1999 are: channel cross-sections, longitudinal thalweg profiles, large wood loading, spawning surveys, summer water temperatures, and canopy closure (Maahs and Barber 2001). Spawning gravel particle size composition and permeability were also measured in 1999, and remeasured in 2004-2005. Instream winter turbidity monitoring coupled with summer gravel sampling were selected as indexes of whether conditions are improving toward numeric TMDL targets and whether watershed restoration is facilitating recovery in the Garcia River basin.

SCOPE OF PROJECT

The focus of this Garcia River Monitoring Project is Trend and Effectiveness Monitoring for both Garcia River Spawning Gravel Quality and Winter Water Clarity (Task 7). The other tasks in the contracted Scope of Work support Task 7, including administration, quality assurance plan, outreach, compilation of a GIS monitoring database for the Garcia River Watershed, and Bluewaterhole Sediment Control Treatments (cost-share under a separate contract in 2003).

Task 7.1: Measure bulk instream gravel composition, embeddedness, and permeability at 4 tributaries for one summer season (South Fork, Mill Creek, Pardaloe and Whitlow creeks). Bluewaterhole Creek has been a monitored sub basin for several years and it has benefited from watershed restoration efforts. Bluewaterhole Creek was thus initially included in the list of tributaries to be monitored. However landowner Stuart Bewley would not allow any further monitoring on his Bluewaterhole Creek property due to existing conflicts he has with NCRWQCB on another property. In the absence of another eligible stream with a cooperative landowner, we felt that examining mainstem turbidities would provide an opportunity to view tributary turbidity in the context of mainstem turbidity. We found a cooperative landowner on the mainstem Garcia, Hugh Brady, and set up a monitoring station there near Eureka Hill bridge instead of at Bluewaterhole Creek. However the streambank at Garcia mainstem underwent undermining lateral erosion toward our station and we felt a second winter risked our instrumentation. We found another cooperative landowner, The Conservation Fund, at Inman Creek tributary. We were compelled to record turbidity conditions there because Inman is targeted for watershed restoration in 2006-2007. In fall 2005 we removed the sampling boom from mainstem Garcia and recycled it into a turbidity monitoring station at Inman Creek. Whitlow Creek could not be monitored for permeability as it was dry on sampling. So instead we sampled permeability at Inman Creek where there was still adequate streamflow in late summer, 2004.

The objectives of task 7.1 were to:
a. Compare with baseline conditions measured in 1999
b. Predict survival-to-emergence based on gravel particle size composition and permeability
c. Correlate gravel particle size composition and embeddedness if possible
d. Have road-related sediment prevention measures improved spawning gravel?
- determine whether percent of in-gravel fines have been reduced to date
- determine whether gravel permeability has increased to date

The body of this report includes an entire section excerpted from the McBain and Trush report entitled "Assessing Salmonid Spawning Gravel Suitability Using Bulk Sediment and Permeability Sampling in the Garcia River Watershed, California." The appendix of our report includes the raw data files and the entire McBain and Trush report from 2005.

Task 7.2: Continuously monitor instream turbidity with recording turbidimeters at the above listed tributaries.

The objectives of task 7.2 were originally to:
a. Locate sediment sources by viewing spikes in turbidimeter readings to determine potential future treatments
b. Observe cause and effect relationships between hillslope activities, hydrologic triggers, and instream conditions by observing turbidity spikes and making cursory channel/hillslope inspections to determine the source of turbidity
c. Determine the total and consecutive days turbidity was sustained over 60 ntus as well as other biologically related turbidity thresholds

It was determined at our first meeting that Tasks 7.2a and 7.2b were unsupported by landowners and land managers in the tributaries of the Garcia River Watershed. Landowners were uncomfortable authorizing unlimited access across their lands to locate sediment sources and would not sign landowner access agreements under those conditions. Therefore, task 7.2a was omitted and task 7.2b was adjusted to replace *site specific* scale channel and hillslope inspections with a *remote* investigation of land management activities (a GIS based review of sub-basins in terms of timber harvest history and road density).

APPROACH

Two approaches were adopted for data analysis. The first approach was to interpret the gravel and instream water clarity data as it relates to salmonid habitat preferences based on criteria used by agencies and found in the literature (Tasks 7.1b and 7.2c). The second approach was to utilize the Garcia River mainstem water clarity (turbidity) data from water year 2003-2004 and gravel particle size distribution and permeability measured in 1999 as background conditions from which to determine:

(1) whether any measurable changes have been recorded.
(2) whether any of these measurable changes might be attributed to land management disturbances or restoration work.

Improvements in water clarity would be indicated where

- chronic exposures to threshold turbidities (hours greater than 30, 60, 150 ntu) were found to be smaller in restored basins or
- in basins having only low levels of timber harvest.

Improvement in gravel quality for anadromous salmonids would be indicated where

- the percent of spawning gravels in the finer categories were smaller in restored basins or in basins having only low levels of timber harvest, or
- if permeability of spawning gravels was more rapid in basins having had restoration or experienced only low levels of timber harvest.

Lack of pre-treatment turbidity data in watersheds with road improvement work prevents conclusions to be drawn directly regarding the effects of roadwork on observed changes. However any measurable improvements found in watersheds having restorative treatments was to be noted.

METHODS

Sampling Locations are Representative of the Garcia River Watershed

The Garcia River Instream Monitoring Plan identified optimal, representative monitoring tributaries of the 2nd and 3rd Strahler stream order (Euphrat et al. 1997). In 1998 this list was refined prior to sampling and based on obtaining written landowner agreements allowing access to the contractors completing the work. Our 2003-05 project utilized a subset of these monitoring study reaches.

Sampling Methods

Spawning gravel quality methods used in this study are well-described in McBain and Trush (1999). Sampling locations were selected using the 1999 procedures, which specified using 8 pool tail crests within the monitoring study reach of each tributary. There was no attempt to relocate the same pool tail crests used in the 1999 study, but the same protocols were followed. Permeability was sampled at 3-5 positions within the pool tail, including the location where bulk gravel was extracted. Gravel was sampled from spawnable (but unspawned upon) pool tail crests located in the monitoring study reaches within Mill, Pardaloe, South Fork, and Whitlow Creek. Gravel was extracted using a toothed cylinder which was pushed into the substrate. All gravel and sediment materials were then removed from the cylinder and transferred into 5-gallon buckets, which were transported to Graham Mathews and Associates' laboratory in Arcata, California for drying and particle size analysis.


Teri Jo Barber measuring permeability in a Garcia River tributary pool tail

Permeability of streambed materials was measured onsite with a backpack sampling apparatus developed by McBain and Trush. This included a notched standpipe driven into the gravel substrate. A long, flexible tube was then inserted into the standpipe and attached to a 12-volt pump. Stream water was pumped from the spawnable gravel (at approximately the 6-8" depth) into a graduated cylinder, where the rate of flow was measured and recorded in centimeters of water pumped in a given time period. Monitoring study reaches are displayed in Figure 1. Bulk spawning gravel was sampled for particle size distribution, permeability, and embeddedness in August and September, 2003.

Winter Water Clarity

The automated turbidity monitoring program is best described at the Redwood Sciences Laboratory website, www.fs.fed.us/psw/topics/water/tts or in the QAPP, written by Teri Jo Barber (see Appendix). The staff at Redwood Sciences Laboratory, particularly Liz Keppeler, Jack Lewis, and Rand Eads, provided technical assistance with installation of the monitoring stations. Kevin Fauchet, Campbell Timberland Management, helped with setting up the ISCO pumping sampler for deployment in the field.

USFS Redwood Sciences Laboratory Turbidity Threshold Sampling (TTS) software enabled a suite of instruments to record turbidity, stage height, water temperature, day #, and time at each "wake-up" scheduled at 10 minute intervals. At the Mill Creek station, this software also prompted the ISCO pumping sampler to read the turbidity and stage height recorded at the last wakeup to determine whether to pump a water sample from the creek.

The equipment used in the Garcia River watershed is similar to that used at the Caspar Creek experimental watershed operated by USFS-PSW and CDF. All five stations had pressure transducers used to monitor stream stage and a DTS-12 digital turbidity sensor with a lens wiper (similar to a windshield wiper used to clear the lens prior to a reading), both of which were wired to a Campbell Scientific CR510 datalogger. We accessed the dataloggers approximately every three weeks with a portable laptop computer, downloaded the data, and performed a variety of troubleshooting and maintenance operations when necessary. This provided automated, constant (every 10 minutes) turbidity records for two water years, 2003-04 and 2004-05 from approximately October through the early summer of each year. However, unlike the Caspar Creek Study, discharge was not determined, so estimates of storm and annual sediment loads were not possible, and there was only one ISCO 6712 automatic pumping sampler, which was placed at the Mill Creek station in early 2005.

LIST OF TASK PRODUCTS SUBMITTED

- 1.2 Progress Reports
- 1.5 Contract Summary Form
- 1.6 MBE/WBE Documentation
- 1.8 Project Survey Form
- 2.2 Permits
- 3.1 Monitoring Plan with Map of Sites
- 3.2 QAPP
- 4.1 Updated Landowner Contact List, notification letters
- 4.2 Pre-project Meeting Minutes
- 4.3 Copies of Landowner Access Agreements
- 4.4 MCRCD newsletter articles
- 4.5 Minutes of Post-project Meeting
- 5.1 CD with complete database
- 7.1 Summary of Instream Gravel Composition, embeddedness, and permeability
- 7.2 Summary of Instream Turbidity
- 8.2 Draft Final Report
- 8.3 Project Final Report

Ben Monmonier pulling bulk gravel sample from the streambed with steel toothed sampling cylinder in pool tail near permeability standpipe driven into same pool tail.

Table 1. Tributary Codes Utilized in this and Preceding Garcia River Monitoring

Tributary code	***Tributary name***
Tributary-1	Whitlow Creek
Tributary-4	Mill Creek
Tributary-5	Pardaloe Creek
Tributary-8	Inman Creek
Tributary-9	South Fork Garcia River

Copyright note: *The following section pertaining to instream gravel quality was reproduced with permission, from McBain and Trush's 2005 Assessing Salmonid Spawning Gravel Suitability Using Bulk Sediment and Permeability Sampling in the Garcia River Watershed, CA and contains numerous references to their previous 2001 report entitled Spawning Gravel Composition and Permeability within the Garcia River Watershed, CA of 1999 baseline gravel sampling available online at the CDF webpage: http://www.bof.fire.ca.gov/board/msg_supportedreports.html . Bulk gravel extraction from the streambed, embeddedness observations, and instream gravel permeability were measured in the field by Ben Monmonier and Teri Jo Barber of Ridge to River. Bulk gravel samples were dried and sieved into particle size classes by Graham Mathews and Associates, Arcata, California.*

2004 sampling results and comparison with 1999 samples

Particle Sizes

Table 2. Comparison between 1999 and 2004 bulk sediment sampling results for fractions finer than 0.85 mm and 8.0 mm. (McBain and Trush's Table 4).

	1999 Bulk Sample				2004 Bulk Sample			
Tributary	0.85 mm mean	0.85 mm standard deviation	8.0 mm mean	8.0 mm standard deviation	0.85 mm mean	0.85 mm standard deviation	8.0 mm mean	8.0 mm standard deviation
Tributary-1	9.7%	0.018	31.0%	0.074	11.2%	0.017	28.9%	0.049
Tributary-4	8.8%	0.021	35.2%	0.085	10.7%	0.051	37.0%	0.137
Tributary-5	8.4%	0.025	37.9%	0.068	7.5%	0.026	36.5%	0.123
Tributary-9	10.1%	0.019	29.8%	0.106	7.8%	0.020	34.5%	0.081

Analysis of the 1999 data focused on the 8.0 mm and 0.85 mm sizes (cumulative percent finer) as indices to characterize the variability of substrate composition within a single tributary. The intent of the 2004 analysis was to use the 1999 results as a basis to detect changes within a tributary over time, and to detect significant differences between tributaries (if they exist). The 1999 results generally showed a large variability of fine sediment percentages for both the 0.85 mm and 8.0 mm fractions within the tributary reaches sampled. The 2001 Report concluded that although certain samples showed similar results, few discernable overall patterns emerged from the data analysis (see the *Analysis of variation in particle size distribution* section in the 2001 Report).

The 2004 sampling collected 8 bulk samples per tributary, approximately the same as the number of samples analyzed in the 2001 Report. For the 2004 bulk sample statistical analysis, Dr. Baker compared the 2004 results to the 1999 results using parametric (two-tailed t-test) and nonparametric statistical testing (order-based statistics). These statistical tests compare the equality of sample means; that is, the null hypothesis is that the two sets of data were drawn from the same distribution. If test results show significance, the null hypothesis is rejected and sample means are not considered equal (the sample means between 1999 and 2004 have changed). Conversely, if significance is not determined, the null hypothesis is accepted and sample means are considered equal (the sample means between 1999 and 2004 for Tributary "X" have not changed). Both tests were performed on the cumulative fractions finer than 0.85 mm and 8.0 mm

The results of the t-test shows significance only for Tributary-1, fraction finer than 0.85 mm - that is, statistically, a change has occurred between means from the 2004 vs. 1999 samples. Therefore the remaining samples can be treated as if no change in their sample means has occurred. But does the lack of significance reflect a real absence of change? Considering the statistics, and acknowledging potential sample biases (e.g., size-distribution variability of individual samples within each tributary; sample mass collected based on Church et al. (1987) criteria; total number of samples collected on each tributary to reduce variability), it is still worth comparing these data to see if an overall trend of the 0.85 mm and 8.0 mm fractions is present? Table 2 presents 1999 and 2004 cumulative percentages finer than both 0.85 mm and 8.0 mm fractions, and summary statistics for each tributary. These results show that overall, some tributaries showed slight decreases in fine sediment (coarsening) where others showed slight increases in fine sediment (fining).

Permeability

2004 permeability data are summarized in Table 3, and includes the log-transformed 1999 permeability results used for comparison.

Table 3. 2004 and 1999 permeabilities and summary statistics from each tributary sample site. (McBain and Trush's Table 5)

2004 permeability

Tributary	Pool-tail site (geometric mean of replicate measurement median permeabilities) (cm/hr)								Geometric mean of all pool-tail sites
	1	2	3	4	5	6	7	8	
Tributary-4	18	1,917	5	22	3,676	3,414	4,815	77	252
Tributary-5	3,808	3,078	2,267	4,473	4,670	3,548	2,955	4,826	3,598
Tributary-8	8,250	3,863	2,455	3,016	9,952	3,647	2,111	1,398	3,551
Tributary-9	2,084	2,572	1,675	1,100	3,545	1,278	2,555	3,588	2,117

1999 permeability

Tributary	Pool-tail site (geometric mean of replicate measurement median permeabilities) (cm/hr)								Geometric mean of all pool-tail sites
	1	2	3	4	5	6	7	8	
Tributary-4	3,300	3,656	2,331	6,940	8,601		3,272	218	2,754
Tributary-5	3,771	853	963	2,237	1,403	539	289	908	1,040
Tributary-8	2,487	4,516	628	1,788		2,091		4,583	2,224
Tributary-9	1,411	95	862	1,113	4,196		3,130	4,937	1,354

Note: Tributary-6 sample 6, Tributary 8 samples 5 and 7, and Tributary-9 sample 6 were omitted from the analysis (see 2001 Report).

Similar to the bulk sediment sample analysis, the 2004 permeability data were compared with the 1999 permeability data using the same statistical tests (parametric and nonparametric) statistics. Results from the t-tests show the change at Tributary-4 was significant at the 95% confidence level ($p = 0.059$), and the change at Tributary-5 was significant at the 99% confidence level ($p = 0.0024$); changes at Tributary-8 and -9 were not significant at any reasonable confidence level (Table 4).

Table 4. Probability value (p) results of two-tailed t-tests for changes in log permeability between 1999 and 2004. The results of this test show that Tributary-4 and Tributary-5 are significant at the 95% and 99% confidence levels, respectively (i.e., the sample means have changed). (McBain and Trush's Table 6)

Site	***1999 mean permeability (geometric) (cm/hr)***	***2004 mean permeability (geometric) (cm/hr)***	***Probability (p) value***	***Significant difference at 95%?***	***Significant difference at 99%?***
Tributary-4	2,754	252	0.059	Yes	No
Tributary-5	1,040	3,598	0.0024	Yes	Yes
Tributary-8	2,224	3,551	0.96	No	No
Tributary-9	1,354	2,117	0.41	No	No

<u>Predicting Survival-to-Emergence from Gravel Particle Size Classes and Permeability</u>

The 2001 Report also used particle size and permeability to predict salmonid egg survival. This was done by estimating salmonid egg survival based on: 1) particle size analysis methods of Tappel and Bjornn (1983), and, 2) a preliminary correlation of permeability and salmonid survival-to-emergence using a relationship developed from studies by Tagart (1976) and by McCuddin (1977). The results showed moderate egg survival using the Tappel and Bjornn analysis (mean survival estimates for chinook salmon ranged from 54% to 82% in all ten tributaries sampled); however, the 95% confidence intervals for these estimates were broad (9% to 93%), and the report noted difficulty in drawing any conclusions of spawning habitat quality based on these predictions. Focusing only on the 4 tributaries sampled in 2004, the 2001 results predicted slightly better egg survival, with mean survival estimates ranging from 66% to 82%, and 95% confidence intervals ranging from 41% to 93%. Using the 2004 data, we performed the same analysis and found similar results (Table 5); survival predictions ranged from 60% to 76%, but 95% confidence intervals remained quite broad (6% to 89%).

Survival estimates using the permeability relationship based on Tagart and McCuddin also show similar results to those presented in the 2001 Report with the exception of Tributary-4, where zero survival is predicted due to a very low mean permeability (the mean permeability for Tributary-4 falls at the bottom of the Tagart and McCuddin regression). Excluding Tributary-4, the 2004 survival estimates are very similar to the 2001 results and have similar 95% confidence intervals (Table 6). Because the data used in the Tagart and McCuddin relationship are based on laboratory studies using two different salmonid species, survival estimates for salmonids in Garcia River tributaries should be considered an index only. Moreover, conclusions of egg survival based on these analyses must be tempered with the ability of spawning salmonids to clean fine sediment from spawning gravels during redd construction (Kondolf et al. 1993).

Table 5. Percent survival of salmonid eggs based on Tappel and Bjornn (1983) particle size analysis methods. (McBain and Trush's Table 7a)

	1999			2004		
Tributary	Mean estimated survival	Lower 95% confidence interval	Upper 95% confidence interval	Mean estimated survival	Lower 95% confidence interval	Upper 95% confidence interval
Tributary-1	74	56	87	76	63	85
Tributary-4	70	41	87	60	6	88
Tributary-5	66	44	81	70	40	89
Tributary-9	82	57	93	73	53	86

Table 6. Percent survival of salmonid eggs based on preliminary permeability relationship from Tagart (1976) and McCuddin (1977). McBain and Trush's Table 7b)

	1999			2004		
Tributary	Mean estimated survival	Lower 95% confidence interval	Upper 95% confidence interval	Mean estimated survival	Lower 95% confidence interval	Upper 95% confidence interval
Tributary-4	43	31	49	0		29
Tributary-5	28	18	33	38	35	41
Tributary-8	40	25	47	38	20	46
Tributary-9	37	27	43	31	24	35

Note: Tributary-4 0% mean survival is caused by low mean permeability (see Table 5); lower 95% confidence interval could not be calculated.

Although the number of samples collected in 2004 did not meet the criteria recommended in 1999, the results can still be used to describe the particle size distributions of the pool-tails sampled. If we assume the same statistical validity with respect to the sample population in each of the tributaries sampled, then the data collection quality between 1999 and 2004 studies was not improved (variability was not reduced). However, if we acknowledge the variability and accept this limitation, the data collected for this study are slightly better than the data collected in 1999 based solely on the sample size analyzed.

Table 7. Comparison of the number of bulk sediment samples analyzed in 1999 versus samples analyzed for this report. (McBain and Trush's Table 8)

Tributary	***Number of pool-tail bulk samples analyzed in 1999***	***Number of pool-tail bulk samples analyzed in 2004***
Tributary-4	6	8
Tributary-5	8	8
Tributary-8	7	8
Tributary-9	7	8

Embeddedness Sampling:

Embeddedness was not measured in 1999 but was included by MCRCD as part of the 2004 field data collection. The intent of the embeddedness sampling was to relate the embeddedness measurements to permeability or to the particle size distribution results (T. Barber, personal communication).

Although the permeability and bulk sediment sample results show large variability between individual sites, we analyzed embeddedness as a function of mean permeability and percent fine sediment finer than 8.0 mm for the embeddedness measured on Tributaries -1, -4, -5, and -9. No apparent trend exists for any of the data; a regression of the permeability data yields an R^2 value of 0.05, and a regression of the sediment data yields an R^2 value of 0.003. This result is somewhat expected: embeddedness is a surface feature, whereas bulk sampling and permeability measure subsurface sediments (see Section 2.1.3). Moreover, embeddedness measurements are subjective, subject to observer bias. More research is needed to determine whether embeddedness can be linked to biological criteria or to detect changes in land management activities (Sylte and Fischenich 2003).

Sediment Quality Relative to Garcia River TMDL

TMDL numeric targets for the Garcia River watershed have been established by the USEPA (1998). More specifically, the TMDL targets the percentage of fine sediments finer than 0.85 mm and 6.5 mm, and the numeric targets are <14% and <30%, respectively. These numeric targets represent the optimal conditions for salmonid reproductive success (USEPA 1988); percentages above these targets constitute an impaired condition. Similar to the 1999 results, the 2004 results indicate that the subsurface sediments finer than 0.85 mm are below the TMDL 14% numeric target; however, three of the four tributaries sampled exceed the 30% numeric target for sediments finer than 6.5 mm (Table 8). Recall that a 6.3 mm sieve screen was used instead of a 6.5 mm screen; the results shown in Table 8 were obtained from the particle size distribution curve.

Table 8. Fraction of bulk sediment samples finer than 6.5 mm and 0.85 mm; TMDL numeric targets are <30% and <14%, respectively. (McBain and Trush's Table 9)

Tributary	***Percent finer than 6.5 mm (TMDL target: < 30%)***	***Percent finer than 0.85 mm (TMDL target: < 14%)***
Tributary-1	26.1%	11.2%
Tributary 4	33.4%	10.7%
Tributary-5	32.3%	7.5%
Tributary-9	30.9%	7.8%

Using South Fork Garcia River Results as an Index for Watershed-Scale Change

Basin-wide erosion control efforts in the South Fork Garcia River watershed prompted the MCRCD to investigate whether any linkages could be established between restorative watershed efforts and improvements in permeability or spawning gravel composition. Has permeability or spawning gravel quality in the South Fork Garcia River (Tributary-9) improved? If so, can these changes be attributed to watershed-scale erosion control efforts?

To address this issue, we reviewed changes in South Fork Garcia River permeability and sediment composition from 1999 to 2004. To summarize:

- Mean permeability increased from 1,354 cm/hr to 2,117 cm/hr, but this change was not significant at any confidence level, i.e., the statistical testing could not demonstrate that a change in the means has occurred.
- The percentage of fine sediment < 8.0 mm increased from 29.8% to 34.5%, and the percentage of fine sediment < 0.85 mm decreased from 10.1% to 7.8%. Similar to the permeability results, these changes were not significant at any confidence level, i.e., the statistical testing could not demonstrate that a change in the means has occurred.

Because a change in the means for the above sampling results could not be demonstrated, using the above results to investigate a relationship between restorative watershed efforts and improvements in permeability or spawning gravel quality was not attempted.

More importantly, however, is understanding the context of the focus of such a comparison, i.e., establishing a cause-and-effect relationship between hillslope processes / land management and fluvial geomorphic processes using bulk sediment and permeability data. Bulk sediment sampling and permeability results can be useful to assess the suitability of the gravels for salmonid spawning habitat within a sampling reach. However, because the data collected for the 1999 and 2004 studies were collected within relatively short channel reaches, extrapolating these results to assess changes in sediment production rate at the watershed scale is not possible unless other factors are considered. For example, changes in sediment particle size distributions can result from a number of causes related to changes in the supply of watershed products. Monitoring efforts must therefore be broadened beyond the current sampling scheme of eight sample sites within single, approximately 1,000 ft reaches to determine how differences in substrate composition in the tributary reaches respond to changes in sediment production at the watershed scale. To do this, a larger-focus investigation would need to be performed, such as a sediment source analysis or a sediment budget. Such an investigation can help identify watershed-scale sedimentation processes (erosion, storage, transportation, deposition) responsible for delivering sediment to, and routing through, the channel. For example, a sediment budget would entail conducting sediment source inventories, calculating transport rates and delivery volumes, examining the interrelationships between transport processes and hillslope form to determine the sediment yield from locations within the basin (these can be tailored to specific monitoring reaches), and repeating the study at a later date to determine changes in the budget. This information, coupled with bulk sediment sampling and/or permeability data, would establish a much stronger linkage between changes in land management and tributary response than using bulk sediment sampling and/or permeability data alone.

In the absence of a sediment budget, sediment yield analysis, or similar watershed-scale monitoring, any changes in South Fork Garcia River substrate based on the 1999 and 2004 data collection (e.g., coarsening or fining) can only be considered as a *possible* result of watershed management efforts, such as upslope sediment reduction from erosion control measures. Presently, other factors such as the magnitude and frequency of storm events, or the number and activity of mass-wasting features in the basin cannot be ruled out as primary causes of change.

Summary and Conclusions

In assessing the relationship between substrate composition and permeability, the 2001 Report focused on sample size (the number of samples per tributary needed to characterize variability). The 2004 sampling focused on collecting samples to compare with the 1999 data, as well as using the results of the comparison to evaluate the effectiveness of the methods for detecting changes. In doing this, we identified additional sources of variability that can affect the sampling results, including: sampler bias (sampling differences between operators), and geomorphic variability (differences between tributaries, reaches, and/or sample sites). Sampler bias was minimized by McBain and Trush and MCRCD field training; however, geomorphic variability persisted, primarily in the form of sample size (the number of samples per tributary reach and collecting a representative sample volume per sample site). Both are given equal weight in terms of their importance, and future sampling efforts should try and meet the sampling criteria described in this report if the objective is to detect change from year to year. Specifically, future data collection should:

- Follow the bulk sediment sampling criteria suggested by Church et al. (1987): the maximum particle size in the sample (D_{max}) should not constitute more than 1% of the total sample mass for particles up to 128 mm (5.0 in), and not more than 5% of the total sample mass for particles greater than 128 mm.

- Follow the sample size recommendations presented in the 2001 Report: to strongly characterize the sampling variance, collect between 15 and 20 bulk sediment samples per tributary to best balance cost and precision. Alternatively, re-evaluate the minimum detectable difference to reduce the number of required samples.

The 2004 data provided a useful comparison of sample means to gauge changes in substrate composition and permeability. Most changes in sample means were not statistically significant. Independent of statistical significance, these results suggest no significant net change has occurred; overall, some of the tributary reaches showed a decrease in fine sediment, whereas others showed a slight increase in fine sediment.

Presently, research to determine a strong relationship between permeability and sediment quality (and to relate the sediment quality to salmonid spawning success) is still developmental. If future sampling is desired to investigate salmonid spawning gravel quality in the Garcia River watershed, the same data collection methods presented herein can be used; however, current literature should be reviewed before developing a study plan, to review advances in the permeability-substrate-spawning habitat relation and to determine how field data collection should be changed. This will aid in determining if permeability sampling is needed in combination with bulk sediment sampling, or if bulk sediment sampling alone will be sufficient to assess changes in spawning gravel quality. Moreover, if future monitoring objectives include a larger-scale understanding of watershed cause-and-effect relationships, monitoring should extend beyond the tributary-reach scale so that the processes responsible for generating changes in spawning gravel quality (e.g., sediment supply, magnitude and frequency of flood events) are identified.

INSTREAM WINTER TURBIDITY AND WATER CLARITY

GOAL

Our goal is to interpret turbidity and suspended sediment concentration data reported in this study to inform CDF, NCRWQCB, Garcia River landowners, and the public about winter water clarity in the tributaries and the mainstem of the Garcia River in terms related to salmonid health and land management.

TURBIDITY DEFINED

Turbidity measures the collective optical properties of a water sample that cause light to be scattered and absorbed rather than passing through a clear sample in straight lines (USGS, 1998). It is generally reported as NTU (Nephelometric Turbidity), FTU (Formazin Turbidity) or JTU (Jackson Turbidity Units). JTUs are no longer commonly used (USGS, 1998). Our study utilized recording Digital Turbidity Sensors (model DTS-12 by Forest Technology Systems) recording in NTUs except for the Salmon Forever lab turbidities from our 2003-04 grab samples, which were recorded in FTUs.

Turbidity measurements represent a gradual occlusion of the clarity of water as NTUs rise. Turbidity is readily observed in North Coast rivers and streams during or after winter storms, when the rivers appear opaquely blue, green or tan after prolonged rains. While turbidity can result from a wide variety of suspended materials, including algae or other organic substances in the water column, turbidity during peak flow events in northwestern coastal California streams is mostly attributable to suspended sediment particles, such as silt and clay (Madej 2005). Organic particles suspended in the water column yield similar turbidity levels but with correspondingly much lighter weight.

SUSPENDED SEDIMENT CONCENTRATION

Suspended Sediment Concentration (SSC) is a laboratory protocol for analysis on a physical water sample. The laboratory procedure for SSC is similar to Total Suspended Solids (TSS) but we duplicated laboratory procedures established for SSC by USFS Pacific Southwest Research Station's Redwood Sciences Laboratory as referenced on their website (www.fs.fed.us/psw/rsl). The basic procedure is to collect suspended particles by vacuum filtration from a water sample of known volume onto a pre-weighed glass-membrane filter. The dry weight of the filtered residue is expressed over the volume of the water sample as milligrams per liter.

INSTRUMENTATION

Sampling of winter water clarity was achieved mostly with the coupling of software and instrumentation that was developed and first implemented by United States Forest Service's Redwood Sciences Laboratory in Arcata, California. Garcia River watershed instrumentation included digital turbidity sensors with a lens wipers (much like a windshield wiper that cleans the lens prior to each measurement, DTS-12, from Forest Technology Systems), pressure transducers (Druck and Water pro) to measure stream stage height, and dataloggers (Campbell Scientific's CR510) for recording the data and downloading into a portable laptop computer. The station at Mill Creek had the benefit of an ISCO 6712 automatic pumping sampler in late winter-spring 2005. Water samples extracted by the ISCO were approximately 330 milliliters in volume. Grab samples from the other streams were generally extracted with a DH-48 Depth Integrated Sampler in similar volumes.

Turbidity sensors were installed and threaded through a long, hollow square aluminum casing (the "turbidity boom"), and suspended into the thalweg from a cable attached to trees on both streambanks (see plates and cross sections 1-5). The mainstem Garcia station required a bank-mounted boom as the river was too wide for a suspension cable installation. The Inman station was installed with the recycled mainstem bank mounted boom and instrumentation. Aluminum parts were fabricated by a Point Arena welder local to the Garcia River Watershed based on a USFS schematic.

CDF AND NCRWQCB COST SHARE

Most of the instrumentation used in this study was purchased under a contract between the MCRCD and CDF. The North Coast Regional Water Quality Control Board supplied an additional contract with the MCRCD to provide the two years of staffing required to set up the stations each winter, download data, maintain the stations, and take them down again. Contract amendments also allowed for data processing and analysis, landowner contacts, and GIS mapping services. In addition, the ISCO sampler and related equipment was provided separately by CDF.

CALIBRATION AND MAINTENANCE OF INSTRUMENTS

Turbidimeters

DTS-12 turbidimeters were calibrated by the manufacturer specifically for ranges expected in the study - the upper resolution of the instrument (1800 ntu, rather than a low range of 1-10 ntu, which would be more interesting to water purveyors for providing pure, clear water for human drinking water supplies). Because our chosen calibration was at the high end of the range, the lowest turbidity readings (clearest streamwater) sometimes resulted in negative numbers. While this may seem problematic, it is reasonable to expect an instrument calibrated for optimum accuracy at the upper limit of its range to be less accurate at the lowest limit of its range. The negative numbers reached a maximum of –2 NTU in the clearest conditions. We tested these waters in the field with a portable HACH 2100P turbidimeter and found the difference between the low negative values and those more accurate HACH readings were less than 2 ntu, which is within the expected range in variability (Jack Lewis, USFS-PSW, Arcata, personal communication, 2004). Each of the five turbidimeters we employed were re-calibrated at the manufacturer in the summer 2004, between the two monitored winter seasons.

Two of our turbidimeters failed due to wiper malfunction during the early summer prior to disconnection for the summer season. We suspect that small pieces of debris or rock lodged between the joint and wiper blade and caused the motor to stick in the on position until the battery failed. In each of the three wiper arm malfunctions, we decided not to disconnect the turbidimeter to send it to the manufacturer for repair during the monitoring season, but rather kept them connected to continue monitoring and waited till the end of the season for repairs. Randy Klein, RNSP, Arcata, suggested that for next season, to disconnect the wiper arm and let the turbidimeter sense the stream clarity without wiping the lens in between measurements. This would enable the station to keep running without depleting the station battery. However, this would definitely NOT be advisable in a mainstem application, where the density of suspended instream algae and periphyton increases dramatically. At the the Garcia mainstem station, algae colonized the turbidimeter lens so prolifically that the lens was visibly occluded and turbidity readings were artificially high as a result.

Pressure Transducers

The Druck pressure transducers are roughly calibrated at the manufacturer, but Redwood Sciences Laboratory instructions specify calibrating pressures sensed to water depth at local atmospheric pressures. A wet lab was set up at the Ridge to River office, where the pressure transducer for each station was installed in a vertical pipe then the water level was raised to different levels (feet of stage) and corresponding pressures were read and recorded to determine the relationship between water depth and pressure readings. Data points were entered into an Excel spreadsheet, and a line was run through the points with the corresponding regression equation providing the slope of the line and intercept. Regression equations were entered as variables in the Turbidity Threshold Sampling (TTS) software provided by RSL and customized for each of our five stations. Customized TTS programs were downloaded into dataloggers at each station.

ISCO Automatic Pumping Sampler

We installed one ISCO pumping sampler at Mill Creek and operated it during the spring of 2005 only. We did not receive any calibration services for the ISCO nor did we perform any ourselves. However, Kevin Fauchet,Campbell Timberland Management, was helpful in the initial wet lab setup of the ISCO and its initial connection with a turbidimeter, pressure transducer, and datalogger in a “dry-run” condition.

WATER CLARITY SAMPLING LOCATIONS

The subset of monitored sub basins from the 1999 baseline monitoring project was selected based on the following criteria:

- presently or historically salmon bearing
- landowners willing to authorize the project on their lands

The table below indicates the location, years of operation, landowner, and identifying codes present in the data (see appendix).

Table 9. Garcia River Water Clarity Monitoring Stations, Water Year, ID, Landowner

STATION NAME	WINTER WATER YEAR	IDENTIFICATION	LANDOWNER
Garcia Mainstem	03-04	GAR - 236	Hugh Brady
Inman Creek	04-05	INM – 249	The Conservation Fund
Whitlow Creek	03-04, 04-05	WHI – 240	The Conservation Fund
Mill Creek	03-04, 04-05	MIL – 241	Maillard Ranch
Pardaloe Creek	03-04, 04-05	PAR – 242	Maillard Ranch
South Fork	03-04, 04-05	SFK – 243	Mendocino Redwood Co

A Garmin V Geographic Positioning System (GPS) was used to establish latitude and longitude of the winter water clarity monitoring stations. Drainage areas refer to the total drainage area of the basin as opposed to the drainage area above the monitoring station. A map of these stations is provided as Figure 1. The table below provides the latitude and longitude of the monitoring stations followed by distances upstream from the Garcia confluence with the Pacific Ocean.

Table 10. GPS Location and Drainage Areas Surrounding Turbidity Monitoring Stations

STATION NAME	LATITUDE/LONGITUDE	DRAINAGE AREA (acres)
Garcia	N 38.90195 W. 123.60787	73,223
Whitlow	N. 38.89720 W. 123.36471	1,221
Inman	N. 38.90778 W. 123.48670	5,481
Mill	N. 38.90700 W. 123.35080	4,846
Pardaloe	N. 38.89720 W. 123.36471	5,626
S. Fork Garcia	N. 38.90188 W. 123.60793	5,598

Inman– Garcia River Confluence is approximately 28 miles from the mouth of the Garcia River. The monitoring site is approximately 700 feet above the confluence, within the mapped study reach (see figure 1).

South Fork – Garcia River Confluence is approximately 18.5 miles from the mouth of the Garcia River. The monitoring site is approximately 1 mile above the confluence within the mapped study reach (figure 1).

Whitlow – Garcia River Confluence is approximately 29.5 miles from the mouth of the Garcia River. The monitoring site is approximately 700 feet above the confluence within the mapped study reach (figure 1).

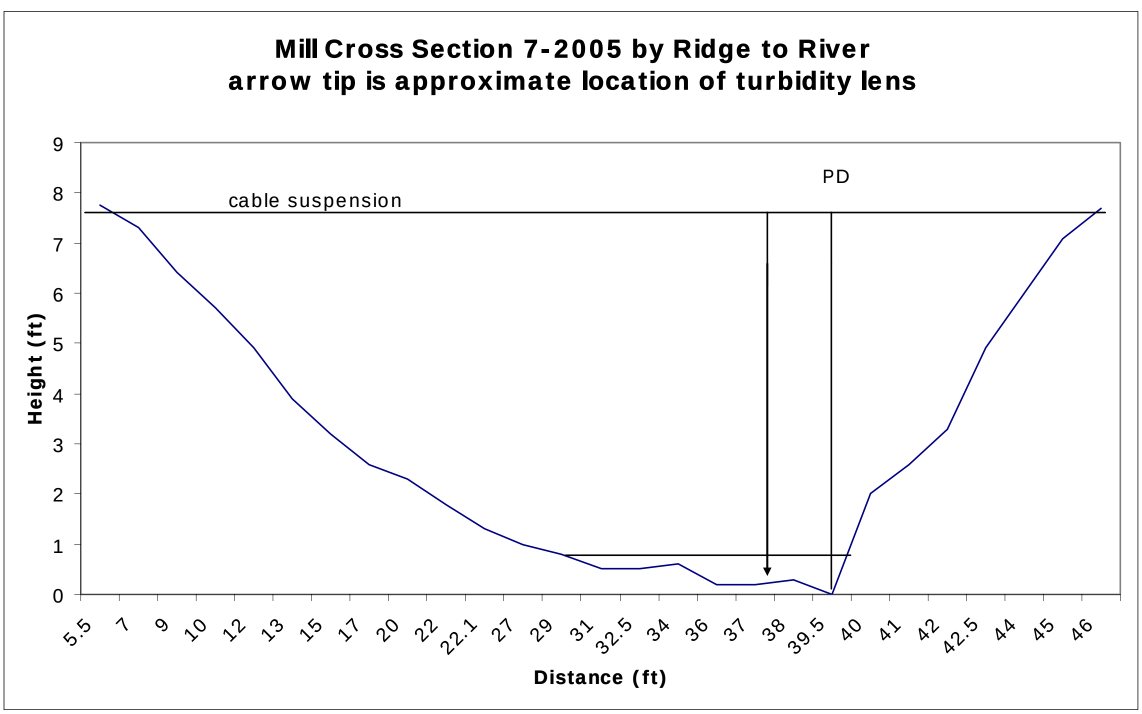
Pardaloe – Garcia River Confluence is approximately 43.5 miles from the mouth of the Garcia River. The monitoring site is approximately 300 feet above the confluence within the mapped study reach (figure 1).

Mill – Garcia River Confluence is approximately 39.5 miles from the mouth of the Garcia River. The monitoring site is approximately 2 miles above the confluence within the mapped study reach (figure 1).

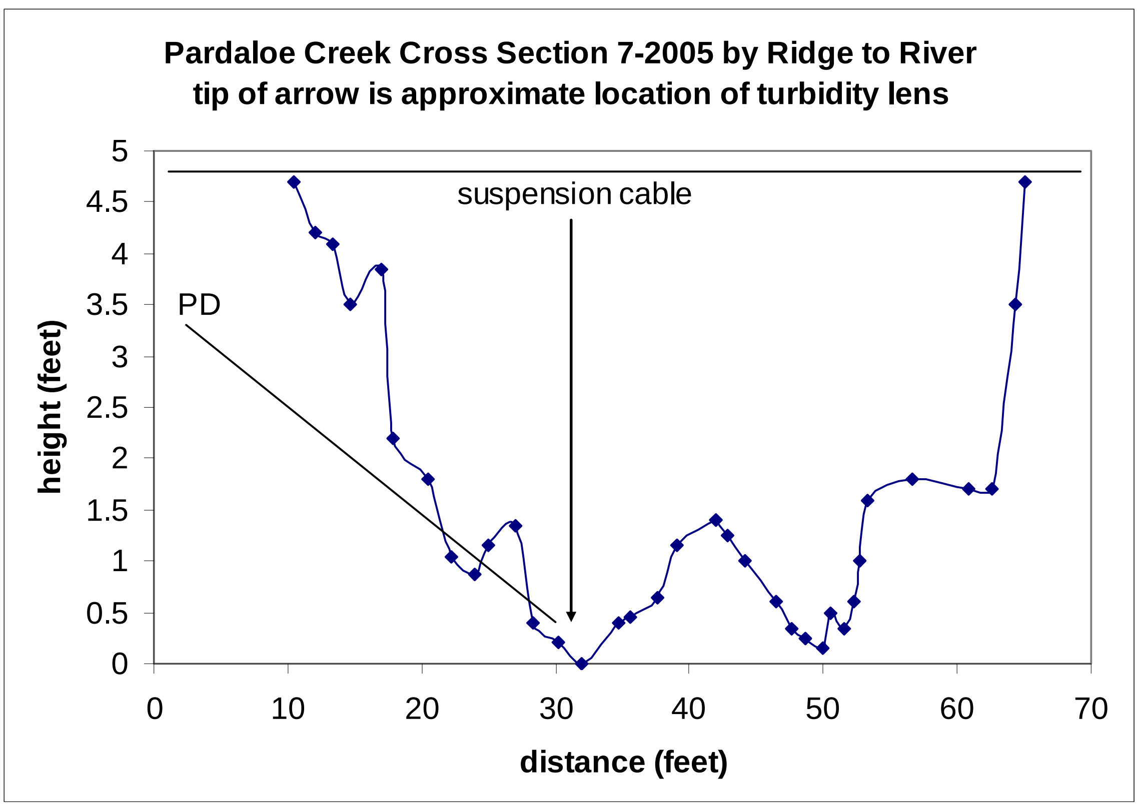
STATION CHANGES BETWEEN WATER YEARS 2004 AND 2005

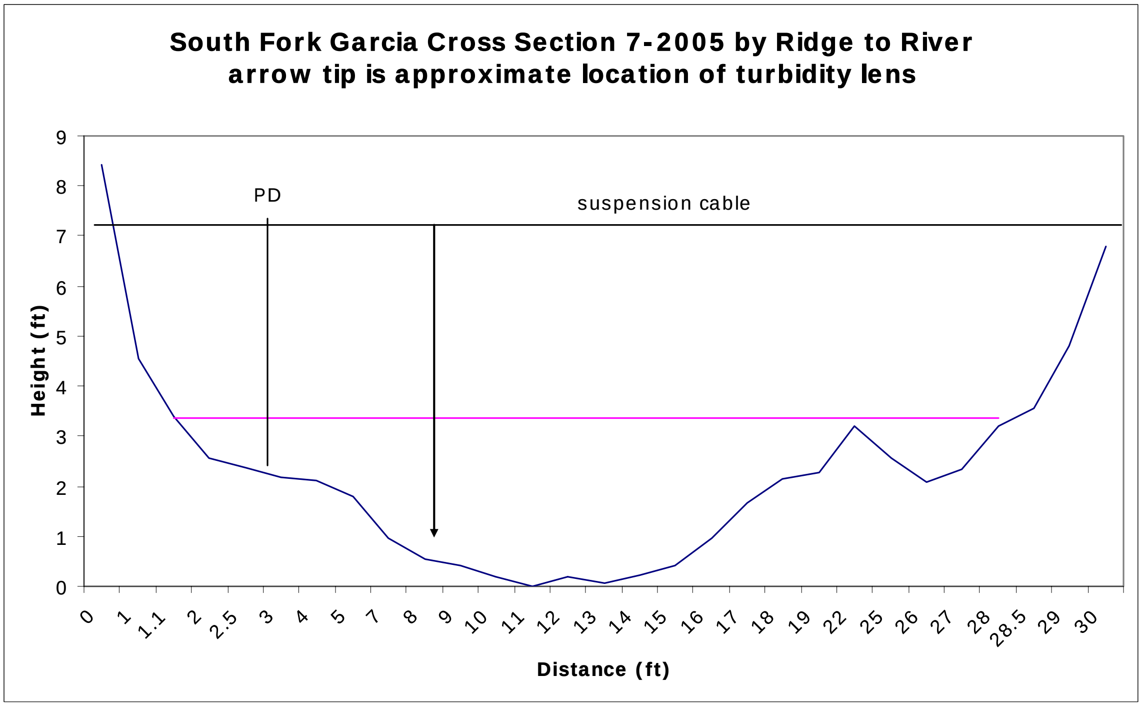
The mainstem Garcia station was disassembled in fall, 2004 and the equipment from that site was re-installed at a new station at Inman Creek. The streambank that held the bank mounted boom in place at the mainstem station was actively eroding and it appeared likely that the equipment would be destroyed during the upcoming winter. Also, a station was needed at Inman Creek to record existing instream conditions prior to erosion control treatments planned for in 2006. The mainstem Garcia station and Inman Creek did not have a properly functioning pressure transducer until CDF provided one for use at the Inman Creek station. This analog instrument, a WaterPro H-310, operated differently than the digital Drucks and required considerable troubleshooting to establish communications between the pressure transducer and the datalogger, which resulted in a lack of accurate stage data at Inman Creek until early 2005.

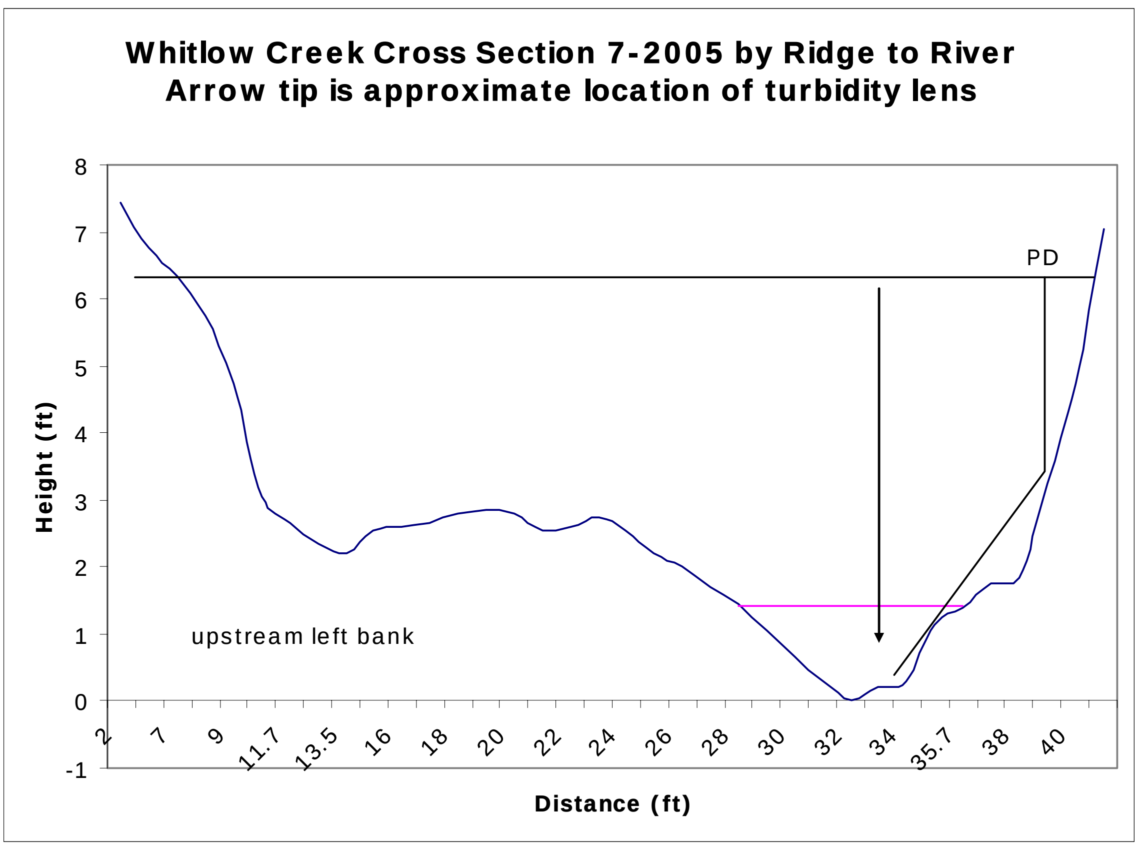
Station Mil: looking upstream at cable-mount turbidity sensor, staff plate, ISCO hose

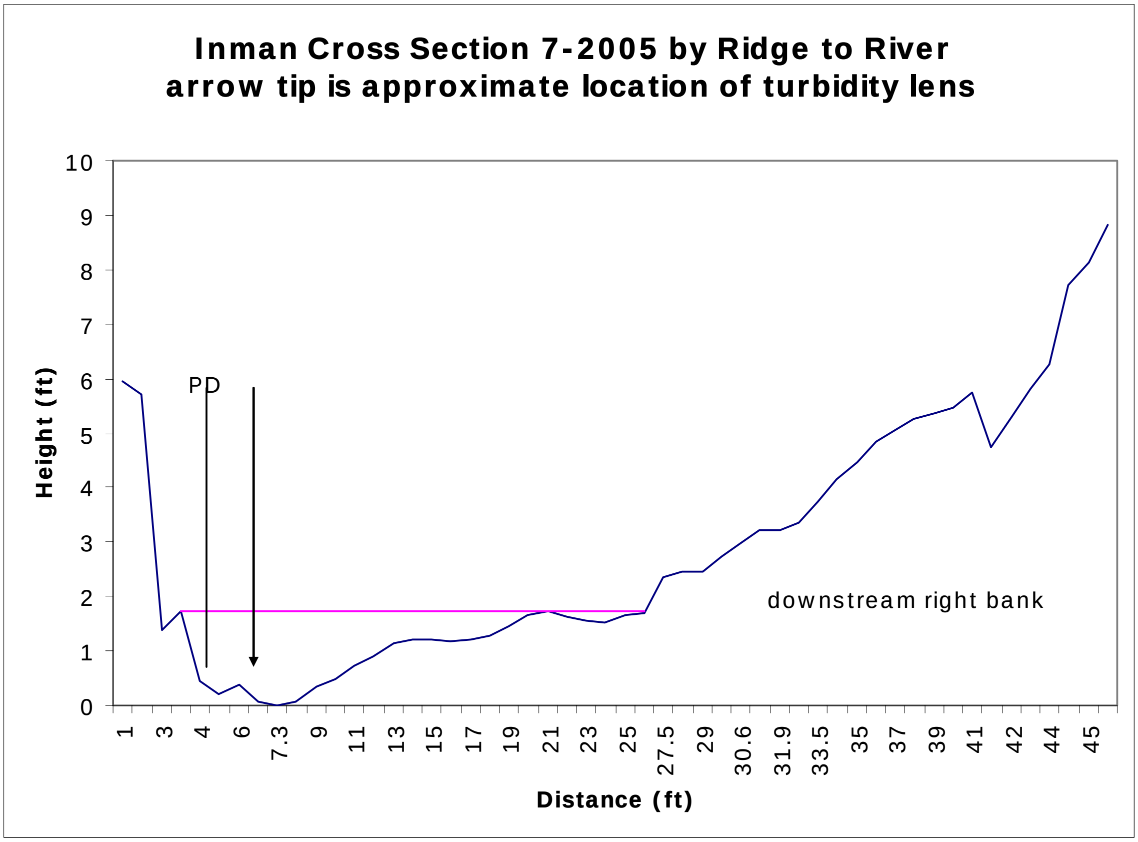


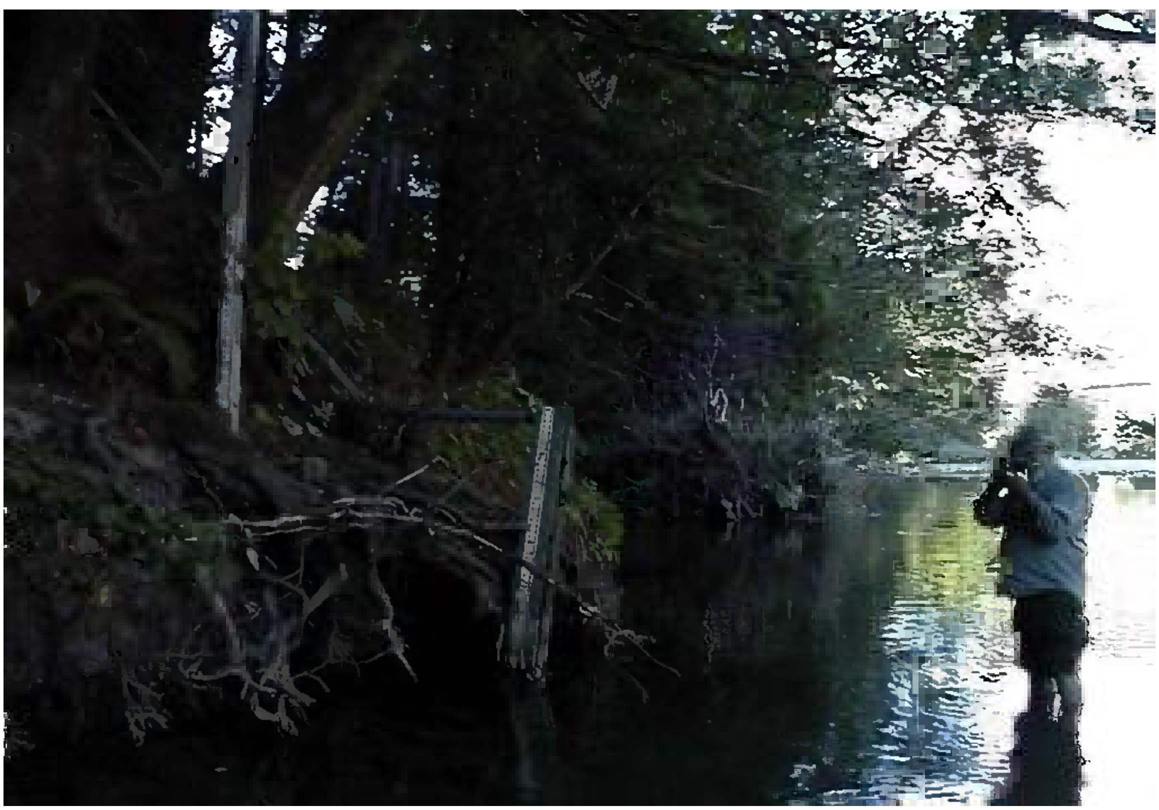
Station Par: looking upstream from Hollow Tree Road view




South Fork Garcia Cross Section 7-2005 by Ridge to River
arrow tip is approximate location of turbidity lens
PD
suspension cable
Height (ft)
9
8
7
6
5
4
3
2
1
0
0
1
1.1
2
2.5
3
4
5
7
8
9
10
11
12
13
14
15
16
17
18
19
22
25
26
27
28
28.5
29
30
Distance (ft)


Whitlow Creek Cross Section 7-2005 by Ridge to River
Arrow tip is approximate location of turbidity lens
PD
upstream left bank
Height (ft)
Distance (ft)
8
7
6
5
4
3
2
1
0
-1
2
7
9
11.7
13.5
16
18
20
22
24
26
28
30
32
34
35.7
38
40





Peter Dobbins at Mainstem Garcia Station observing active bank erosion there

FIELD METHODS

Ben Monmonier, Anna Birkas, and Teri Jo Barber conducted the field setup, downloading data from the dataloggers, and troubleshooting and maintenance of the sensor instruments, dataloggers, and batteries. Field methods included site visits approximately every three weeks to each monitoring station. Field books containing details of the site visits were maintained and kept inside a data box where the datalogger was stored. Standard maintenance included examining the wiper action on the turbidimeter lens, checking the battery voltage, and checking that the pressure transducer was recording accurate stage heights as referenced by the staff plate installed at each station. Examples of problems and maintenance include: Pardaloe's pressure transducer was scoured out by a high flow and had to be reinstalled, bolts holding the angle for the turbidimeter housing had to be changed out on occasion, and batteries had to be recharged on a regular basis. Water samples pumped from Mill Creek by our ISCO were pulled from the sampler approximately every 2 weeks depending on the timing of storms. Grab samples pulled from other streams by hand were also collected. Water samples were stored in a dark cool cabinet prior to processing them in the laboratory.

DATA PROCESSING

Datalogger software PC208w supplied by Campbell Scientific was used for this project. The TTS program from Redwood Sciences laboratory worked well with the statistical program R, to facilitate plotting of the data, but neither software is widely available nor user friendly. Therefore, data files were imported into an Excel spreadsheet and then day numbers 1-352 were translated into a standard date format (e.g., 12-6-2004), which was added to the data files. Raw data files were provided to landowners and are shown in the Appendix.

Raw data was also provided for "sanitization" to Randy Klein, Redwood National and State parks Hydrologist, Arcata, California. There are several sources for erroneously high turbidity including

- bubbles
- turbidimeter too close to streambed
- algal growth on lens
- macroinvertebrates clinging to the turbidimeter
- grass, leaves, etc clinging to the turbidimeter

Mr. Klein manipulated these data to correct for the suspected errors, most of which were confirmed during conversations between Randy Klein and Teri Jo Barber while referencing fieldbooks that contained references to problems encountered during site visits. The resultant finished data set was used for analysis. A summary of techniques used by Randy Klein is described in a document currently under development (Klein, 2003b).

COLLECTING STREAM WATER SAMPLES: GRABBED AND PUMPED

In 2003-04, 14 grab samples were obtained from the monitored tributaries and processed by the Salmon Forever laboratory in Arcata, California. No pumping samplers were available at this time and grab samples were to be taken when streamflows were highest.

As has been shown repeatedly by similar efforts to take samples at high flow, it is very difficult to predict the date and time of peak streamflows. These often occur in the middle of the night when personnel are often unavailable or reluctant to travel long distances to remote places. Therefore, we were not able to obtain samples during peak discharges, as desired. Grab samples were taken with a DH-48 Depth Integrated Sampler and were width integrated as well when the stream could be waded. When flow velocities made streams unsafe to wade, channels were sampled from as close as possible to the turbidity sensor. Pumping samplers alleviate these types of problems.

In 2004-05, 94 water samples were extracted from Garcia river tributaries. Twelve of these were “grab samples” and 82 were pumped by the ISCO pumping sampler at Mill Creek. Pumped samples from Mill Creek were made possible by the ISCO 6712 automatic pumping sampler provided by CDF. These are not depth or width-integrated, but originate at the same point in the water column as the turbidity sensor.

LABORATORY METHODS

Water samples taken from either the ISCO pumping sampler, or the DH 48 sampler were refrigerated out of the light until processing at the laboratory of Salmon Forever (2003-04 samples) or the Ridge to River laboratory (2004-05 samples). The USGS guide (1998) states that turbidity is optimally measured immediately on collecting a sample, preferably in the field. Holding the sample longer than 24 hours predisposes the measurement to biofouling which tends to bring the ntu value artificially down (USGS, 1998). Avoiding sunlight and keeping the sample cold limits biofouling. Turbidity values can be artificially high if the glass cuvette bottle becomes scratched or is not inserted with the same orientation for each reading. Care was taken to avoid these pitfalls by cleaning the cuvette with the special oil provided and wiping with a lint-free cloth before and after each measurement.

ISCO samplers collect 24 sample bottles over a period typically varying from two weeks to one month, making the 24-hour shelf-life untenable. Hydrologic year 2004-05 water samples were processed several months after collection, due both to the ISCO collection delay period and because the CDF contract providing funds for analysis was not effective until July 18, 2005. Biofouling is often indicated by the presence of algae in the sample, but none of the samples contained any visible algae. Our laboratory manager stated that algae in the sample behaves consistently regardless of color, such that it swirls when stirred. In contrast, suspended sediments that we expected in our sample did not swirl, but rather settled out on the bottom after stirring (Susan Wright, personal communication). Mrs. Wright performed approximately half of the turbidity measurements in the lab from sample bottles. A Hach 2100P turbidimeter was used, the qualities of which are reported in the USGS 1998 guide.

All 2004-05 water samples were processed for lab grade turbidity in October, 2005 with a HACH 2100P turbidimeter at the Ridge to River laboratory. Water samples remained in dark, cool storage until December when they were analyzed for SSC by Chemist Ruth Dobberpuhl. Labeled glass filters with residues intact are stored in the laboratory and were examined while preparing the section in this report addressing water sample results.

The ISCO with its hard hat off, exposing programmable brain, installed at Mill Creek


Mill Creek water samples, 2 from the ISCO (photo left) and 1 depth integrated sample

CHRONIC TURBIDITY IN RELATION TO BIOLOGICAL THRESHOLDS

Sediment is produced naturally from the watershed's hillslopes, river banks, and streambed during episodes of erosion from high intensity rainfall. This is commonly referred to as "background erosion." Accelerated erosion caused by human activities has been widely documented. Primary causes of accelerated erosion include alteration of natural drainage patterns by roads, removing soil cover, and by placement of unstable fill material in a position where it can be delivered to streams.

Once the products of erosion are delivered to the stream channel, entrained course sediments deposit in point bars or in the active streambed. The smaller particle fractions (sand, silt, and clay) stay suspended in the water column for longer periods of time (the smaller the particle, the longer they will be suspended) and settle in pools or on banks and in riffles when higher flows recede.

Land management practices that disturb soil and drainage patterns have increased erosion and the delivery of sediment to rivers and streams, to the detriment of salmonid habitat. Deep rearing pools have been shallowed and spawning gravels have been degraded by sand, silts, and clays that clog pore spaces. Winter water clarity between winter storms is important for anadromous fishes so that they can keep gills clean, see to feed, and have gas exchange for alevins developing in redds. When between storm water clarity decreases due to chronic recession limb turbidity, the lack of water clarity inhibits a fish's ability to see and capture prey, suspended sediments can clog sensitive gill tissue, and light is inhibited from reaching the benthos where periphyton grow as primary food production.

Sands, silts, and clays are the finer particle sizes found in sediment. Silt and clay particles become entrained during the rise of a stream in response to a rainstorm and stay suspended longer than do the larger sediment size classes. When these smaller particles remain in suspension between major storms, this creates a condition that is often referred to as "chronic turbidity". Chronic turbidity refers to long durations of turbidity exposure, while acute turbidity exposures refer to very short periods of high turbidity.

Acute turbidity from elevated suspended sediment concentrations in the water column occur during relatively brief periods of high runoff in response to intense, prolonged rainfalls that can also cause flooding. These episodes of intense rainfalls and flooding can also provide the opportunity for alluvial rivers to form or reform their beds and banks, which may include lateral bank erosion, changes meander patterns, and sorting and cleaning of gravel. This natural disturbance pattern is typical for undammed alluvial rivers like the Garcia and illustrates that rivers are changing features in terms of their position within floodplains and terraces between valley walls, rather than static features on the landscape.

One example at the reach scale is the immediate flush of suspended sediment derived turbidity that occurs immediately downstream of a bank collapse. This type of sediment pulse includes the short term entrainment of sediments, downstream transport of the finer sediment particles in the washload, and local deposition of the coarsest sediment materials into the bedload. The short duration of these events is tolerated by anadromous fishes, owing to adaptations developed in co-evolution with their native streams over geologic time scales.

Elevated chronic exposure to turbidity and suspended sediment between storms is perhaps a more recent experience for salmonids. We assume that chronic turbidity exposure is elevated over background conditions in duration and magnitude due to watershed disturbance from land management practices. Klein (2003) recently reported that for eight turbidity-instrumented small streams from Mendocino County to the Oregon border from water years 2000 to2002, land use practices explained more of the chronic turbidity variation than did the natural sources of variation combined (geology, climate, vegetation). Land use most directly correlated to chronic turbidity in that study was summarized in two measurements: percent of watershed area harvested per year and road density in miles of road per square mile of watershed area (Klein, 2003).

Sublethal effects of chronic turbidity on aquatic organisms has been explained and documented by a variety of researchers. In the report of the Scientific Review Panel on California Forest Practice Rules and Salmonid Habitat (Ligon et al. 1999), the sub-lethal effects of chronic turbidity on salmonids were listed as reduction in feeding by juvenile salmonids, thereby reducing growth rates, irritation and abrasion of gill tissues, avoidance behaviors, and mortality at high concentrations as documented and reported by Noggle (1978).

Barber (1997) reported a series of biological responses of salmon and steelhead to varying concentrations of suspended sediment and turbidity in fresh water based on a review of the literature as follows. It was stated that water appears clear to 2 NTU. Fishes avoid turbid water sources and begin seeking of cover when instream turbidities rise to 5 NTU. At 6, streamwater is often colored, appearing blue or green and is barely clear enough to see through. At about 10, water clarity appears noticeably diminished to the human eye, rainbow trout growth rates slow, and at 1 meter depth light in the water column is 77% of what is available at the surface. At 20 NTU, fishermen go elsewhere, perhaps because coho salmon (and other fishes) reactive distance is reduced by 52%. Sigler et al (1984) reported that at 25 NTU, steelhead juveniles ceased to grow. Trush (2001) reported that when turbidity reaches 25-38 NTU, a fish's ability to feed is cut in half. Hadden (et al. 2004) reported that both field and laboratory studies revealed that while the foraging efficiency of juvenile salmonids was decreased by increased turbidities, fish continued to capture prey at turbidity levels in the range of 40-50 NTUs.

A turbidity value of 30 NTU stimulates a major increase in macroinvertebrate drift downstream, which may be an avoidance technique. At 50 light is reduced to 60% of what is available at the surface and streamwater appears like chocolate milk. At 60 reduced capture success of prey by fishes is exhibited. At 86 a marked reduction in growth rate in brook trout juveniles was reported. At 100 coho juveniles avoid turbid waters by swimming away to clearer waters if they are available. At 200, steelhead exhibit the same avoidance technique. When turbidities reach 1000-2000, light at 0.1 meter in depth is reduced to only 4% or what is available at the surface. No lethal effects of turbidity have been reported, however lethal effects on fishes due to extremely high suspended sediment concentrations have been documented at 500-2000 mg/l (Barber, 1997).

Klein (2003) determined turbidity durations greater than or equal to the following thresholds: 25, 50, 70, 100, 150, 200, 500, 1000, 2000 NTUs during a single water year. Of these, thresholds of 25, 60, and 150 NTUs were selected for use in this study because the biological effects of these turbidity thresholds are well-reported in the literature.

Turbidity peaks are usually related to rises in stream heights, which, in turn, can be traced to periods during and immediately after rainfall. Long durations of turbidity not explained by rainfall and stream stage rises might be caused by emergency road repairs installed in the rain or winter vehicle traffic on a dirt road, etc. Klein's (2003) results indicated that two basins in his study expressed larger chronic turbidities, reflecting a high percent of roaded drainage area and high frequency of post 1988 timber harvest expressed as a percent of basin area.

WATER CLARITY RESULTS

Figures 7-16 display the turbidity values recorded in NTU for the Garcia River mainstem and tributary stations by water year. Raw digital turbidity data was utilized to prepare the graphs for presentation and for analysis except where "suspect data" (as identified by Randy Klein) were corrected. The red line indicates the whole corrected dataset and the green line indicates which suspect raw data were corrected. Tributary graphs also feature a horizontal line across the data indicating the three turbidity thresholds we selected as an index of water clarity due to biological differences in overwintering salmonids: 30, 60, and 150 NTU. Tables 11 and 12 and figures 18 and 19 summarize duration of chronic turbidity, the hours of duration by water year and tributary sustained in each creek above the 30, 60, and 150 threshold levels. Physiological and behavioral changes exhibited at these thresholds are summarized above.

Figure 7. Garcia Mainstem Turbidity 2003-2004

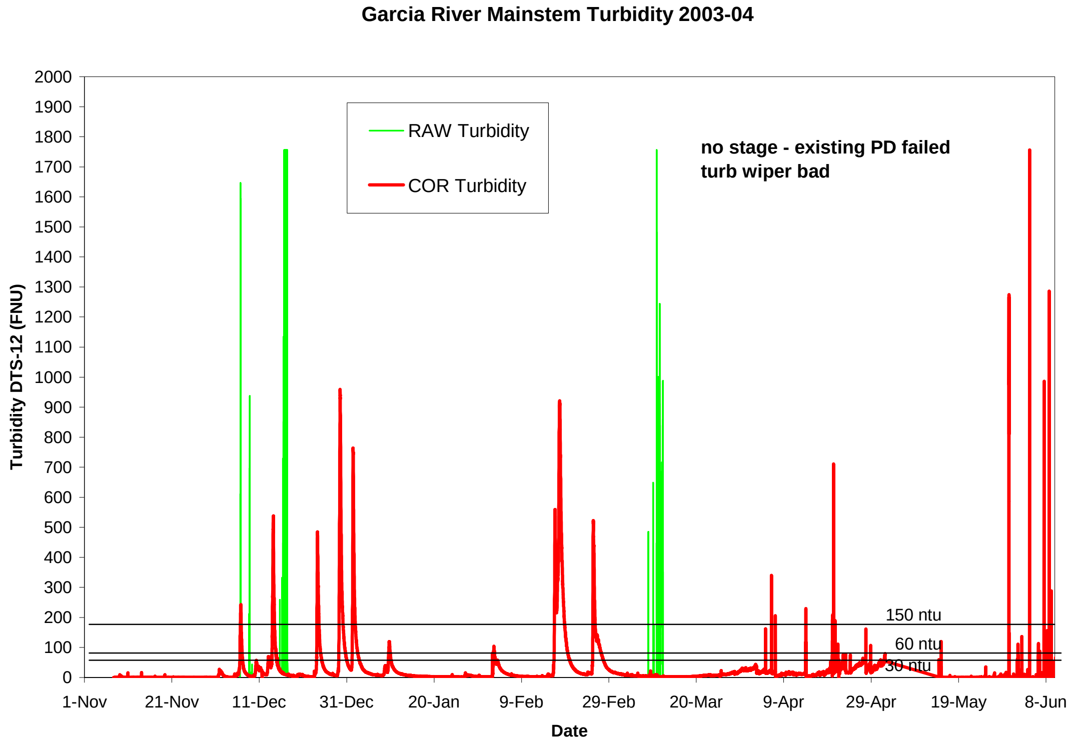


Figure 8. Mill Creek Turbidity 2003-2004

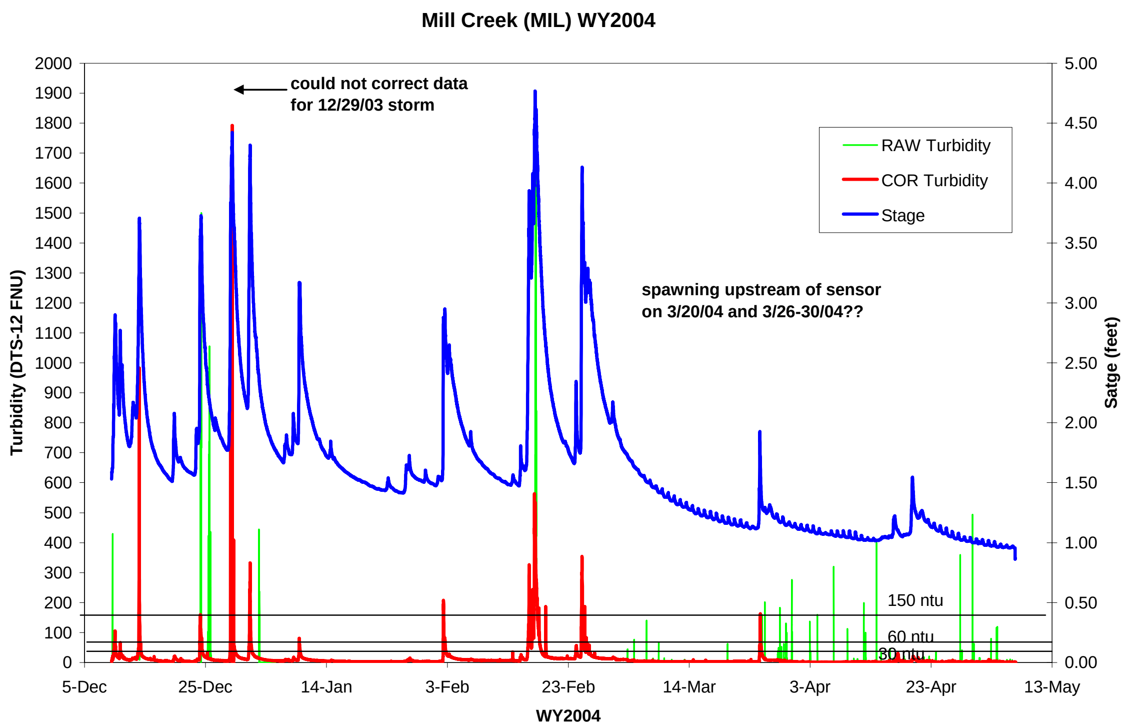


Figure 9. Mill Creek Turbidity 2004-2005

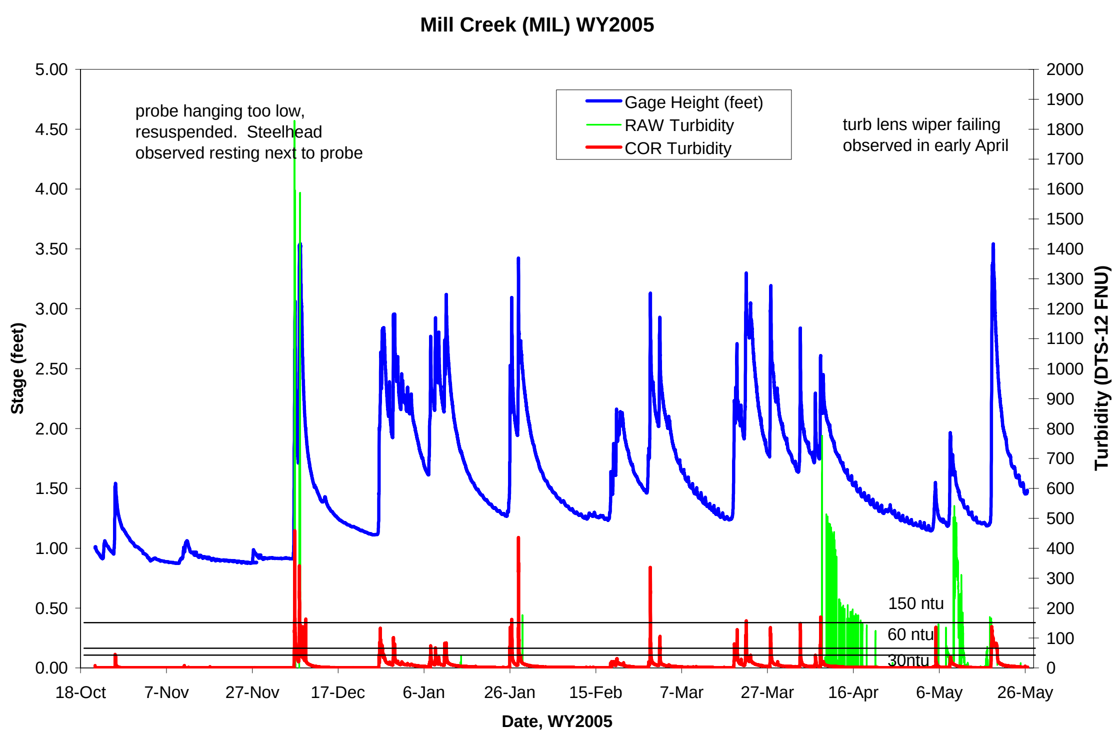


Figure 10. Pardaloe Creek Turbidity 2003-3004

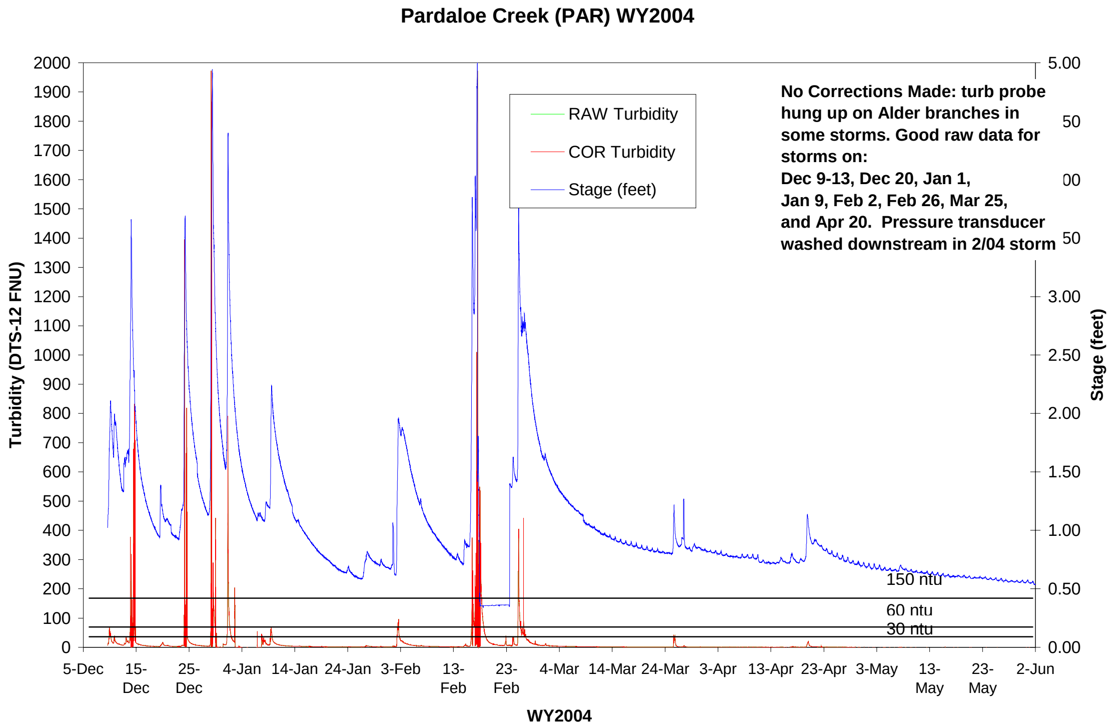


Figure 11. Pardaloe Creek Turbidity 2004-2005

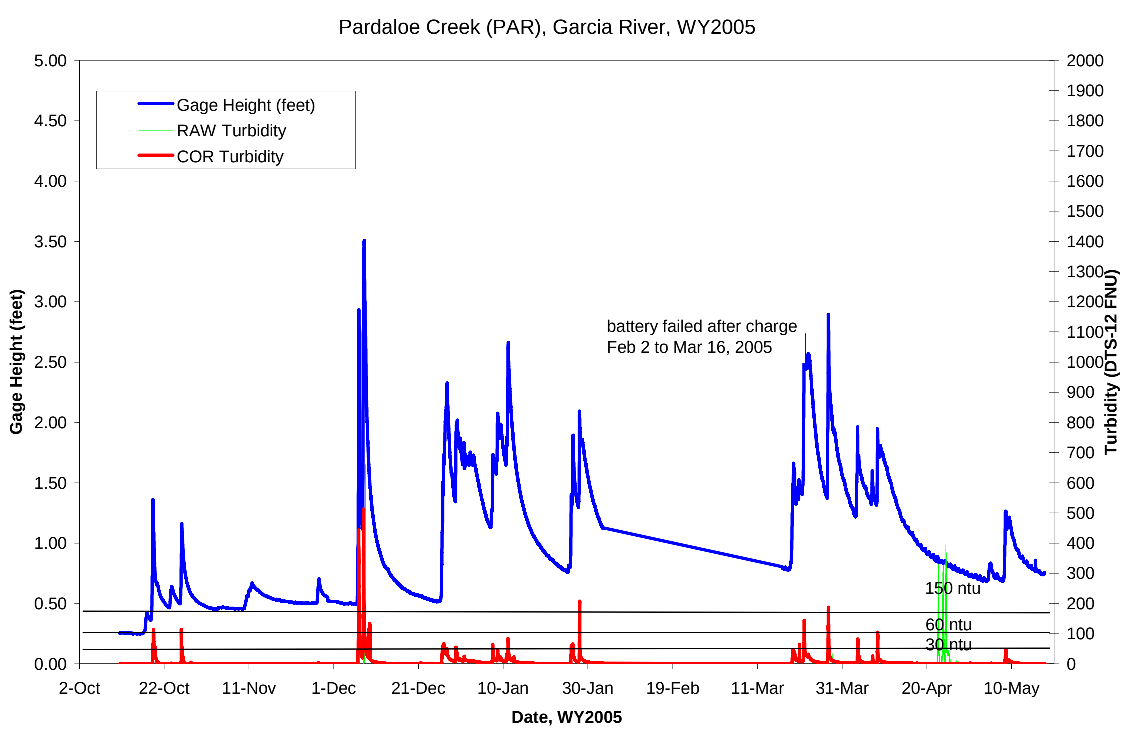


Figure 12. Whitlow Creek Turbidity 2003-2004

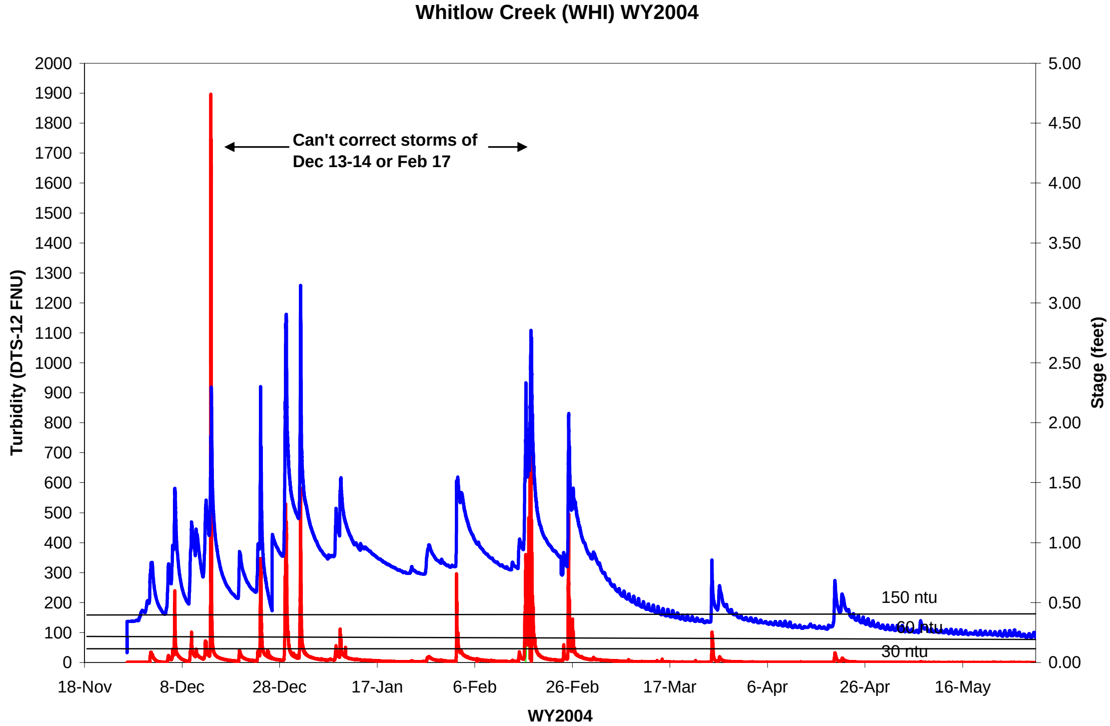


Figure 13. Whitlow Creek Turbidity 2004-2005

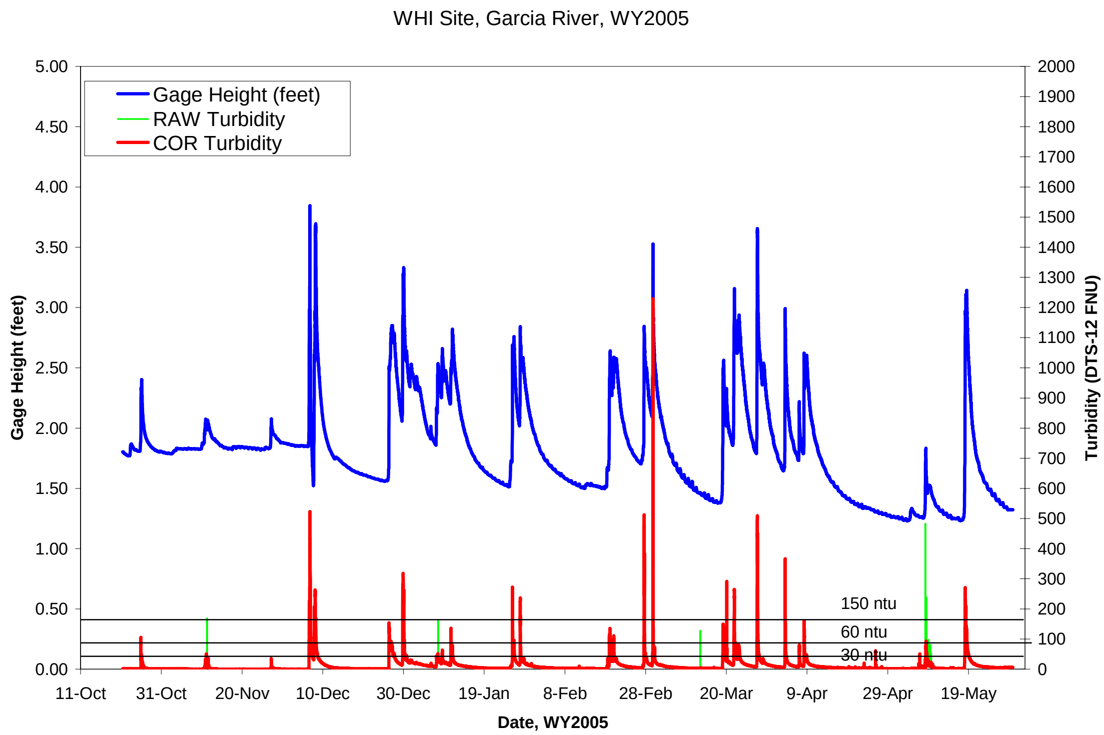


Figure 14. South Fork Turbidity 2003-2004

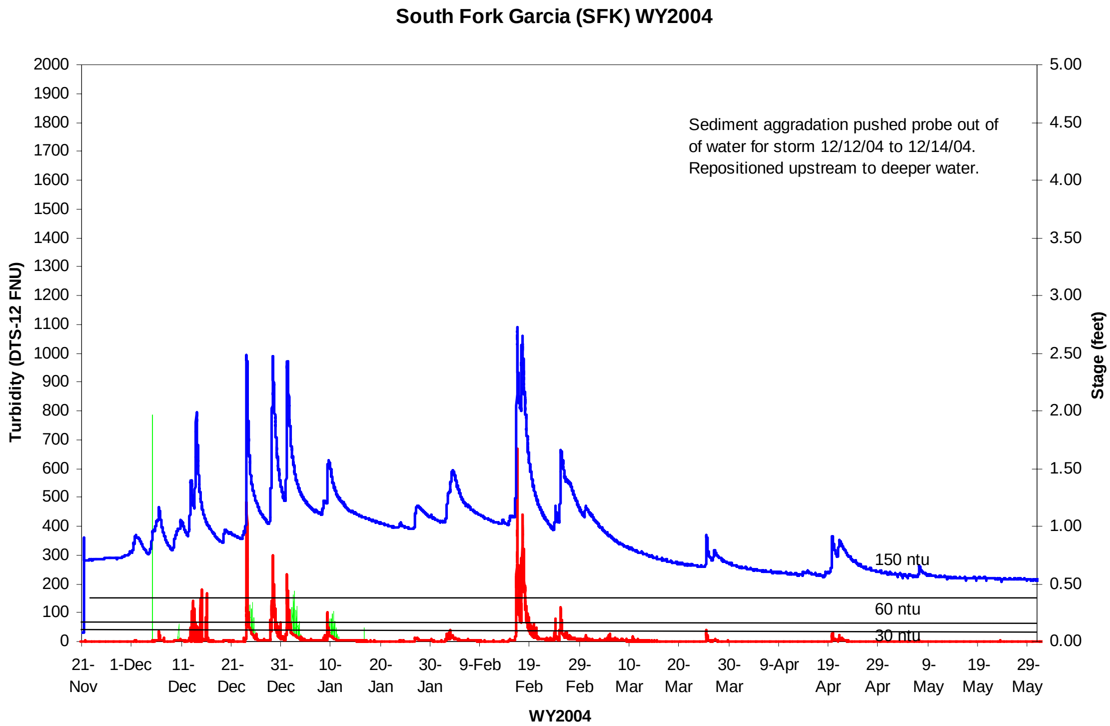


Figure 15. South Fork Turbidity 2004-2005

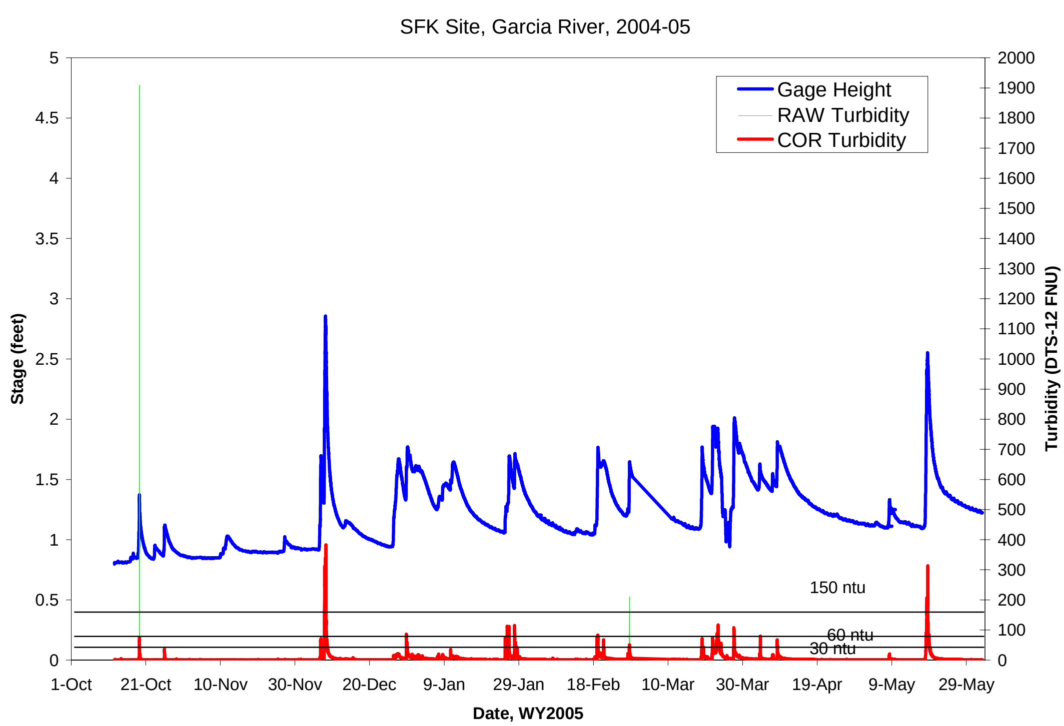
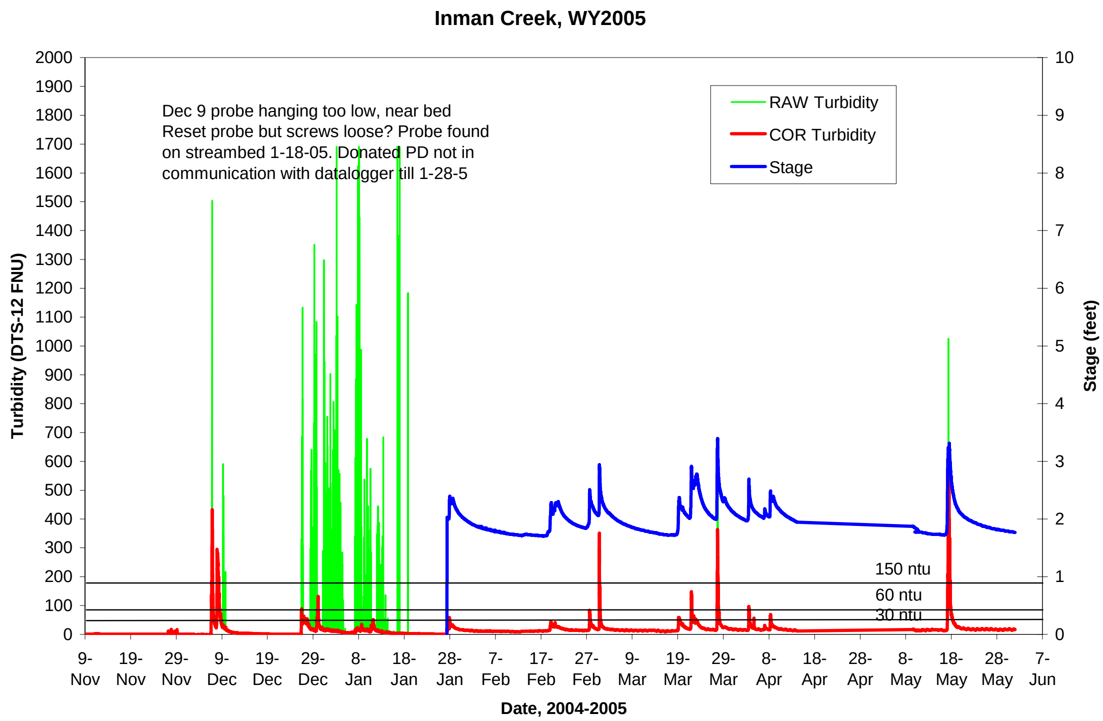


Figure 16. Inman Creek Turbidity 2004-2005



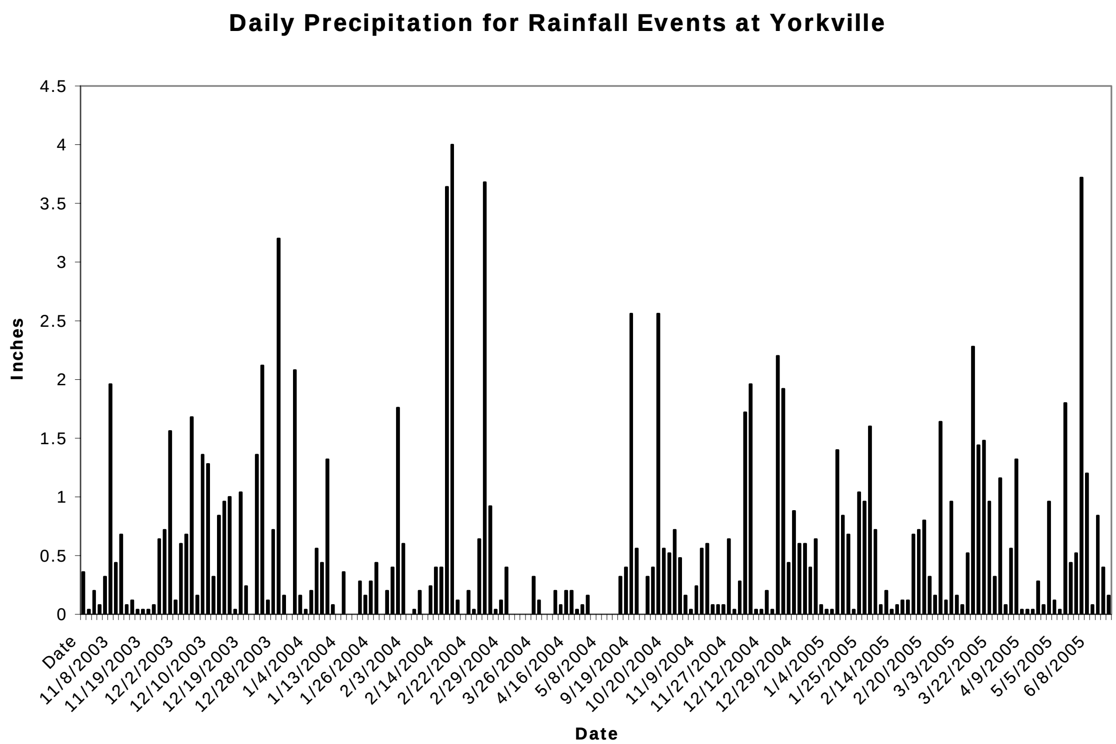
PHYSICAL FEATURES OF MONITORED GARCIA RIVER SUB BASINS

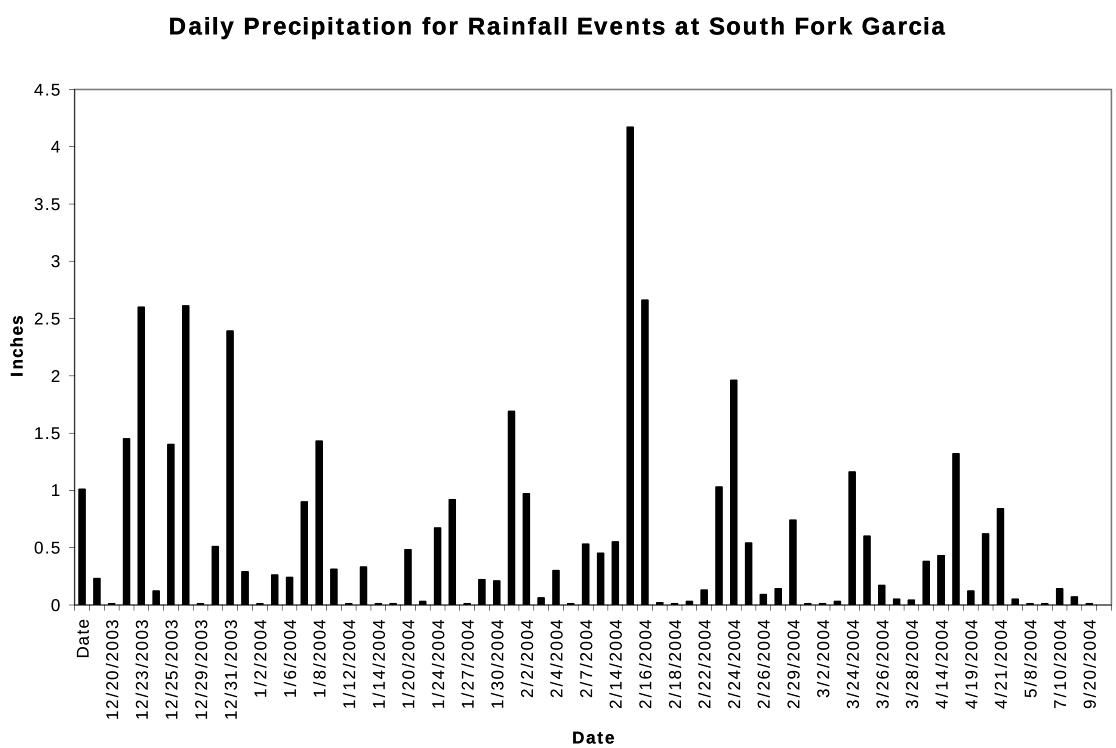
RAINFALL

No rainfall stations were installed in Garcia River watershed tributary basins. Data from the Yorkville weather station is attached, courtesy of the California Department of Water Resources' web site. This station resides in the Dry Creek watershed, and is a tributary to the Russian River. It shares a ridge with the Garcia in the upper reaches beyond Mill and Pardaloe Creeks. Precipitation information was provided from a gauge operated by Mendocino Redwood Company in the South Fork Garcia basin (see Data Appendixes). Referencing rainfall from the two extreme ends of the basin provide a context with which to make assumptions about the rainfall experienced in the individual tributaries.

Average annual precipitation from the Yorkville station is 43.5 inches based on records from 1948-1951 and from 1998-2005. During our study period above-average annual precipitation occurred in Yorkville: 60.5 inches in WY2003, 52.2 inches in WY2004, and 60 inches in WY2005 suggesting a rather wet study period. Precipitation data from South Fork station was not complete, measuring 40.8 inches between December 2003 and October 2004.

Figure 17. Rainfall Events Surrounding Garcia River Monitoring WY 04-05





LAND MANAGEMENT HISTORY IN THE GARCIA RIVER WATERSHED

The most comprehensive history of land management in the Garcia River Watershed is Monschke and Caldon's (1992) Garcia River Watershed Enhancement Plan. It includes old maps and photos, accounts of Spanish, Native American, early white settler history, a logging history, and interviews with several long-time landowners.

LEGACY CONDITIONS

Historical logging photographs reveal the extreme impacts of logging that took place, especially near mills, at the turn of the last century and through the early 1900s. Even more detrimental effects of timber harvest on instream ecology, especially in the Garcia River, occurred following World War II, as the use of tractors for timber harvest came into full swing. Erosion rates were accelerated well-beyond background levels due to extremely poor logging practices in and near stream channels. Pacific Watershed Associates estimated the rate of erosion to be 1380 tons per square mile per year in the period 1952-1997 for the Garcia River watershed (USEPA, 1998). The cumulative damage across the 1950s and into the early 1970s as a result of unregulated tractor logging, combined with erosive effects of the 1955 and 1964 record floods, had an enormous impact on the Garcia River's sediment regime. The combination of these past impacts are referred together by the term "legacy conditions." Most scientists agree that the stream channels today still reflect these legacy conditions (Knopp 1993).

Table 11. Total hours above turbidity thresholds: 30, 60, and 150 ntus Water Year 2004

Turbidity (ntu) thresholds	Garcia River Stations in Water Year 2004				
	Garcia	Mill	Pardaloe	South Fork	Whitlow
>30	759.8	207.3	205.7	203.8	343.7
>60	335.7	101.5	110.3	125.3	152.5
>150	156.8	39.5	42.2	39.8	62.8

The mainstem Garcia station consistently experienced twice as many hours over turbidity thresholds than did any tributaries, yet turbidity peaks were about the same. Elevated Garcia mainstem exposures could be a cumulative effect but certainly suspended algae is a prominent feature of a mainstem river due to direct insolation and lack of overhead canopy. The suspended algae source is consistent with the river continuum concept where insolation stimulates primary algal production in open rivers rather than the allochthenous detritus driven food web of creeks under canopy. When the turbidity lens wiper arm failed in the mainstem, the lens was quickly colonized by periphyton which supports the concept of greater algae derived turbidity in the mainstem. Conversely, wiper arm failure at Mill Creek resulted in no noticeable lens colonization.

In WY 2004, South Fork experienced the least number of hours where turbidity exceeded 30 ntu while Mill Creek experienced the least number of hours where turbidity exceeded 60 and 150 ntu.

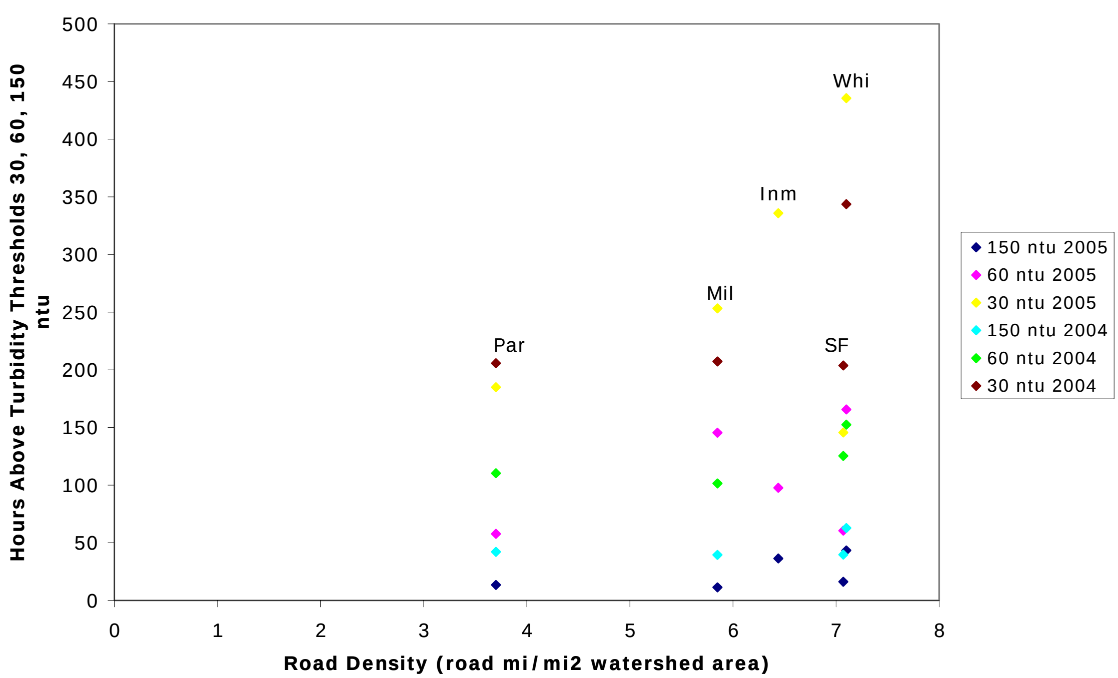
Table 12. Total hours above turbidity thresholds: 30, 60, and 150ntus Water Year 2005

Turbidity (ntu) thresholds	Garcia River Stations in Water Year 2005				
	Inman	Mill	Pardaloe	South Fork	Whitlow
>30	335.8	253.3	184.8	145.7	435.5
>60	97.7	145.5	57.8	60.5	165.7
>150	36.5	11.3	13.5	16.2	43.5

In 2005, South Fork experienced the least number of hours where turbidity exceeded 30 ntu, Pardaloe Creek experienced the least number of hours where turbidity exceeded 60 ntu, and Mill Creek experienced the least number of hours where turbidity exceeded 150 ntu.

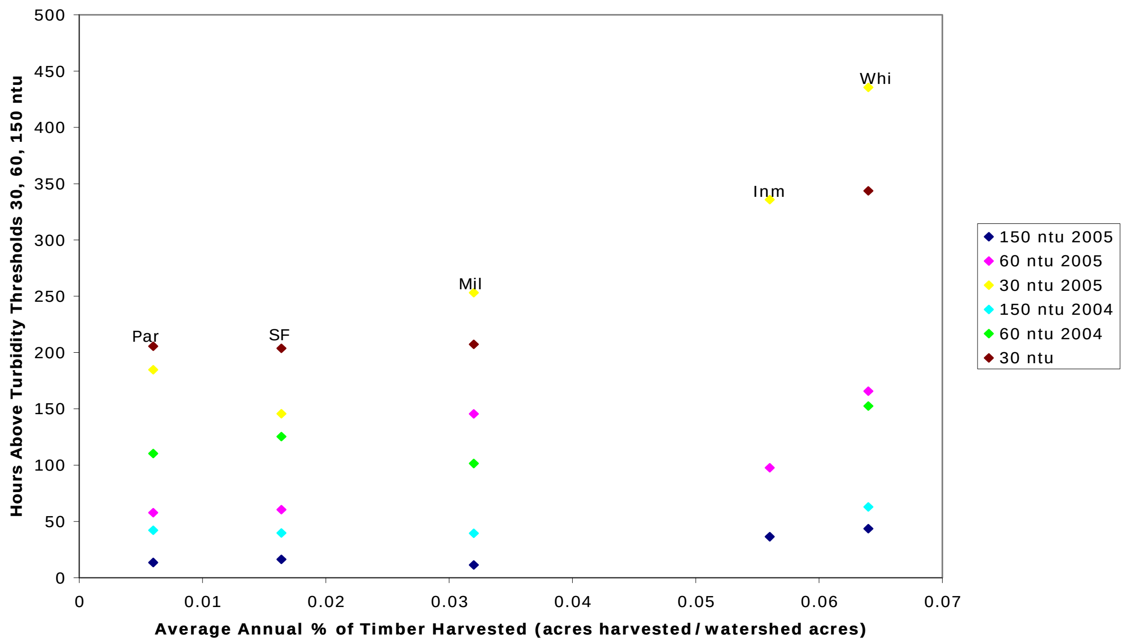
Pardaloe Creek Watershed experienced the lowest Road Density at 4 miles of road per square mile of watershed area while the other watersheds experienced road densities of between 5.5 and 7.5. Pardaloe Creek also experienced the lowest Timber Harvest Intensity. South Fork experienced slightly more Timber Harvest Intensity than Pardaloe. Mill experienced a medium level of road density and Timber Harvest Intensity. Whitlow and Inman experienced relatively intense Timber Harvest Intensity.

Figure 18. Hours Above Turbidity Thresholds vs. Road Density



There seems to be a positive linear relationship between Hours above Turbidity Thresholds (tables 11 and 12) and both Road Density (Figure 18) and Timber Harvest Intensity (Figure 19). The exception to this trend is at South Fork where road density is moderately high relative to the other basins (over 7 miles per square mile of watershed area) yet hours sustained above turbidity thresholds were relatively low.

Figure 19. Hours Above Turbidity Thresholds vs. Timber Harvest Intensity



The storms that produced the highest stages did not necessarily produce the highest turbidity. For the biggest storms of each water year, we pulled the highest turbidity experienced by each tributary. These are presented in Table 13.

Table 13. Largest Storms in the study period over ½ Bankfull by Date and Turbidity (ntu)

Storm Date	**South Fork**	**Pardaloe**	**Mill Creek**	**Whitlow Cr.**	**Inman Cr.**
12-11-2003	7	70	105	102	NA
12-14-2003	180	NA	982	NA	NA
12-24-2003	485	NA	158	495	NA
12-29-2003	298	NA	1792	529	NA
1-1-2004	229	791	332	582	NA
2-16-2004	669	NA	563	483	NA
2-25-2004	114	443	349	495	NA
12-8-2004	381	513	459	523	NA
12-26-2004	86	67	132	318	NA
1-11-2005	34	84	68	136	NA
1-29-2005	116	203	436	272	59
3-1-2005	50	NA	330	1229	351
3-22-2005	117	31	155	291	148
3-27-2005	99	188	135	509	364
5-17-2005	298	48	138	264	611
Average ntu	*210.8*	*221.6*	*383.3*	*415.2*	*255.5*

In both 2004 and 2005 South Fork experienced the lowest average peak turbidity during storms, followed by Pardaloe. Figures 20 and 21 compare these average peak turbidities with Road Density and Timber Harvest Intensity. Again, South Fork has a relatively high density of roads yet has lower peak turbidity.

Figure 20. Average Peak Turbidity vs. Road Density

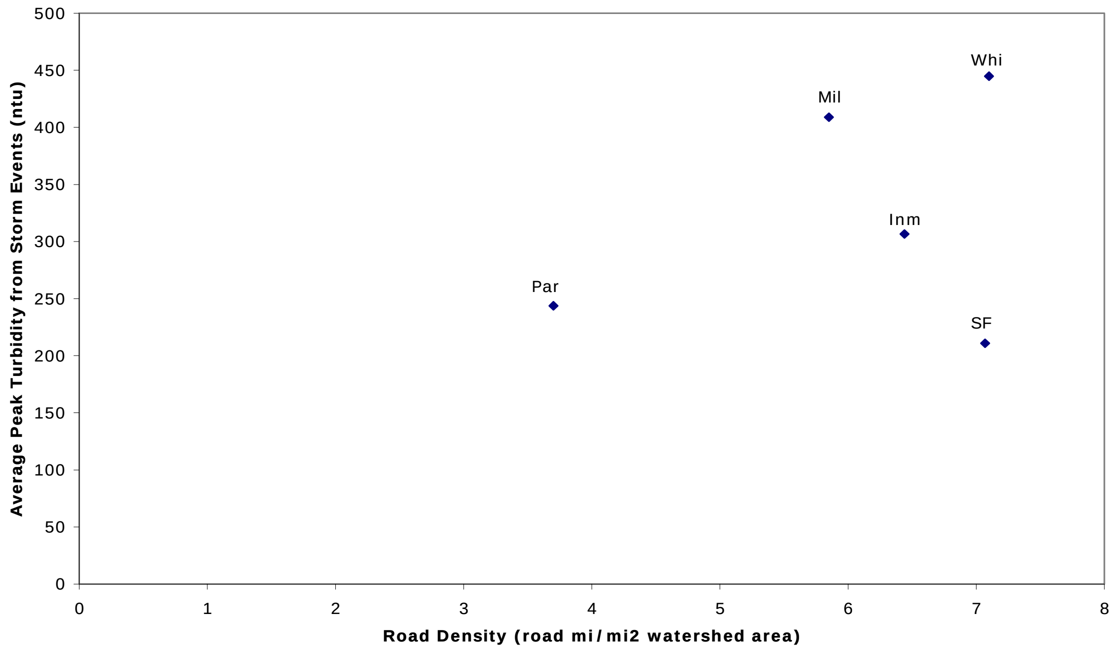
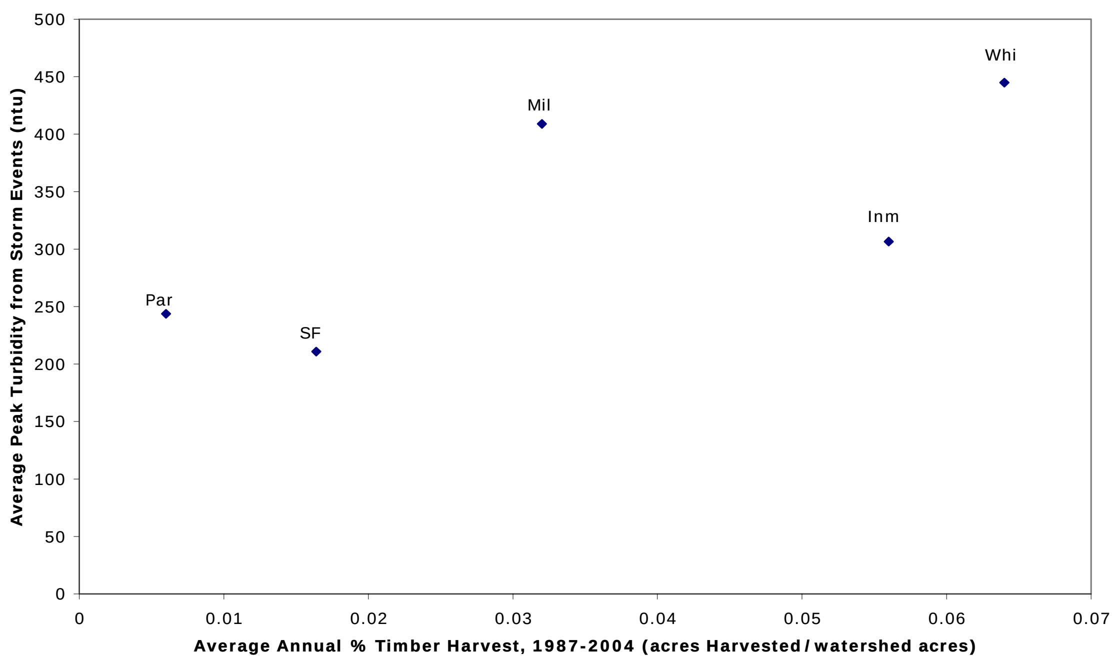


Figure 21. Average Peak Turbidity vs Timber Harvest Intensity



LAND USE IMPACTS VERSUS NATURAL VARIATIONS

A description of land uses that may have contributed to current instream conditions is provided below for each tributary watershed. It is understood that current conditions are a result of interacting watershed –related parameters, including precipitation patterns, geology, vegetation, and soil types, which vary naturally. Therefore, variation in instream conditions is a constant in watershed science that must be expected, tolerated, and embraced. This broad spectrum of variation could be used to explain or mask virtually any instream conditions. However, we hope to pull back the curtain of natural variation on the question of whether there is a measurable difference in spawning gravel quality or water clarity after road based erosion control restoration based on the data collected by this project. In doing this, we are looking directly at two types of land uses: restoration and disturbance.

SUB WATERSHED RESTORATION

Support for restoration of salmon bearing watersheds and funds for granting agencies to implement restoration projects ultimately comes from taxpayers. Politicians who bring forth the spending bills that support watershed restoration are looking for evidence that watershed restoration can be used effectively to repair critical conditions in salmon bearing streams to increase their productivity.

South Fork Garcia River

Craig Bell, active Garcia River Coordinator, was pivotal in the restoration of the South Fork Garcia tributary on property owned and managed by Mendocino Redwood Company. He summarized the watershed and instream restoration implemented there as follows (Bell, personal communication, 2004):

> *"In 1998 the South Fork Garcia River was chosen as the trial project for Trout Unlimited's North Coast Coho Project. The work involved a comprehensive assessment and implementation plan developed by Pacific Watershed Associates for upslope road-based erosion control and drainage improvements as well as the design and implementation of instream installation of woody debris as fish habitat structures by Craig Bell and Trout Unlimited. The upslope assessment mapped 148 individual sites had the risk of delivering sediment to stream channels, along 22 road segments totaling nearly 25 miles. The assessment documented 39,700 cubic yards of future sediment delivery if no efforts were made to correct road conditions.*
>
> *In 1999 TU submitted and was awarded a DFG grant to treat 8.75 miles in length (36%) of the total inventoried miles), 82 of the inventoried sites (55% of the total), and prevent over 28,000 cubic yards of sediment delivery (72% of the estimated total delivery). In addition ten instream habitat sites were funded for implementation. The work was carried out in 2000 and 2001.*

In 2005 Trout Unlimited in partnership with Landowner Mendocino Redwood Company has submitted a grant to DFG for implementation of measures to prevent the remaining 25% of the controllable sediment delivery from entering the South Fork Garcia. If funded, erosion control work is expected to be carried out in the summer of 2006. Mendocino Redwood Company owns virtually all of the South Fork Garcia sub basin. Ongoing monitoring is important to answer the question of response time to comprehensive upslope erosion control measures"

.

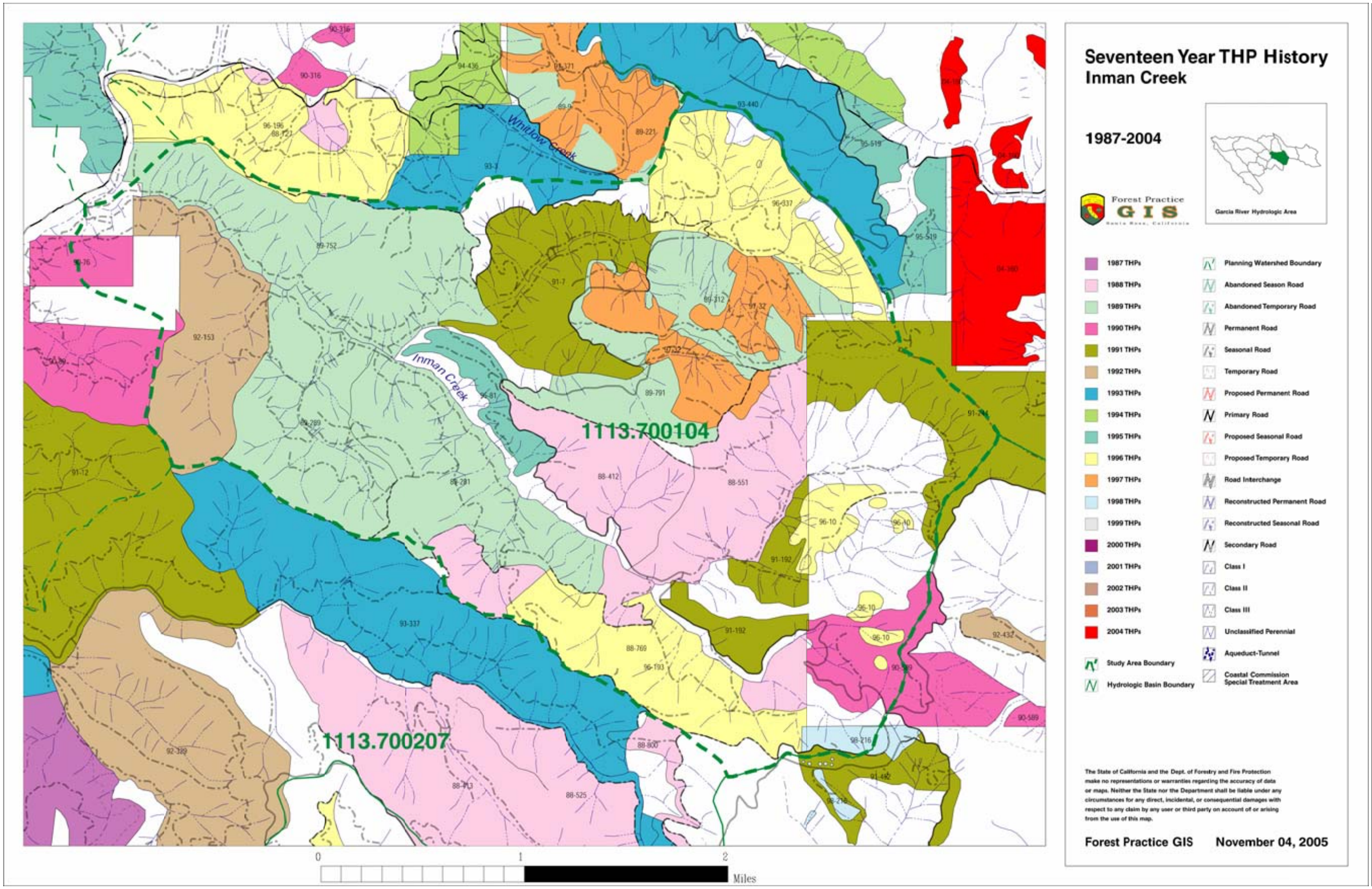
<u>*Inman Creek*</u>
Because Inman Creek has been an important tributary for reproduction of anadromous salmonids, The Conservation Fund and Trout Unlimited developed a grant proposal to the California Department of Fish and Game to prepare a subwatershed-wide roads assessment by Pacific Watershed Associates for the Inman Creek subwatershed. If granted, implementation is expected as early as 2006.

SUB WATERSHED TIMBER HARVEST AND ROAD DENSITY DISTURBANCES
Documenting disturbance is a broad goal that changes from year to year, place to place, and at the discretion of the landowner. As a result, we are documenting just three land use disturbance measures that are publicly available through CDF and have been used by watershed scientists in the past to gauge watershed disturbance. These are: percent of watershed acres of timber harvested in a 10- year period (Trush 2005), percent of watershed area harvested annually (Klein 2003), and road density expressed as miles of road per square mile of drainage area (Trush, 2005; Klein, 2003). For our study we received GIS services provided by Ms. Suzanne Lang at CDF's Santa Rosa office, who mapped timber harvest history and road density from1987-2004 (see figures 18-27 supplied in hard copy).

Road density (miles of road per square mile of watershed area) and timber harvest intensity expressed as a percentage of sub- watershed harvested in a 1 year or 10 year period are units of land use that may influence changes in water clarity (Klein 2003). CDF provided a 17-year timber harvest history map showing 1987-2004 THPs, complete with roads, for use in our analysis (figures 18-27). Two sets of maps were provided: THPs color-coded by year and THPs color coded by silvicultural method. The "No Harvest" portion of the map is white and stands out in contrast to the areas that have been harvested, which are represented in color. White areas of the Pardaloe and South Fork dominate maps indicating a low rate of harvest within the 17-year period compared to the other basins. Based on this information, the intensity of timber harvest was ranked qualitatively with a number varying from 1 to 5, with 1 being the lowest harvest intensity and 5 being the highest.

Table 14. Road Density, Timber Harvest History 1987-2004 (data courtesy CDF)

Stream	PAR	SF	MIL	WHI	INM
Drainage Area - acres	5626	5598	4846	1221	5481
Road Density mi/mi^2	3.70	7.07	5.85	7.09	6.44
THP 1987 a	21.56	54.4	315.68	0	0
THP 1988 a	0	627.15	0	272	1019.83
THP 1989 a	0	224.31	13.07	217.71	1952
THP 1990 a	0	4.27	0	42.98	50.76
THP 1991 a	27.29	0	177.19	7.88	714.9
THP 1992 a	0	0	0	42.52	329.23
THP 1993 a	0	0	139.75	182.78	54.15
THP 1994 a	0	63.45	192.11	137.12	83.12
THP 1995 a	0	0	0	0	0
THP 1996 a	175.32	0	720.84	256.59	746.19
THP 1997 a	0	0	0	172.95	259.78
THP 1998 a	3.62	143.35	344	0	28.15
THP 1999 a	0	66.88	0	0	0
THP 2000 a	187.39	0	351.91	0	0
THP 2001 a	0	126.68	171.92	0	0
THP 2002 a	0	0	0	0	0
THP 2003 a	0	0	0	0	0
THP 2004 a	164.32	250.94	191.59	0	0
Total acreage harvested in 17 years	579.5	1561.43	2618.06	1332.53	5238.11
proportion of watershed area annually harvested (17 year avg)	0.0061 = 0.61%	0.0164 = 1.64%	0.0318 = 3.18%	0.0642 = 6.42%	0.0562 = 5.62%



Seventeen Year THP History
Inman Creek
1987-2004
Garcia River Hydrologic Area
Forest Practice GIS
1987 THPs
1988 THPs
1989 THPs
1990 THPs
1991 THPs
1992 THPs
1993 THPs
1994 THPs
1995 THPs
1996 THPs
1997 THPs
1998 THPs
1999 THPs
2000 THPs
2001 THPs
2002 THPs
2003 THPs
2004 THPs
Study Area Boundary
Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
Abandoned Temporary Road
Permanent Road
Seasonal Road
Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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Forest Practice GIS
November 04, 2005
Whitlow Creek
Inman Creek
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1113.700207
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1
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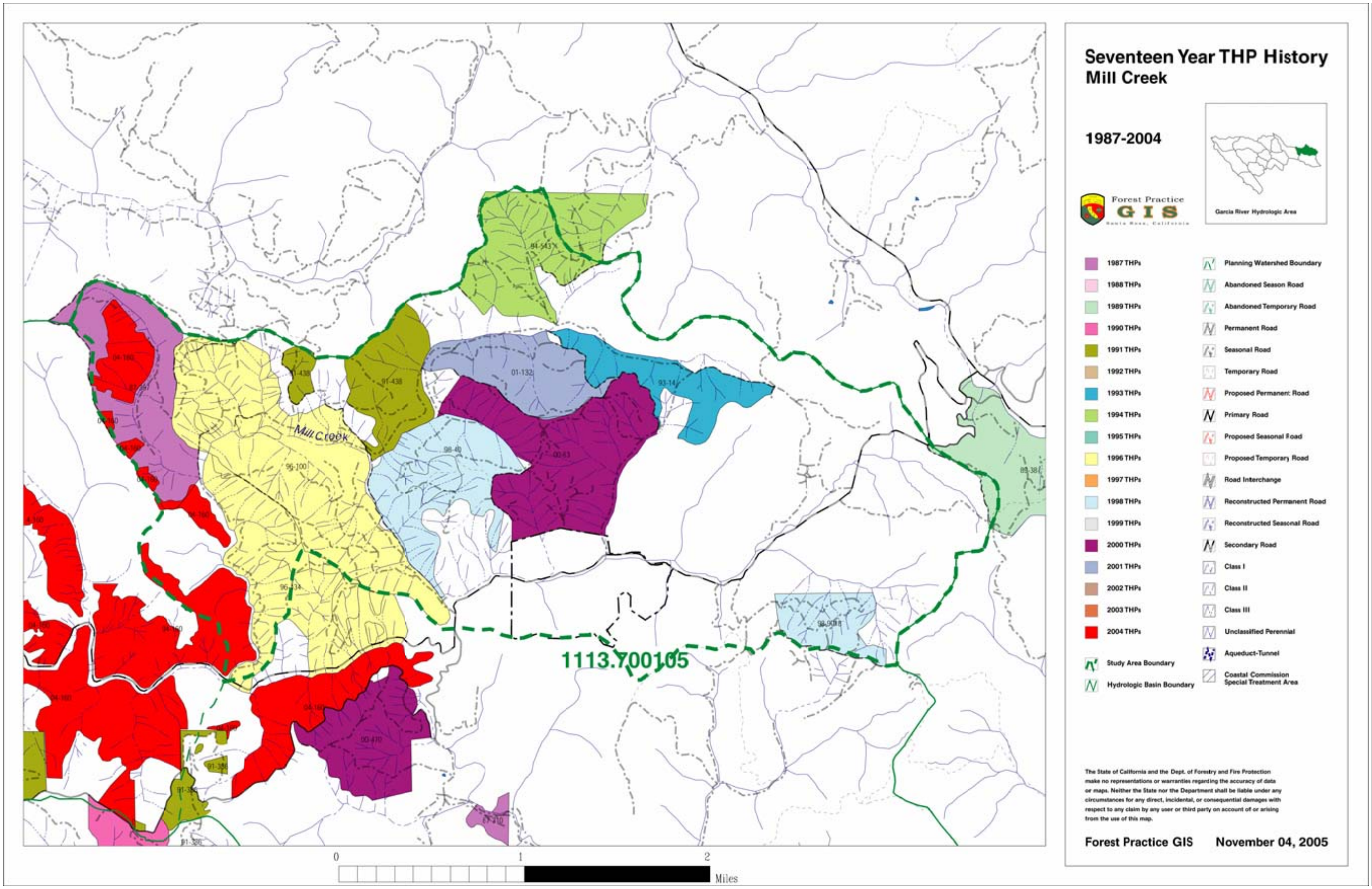
Miles



Seventeen Year THP History
Mill Creek
1987-2004
Garcia River Hydrologic Area
Forest Practice GIS
1987 THPs
1988 THPs
1989 THPs
1990 THPs
1991 THPs
1992 THPs
1993 THPs
1994 THPs
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1999 THPs
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2003 THPs
2004 THPs
Study Area Boundary
Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
Abandoned Temporary Road
Permanent Road
Seasonal Road
Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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November 04, 2005
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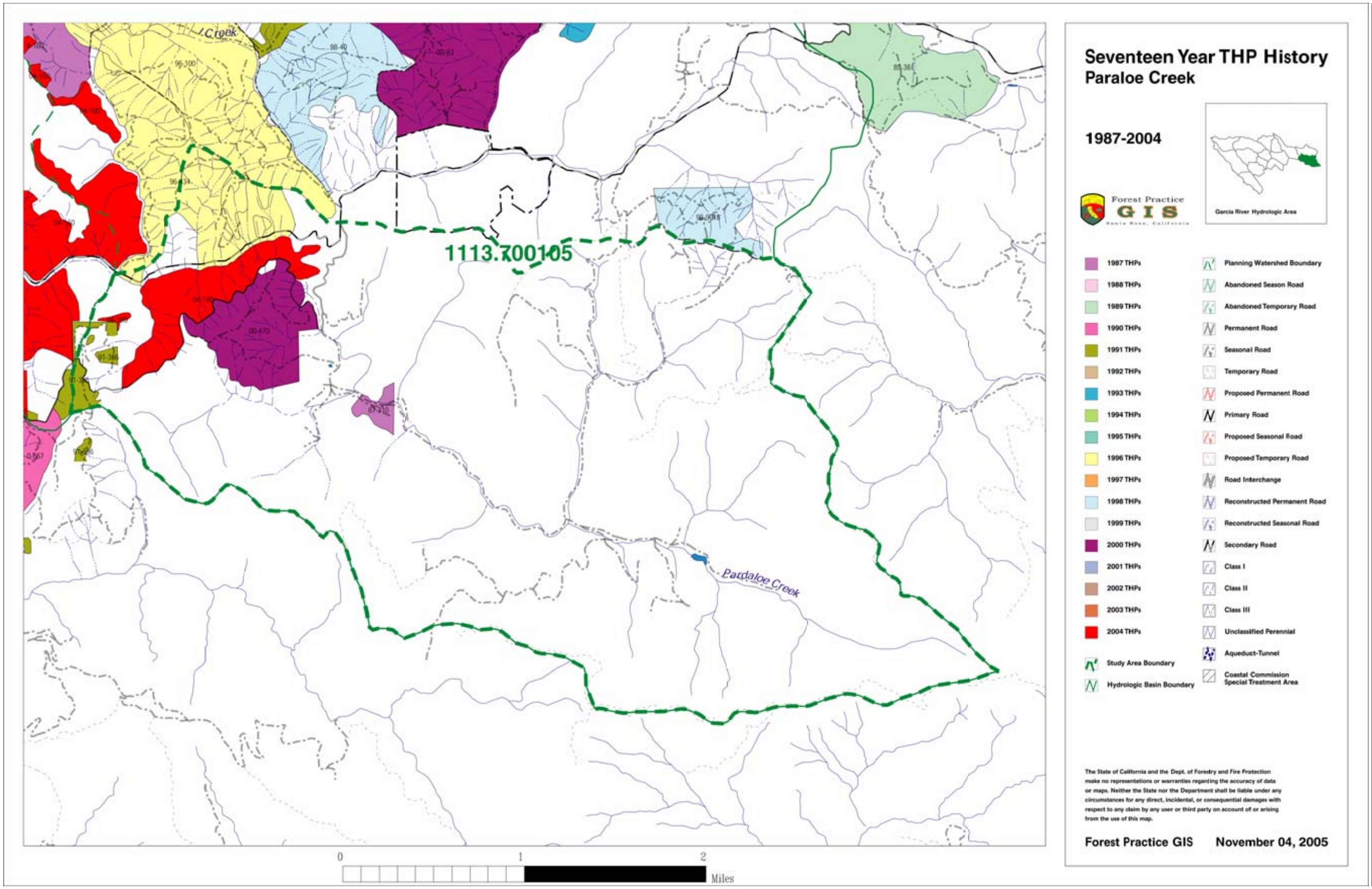
Miles



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1987-2004
Garcia River Hydrologic Area
Forest Practice GIS
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Paraloe Creek
1987 THPs
1988 THPs
1989 THPs
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1993 THPs
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1997 THPs
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1999 THPs
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2002 THPs
2003 THPs
2004 THPs
Study Area Boundary
Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
Abandoned Temporary Road
Permanent Road
Seasonal Road
Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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November 04, 2005
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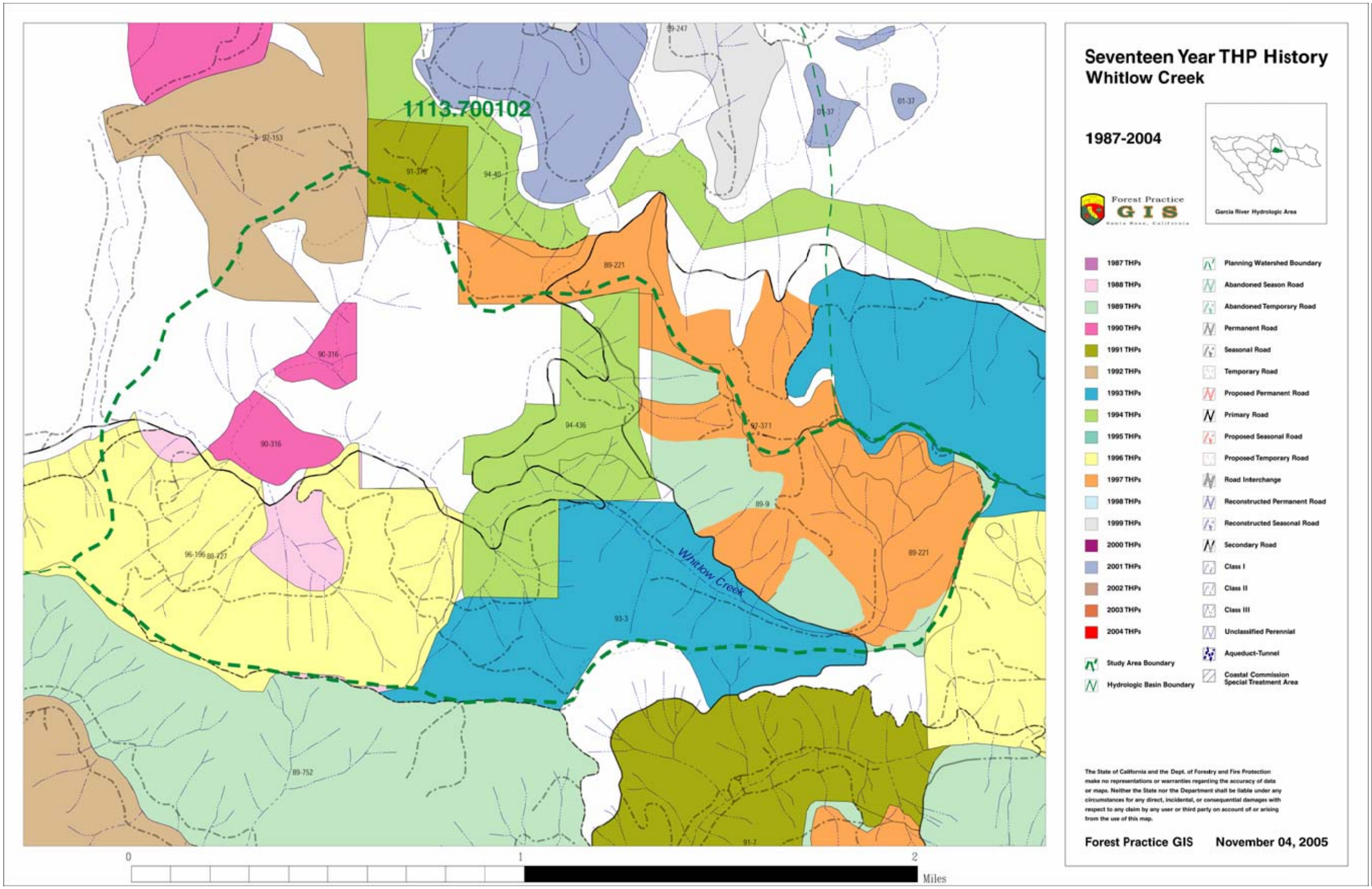
Miles


Seventeen Year THP History
Whitlow Creek
1987-2004
Garcia River Hydrologic Area
Forest Practice GIS
Santa Rosa, California
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1988 THPs
1989 THPs
1990 THPs
1991 THPs
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1994 THPs
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2002 THPs
2003 THPs
2004 THPs
Study Area Boundary
Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
Abandoned Temporary Road
Permanent Road
Seasonal Road
Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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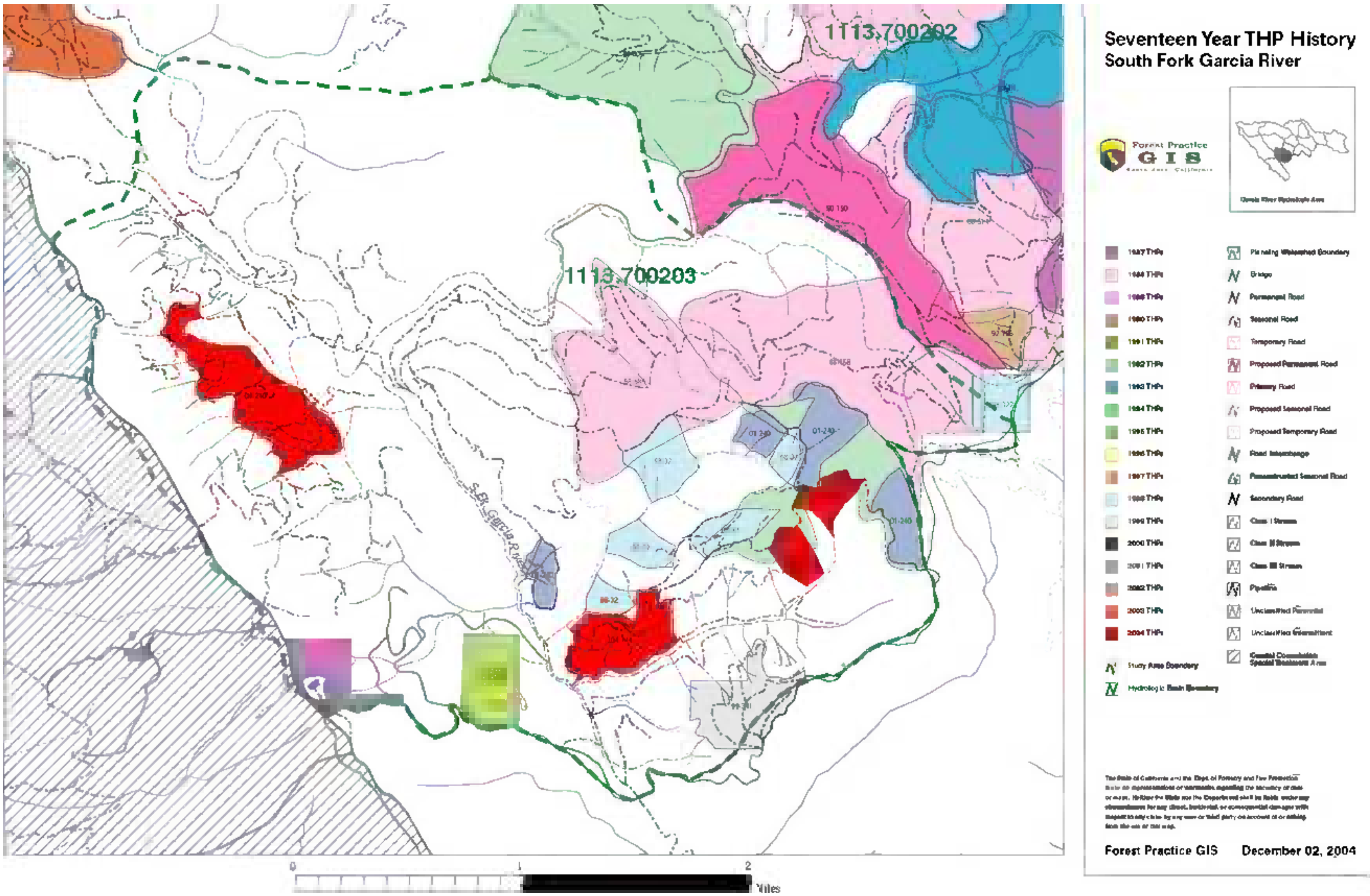
Miles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S.F. Garcia River
Forest Practice GIS
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Miles
1991 THPs
1995 THPs
1999 THPs
2000 THPs
2001 THPs
2003 THPs
2004 THPs
Bridge
Permanent Road
Seasonal Road
Temporary Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
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Class II Stream
Class III Stream
Pipeline
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Hydrologic Basin Boundary
Forest Practice GIS
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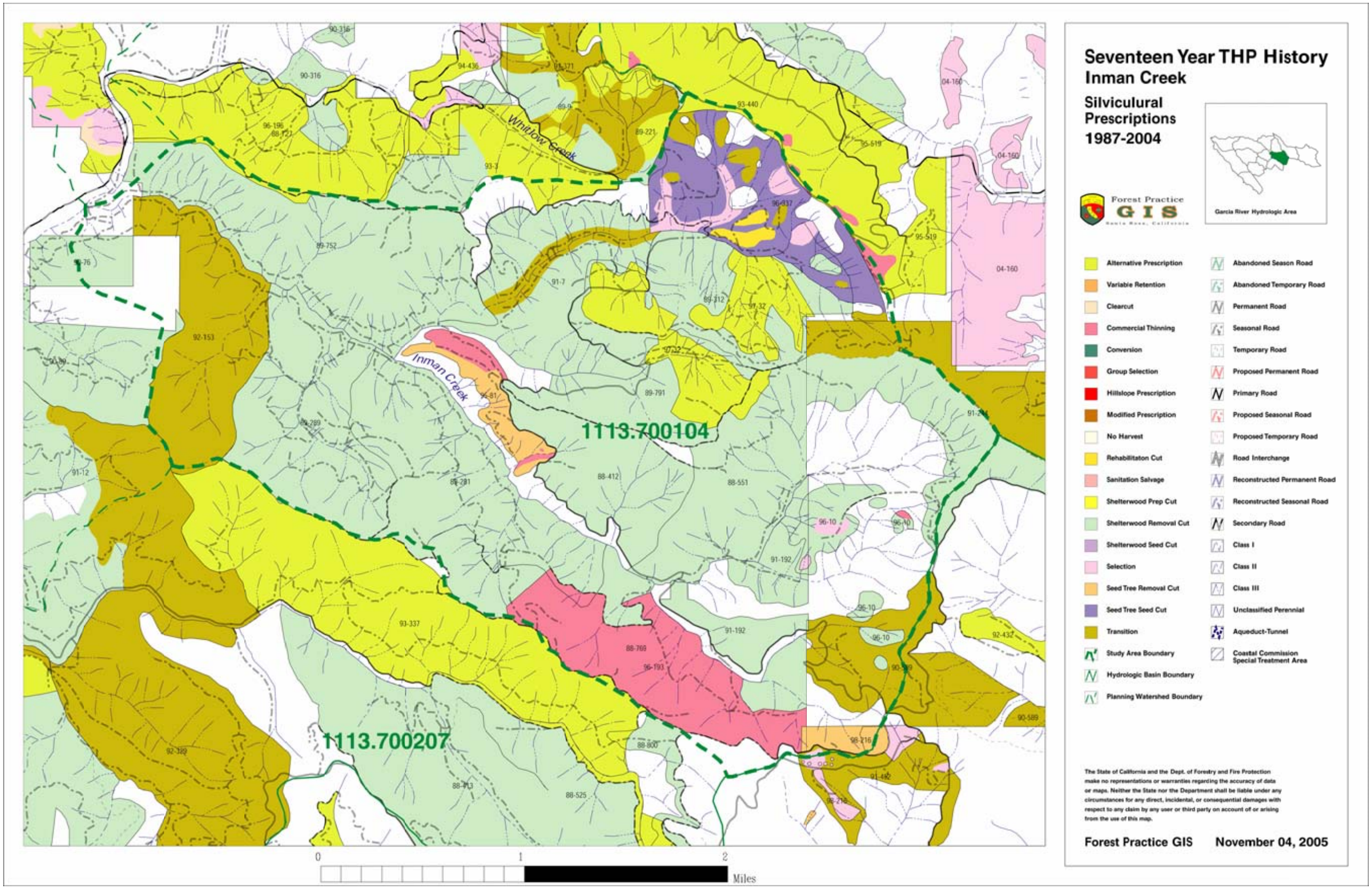

Seventeen Year THP History
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Silviculural Prescriptions
1987-2004
Forest Practice GIS
Garcia River Hydrologic Area
Alternative Prescription
Variable Retention
Clearcut
Commercial Thinning
Conversion
Group Selection
Hillslope Prescription
Modified Prescription
No Harvest
Rehabilitaton Cut
Sanitation Salvage
Shelterwood Prep Cut
Shelterwood Removal Cut
Shelterwood Seed Cut
Selection
Seed Tree Removal Cut
Seed Tree Seed Cut
Transition
Study Area Boundary
Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
Abandoned Temporary Road
Permanent Road
Seasonal Road
Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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November 04, 2005
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Whitlow Creek
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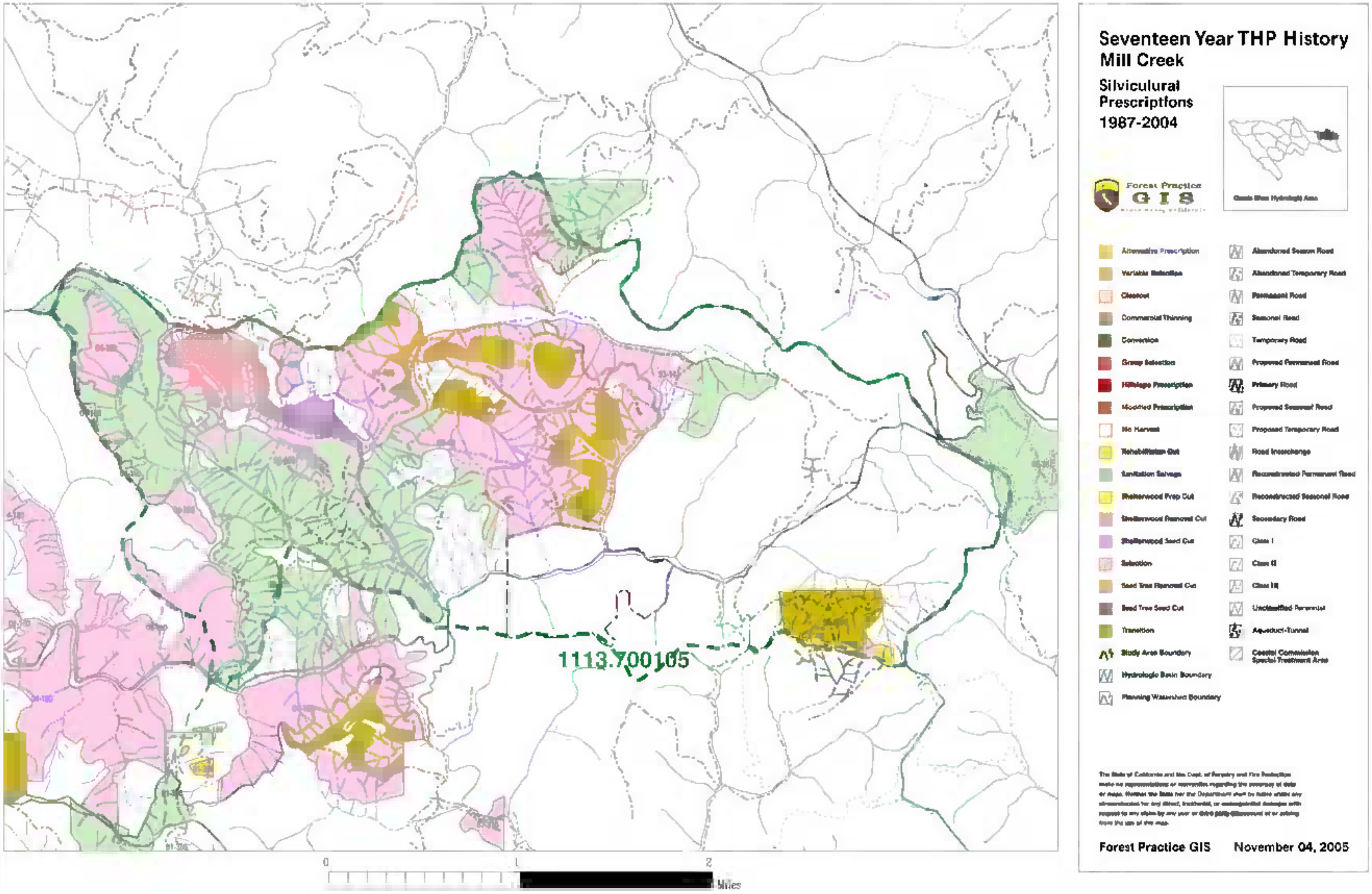
Miles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Commercial Thinning
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Rehabilitation Cut
Sanitation Salvage
Shelterwood Prep Cut
Shelterwood Removal Cut
Shelterwood Seed Cut
Selection
Seed Tree Removal Cut
Seed Tree Seed Cut
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Primary Road
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Proposed Temporary Road
Road Interchange
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Reconstructed Seasonal Road
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Class III
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Coastal Commission
Special Treatment Area
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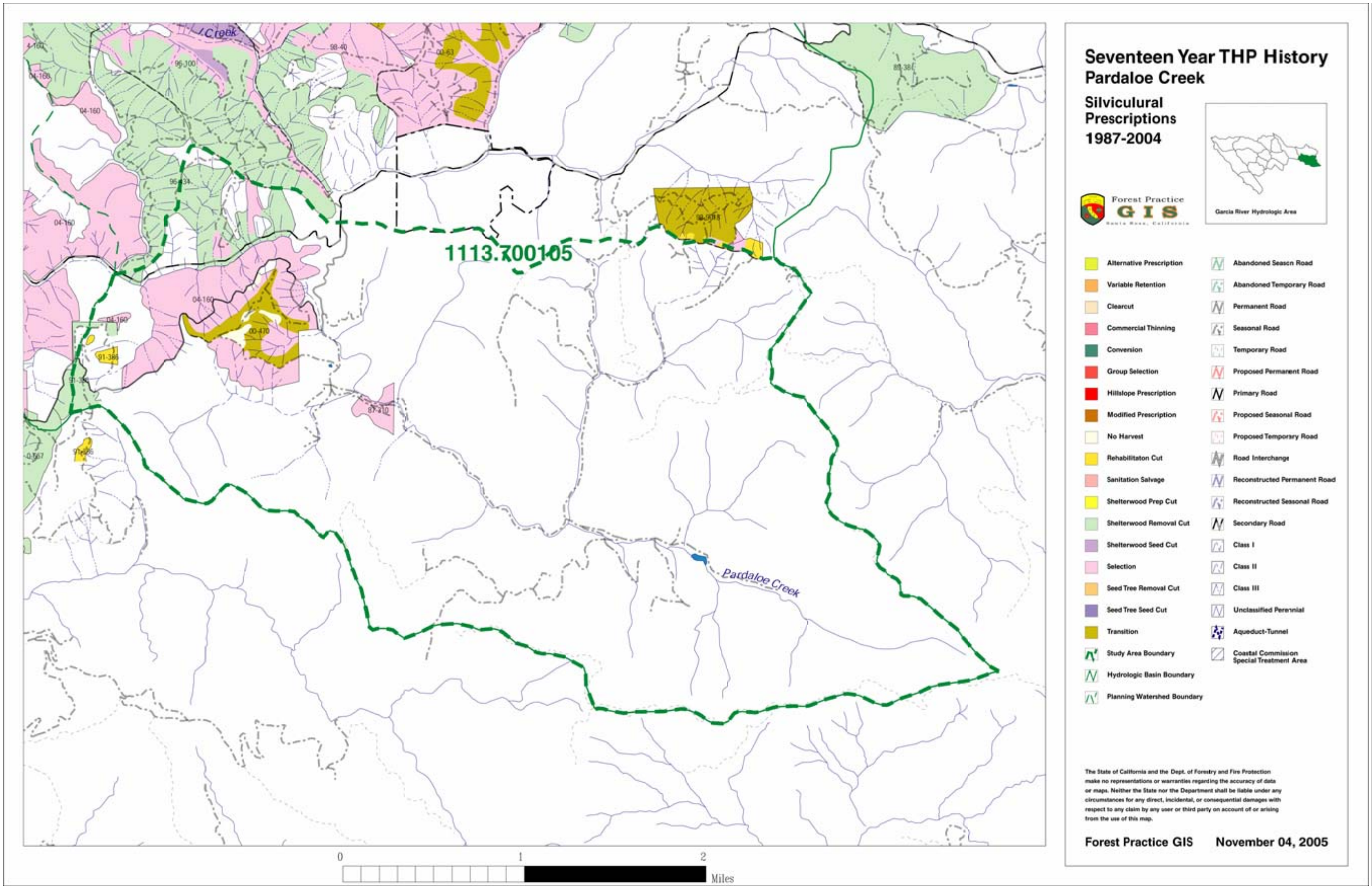
Miles
Forest Practice GIS
November 04, 2005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Variable Retention
Clearcut
Commercial Thinning
Conversion
Group Selection
Hillslope Prescription
Modified Prescription
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Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
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Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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Forest Practice GIS
November 04, 2005
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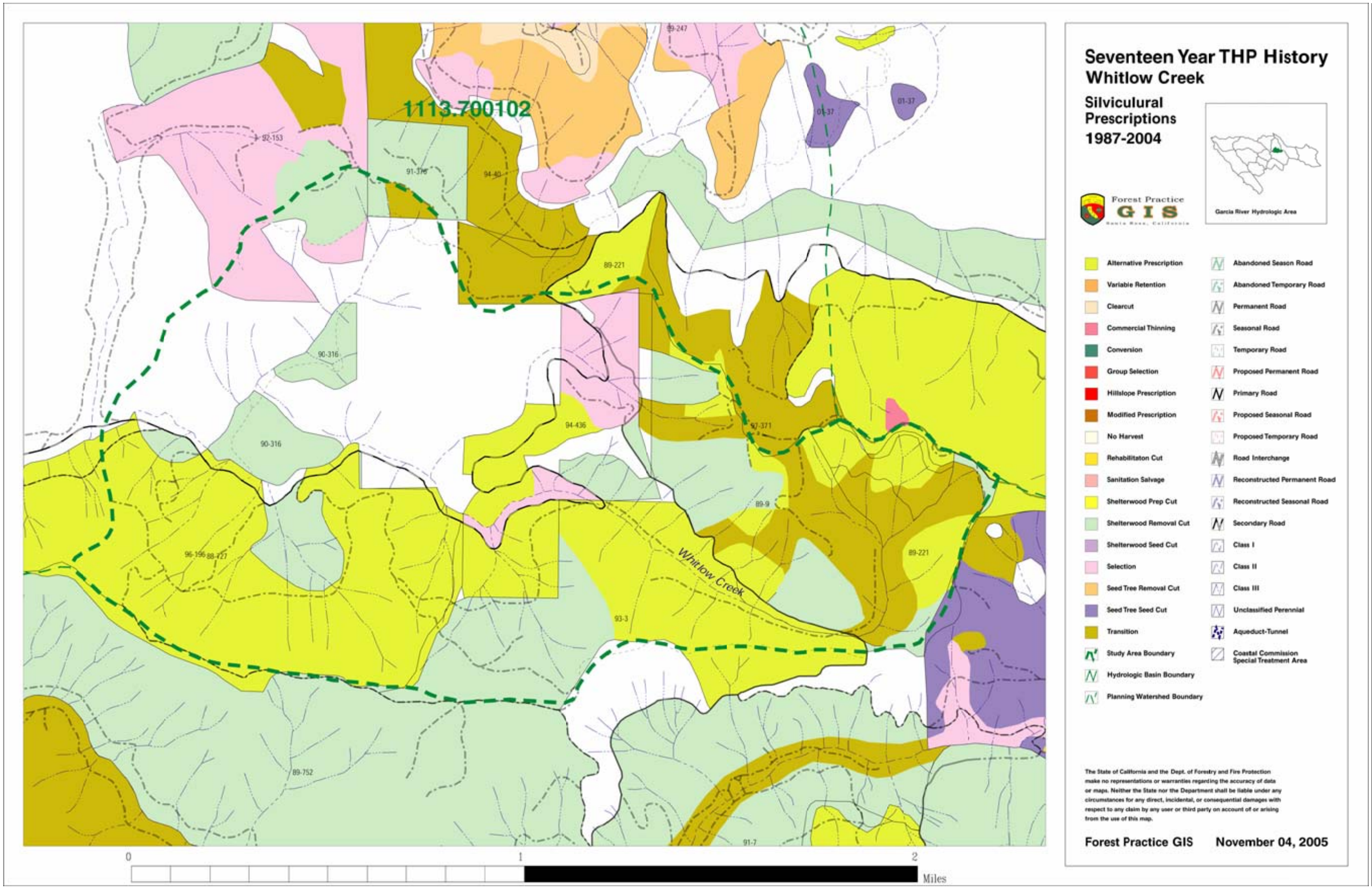

Seventeen Year THP History
Whitlow Creek
Silvicultural Prescriptions
1987-2004
Forest Practice GIS
Santa Rosa, California
Garcia River Hydrologic Area
Alternative Prescription
Variable Retention
Clearcut
Commercial Thinning
Conversion
Group Selection
Hillslope Prescription
Modified Prescription
No Harvest
Rehabilitaton Cut
Sanitation Salvage
Shelterwood Prep Cut
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Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
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Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
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November 04, 2005
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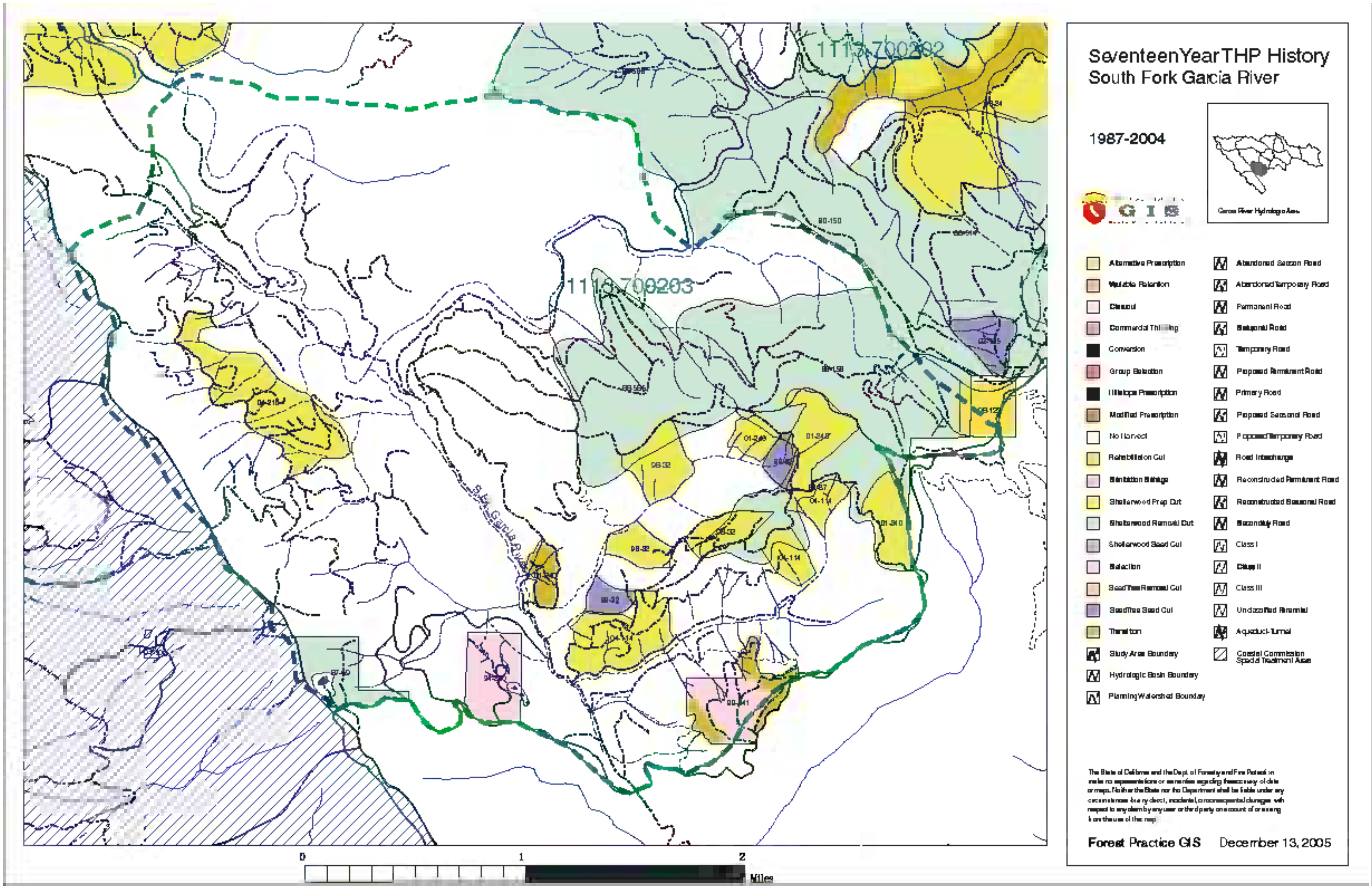
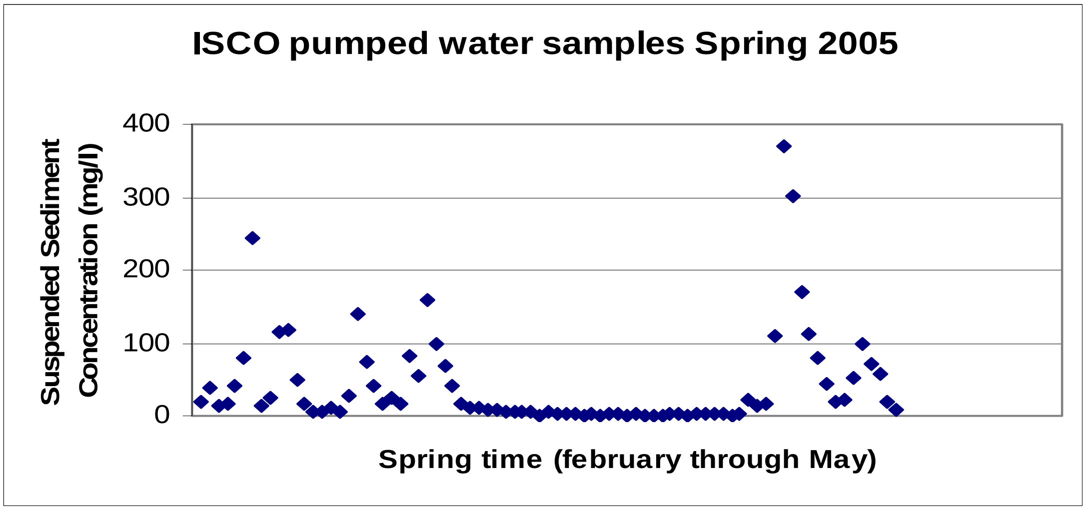
Miles


Seventeen Year THP History
South Fork Garcia River
1987-2004
Garcia River Hydrologic Area
1113.700202
1113.700203
S.F. Garcia River
Alternative Prescription
Variable Retention
Clearcut
Commercial Thinning
Conversion
Group Selection
Modified Prescription
No Harvest
Rehabilitation Cut
Shelterwood Prep Cut
Shelterwood Removal Cut
Shelterwood Seed Cut
Selection
Seed Tree Removal Cut
Seed Tree Seed Cut
Transition
Study Area Boundary
Hydrologic Basin Boundary
Planning Watershed Boundary
Abandoned Season Road
Abandoned Temporary Road
Permanent Road
Seasonal Road
Temporary Road
Proposed Permanent Road
Primary Road
Proposed Seasonal Road
Proposed Temporary Road
Road Interchange
Reconstructed Permanent Road
Reconstructed Seasonal Road
Secondary Road
Class I
Class II
Class III
Unclassified Perennial
Aqueduct-Tunnel
Coastal Commission Special Treatment Area
Forest Practice GIS
December 13, 2005
0
1
2
Miles

LABORATORY ANALYSIS AND INTERPRETATION OF WATER SAMPLES

ISCO water samples from Mill Creek were collected at various dates, times, and stages between February and May, 2005. The following text describes our analysis and interpretation of the water samples we collected on Mill Creek by the ISCO automatic pumping sampler and from the grab samples we collected from the other Garcia River streams in our study.

Figure 32. ISCO Water Samples extracted from Mill Creek water column over time



ANALYSIS A: DTS-12 TURBIDITY ON LABORATORY TURBIDITY
This analysis was performed to examine whether field turbidity equipment (DTS-12) remained well-calibrated over time by comparing DTS-12 turbidity to lab grade turbidity. Mill Creek water samples were analyzed for laboratory grade turbidity (superior in accuracy to field grade turbidity). Results were compared to samples measured at the same time and place with the field grade instream turbidimeter.

ANALYSIS B: LAB TURBIDITY ON SUSPENDED SEDIMENT CONCENTRATION
We also utilized the ISCO samples to compare turbidity with suspended sediment concentration (SSC). In similar studies, SSC is the real variable of interest and turbidity is used as a surrogate variable to predict suspended sediment concentration (SSC). Turbidity can be measured digitally in the field by machines (relatively efficient) but SSC requires actual water samples and laboratory analysis (relatively expensive). Correlation can be established between SSC and NTU when single water samples are examined for both parameters: SSC values were plotted against turbidity values from each water sample yielding a regression equation that defines the relationship whereby SSC is predicted from ntu.

The subset of points swarming upwards from the origin in relation to other points may represent a sub-population. We took the two populations apart from each other and graphed them separately. In attempt to explain why the low range in turbidity would have a higher concentration of suspended sediment, we looked to Hysteresis. We divided the whole group of points again and separated them into falling limb and rising limb points. For example, if a sample was taken as the stream was rising in stage, it was grouped with rising limb points. The outlying data point had relatively high sand content (see photo).

ANALYSIS C: SSC ON DIGITAL FIELD TURBIDITY
If we were to utilize the field NTU recorded at Mill Creek to predict SSC, we would regress SSC on field turbidity and use that regression equation to calculate SSC for each turbidity measurement recorded. The regression equation technique can be applied to all Mill Creek turbidity data in order to predict corresponding SSC. This might be done should we desire to perform an "area under the curve" type analysis and sum the areas to get to total suspended sediment yield.

ANALYSIS D: COMBINED GRAB SAMPLES; LAB TURBIDITY ON SSC
This analysis presents the results of water sample analysis from the grouped "grab" samples we pulled from each of the Garcia River Tributaries we sampled, combined. One outlier was pulled from the main graph. We could not explain why this South Fork sample had a much higher SSC than expected from a moderately turbid sample. We concluded that the sample was not kept in a cool dark location post-sampling or there was an error in the lab analysis. Although efforts were made to obtain grab samples from streams at high flow levels, all grab samples recorded suspended sediment concentrations below 60 mg/L and under 40 ntu. Many samples collected by the ISCO recorded much higher values than our grab samples. This is because the ISCO is stationed on site and works nights, weekends, and in extremely wet weather without need of travel or sleep.

ANALYSIS E: INDIVIDUAL TRIBUTARIES; LAB TURBIDITY ON SSC
This analysis examines the tributaries individually. Grab samples are divided according to which tributary stream they were pulled from, then SSC was graphed on turbidity (ntu). This is appropriate in that streams from the same watershed can have different sediment weights for a given particle size. This would reflect in different SSC for a given turbidity.

Analysis A: Figure 33: Laboratory Turbidity on Field Turbidity, February 2005

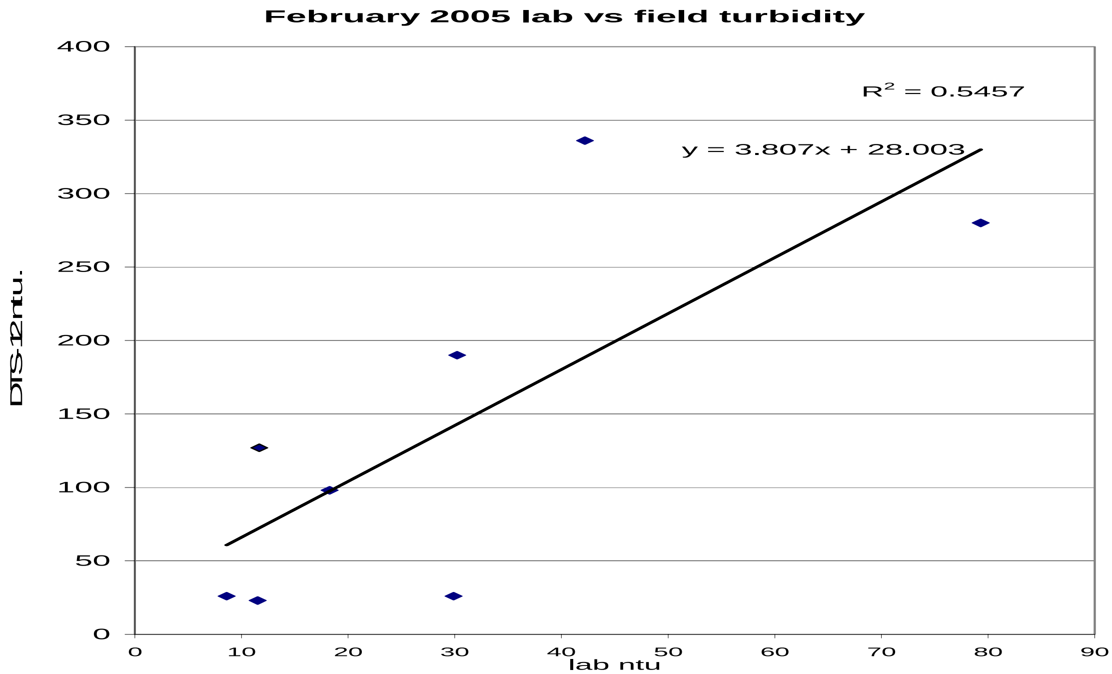
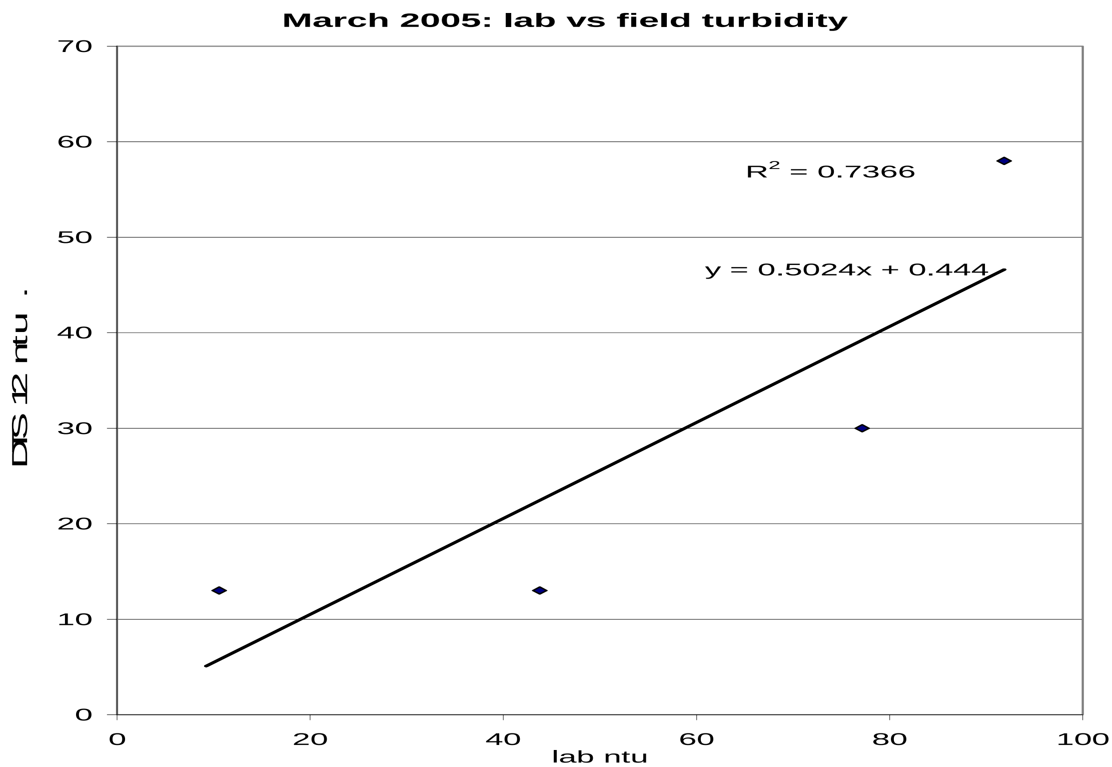
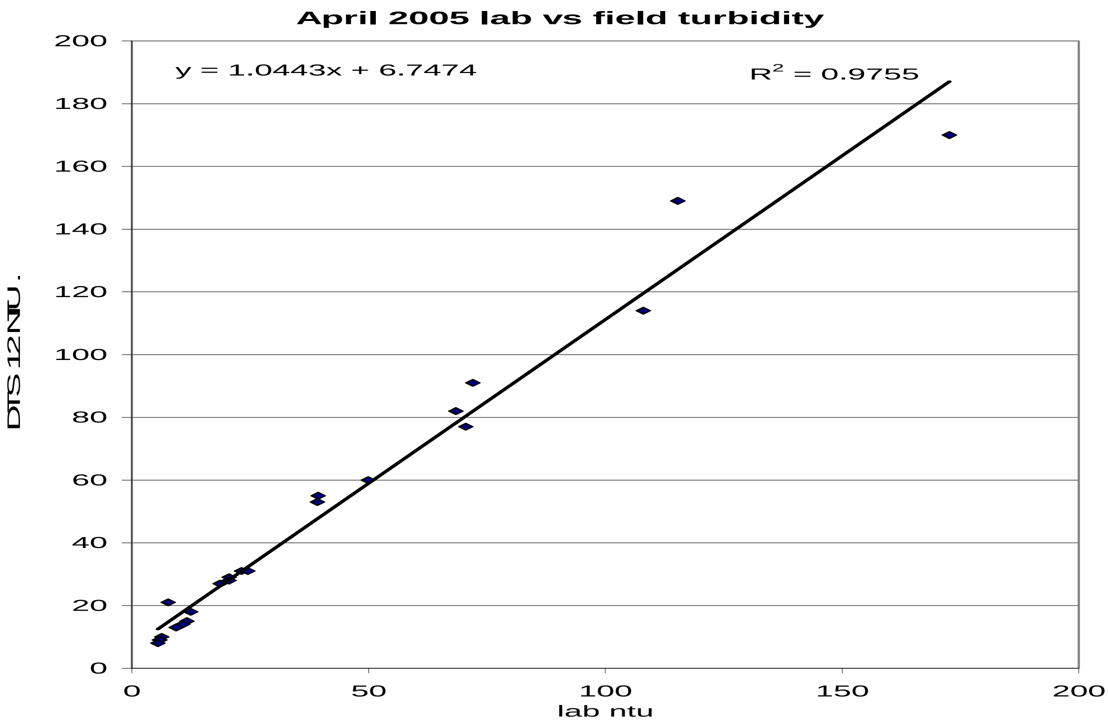


Figure 34: Laboratory Turbidity on Field Turbidity, March 2005



Analysis A: 35: Laboratory Turbidity on Field Turbidity, April 2005

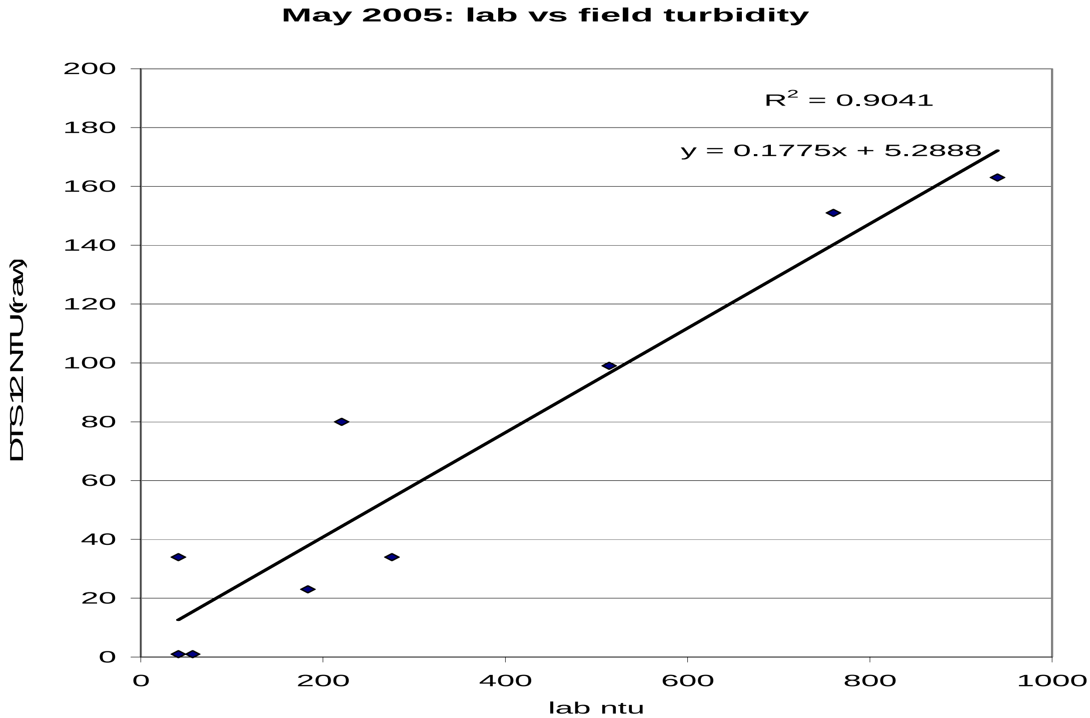


Analysis A

On Mill Creek turbidity sensing equipment appeared to remain accurate during the ISCO phase of our experiment. Correlation values from ISCO-pumped lab turbidity and DTS-12 field turbidity appeared poorly correlated as a group but when we separated them into chronological groups, R^2 values increased to between 0.55 and 0.98. Why would the lab analyzed turbidity vary from DTS-12 logged turbidity differently over time if not for sensor “drift”? In discussing this issue with Jack Lewis, Statistician at USFS Redwood Sciences Lab, it appears rather commonly that field sensors read somewhat higher or lower as the source of the turbidity changes in a creek (Jack Lewis, 2006 personal communication).

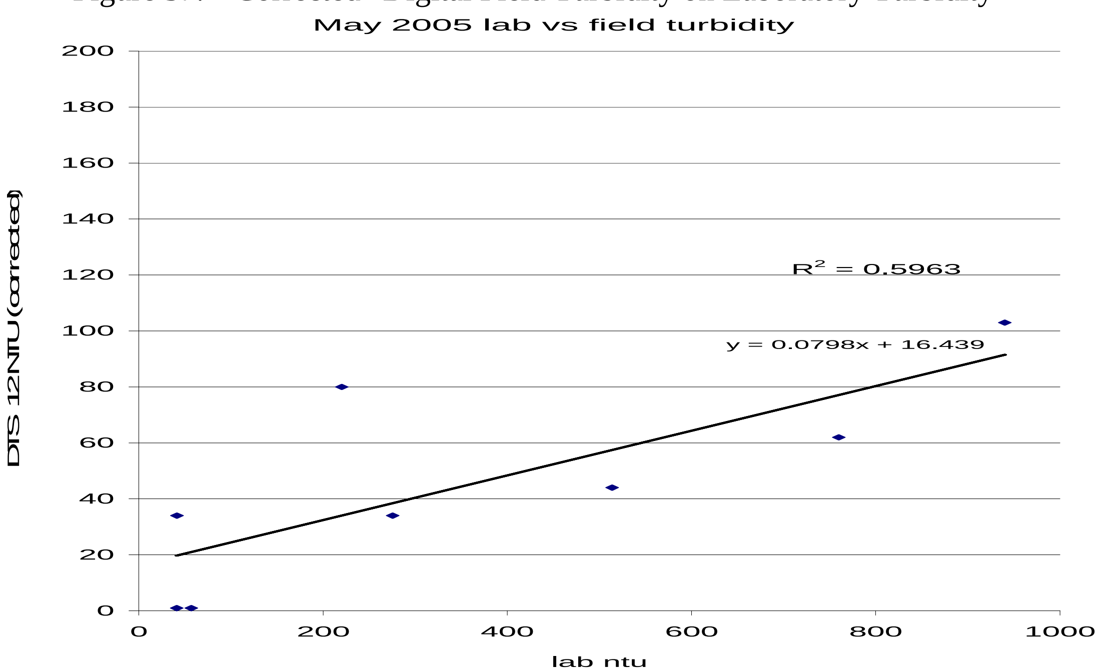
In May, 2005, turbidity data from Mill Creek was corrected because the “windshield wiper” arm on the turbidity sensor failed. In comparing raw and corrected turbidity values with laboratory tested turbidity (assumed to have greater accuracy) (Figure X) it appears that raw data was more accurate with a R2 value of 0.90 vs 0.60 for corrected data. Data were corrected at Mill Creek and at other tributaries where digital turbidity data was suspected to be erroneously high and so DTS-12 reported turbidity was reduced to be more in line with expected turbidity. This experience suggests that we may have reduced Mill Creek turbidity too far in the correction process. In this case, despite the wiper arm failure, the raw DTS-12 field turbidity was better correlated to lab turbidity than was the “corrected” turbidity. The wiper arm is critical where periphyton populations are high as in a solar production autochthenous food web but Mill Creek has a dense canopy, and is allochthenous in nature.

Analysis A: Figure 36. Raw Digital Field Turbidity on Laboratory Turbidity



Uncertainty in some displayed raw data suggested it was not realistic. But in this case, it appears that the raw data was actually more accurate than the "corrected" data. Water sample analysis was not available at the time the corrections appeared necessary and insufficient water samples negate the opportunity to examine accuracy of corrected data from the other streams.

Figure 37. "Corrected" Digital Field Turbidity on Laboratory Turbidity



Analysis B Figure 38. All Mill Creek ISCO Water Samples: SSC on Turbidity

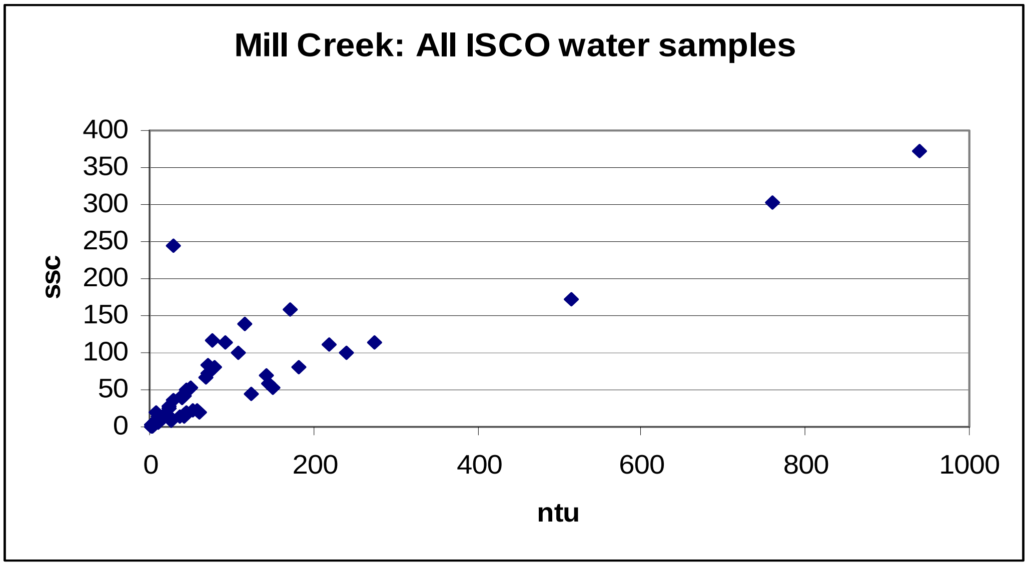
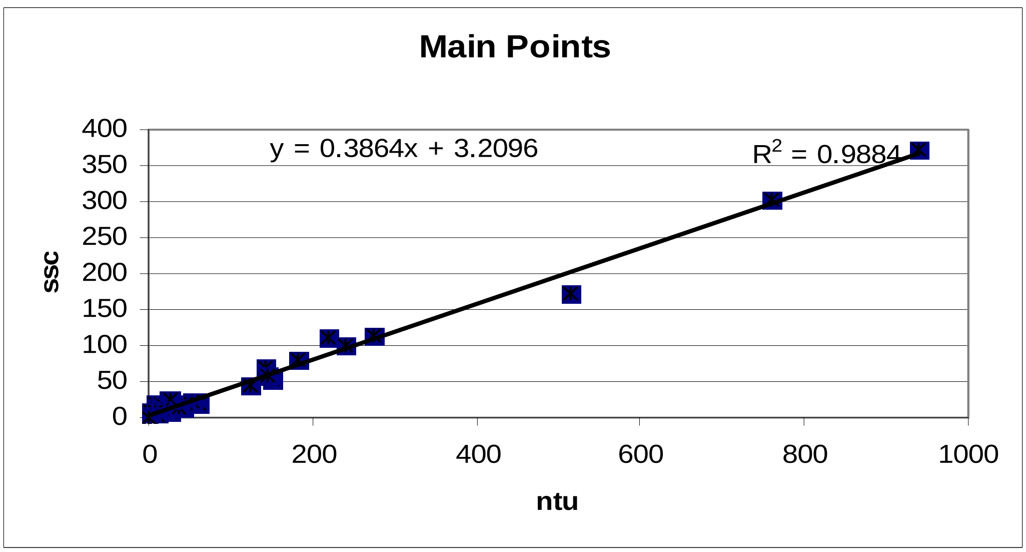


Figure 39. Mill Creek ISCO Water Samples: SSC on Long Range Turbidity



The two graphs above present all the Mill Creek water samples pumped from the ISCO automated sampler between February and May, 2005. The top graph presents them all together, including the outlier referred to earlier. The lower graph has "main points" that draw a diagonal between the graph's origin at 0 and the high ends of the data range.

Next we exhibit the "short range" (middle cloud of points) points graphed with both the outlier included and with the outlier omitted, and separates the "short range points" from the data set.

Figure 40.
Mill Creek ISCO Water Samples: SSC on Short Range Turbidity (outlier removed)

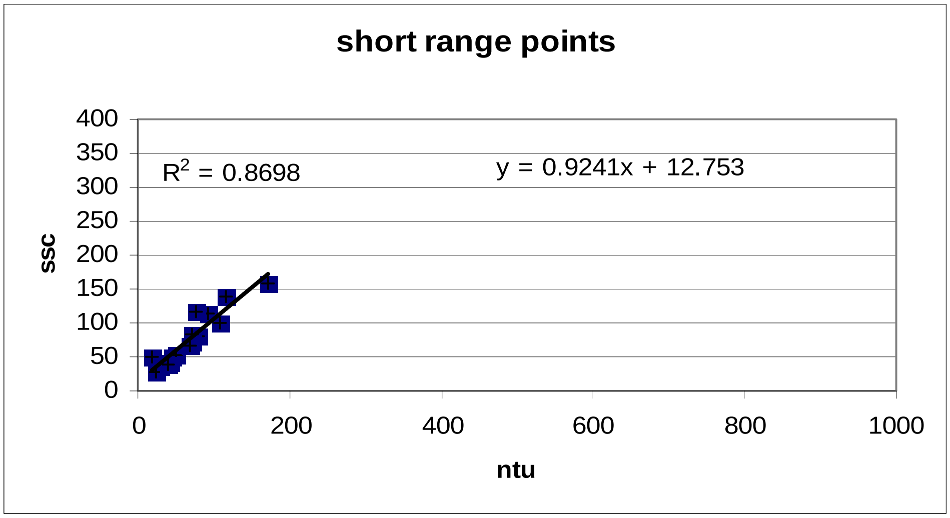
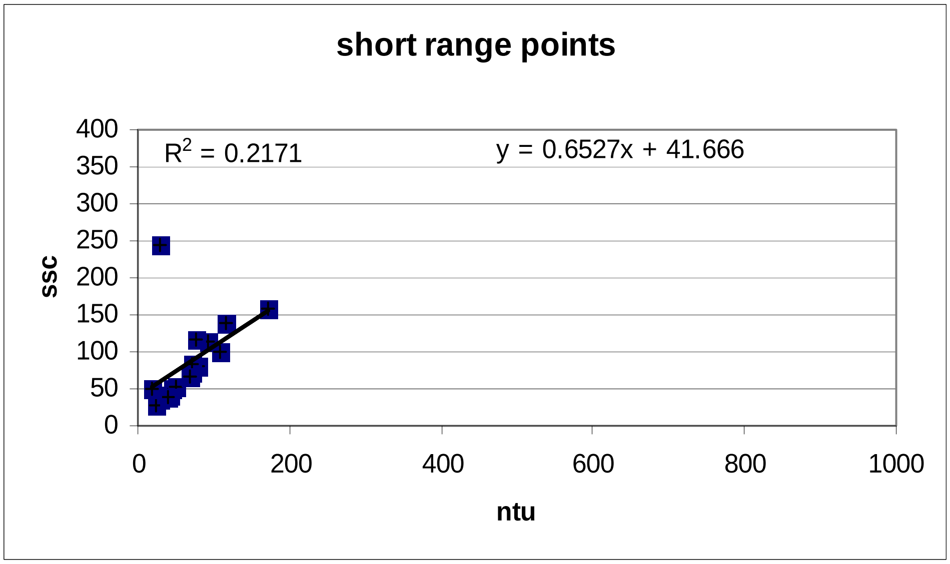


Figure 41. Mill Creek ISCO Water Samples: SSC on Short Range Turbidity



Correlations between SSC and Turbidity analyzed from the same ISCO samples show two distinct relationships and a prominent outlier. The long range or "Main" points show a lower SSC with a given turbidity than the short range points. This could be explained perhaps if the short range sub-populations happened to correspond to rising limb samples on a storm hydrograph while long range points represented falling limb samples on a storm hydrograph, a phenomenon termed "hysteresis". Alternately, long range points might have organic matter suspended as well as inorganic particulate. Inorganic particles would result in a higher SSC than would organic particles in suspension when turbidity is equivalent. One prominent outlier exists within the short range samples. This outlier had visibly more sand in it than the other samples from this analysis. This resulted in a much higher SSC for a moderate turbidity (see photo).

Analysis B: Hysteresis

Figure 42. Mill Creek Hysteresis: Rising Limb Water Samples

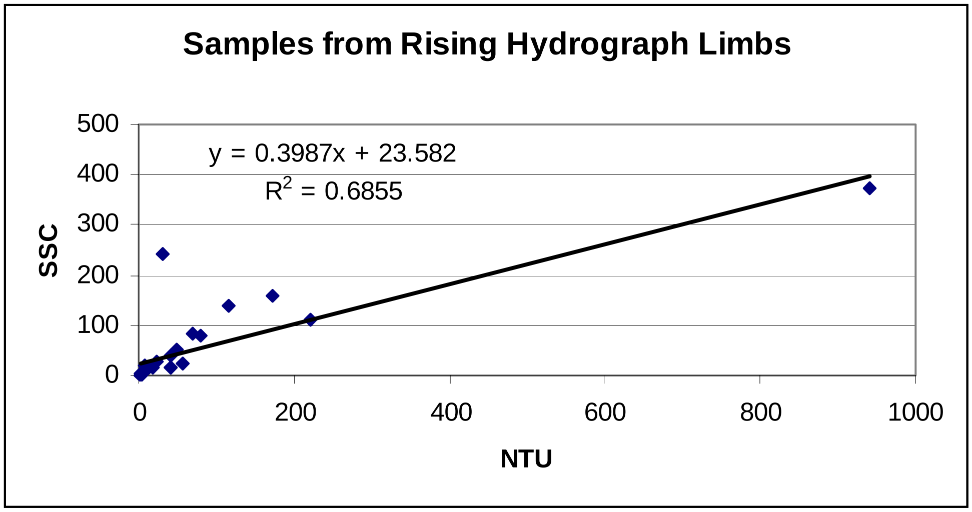
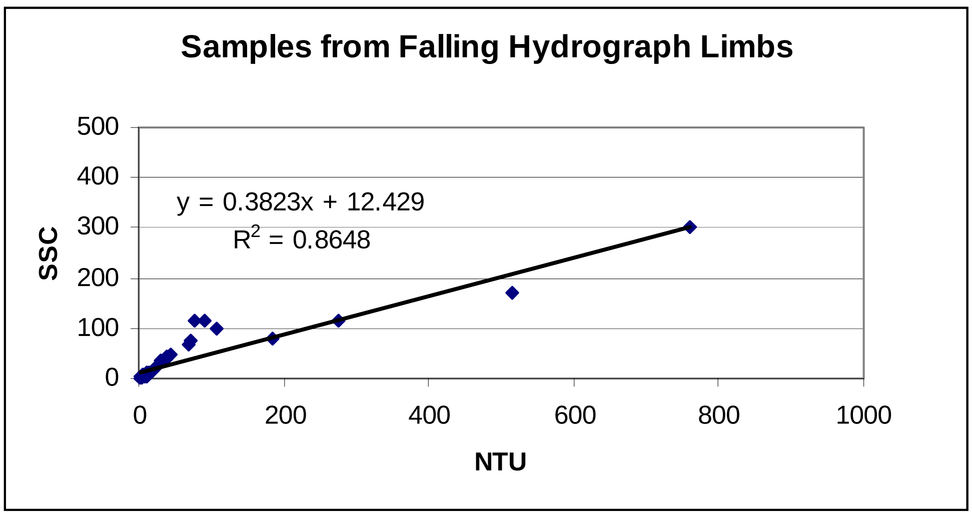


Figure 43. Mill Creek Hysteresis: Falling Limb Water Samples



It appears that short range and long range points occur in both rising and falling limb categories. However examining sediment residues collected from water samples on the glass membrane filters suggests long range points have more organic residue than short range points. Some short range residues appeared to contain a small sand fraction and one of these has a large sand fraction (the one prominent outlier presented in Analysis B). The outlier with unusually high suspended sediment concentration did occur on the rising limb of the hydrograph, as might be expected.

Analysis B: Photographs of filtered sediment residues from winter water samples

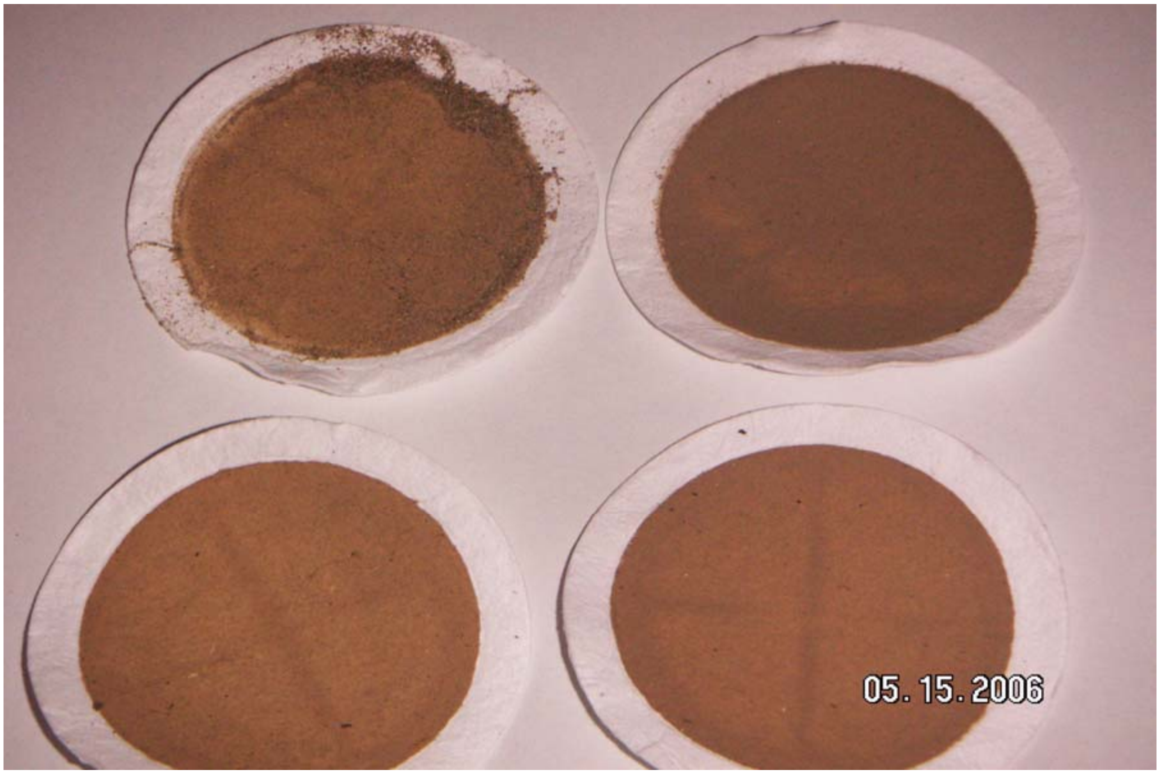
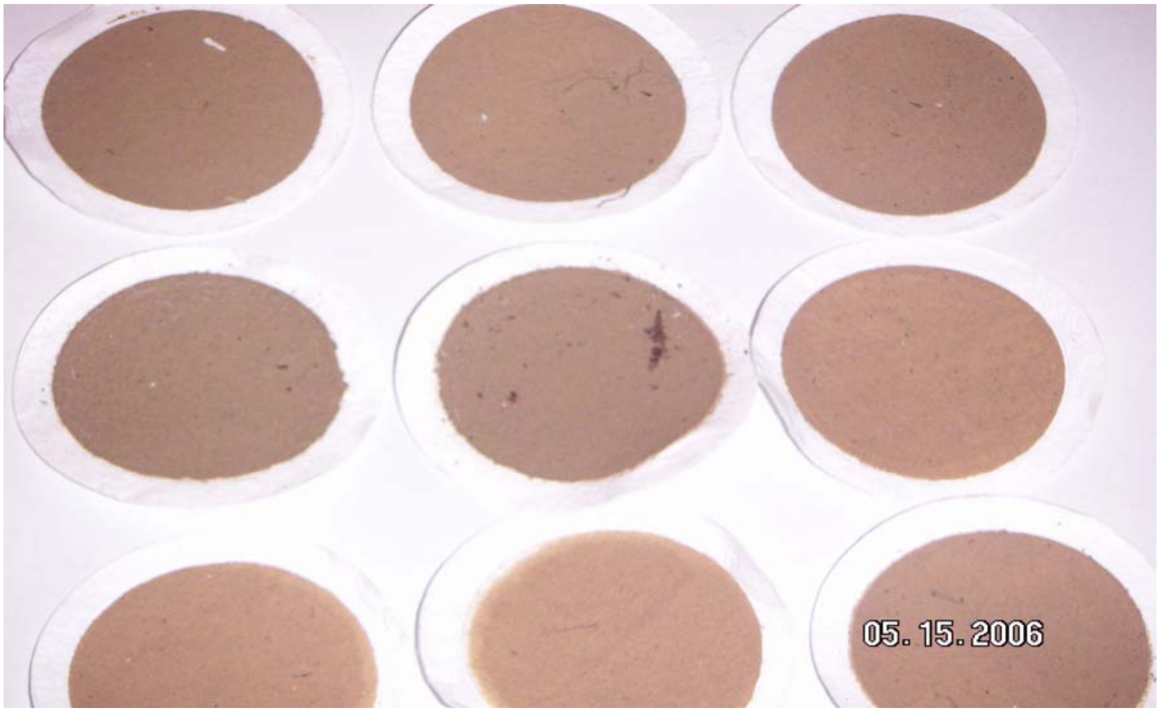
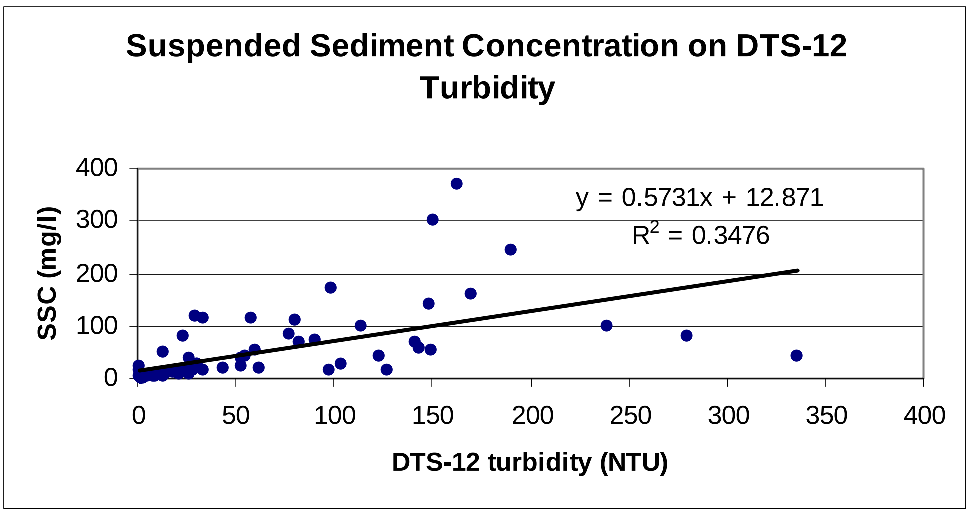


Photo 12. Mill Creek filtered suspended sediment residues: Above: Short Range example. The sample in top left corner includes a heavy sand fraction unlike others in the group having some sand but mostly finer silts and clays. Below: Long Range Points, including organic particles, particularly in the center sample.



Analysis C
If we were to want to convert all the turbidity recorded in Mill Creek to Suspended Sediment Concentration, perhaps to estimate total suspended load, we would first regress SSC on field turbidity. The R-squared would evaluate how accurate the prediction of SSC is likely to be from turbidity. This is the relationship graphed as Analysis C. In this graph the mid-range turbidities have wide variation in corresponding SSC, and after consulting our filters and their residues, we believe the breadth in SSC stems from the relative fraction of organic matter suspended in the water column. With this level of precision, the error imparted to a suspended sediment yield calculated from this data source would have to be considered a "ball park" figure.

Figure 44. Mill Creek Suspended Sediment Concentration on Digital Field Turbidity



Analysis D:

Figure 45. Grab Samples from All Garcia Tributaries Sampled

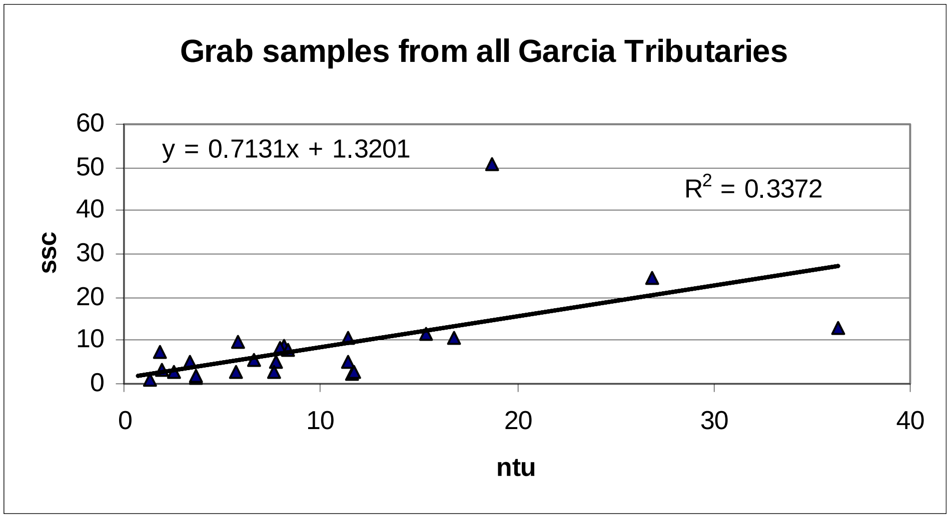
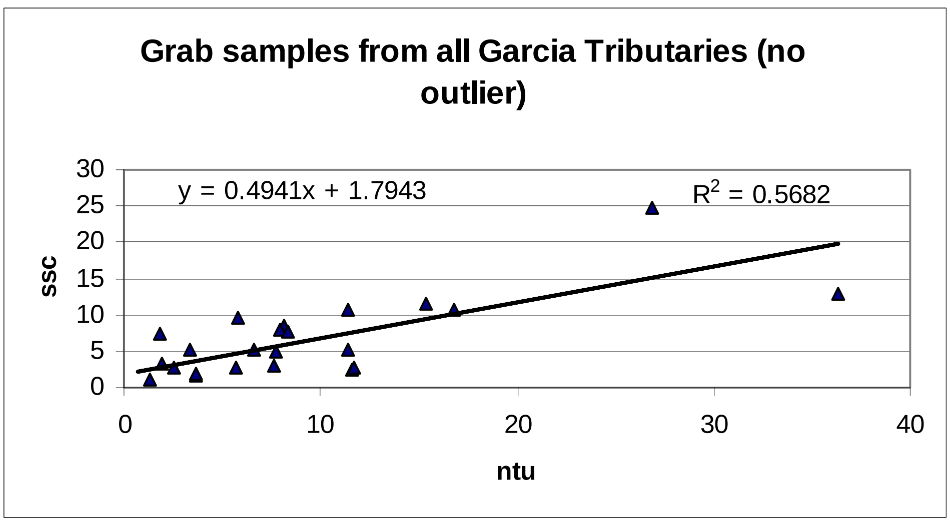


Figure 46. Grab Samples from All Garcia Tributaries Sampled (outlier omitted)



Although efforts were made to obtain grab samples from streams across the range of conditions and especially at high flow levels, all grab samples obtained contained suspended sediment concentration levels below 60 mg/L and turbidity levels below 40 ntu. The R^2 value for grab sample suspended sediment concentrations on lab turbidity is 0.34. One outlier had a higher SSC value than others for the given turbidity. A visual assessment of this sample's filtered residue indicated that it had more sand than other samples. When this outlying sample was removed the R^2 value improved to 0.57. It is logical to presume that tributaries in the same watershed would exhibit similar relations between turbidity and SSC. Yet to isolate any differences between tributaries and improve our R-squared correlation coefficient, we separated them out in Analysis E.

Analysis E:

Figure 47. Mill Creek Grab Samples: SSC on Lab Turbidity

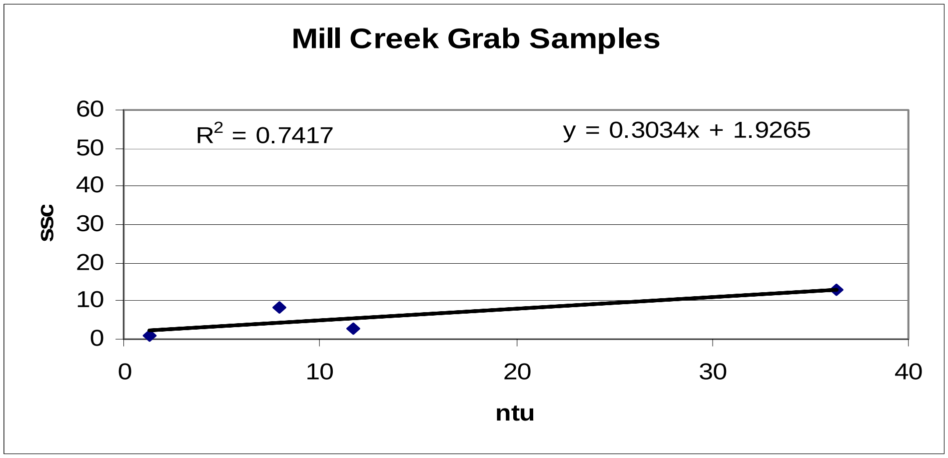


Figure 48. South Fork Grab Samples: SSC on Lab Turbidity

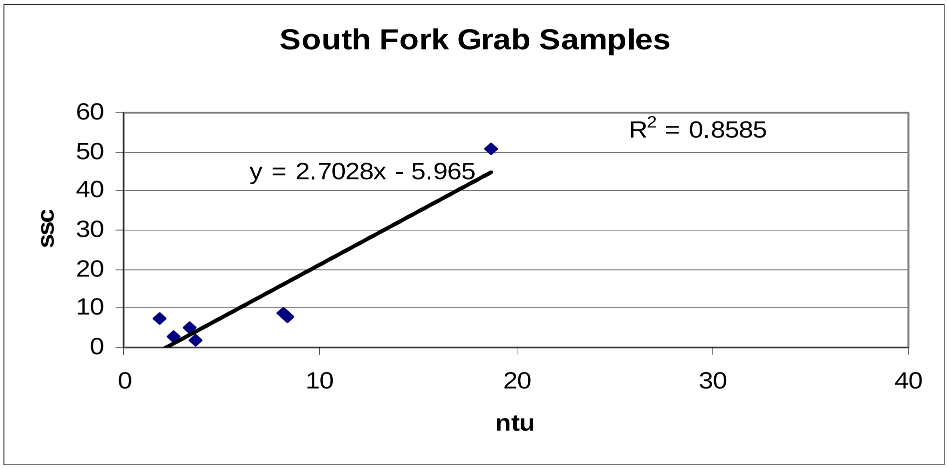


Figure 49. Pardaloe Grab Samples: SSC on Lab Turbidity

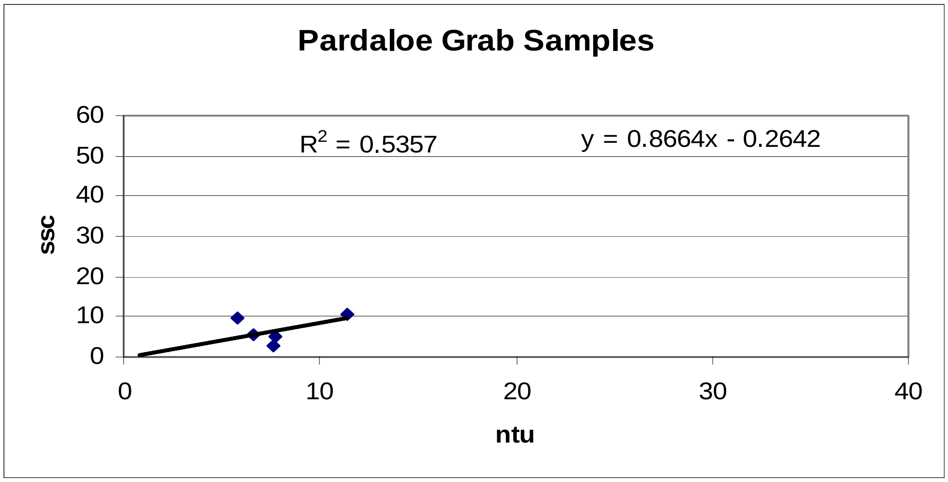


Figure 50. Whitlow Creek Grab Samples: SSC on Lab Turbidity

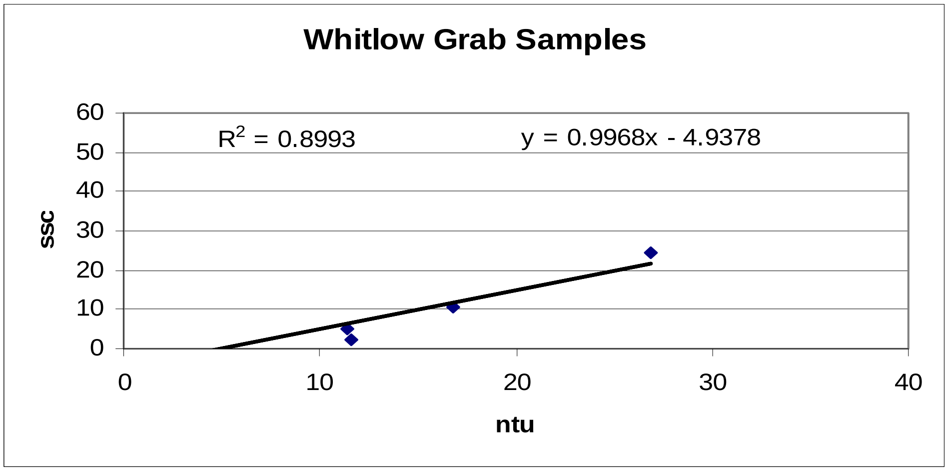


Figure 51. Inman Creek Grab Samples: SSC on Lab Turbidity

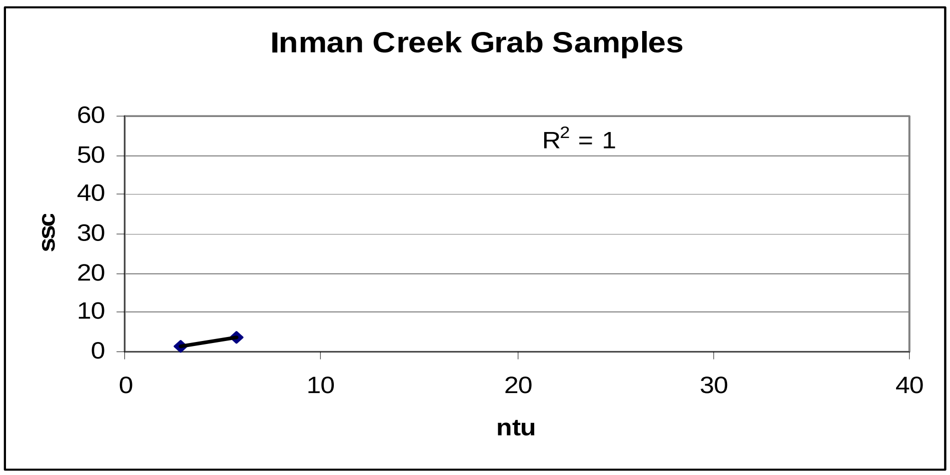
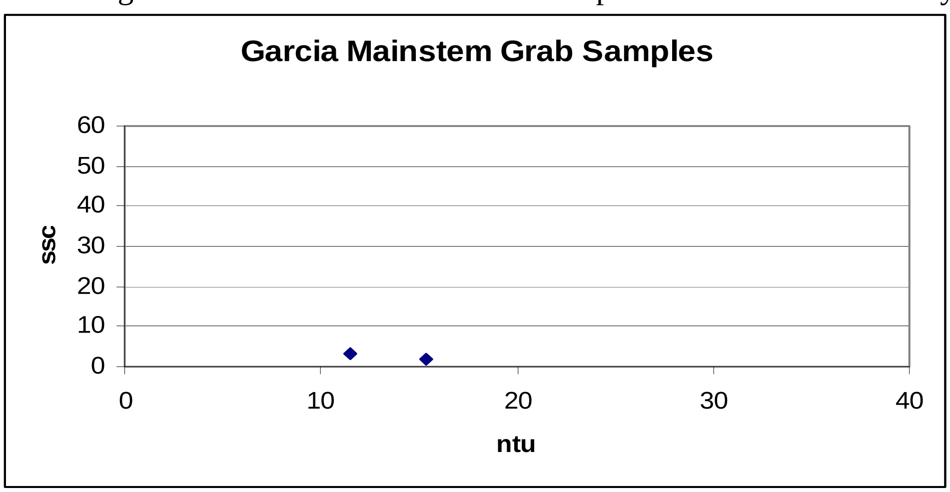


Figure 52. Garcia Mainstem Grab Samples: SSC on Lab Turbidity



Analysis E
Separating the tributaries certainly did increase the R-squared correlation coefficients indicating tighter correlation. In looking at the correlation between SSC and turbidity by tributary we can see that South Fork of the Garcia stands out because the relationship is much steeper indicating heavier particulate for a given turbidity. The presence of the outlier at South Fork contributes to the steepness of this relationship; however, we saw no reason to remove it especially because this sample reflects the highest turbidity we sampled at South Fork. This outlier appeared to have more sand in it than other samples, which suggests that it would have a higher SSC value for the given turbidity. It may be that at higher turbidity levels there is more sand present within the water column on this tributary. On the other hand, there may have been an undetected sampling mistake, which would result in a shallower correlation more similar to those from other tributaries.

In looking at the two grab samples from the Mainstem Garcia, one can see that SSC was surprisingly low for the two moderate turbidities sampled. This is probably due to the relatively high density of algae suspended in the mainstem compared with algae levels in tributaries. Visual assessment of lens colonization on the mainstem suggests that there is much more periphyton present there than in tributaries where no lens colonization occurred. Suspended algae that lend the rather green water color of the mainstem captured in the cover photo may also be experienced as turbidity. The greater solar exposure experienced by the river given the high fraction of open canopy above it explains why more photosynthetic organisms live there as compared to the limited open water surface in tributaries where riparian trees nearly close the canopy.

SUMMARY INTERPRETATION OF WATER SAMPLES ANALYSES A-E

A: Dts-12 Turbidity On Laboratory Turbidity
Data from the ISCO sampler shows more accurate correlations between digital field grade turbidity and the more precise laboratory turbidity, at higher turbidity values. This suggests that the higher turbidity values obtained with the DTS-12 Digital Turbidity Sensor should be more accurate than low turbidity values. This was by design. We requested factory calibration to the higher end of expected values because we thought it was more important to get the high values correct than the lower values in this application. The relationship of lab grade turbidity to field grade turbidity changes over time. One regression equation is not adequate to describe this relationship as is suggested by changes in the curves we plotted during consecutive springtime months. Data correction should be approached very carefully. In this situation Mill Creek ISCO water samples indicate that corrections made to improve accuracy at Mill Creek (and perhaps at others but we have an insufficient number of water samples with which to test) actually reduced accuracy: In this case raw field turbidity predicted lab grade turbidity more closely than corrected field turbidity.

B: Lab Turbidity On Suspended Sediment Concentration
Two sub-populations appeared when SSC was plotted on turbidity: the short range group had heavier SSCs for low to moderate turbidities and the long range group had lighter SSCs across the turbidity range experienced. One grab sample from South Fork plotted as

a clear outlier (30.2 ntu, 243.06 mg/l). It had much higher SSC for a moderate turbidity. We examined hysteresis in the data set as well as the filtered sediment residues (see photo). The South Fork grab sample outlier had an unusually high sand fraction for a low turbidity. None of the other filtered residues had nearly as much sand. Long range residues had organic content visible in the filtered residue, which would add less weight (higher SSC) for a given turbidity. The difference in organic content versus heavier inorganic particulate explains the presence of two sub-populations.

C: SSC On Digital Field Turbidity

A regression equation is presented graphically and mathematically that defines the relationship correlating SSC with field grade turbidity. We presented the graph and regression statistics for information purposes. The tighter the correlation between SSC and turbidity, the better in predicting accurate SSC from turbidity. This relationship is not particularly linear, and the wide scatter at moderate turbidity levels suggests that without considerable effort to refine the data, pull outliers, and explain wide variation around the center, this analysis does not appear to justify a plan to predict SSC (and then predict suspended sediment yield) from field grade turbidity. Perhaps efforts to refine this dataset could improve the certainty and accuracy with which Mill Creek SSC could be predicted from Mill Creek field grade turbidity.

D: Combined Grab Samples; Lab Turbidity On SSC

The R^2 value for combined grab samples for suspended sediment concentrations on lab turbidity was 0.34, a very poor correlation. One outlier had a higher SSC value than others for the given turbidity. A visual assessment of this sample's filtered residue indicated that it had more sand than other samples. When this sample was removed the R^2 value increased to 0.57 but still SSC appears poorly correlated to turbidity when grab samples from all Garcia River tributaries were combined.

E: Individual Tributaries; Lab Turbidity On SSC

Once the grab samples were divided so as to represent single, independent tributaries, then variation in the relationship of SSC on lab turbidity is reduced and the R-squared improved substantially. This indicates that the Garcia tributaries should not be represented as a group but rather in scientific discussion pertaining to water quality, the tributaries should be treated individually to improve on accuracy in predictions and decision making.

CONCLUSIONS

The data collected in this study was used to address the following questions:

- Does this recent data suggest a trend of improvement in spawning gravel quality toward attainment of regulatory targets? IN SOME STREAMS
- Is there a measurable improvement in spawning gravel quality at South Fork Garcia as a result of watershed restoration work conducted between the 1999 and 2004 measurements? MEASURABLE IMPROVEMENT, YES
- Does between-storm turbidity impose a threat to over wintering juvenile salmonids? YES
- Has winter water clarity improved following watershed restoration at the South Fork? YES
- Do land management practices exert a measurable influence on winter water clarity? YES
- If so, how problematic is that influence on anadromous salmonids?

GRAVEL CONDITIONS IN 2004 COMPARED TO 1999 BASELINE

Particle Size Distribution

Table 4 indicates Whitlow Creek had more streambed fines smaller than 0.85mm in 2004 than in 1999. That conclusion is validated by Dr. Baker's parametric tests. His parametric tests also conclude that there was no statistically significant difference between 1999 and 2004 samples from the other tributaries yet there were measurable differences: Compared with 1999 samples, in 2004: Whitlow Creek had less fines in the <8.0mm size class; Mill Creek had more fines in both the <0.85mm and the <8mm size classes; South Fork had more fines < 8.0mm size class but less fines <0.85mm; Pardaloe has less fines in both the <0.85 and <8.0mm size classes.

Permeability

Spawning gravel permeability at Mill Creek was slower in 2004 than in 1999. At Pardaloe Creek permeability was faster in 2004 than in 1999 which is desirable because permeability through gravel interstices brings more dissolved oxygen to the salmon redd as well as more flushing of metabolic wastes. Both of these findings were statistically significant. Inman and South Fork permeabilities were measurably faster in 2004 but these differences were not found to be significant statistically.

Embeddedness

No embeddedness measurements were taken in 1999 with which to compare 2004 results. Embeddedness measurements in 2004 were tried as an attempt to correlate this relatively simple technique which has been recorded for a large number of California streams (Doug Albin, personal communication) to particle-size distributions and/or permeability. McBain and Trush's correlation analysis indicated embeddedness explains 0.3% of the variability in the percentage of gravel smaller than 8.0mm and explained 5% of the variability in the percentage of the gravel smaller than 0.85mm. This lack of correlation may be explained in that embeddedness is measured at the bed surface but both bulk gravel and permeability are measured several inches below the surface.

Sediment Quality Relative to Garcia River TMDL

EPA in its TMDL targets the percentage of fine sediments in spawning gravels finer than 0.85 mm and 6.5 mm at <14% and <30%, respectively. As in 1999, 2004 gravels finer than 0.85 mm fall below the 14% numeric target, indicating less of the bedload is in that very fine category. However Mill and Pardaloe creeks exceeded the 30% numeric target for sediment finer than 6.5 mm suggesting spawning gravels are still impaired with by too many fines in pea gravel category.

What is the effect of these trends on Salmon using these Spawning Gravels

McBain and Trush used gravel particle size composition and permeability data to predict egg survival to emergence in gravel salmon redds. In 2004, survival to emergence was predicted to be higher at Whitlow and Pardaloe creeks than in 1999 but lower at Mill and South Fork based on particle size composition. Survival to emergence was predicted to be higher at Pardaloe Creek in 2004 but lower on the other tributaries based on permeability data.

As is shown in the table below, the gravel particle size distribution suggests that all monitored tributaries now have spawning gravels that meet the targeted proportion of fines smaller than 0.85 millimeters (target is <30%). For particles smaller than 6.5 millimeters, spawning gravels are hovering around the target established by EPA (target is <14%). There has been discussion as to whether these targets are applicable when discussing gravels that are dried prior to sieving and weighing. Drying gravels prior to sieving reduces water tension between particles and reduces water weight held by wet particles.

Do South Fork Gravel Measurements reflect Watershed Restoration Efforts?

Between 1999 and 2004 mean permeability increased from 1354 cm/hr to 2117 cm/hr and the percentage of fine sediment <0.85 mm decreased from 10.1% to 7.8%. These changes were measurable, but they were found not to be statistically significant. In the Author's opinion, measured spawning gravel quality improvements may be a result of watershed restoration.

Winter Water Clarity

Mill Creek, Pardaloe, and South Fork had the clearest winter water in the study period (tables 11 and 12). South Fork Garcia had the lowest peak turbidity for most of the largest storms we extracted from the data set (followed by Pardaloe) and had the lowest turbidity when we averaged those peak turbidities (table 13). In Water Year 2004 South Fork experienced the least number of hours above the >30 ntu turbidity threshold while Mill Creek experienced the lowest number of hours above the 60 and the 150 ntu turbidity thresholds. In WY 2005 South Fork experienced the lowest number of hours above 30 ntu, Pardaloe Creek experienced the least number of hours where turbidity exceeded the 60 ntu threshold, and Mill Creek experienced the least number of hours where turbidity exceeded 150 ntu.

Does clearer winter water quality reflect Watershed Restoration Efforts?

The water samples we collected from tributaries indicate that at low and moderate turbidity levels, filtered suspended matter consists mostly of inorganic particles of clay, silt, and some sand. This is the type of sediment one might expect in run off from road surfaces during small to moderate storms or sustained light rainfall. It follows logically that after road-based erosion control that sediment delivery would be lessened and water clarity would improve between storms (less hours of turbid water exceeding 30 ntu). During storm flows, road based erosion is perhaps more likely to result from problematic stream crossings than road surfaces. Upgraded stream crossings like those implemented in South Fork are designed to reduce sediment delivery to streams and therefore would increase water clarity in storms. The improved water clarity exhibited in South Fork over other streams with equivalent or higher road density suggests that watershed restoration has been effective in reducing sediment delivery that impairs winter water clarity. However, we have no pre-treatment data from South Fork to support this conclusion, and are relying only on comparative data between Garcia River Tributaries.

Most surface erosion from areas disturbed by road erosion control work typically occurs during the first three years following implementation. For the South Fork Garcia River, that period was from 2002-2003. Our 2004-05 data suggest that South Fork turbidity exposures, in hours above 30, 60, and 150 NTU, did decline in duration between WY04 and WY05, as would be hoped in the years following erosion control restoration. In WY04, South Fork experienced the fewest hours above the 30 NTU threshold. In WY05 South Fork experienced fewer hours above the 30 and 150 NTU threshold exposures than did the other monitored tributaries, as well as recording the lowest peak turbidities in monitored storm events.

This information is likely well received for those planning to reduce sediment delivery to fish bearing streams by road rehabilitation. The Author is excited to bring these conclusions to light. However, lack of pre-treatment data, and confounding influences supplied by natural sources of variation, does dampen such conclusions that might be drawn regarding causation until more focused and detailed studies emerge. We have applied for additional funding for that purpose.

Can we attribute clear waters to less disturbance?
The timber harvest maps included (figures 22-31) and Timber Harvest Intensity data (Table 14) indicate that both Pardaloe and South Fork have had comparatively lower rates of timber harvest in the last 17 years compared to other Garcia Rive tributary watersheds included in the study.

What is the Winter Water Clarity like in the other Garcia River Tributaries?
Mill Creek experienced the least number of hours where between storm instream turbidity was higher than the 150 ntu threshold during both water years. Yet Mill Creek

showed an increase in duration of threshold turbidity exposures from WY-04 to WY-05 by about one-third for the >30 and >60 ntu thresholds, without any obvious change in land use practices or restoration efforts. This illustrates the difficulty in trying to draw conclusions based on very limited datasets.

Whitlow and Inman showed the longest duration of exposures over chronic turbidity thresholds, and these watersheds had the most upstream timber harvesting of the monitoring sub basins. The Garcia River mainstem had the highest chronic turbidity in WY04, possibly as a result of cumulative watershed effects, but the weight of suspended sediment was unusually low compared to tributaries. We observed lens fouling by periphyton in a 24-hour period there on the mainstem but not for weeks on tributaries. This observation suggests that mainstem turbidity is often algal in nature and doesn't weigh very much. The greenish color of the mainstem between storms (as shown in cover photo) supports the observation of periphyton or suspended algae in the mainstem as a dominant source of mainstem turbidity.

<u>Is chronic winter turbidity a hardship on overwintering salmon and steelhead?</u>
This question is certainly relevant but outside the scope of this project. The expertise of fisheries biologists is required to answer these questions. What we can say is that the hours spent in streams over 150 ntu are likely to be hours spent without feeding. Hours spent over 60 ntu are challenging for feeding in that it is difficult for fishes to see due to lack of light and suspended particles and also because macroinvertebrates pick up off the bottom and drift to escape turbidities over 60 ntu. The hours spent over 30 ntu are non-growth periods, perhaps because the energy it takes to capture food balances the nutrition provided by that food. Our analysis followed the procedure used by Klein, 2003 in that the hours over any one turbidity threshold are total but not consecutive hours.

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APPENDIX 11

OVERVIEW OF EXISTING SEDIMENT STUDIES RELEVANT TO THE JDSF EIR

Introduction

This document provides an overview of the sediment yield estimates that have been compiled for the Jackson Demonstration State Forest (JDSF) EIR assessment area, and discusses the implications of this information for JDSF management. The main sources of information for sediment yield in the JDSF EIR assessment area are: 1) watershed assessments completed by privately held timber companies, 2) Stillwater Sciences' watershed analysis for the draft JDSF HCP/SYP, 3) Total Maximum Daily Load (TMDL) documents produced by the U.S. EPA, 4) sediment source area investigations conducted to support TMDL development, 5) data collected by the USDA Forest Service—Pacific Southwest Research Station at Caspar Creek, and 6) recent cosmogenic radionuclide data for long-term average erosion rates.

Numerous sediment studies have been conducted within the JDSF EIR assessment area over the past several years. At the large watershed scale (i.e., the Noyo and Big River basins), this work has generally consisted of office-based watershed assessments using techniques such as aerial photograph reconnaissance with limited field data collection. Some sediment data has been collected in the South Fork Noyo River, which provides a context for watershed assessment conclusions reached in the Noyo River basin. In contrast, research-level sediment data has been collected for 40 years in the headwater basins of the North and South Forks of Caspar Creek, a small coastal watershed located on Jackson Demonstration State Forest (JDSF) that is situated between the much larger Noyo and Big River basins.

Sediment yield estimates are summarized and discussed by individual watershed first, and then for the assessment area used for the JDSF draft HCP/SYP. Discussion of the results of the various assessments follow, along with management implications. Tables 1 and 2 in the Comparison of Sediment Yields section provide a summary of sediment yield estimates completed in the JDSF EIR assessment area.

Noyo River Watershed

Graham Matthews and Associates (1999) developed a preliminary sediment budget from a reconnaissance-level sediment source area analysis for the Noyo River basin. The study was based on analysis of air photos and digital mapping data. For the 67-year period between 1933 and 1999, total sediment inputs were estimated to be 590 t mi^{-2} yr^{-1} (Figure 1). Total sediment input was estimated to be 658 t mi^{-2} yr^{-1} for the period from 1958 through 1999. Matthews states that sediment input sources are likely to be underestimated due to the information available and the limitations of the analytic techniques employed. Under current conditions, it was estimated that about 35% of the sediment inputs for which estimates were developed are management related.

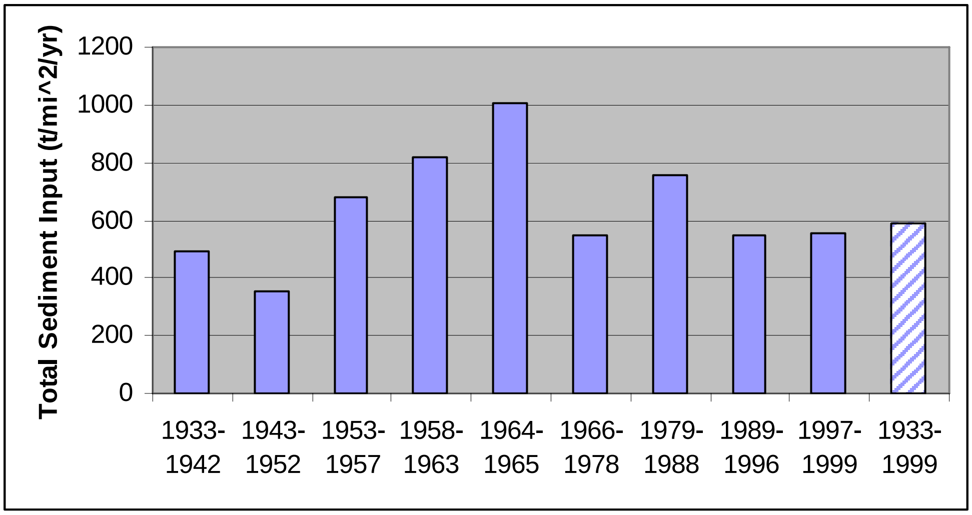

Figure 1. Estimated total sediment input values for the Noyo River watershed for varying time periods (Matthews and Associates 1999).

Graham Matthews and Associates' (1999) sediment source area analysis was the basis of the sediment data for the Noyo River TMDL (US EPA 1999). It is restated that the average annual sediment input over the 67 year period is 589 t mi^{-2} yr^{-1}. This document combines Matthews' periods of observation from the original nine (1933-1942; 1943-1952; 1953-1957; 1958-1963; 1964-1965; 1966-1978; 1979-1988; 1989-1996; and 1997-1999) to three (1933-1957; 1958-1978; and 1979 to 1999). With this grouping, the TMDL concludes that sediment delivery has generally increased over time, including an estimate of 667 t mi^{-2} yr^{-1} for the period from 1979 to 1999. Background sediment yield was stated as 374 t mi^{-2} yr^{-1} (56%), timber harvest[1] 36 t mi^{-2} yr^{-1} (5%), roads 251 t mi^{-2} yr^{-1} (38%), and railroad 6 t mi^{-2} yr^{-1} (1%) (see Table 14 in the Attachments to this appendix, reproduced from the Noyo River TMDL, for more detailed information).

The Noyo River TMDL states that the practices used during the Forest Practice Act period of 1979 to 1999 appear to have contributed to a deceleration in the rate of sediment delivery from management-related sources, but have not controlled them. For this period, 43% of the sediment yield is estimated to have come from timberland management, 1% from other management related sources, and 56% is attributed to natural/background sources.

[1] The timber harvest category includes hillslope mass wasting (landslides) and "in-unit" surface erosion (e.g., surface erosion from skid trails).

Because the US EPA estimates for the Noyo River TMDL were developed using office methods rather than field measurements, geomorphic mapping of the South Fork Noyo River valley floor was undertaken to quantify the volume of sediment stored in the watershed, and 10 streamflow and suspended sediment sampling stations were established for water year 2001 (Koehler and others 2001, 2002, 2004). These field measurements showed that large amounts of historic logging-related sediment trapped in long-term storage along the South Fork channel are transported downstream during high discharge events. This sediment increases the overall suspended sediment load and was not accounted for in the previous TMDL calculations, indicating that the TMDL overestimated sediment generated by upslope management practices (Koehler and others 2002, 2004). This study concluded that accurately quantifying channel sediment storage is a critical step for assessing sediment budgets, especially in TMDL documents attempting to relate upslope management to suspended sediment production.

Similarly, Benda and Associates (2004a) estimated bank erosion rates to be 2.75 in/yr in the Little North Fork Noyo River watershed as part of comprehensive field study. This high rate was thought to possibly reflect continuing channel incision and lateral migration of the channel related to historical logging (prior to 1970) that either filled the channel with sediment and wood, or otherwise changed their hydraulic geometry. The calculated sediment flux from bank erosion for third and higher order channels was reported as approximately 1060 t mi^{-2} yr^{-1}, which is inconsistent with the US EPA (1999) TMDL estimate of 200 t mi^{-2} yr^{-1}, developed primarily with office techniques. Benda and Associates (2004a) state that this suggests that the EPA TMDL for the Noyo River underestimated the bank erosion component of the sediment budget by approximately 500% and consequently the “background” sediment yield by 250%. They caution that while this analysis is preliminary and deserving of additional analysis, it suggests the EPA TMDL for the Noyo River is inaccurate and quantitative values obtained from it should be treated with caution. If Benda and Associates (2004a) estimate of bank erosion is used with the other US EPA TMDL estimates, it reveals that management-related sources are responsible for approximately 20% of the total sediment yield of 1527 t mi^{-2} yr^{-1}. [2] And the proportion of management-related sediment could be reduced even more by accounting for the downstream channel deposit erosion reported by Koehler (2001).

The watershed analysis conducted by Mendocino Redwood Company (2000) for the Noyo River watershed analysis unit (WAU)[3] estimated that the average sediment input for the past 40 years was 470 t mi^{-2} yr^{-1}. Watershed analysis was conducted based on a modified version of the Washington Forest Practice Board watershed analysis procedure (WFPB 2001). Inputs were attributed to hillslope mass wasting (42%), road mass wasting (24%), road surface and fluvial erosion (24%), and skid trail erosion (10%) (see Figure 2). Road associated erosion was found to be the dominant sediment contributing process in the Noyo watershed assessment area, with road associated mass wasting, surface and fluvial erosion combined accounting for 48% of the

[2] Benda and Associates (2004b) report similar results for two subbasins in the Ten Mile River basin.
[3] The Mendocino Redwood Company watershed analysis work was only conducted on the portion of the watershed within their ownership.

estimated sediment inputs. Mass wasting from roads and hillslopes accounted for 66% of sediment inputs.

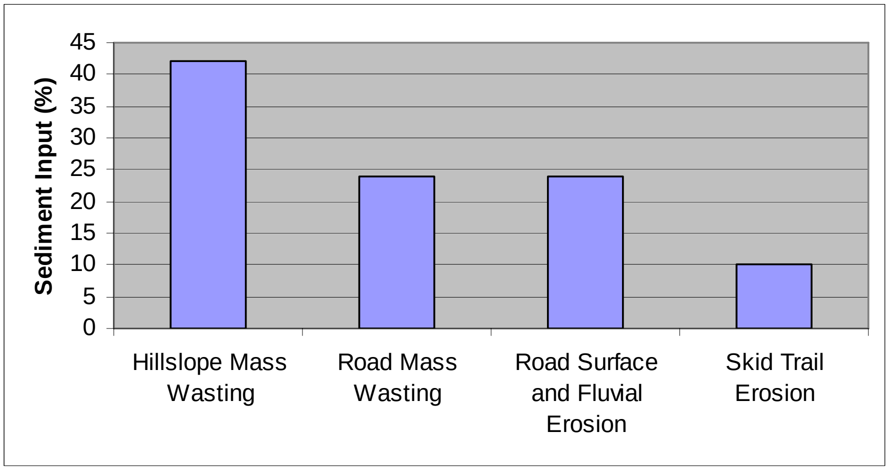

Figure 2. Sediment input percentages for the Mendocino Redwood Company's Noyo River watershed analysis unit (MRC 2000).

In contrast to the approaches that rely on air photo analysis as part of a watershed assessment process, Griggs and Hein (1980) estimated the suspended sediment yield of the Noyo River based on a combination of regional sediment yields, basin drainage areas, LANDSAT imagery, and existing sediment data. They reported a higher sediment yield of 1510 t mi^{-2} yr^{-1} for the period from approximately 1955 to 1980. Suspended sediment rate was used to estimate denudation rates, but may be 15 to 30% too low since bedload and dissolved loads were not taken into account. The average erosion (or denudation) rate for the Noyo River watershed was reported as slightly greater than 0.25 mm/yr, or approximately 1930 t mi^{-2} yr^{-1} [assuming that the an erosion rate of 1 mm/yr is approximately 7710 t mi^{-2} yr^{-1} of sediment, using a density of rock of 2.7 g cm^{-3} (Ferrier and others 2003a)].

In summary, these sediment studies indicate that sediment input estimates for the Noyo River basin over the past 20 to 40 years range from approximately 500 to 1900 t mi^{-2} yr^{-1}, depending on the methods and time frames considered.

Big River Watershed

As with the Noyo River basin, Graham Matthews and Associates (2001) developed a preliminary sediment budget for the Big River basin using a rapid reconnaissance-level sediment source area analysis. Limited streamflow and sediment data were collected for water years 2000 and 2001. Matthews (2001) reported improvements in management practices since 1974 have resulted in decreases in road-related mass wasting and harvest-related surface erosion, but sediment delivery from these processes is still well above estimated background rates. Significant construction of new roads has led to increasing sediment yields from road surface erosion, despite improved practices (see Table 27 in the Attachments, reproduced from Matthews 2001).[4] Combined management-related sediment sources (management-related landslides, skid trail and road surface erosion) were estimated to be producing 51.7% of the current sediment loads, while non-management related sediment sources comprise the remaining 48.3%. Due to greater levels of disturbance in earlier periods, the overall average for the 80-year period was estimated to be 66.4% management-related and 33.6% non-management related.

The TMDL document for Big River (US EPA 2001) uses Graham Matthews and Associates' (2001) sediment estimates and states that estimated average sediment delivery in the watershed from 1921 to 2000 was 944 t mi^{-2} yr^{-1} (Figure 3). Sediment production was lower during the periods of 1966 to 1978 (594 t mi^{-2} yr^{-1}), 1979 to 1988 (618 t mi^{-2} yr^{-1}), and 1989 to 2000 (600 t mi^{-2} yr^{-1}) (Figure 3). The average rate from 1979 to 2000 was estimated to be 609 t mi^{-2} yr^{-1} (US EPA 2002). Background during this period was estimated at 261 t mi^{-2} yr^{-1} (43%), timber harvest 115.5 t mi^{-2} yr^{-1} (19%), roads 202 t mi^{-2} yr^{-1} (33%), and grassland landslides 30.5 t mi^{-2} yr^{-1} (5%). During this period, 52% of the sediment yield was estimated to be related to timberland management, 5% related to other management related sources (i.e., grassland related landslides), and 43% to natural/background sources (see Table 7, reproduced from the Big River TMDL, in the Attachments for more detailed information).

The Mendocino Redwood Company (2003) draft Big River watershed analysis states that the average estimated sediment input for the past 30 years for the Big River watershed analysis unit (WAU) is 880 t mi^{-2} yr^{-1}.[5] The inputs in the Big River WAU over the last 30 years have come from mass wasting (48%), and surface and point source erosion (52%). Road associated erosion is reported to be the dominant sediment contributing process in the Big River assessment area. Road associated mass wasting, surface and point source erosion combined accounts for 65% of the estimated sediment inputs in the Big River WAU. When skid trail erosion is included with the road sediment inputs, the combined amount totals 81% of the sediment inputs to the Big River assessment area. Specifically, road associated mass wasting was estimated to produce 255 t mi^{-2} yr^{-1} (29%), road surface erosion 190 t mi^{-2} yr^{-1} (22%), road point source erosion 130 t mi^{-2} yr^{-1} (15%), hillslope mass wasting 170 t mi^{-2} yr^{-1} (19%), and skid trail erosion 135 t mi^{-2} yr^{-1} (15%) (Figure 4).

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[4] Road building rates increased in the late 20th century partly due to the need to convert from ground-based logging systems (i.e., tractor logging) to aerial yarding systems (i.e., skyline cable yarding). Tractor logging used roads located in the bottoms of drainages, while cable yarding requires roads located near ridgelines.

[5] The Mendocino Redwood Company watershed analysis work was only conducted on the portion of the watershed within their ownership.

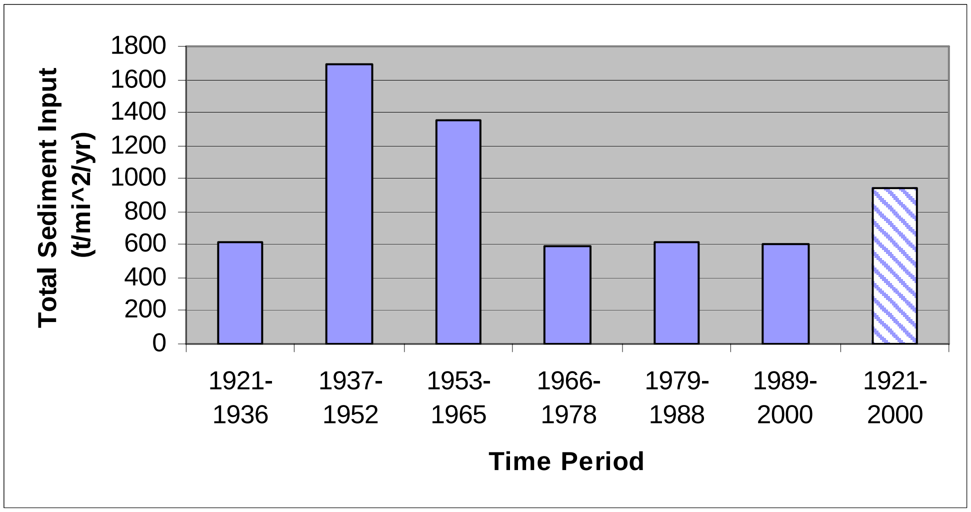

Figure 3. Estimated total sediment input values for the Big River watershed for varying time periods (U.S. EPA 2001 and Matthews and Associates 2001).

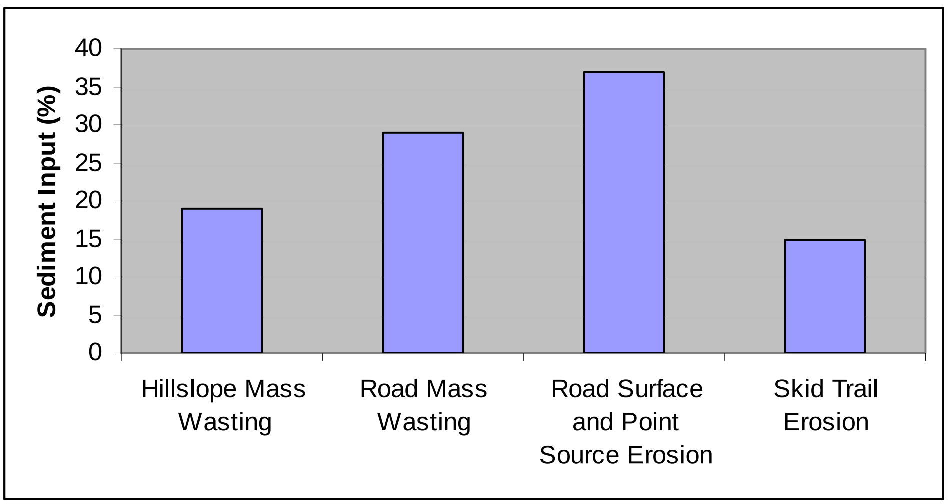

Figure 4. Sediment input percentages for the Mendocino Redwood Company's Big River watershed analysis unit (MRC 2003).

As in the Noyo River basin, Griggs and Hein (1980) estimated the suspended sediment yield of Big River based on a combination of regional sediment yields, basin drainage areas, LANDSAT imagery, and existing sediment data. They reported a yield of 940 t mi^{-2} $yr^{-1}$for the period from approximately 1955 to 1980, which is nearly identical to the Matthews (2001) total sediment yield estimate from 1921 to 2000. Suspended sediment rate was used to estimate denudation rates, but may be 15 to 30% too low since bedload and dissolved loads were not taken into account. The average erosion (or denudation) rate for the Big River basin was reported as about 0.20 mm/yr, or approximately 1540 t mi^{-2} yr^{-1}. As stated above, an average erosion rate of 1 mm/yr is approximately 7710 t mi^{-2} yr^{-1} of sediment, assuming a rock density of 2.7 g cm^{-3}.

In summary, the GMA (2001) and U.S. EPA (2001) documents conclude that sediment yields in Big River have averaged about 950 t mi^{-2} yr^{-1}over an 80 year period. Sediment production was estimated to be lower for the periods beginning in 1966. Griggs and Hein (1980) reported a similar suspended sediment yield for approximately a 25 year period ending about 1980, with a total sediment yield up to 1540 t mi^{-2} yr^{-1}. The TMDL document (US EPA 2001) found an average of about 610 t mi^{-2} yr^{-1}over the past 30 years, while MRC's (2003) sediment yield estimate is 880 t mi^{-2} yr^{-1}over the past 30 years. Therefore, there is general agreement that sediment input ranges from about 600 to 1540 t mi^{-2} $yr^{-1}$for the Big River basin.

Caspar Creek Watershed Study

Annual sediment loads for suspended sediment and bedload have been measured at the North and South Forks of Caspar Creek, a small coastal watershed situated between the Noyo and Big River drainages, for the past 40 years.[6] Mean annual sediment yields in the North and South Forks from 1963 to 2002 are 440 t mi^{-2} yr^{-1} and 495 t mi^{-2} yr^{-1}, respectively (using data from USFS-PSW 2004).[7] Lewis (1998) reported that approximately 70% of the total sediment load is transported as suspended sediment and 30% is bedload. Extremely high annual variability in sediment yield has been documented, based on number and size of storm events for a given winter, as well as watershed treatments applied (Figure 5). 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). The long-term average sediment yields for Caspar Creek are of great value and provide a benchmark for comparison with office-based sediment budget values developed for JDSF and the larger river basins to the north and south.

Recent work using cosmogenic radionuclides in Caspar Creek has determined the average erosion rate over approximately 5500 to 8900 years for this basin (Ferrier and others 2004). They report an average denudation rate of 0.09 $\pm$0.02 mm/yr, or approximately 695 t mi^{-2} yr^{-1} (physical erosion plus chemical weathering fluxes). This figure is somewhat greater than the mean erosion rate measured over the past 40 years (0.057 $\pm$0.015 mm yr^{-1} and 0.064 $\pm$0.012 mm yr^{-1}) for the North and South Forks, respectively (~440 and 495 t mi^{-2} yr^{-1}) (Ferrier and others 2004) (see Figure 6).[8]
If it is assumed that the 1963-1975 suspended sediment sampling at Caspar Creek over-estimated sediment yields by a factor of 2-3 times (Lewis 1998) due to over sampling the rising limb of the storm hydrograph, short-term suspended sediment yields

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[6] Sediment data are available for water years 1963 through 2002, with the exception of water year 1977.
[7] Solute erosion is assumed to be approximately 40 t mi^{-2} yr^{-1} for the South Fork of Caspar Creek based on an estimate of 8 percent of total sediment yield provided in Griggs and Hein (1980). Solute erosion is not included in these total erosion estimates.
[8] Note that this does not include chemical weathering fluxes.

can be reduced 40% to account for sampling bias (Ferrier and others 2004). With this revision, erosion rates at the North and South Fork weirs are 0.044±0.009 mm yr^{-1} and 0.046±0.007 mm yr^{-1}, respectively (~340 and 355 t mi^{-2} yr^{-1}).

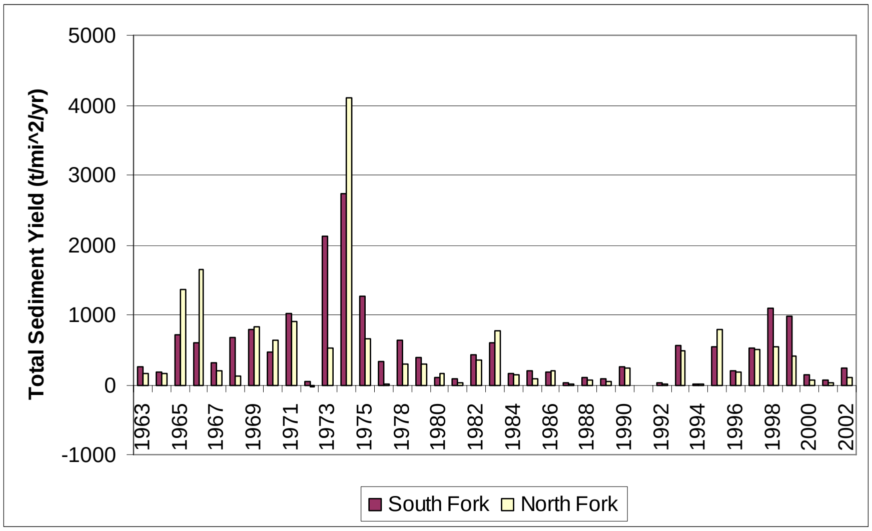

Figure 5. Total annual sediment yield measured for both the South and North Forks of Caspar Creek (USFS-PSW 2004).

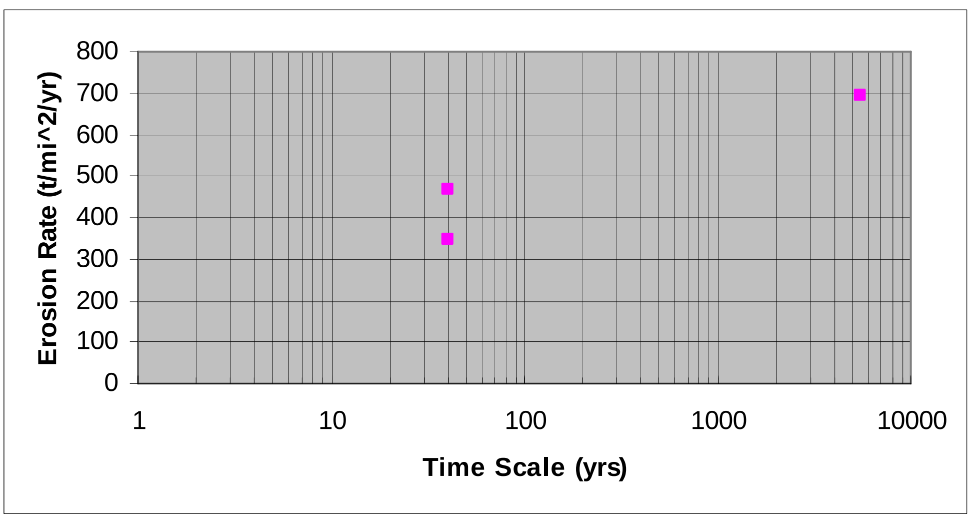

Figure 6. Erosion rates over different time scales at the North and South Forks of Caspar Creek (Ferrier and others 2004). Average erosion rates over the past 40 years are displayed as measured (average of the North and South Fork values), and with a 40% reduction for the years from 1963-1975 due to oversampling the rising limb of the hydrograph. Long-term physical erosion and chemical weathering fluxes are shown at the 5500 year time scale.

With this data modification, Ferrier and others (2004) state that the long-term measurements imply that Caspar Creek has experienced erosion rates that are approximately 2 times higher than decadal erosion rates (i.e., measured from 1963 through 2003). As a potential explanation, they state that sediment delivery to streams is highly episodic and that over 40 years of monitoring, relatively few large storm events that dominate long-term average erosion rates will have occurred. Evidence that indicates that the long-term erosion rates are dominated by large mass movement events with long recurrence intervals at Caspar Creek includes the presence of a large landslide (1,000,000 to 5,000,000 cubic yard) that dammed the North Fork of Caspar Creek and initiated sediment deposition in the upper part of the watershed (Cafferata and Spittler 1998). The landslide dam initiated sedimentation about 7000 years BP as determined by radio carbon (^{14}C) dating by Reneau (1989). Other landslides in the watershed are also substantially larger than any that have failed during the past 40 years (Spittler and McKittrick 1995).

Kirchner and others (2003) have used cosmogenic radionuclide geochemistry at dozens of sites in the western hemisphere and found that in the northern California Coast Range, long-term erosion rates are within a factor of two to three of modern day sediment yield measurements, suggesting that sediment delivery over decadal timescales is broadly consistent with the long-term average rate of sediment production in these watersheds (Ferrier and others 2003a, 2004).[9]

Ferrier and others (2004) state that the 5500- to 8900-year average erosion rate at Caspar Creek (0.09 mm/yr) is less than the local uplift rate, which is 0.3 to 0.4 mm/yr averaged over the past 300,000 years. Merritts and Bull 1989 reported that the average tectonic uplift has been relatively uniform throughout the Holocene (10,000 years ago to the present) and is approximately 0.3 mm/yr off the Mendocino County coast (inferred from marine terrace ages). These data imply that the mean elevation of the North Fork of Caspar Creek is still increasing (Ferrier and others 2004).

While the Caspar Creek watershed is considerably smaller than the Noyo and Big River basins (9 mi^2 vs. 113 mi^2 and 181 mi^2, respectively), it has a comparable land use history.[10] Additionally, the geology, soils, climate, and vegetation are grossly similar to those found in the larger basins. Old-growth redwood and Douglas-fir were logged from Caspar Creek from 1864 to 1904 (Napolitano 1996). Young-growth harvesting began in the late 1950s utilizing crawler tractors on steep slopes and roads located near channels, with improved forest practices occurring after the mid-1970s. In the experimental watersheds, the South Fork was logged from 1971 to 1973 with practices used prior to the implementation of the modern California Forest Practice Rules, while portions of the North Fork were logged from 1985 to 1992 using modern forest practices

[9] In contrast, comparison of long-term erosion determined with cosmogenic radionuclides to present day erosion in a largely deforested tropical highland in Sri Lanka shows that there has been a 10 to 100 fold recent increase in average erosion rates. Soil is being lost 10 to 100 times faster from agriculturally utilized areas than it is being produced in this location (Hewawasam and others 2003).

[10] The entire Caspar Creek basin where it enters the ocean is 9 mi^2, but the North and South Fork watersheds above the weirs, where sediment has been measured, are only 1.9 mi^2 and 1.7 mi^2, respectively.

(Henry 1998). Numerous landslides occurred after road construction and logging in the South Fork due to inadequate road, skid trail, and landing design, placement, and construction (Cafferata and Spittler 1998). Similar problems were not observed in the North Fork following harvesting that primarily utilized skyline cable yarding and roads located near ridges.

A complete sediment budget has yet to be prepared for the entire Caspar Creek watershed. Napolitano (1996), however, completed a sediment budget for the mainstem of the North Fork of Caspar Creek (from the weir to the old splash dam site). This work revealed that the main channel is still adjusting to severe channel impacts caused by splash dam operations and log drives that occurred in the nineteenth century.[11] In addition, large wood loading was greatly diminished by these historical logging activities (Napolitano 1998). Old-growth logging appears to have produced lasting channel impacts, including channel incision, simplification of channel form, and reduction in sediment storage capacity. During the 1980 to 1988 water years covered by Napolitano's (1996) sediment budget, average annual sediment yield was only approximately 200 t mi^{-2} yr^{-1}, reflecting the relatively low discharge events experienced during this period. Changes in sediment storage were measured, but sediment inputs from various types of hillslope erosion features (e.g., shallow rapid landslides, deep slow landslides, and road surface erosion) were not estimated. Lewis' (1998) subsequent work, however, concluded that roads were relatively unimportant sediment sources in the North Fork due to their location on ridges away from channels. Sediment increases from channel erosion associated with unbuffered intermittent small streams in burned and, to a lesser degree, in unburned areas, were found to be a significant sediment source in recently harvested areas. Increased erosion was attributed to increased gullying of headwater channels (Keppeler and others 2003).

Recent work in the Caspar Creek watershed also has found bank erosion to be an important sediment source (similar to that reported by Benda and Associates 2004a for the Little North Fork Noyo River watershed). Tributary and headwater valleys show signs of incision along much of their lengths, and Dewey and others (2003) report that ongoing levels of suspended sediment delivery correlate well with total amount of exposed channel bank. On an annual to decadal time-scale, they found that rates of suspended sediment delivery per unit area of watershed area correlate better with the amount of exposed bank area in reaches upstream of stream gages, than with the volume of sediment delivered by landslide events, with total basin area, or with peak storm flow per unit area.

[11] It may be that some of the channel adjustment is due to the establishment of an effective channel through the old landslide dam, as well as active downcutting that is in response to the regional tectonic uplift (T. Spittler, CGS, Santa Rosa, electronic communication).

JDSF Draft HCP/SYP Sediment Budget

In 1999, Stillwater Sciences developed a rapid sediment budget for the JDSF Habitat Conservation Plan (HCP)/Sustained Yield Plan (SYP) watershed assessment area (approximately 156 mi^2).[12] This sediment budget included estimates of hillslope erosion, sediment yield to channels, and changes in sediment storage within channels (CDF 1999). Results from the surface erosion and mass wasting modules completed for the watershed assessment were used for this work. A sediment yield estimate was provided for the period from 1958 to 1997, because 1978 and 1996 air photos provided a record of landsliding covering the period from 1958 to 1996. Separate sediment budgets could not be constructed for the periods from 1958 to 1978 and 1978 to 1997 because rates of road and natural surface erosion, deep-seated landslides, and soil creep sediment production could not be differentiated, but discrete rates of landsliding for the two periods were produced. Therefore, the overall sediment yield estimate encompasses a very wide range of forestry practices.

The rapid sediment budget indicated that road-related surface erosion accounted for 45% of hillslope erosion, road-related shallow landslides produced 27%, deep seated landslides 2%, soil creep 2.5%, hillslope shallow landslides 21%, and background surface erosion 2.5% (Figure 7). Combined road-related erosion (surface and mass wasting) accounted for 72% of the total hillslope erosion. The remaining 28% of the hillslope erosion was associated with natural and management related sources (e.g., in-unit landslides) on hillslopes and inner gorges. Average sediment yield was estimated at 856 t mi^{-2} yr^{-1} for the period from 1958 to 1997, which is a 2.5 fold increase over estimated background rates (342 t mi^{-2} yr^{-1}).

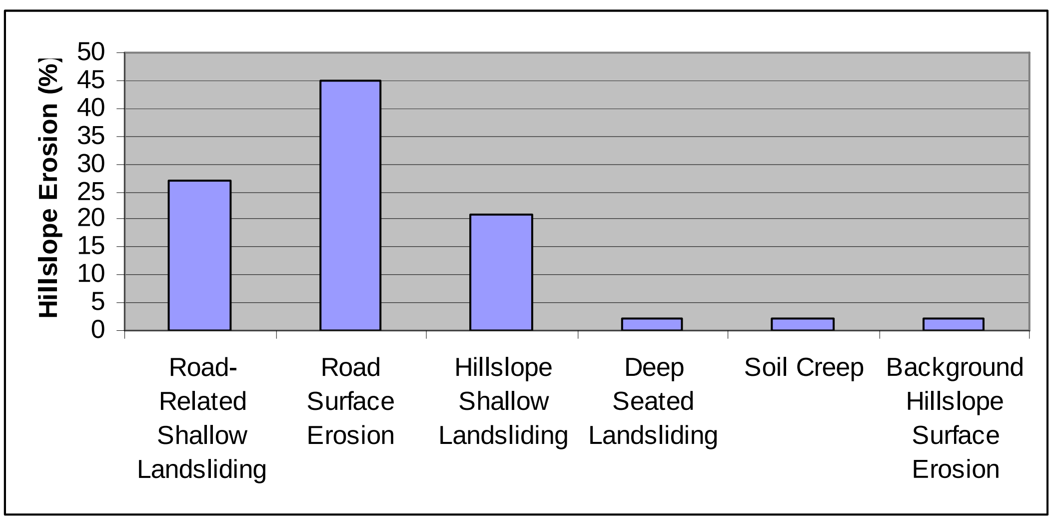

Figure 7. Rapid sediment budget produced for the draft JDSF HCP/SYP assessment area, 1958-1997 (CDF 1999).

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[12] The assessment area for the draft JDSF HCP/SYP included the South Fork of the Noyo River, four small coastal watersheds (Hare, Mitchell, Caspar Creeks, and Russian Gulch), Lower Big River, and the North Fork of Big River.

Stillwater Sciences reported two results that show improved forestry practices required by the modern Forest Practice Rules have significantly reduced sediment yields in the past two decades (CDF 1999). First, the amount of sediment from road-related shallow landslides from 1979 to 1996 was approximately half that found during 1958 to 1978. Second, logging in the South Fork of Caspar Creek that was conducted prior to the implementation the modern rules produced from 2.4 to 3.7 times more suspended sediment than was produced in the later North Fork logging (Lewis 1998, Lewis and others 2001). Most of the increase in suspended sediment measured at the North Fork weir resulted from one large landslide that occurred in January 1995 (Lewis and others 2001).

Comparison of Sediment Yield Estimates

Due to the differences in methodologies, time periods, and watershed scales, sediment input estimates for the JDSF EIR assessment area range from approximately 400 t mi^{-2} $yr^{-1}$to about 1900 t mi^{-2} yr^{-1} (Table 1). It is likely that actual sediment yield for the EIR assessment area during varying time periods is within this range, since the erosion rate measured over 40 years at Caspar Creek is almost 500 t mi^{-2} $yr^{-1}$and the long-term average erosion rate estimated over about 5500 to 8900 years at Caspar Creek is approximately 695 t mi^{-2} yr^{-1} (Ferrier and others 2004).

Table 2 puts the results of the above described sediment production studies into groups of planning watersheds, time periods, and sediment source areas for comparison of methodologies and sediment yields. Listed sediment yields vary in some cases from previous values because the results cover different time periods, the addition of estimated bedload transport, or subtraction of soluble load to give more comparable expressions of total sediment load.

This comparison shows the great amount of variation in the types and combinations of sediment sources used to arrive at overall estimates. None of the sediment budget methods (MRC 2000, MRC 2003, U.S. EPA 1999, U.S. EPA 2001, and CDF 1999) use the same combinations of erosion processes and sediment sources. For example, streambank erosion is only considered by U.S. EPA (1999 and 2001), and only U.S. EPA (2001) and CDF (1999) include soil creep in their sediment budget. However, considering the differences in methods, watershed sizes, management history, and time periods, results from these sediment budget approaches all fall within the mid-range of estimated sediment production (398 to 974 t mi^{-2} yr^{-1}). Other approaches that rely on physical estimates of sediment inputs, including streambank erosion (Benda 2004a) or measurement of instream sediment transport (Koehler 2001, Lewis 1998, and above in this EIR) report sediment yields that are both well above and well below the range of sediment budget results.

Table 1. Summary of Sediment Input Values, Sediment Yield Estimates, and Long-Term Erosion Rates Completed in the JDSF EIR Assessment Area.

Source	Time Period Considered	Primary Method	Noyo River Watershed ($t\ mi^{-2}\ yr^{-1}$)	Big River Watershed ($t\ mi^{-2}\ yr^{-1}$)	NF/SF Caspar Cr. Watershed[13] ($t\ mi^{-2}\ yr^{-1}$)	JDSF Draft HCP/SYP Assessment Area ($t\ mi^{-2}\ yr^{-1}$)
Matthews (1999)	1958-1999	Office Assessment	658			
U.S. EPA (1999)	1979-1999	Office Assessment	667			
MRC (2000)	1960-2000	Office Assessment	470			
Griggs and Hein (1980)	~1955-1980	Office-based Estimate[14]	1930			
Matthews (2001)	1989-1999	Office Assessment		561		
U.S. EPA (2001)	1979-2000	Office Assessment		609		
MRC (2003)	1971-2001	Office Assessment		880		
Griggs and Hein (1980)	~1955-1980	Office-based Estimate[12]		1540		
USFS-PSW (2003)	1963-2002	Sediment Data Measurement			~440/495 [~340/355]	
Ferrier and others (2004)	~5500 to 8900 yrs	Cosmogenic Radionuclides			~695	
CDF/Stillwater Sciences (1999)	1958-1997	Office Assessment				856

Measurements made in water year 2001 by Koehler (2001) in the South Fork Noyo River indicate suspended sediment transport of 25 t $mi^{-2}\ yr^{-1}$ in 2001. Assuming that 30 percent of the total sediment load is transported as bedload, this gives a sediment yield of only 36 t $mi^{-2}\ yr^{-1}$. Koehler (2001) also found that 50 percent of the total suspended sediment load originated in a downstream stretch of river channel between major tributaries that includes only 10 percent of the watershed area, which indicates the potential importance of stream channel sediment sources.

In contrast to the relatively small sediment yields reported by Koehler (2001), Benda (2004a) determined that streambank and creep erosion in an upstream tributary to the South Fork Noyo River was producing 1,376 t $mi^{-2}\ yr^{-1}$ of stream sediment. This difference between up and down stream measurements can be partly explained by the recognizing that Koehler's measurements were limited to a single, relatively dry year, while Benda's results represent an average over several decades that include some of the largest storms on record. In effect, the large, upstream sediment yields reported by

[13] Estimates do not include the solute erosion component.
[14] Griggs and Hein (1980) used a combination of regional sediment yields, basin drainage areas, LANDSAT imagery, and existing sediment data to estimate sediment loads.

Table 2. Sediment Yield Summary.

Watershed Name	Source of Estimate	Period (years)	Area (mi^2)	Mass Wasting Erosion Sources ($t\ mi^{-2}\ yr^{-1}$)									
				Hillslope	Road	Skid Trail	Railroad	Harvest	Grass.	Shallow	Deep	Bkgrd.	Total
Little NF Noyo	Benda (2004a) (1)	<37	13.18	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
South Fork Noyo	Koehler (2001) (2)	2001	27.32	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Upper Noyo	MRC (2000)	1958-1998	55.64	189	53	NA	NA	NA	NA	NA	NA	NA	242
Upper Noyo	U.S. EPA (1999)	1953-1978	52.24	NA	81	NA	4	30	NA	NA	NA	104	219
Upper Noyo	U.S. EPA (1999)	1979-1999	52.24	NA	106	NA	5	7	NA	NA	NA	148	266
Noyo River	U.S. EPA (1999) (3)	1953-1978	113.00	NA	53	NA	33	14	NA	NA	NA	83	183
Noyo River	U.S. EPA (1999) (3)	1979-1999	113.00	NA	76	NA	6	20	NA	NA	NA	99	201
Noyo River	Griggs (1980) (2)	1955-1980	113.00	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
S+U Big River	MRC (2003) (4)	1970-2000	69.70	159	196	NA	NA	NA	NA	NA	NA	NA	355
Big River	U.S. EPA (2001)	1953-1978	181.00	NA	225	64	NA	195	49	NA	NA	199	732
Big River	U.S. EPA (2001)	1979-2000	181.00	NA	116	18	NA	87	30	NA	NA	146	397
Big River	Griggs (1980) (2)	1955-1980	181.00	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
JDSF WWAAs	CDF (1999)	1978-1997	156.00	NA	NA	NA	NA	NA	NA	195	26	NA	221
SF Caspar Creek	This EIR	1963-2002	1.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SF Caspar Creek	Lewis (1998) (5)	1972-1978	1.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
NF Caspar Creek	This EIR	1963-2002	1.83	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
NF Caspar Creek	Lewis (1998) (5)	1990-1996	1.83	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Caspar Creek	Ferrier (2004) (6)	>5000	8.38	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Watershed Name	Source of Estimate	Period (years)	Creep ($t\ mi^{-2}\ yr^{-1}$)	Surface Erosion Sources ($t\ mi^{-2}\ yr^{-1}$)					Stream ($t\ mi^{-2}\ yr^{-1}$)		Load ($t\ mi^{-2}\ yr^{-1}$)		Sed. Yield ($t\ mi^{-2}\ yr^{-1}$)
				Road	Skid Tr.	Harvest	Bkgrd.	Total	Bank	Channel	Pre-Log	Increase	
Little NF Noyo	Benda (2004a) (1)	<37	316	NA	NA	NA	NA	NA	1060	NA	NA	NA	1,376
South Fork Noyo	Koehler (2001) (2)	2001	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	36
Upper Noyo	MRC (2000)	1958-1998	NA	102	54	NA	NA	156	NA	NA	NA	NA	398
Upper Noyo	U.S. EPA (1999)	1953-1978	NA	149	50	NA	75	274	ND	ND	NA	NA	493
Upper Noyo	U.S. EPA (1999)	1979-1999	NA	172	19	NA	75	266	ND	ND	NA	NA	532
Noyo River	U.S. EPA (1999) (3)	1953-1978	NA	136	26	NA	75	237	200	ND	NA	NA	620
Noyo River	U.S. EPA (1999) (3)	1979-1999	NA	175	16	NA	75	266	200	ND	NA	NA	667
Noyo River	Griggs (1980) (2)	1955-1980	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	2,157
S+U Big River	MRC (2003) (4)	1970-2000	NA	271	90	NA	NA	361	NA	NA	NA	NA	716
Big River	U.S. EPA (2001)	1953-1978	85	55	28	NA	NA	83	74	NA	NA	NA	974
Big River	U.S. EPA (2001)	1979-2000	63	84	10	NA	NA	94	54	NA	NA	NA	608
Big River	Griggs (1980) (2)	1955-1980	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	1,343
JDSF WWAAs	CDF (1999)	1978-1997	29	428	NA	NA	29	457	NA	57	NA	NA	764
SF Caspar Creek	This EIR	1963-2002	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	440
SF Caspar Creek	Lewis (1998) (5) (6)	1972-1978	NA	NA	NA	NA	NA	NA	NA	NA	171	787	486
NF Caspar Creek	This EIR	1963-2002	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	495
NF Caspar Creek	Lewis (1998) (5)	1990-1996	NA	NA	NA	NA	NA	NA	NA	NA	76	59	135
Caspar Creek	Ferrier (2004) (6)	>5000	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	639

(1) Creep and bank erosion only. (2) 30% bedload added. (3) Includes streambank sediment. (4) Excludes Leonardo Lake and Martin Creek. (5) Calculated from reported loads and post-harvest % increases. (6) Corrected for sampling bias. (7) Corrected for 8% soluble load. S+U = South + Upper. NA = Not applicable. ND = No data.

Benda may be providing the stored sediments that are the source of Koehler's downstream findings. These findings also serve as an example, and warning, about the extreme variability of annual sediment production and the difficulty of representing sediment yield with a single number

Results of the Caspar Creek Studies reported by Lewis (1998) also give contrasting pictures of sediment yields from similar, small watersheds. Sediment production in the South Fork of Caspar Creek following 1970s tractor logging, with roads located near streams, was increased by nearly 212 percent to a total of over 486 t mi^{-2} yr^{-1}. In the nearby North Fork of Caspar Creek, a combination of skyline and tractor logging in the late 1980s, using modern Forest Practice Rules and upslope roads, increased the rate of sediment production by 89 percent to a total of only 135 t mi^{-2} yr^{-1}. Unfortunately, the loads measured in these two time periods cannot be compared directly because of generally higher flows during the earlier South Fork Study, but the percentage increase in sediment production from the South Fork logging was 2.4 times greater than in the North Fork Study. It is also interesting to note that the difference between longer-term estimates of sediment yield from the North and South Forks of Caspar Creek reported in this EIR is much smaller.

Discussion

Based on the work completed by Koehler and others (2001) and Benda and Associates (2004a), the percentages of sediment yield attributed to recent timber management in the Noyo and Big River basins as part of the TMDL documents appear to be too high. In a comprehensive review of the U.S. EPA's TMDL sediment yield estimates for seven North Coast watersheds (including Noyo and Big Rivers), Bedrossian and Custis (2002) reported that the natural/background rates of sedimentation in some cases were underestimated by at least an order of magnitude. The reasons for this included: 1) underestimates of erosion and sedimentation from deep-seated landslides, 2) tectonic uplift and erosion rates not considered, 3) lack of reference to past regional sediment source studies, 4) under-representation of legacy effects from past land use, 5) areas of significant natural/background sediment generation not included in analysis, and 6) inadequate consideration of impacts from land uses other than timber management.

For the Noyo River TMDL studies, Bedrossian and Custis (2002) state that even though deep-seated landslides were mapped in conjunction with TMDL development, deep-seated landslides were assumed to contribute little sediment except that derived from sheetwash or gullying processes (Matthews 1999). Virtually all large, dormant landslides were eliminated during the sediment source analysis. Bedrossian and Custis (2002) found that similar methods were used for the Big River sediment source analysis.

Kramer and others (2001) analyzed timber harvest and sediment loads in nine TMDL studies on the North Coast (including the Noyo River) and concluded that the TMDL sediment source analyses cannot distinguish whether post-Forest Practice Rule road-related sediment delivery originated from older roads or roads constructed under the current rules. Kramer and others (2001) state: "Although the degree of uncertainty depends upon the methodology used, the range of uncertainty in sediment source analyses is generally on the order of 40-50% (Raines and Kelsey 1991, Stillwater Sciences 1999). Methodological constraints (e.g., estimates of landslide frequency, areal extent, depth, age, bulk density, estimates of landslide delivery ratio, and natural temporal variability in erosion-triggering storms events) suggest that "this uncertainty may be too high to reliably detect differences between land uses or recent changes in land use practices such as those introduced in 1973 under the Z'berg-Nejedly Forest Practices Act (FPA) of 1973 (CCR 14 Chapters 4 and 4.5)." Similarly, Knopp (1993) found that post-modern Forest Practice Rule impacts cannot be easily or accurately extricated from legacy conditions after conducting a study of factors affecting coldwater fish habitat on the North Coast.

In addition to the work of Ferrier and others (2004) and Kirchner and others (2003), Constantine and others (2003) have studied very long-term sediment rates in a western Mendocino County river basin. They recently cored floodplains in the Navarro River basin to report on long-term sedimentation rates and found that land use change (i.e., old-growth and young-growth logging) has not had a significant impact in altering long-term average sediment deposition rates in this watershed.[15] Rather, climate and tectonics are suggested as the dominant controls on the evolution of Navarro River floodplains over hundreds to thousands of years. Constantine and others (2003) caution, however, that roads and large wood in low order drainages are storing large quantities of logging related sediment, preventing this material from escaping the lowest order tributaries and being deposited on established floodplains.

The work of Koehler and others (2001), Benda and Associates (2004a), Bedrossian and Custis (2002), Constantine and others (2003), each of which used different approaches to estimate sediment loading rates, combined with actual measurements made in Caspar Creek (USFS-PSW 2003, Dewey and others (2003), Ferrier and others 2003a,b), document that current timber operations under the modern Forest Practice Rules are unlikely to be responsible for producing 43 to 52% of the current sediment load, as reported by the TMDL work for Noyo and Big Rivers, respectively. The natural/background sediment generation in these North Coast watersheds is shown to be a considerably higher proportion of the total sediment load than that stated in the TMDL documents, since: 1) sediment from historic logging practices stored in low gradient channel networks that is being attributed to modern timber operations in the TMDLs is not supported by qualitative observational evidence or quantitative measurements; 2) channel bank erosion measured by researchers is substantially higher than that assumed in the TMDLs; and 3) long-term erosion rates associated with large mass movement events with relatively long recurrence intervals are documented to be a significant contributor to the background rate of sediment generation.
It is apparent from sediment source area studies completed for these watersheds and other hillslope monitoring work conducted in California that a majority of sediment

[15] The Navarro River mouth is about 8 miles to the south of the mouth of Big River.

related to current forest management activities is coming from road surfaces, road-related landslides, and road stream crossings (CDF 1999, Cafferata and Munn 2002, Bawcom 2003, Maahs and Barber 2001, MacDonald and others, in press).[16] Mass wasting (i.e., in-unit landslides) and surface erosion associated with timber harvesting appears to be a much smaller source of sediment (Figure 8). For example, Bawcom (2003) evaluated fifty clearcut units harvested from 1982 to 1994 in four watersheds to determine landslide occurrence in even-aged management units on JDSF following storms with the ability to trigger shallow rapid landslides. Of the 32 landslides identified in this study, all but four were associated with older roads, landings, and skid trails, and there was little evidence that vegetation removal associated with even-aged management in these coast redwood dominated watersheds was a significant contributor to slope instability or reactivations of dormant landslides for operations conducted under the modern Forest Practice Rules. In general, data collected to date in northwestern California areas with sprouting coast redwood does not show a clear relationship between clearcutting under the current FPR regime (sometimes in combination with requirements included in landscape level documents) and landslide rates. Most of the recent mass wasting features are related to roads and landings (Cafferata and Spittler 2004). Similarly in the central Sierra Nevada, MacDonald and others (in press) found surface erosion from roads was nearly an order of magnitude higher than that generated from harvested areas. On a national scale, Toy (1982) documents that the erosion rate associated with roads is an order of magnitude greater than that for harvesting and ground-based yarding.

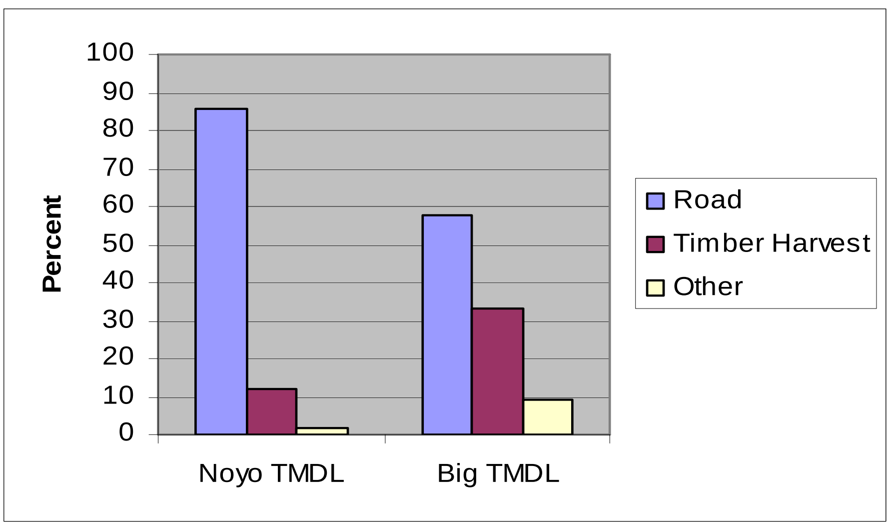

Figure 8. TMDL estimates of the percentage of management-related sediment resulting from roads, timber harvesting, and other sources for the Noyo and Big River basins (U.S. EPA 1999, 2001).

[16] This was not found to be the case for the North Fork of Caspar Creek, due to the fact that the roads were located high on the ridges, and only small spur roads off of ridges were built for the recent timber harvesting. It has also not been found for highly erodible watersheds such as Bear and Jordan Creeks in southern Humboldt County with very high rates of inner gorge landsliding (PWA 1998, 1999).

The main lesson to be learned from the sediment studies completed to date in the JDSF EIR assessment area is that roads and watercourse crossings need to be designed, constructed, surfaced, and maintained in a manner that will reduce long-term sediment yield. This can best be accomplished by application of a road management plan, which has been included as part of the JDSF Management Plan. Much of management-related sediment originates from points at or near where streams are crossed by roads, from roads with inside ditches, and from large road fill failures (Furniss and others 1991, Weaver and Hagans 1994). Inventorying and improving JDSF's roads to reduce sediment yield is needed due to the legacy of a road network partially relying on out-dated drainage systems and old segments located along watercourse channels. The road management plan will provide a systematic program to ensure that the design, construction, use, maintenance, and surfacing of the Forest's roads, road landings, and road crossings will be conducted to avoid, minimize, or mitigate adverse impacts to the aquatic habitats supporting anadromous fish, amphibians, and other aquatic organisms. Watercourse crossing inventories are an important component of this road management plan and can reduce sediment yield to streams by locating and prioritizing repair for high risk structures (Flanagan and others 1998, Flanagan and Furniss 1997).

The sediment data also support avoiding or intensively mitigating timber management activities (i.e., road building, tractor skidding, and other ground disturbing activities) on high risk portions of the landscape, such as unstable features, inner gorge areas along stream channels, and headwall swales located near ridges. Mass wasting avoidance strategies that include on-the-ground site review and recommendations by qualified professionals are included in the JDSF Draft Forest Management Plan (DFMP) and have been shown to be effective in reducing shallow, rapid landslide features in terrain that is more unstable than the JDSF EIR assessment area in Humboldt County (Marshall 2002, Marshall 2003, Smelser 2001).[17] The DFMP also calls for making use of the California Geological Survey's Relative Landslide Potential maps for identifying potentially unstable areas that should be avoided or carefully evaluated in the field prior to conducting potentially destabilizing activities such as road construction.

Similar efforts are underway on the industrial timberlands owned by Mendocino Redwood Company (MRC) and Hawthorne Timber Company (managed by Campbell Timberland Management) in the JDSF EIR assessment area. MRC has nearly completed a Habitat Conservation Plan (HCP) and Natural Communities Conservation Plan (NCCP) for its ownership in western Mendocino County. Both companies are actively improving their road network to reduce sediment yields to watercourses. Improved road-related practices and riparian zone protection throughout the vast majority of the JDSF EIR assessment area have been mandatory since the passage of the BOF's Threatened and Impaired Watersheds Rule Package, which became effective on July 1, 2000. The extensive efforts that JDSF, MRC, HTC, and other landowners have taken to reduce road sedimentation are documented in section VIII.2.1 in the EIR.

[17] Marshall (2002, 2003) and Smelser (2001) report on the presence of landslide features in watersheds located in southern Humboldt County on Pacific Lumber Company timberlands.

Conclusions

Key conclusions that can be drawn from this review of sediment studies conducted within the JDSF EIR assessment area include:

- Sediment production estimates have ranged from approximately 400 to 1900 t mi^{-2} yr^{-1}, depending on the watershed being considered, time frame analyzed, and analysis method used.
- Average annual sediment yield measured over 40 years at Caspar Creek is within this range (approximately 500 t mi^{-2} yr^{-1}). Recent work using cosmogenic radionuclides in Caspar Creek has determined the average erosion rate over 5500 to 8900 years for this basin is approximately 695 t mi^{-2} yr^{-1}, which is about twice the mean of sediment measurements made over the past 40 years when corrections for oversampling from 1963 to 1975 are incorporated (Ferrier and others 2004). Sediment delivery to Caspar Creek over the past four decades is somewhat lower than the long-term average rate of sediment production in this watershed, likely to due to the fact that sediment delivery to streams is highly episodic and few exceptionally large storm events that dominate long-term average erosion rates have occurred over the past 40 years (Ferrier and others 2004).
- Multiple sources of information (Noyo and Big River TMDL, Caspar Creek watershed study, JDSF draft HCP/SPY watershed assessment) indicate that improvements in management practices since implementation of the modern Forest Practice Act (i.e., after 1974) have resulted in decreases in road-related mass wasting and harvest-related surface erosion.
- While TMDL studies have estimated that 43 to 52% of the total sediment yield is produced by timber operations in the Noyo and Big River basins, respectively, more recent analysis and data show that: (1) natural/ background sediment production rates are probably much higher than reported in the TMDL documents, and 2) in-channel storage of sediment from historic logging operations is a likely source of some of the sediment that TMDLs have attributed to current timber management practices.
- Sediment budgets prepared for Noyo and Big River watershed assessments shows that road-related sediment (both from road surface erosion and road-related landslides) is a dominant source of sediment from current management activities, while in-unit hillslope erosion is a much smaller contributor.
- The Road Management Plan and the mass wasting avoidance strategy included in the JDSF Management Plan are expected to significantly reduce sediment yield associated with JDSF timber management activities.

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Attachments

Table 7—Big River Watershed Sediment Source Analysis, Preliminary Sediment Budget, Sediment Input Summary, Average Annual Unit Area Rates, reproduced from the Big River TMDL for sediment (U.S. EPA 2001)

Table 27—Big River Sediment Source Analysis, Road Construction History by Planning Watershed and Sub-Watershed, reproduced from Matthews (2001).

Table 14—Summary of Sediment Inputs to the Noyo River Watershed as Derived from Data Presented by Matthews (1999), reproduced from the Noyo River TMDL for sediment (U.S. EPA 1999).

Table 1—Preliminary sediment inputs and road density by planning watershed for Mendocino Redwood Company timberlands in the Big and Noyo River basins (C. Surfleet, Hydrologist, Mendocino Redwood Company, Ft. Bragg, electronic communication).

TABLE 7
BIG RIVER WATERSHED SEDIMENT SOURCE ANALYSIS
Preliminary Sediment Budget -- Sediment Input Summary -- Average Annual Unit Area Rates

STUDY PERIOD	NO. OF YEARS	INPUTS: BKGRND LANDSLIDES (t/mi2/yr)	TOTAL MGMT LANDSLIDES (t/mi2/yr)	HARVEST RELATED LANDSLIDES (t/mi2/yr)	ROAD RELATED LANDSLIDES (t/mi2/yr)	SKID TRAIL LANDSLIDES (t/mi2/yr)	GRASSLAND LANDSLIDES (t/mi2/yr)	TOTAL LANDSLIDES (t/mi2/yr)	SURFACE EROSION: BKGRND (t/mi2/yr)	SURFACE EROSION: SKID TRAIL (t/mi2/yr)	SURFACE EROSION: ROAD (t/mi2/yr)	FLUV/BANK EROSION (t/mi2/yr)	TOTAL NON-MGMT INPUTS (t/mi2/yr)	TOTAL MGMT INPUTS (t/mi2/yr)	TOTAL INPUTS (t/mi2/yr)
1921-1936	16	175	284	284	0	0	0	459	75	16	6	65	315	306	621
		28%	46%	46%	0%	0%	0%	74%	12%	3%	1%	10%	51%	49%	100%
1937-1952	16	179	1,336	847	364	3	122	1,515	75	8	23	65	319	1,367	1,686
		11%	79%	50%	22%	0%	7%	90%	4%	0%	1%	4%	19%	81%	100%
1953-1965	13	203	915	341	397	94	83	1,118	87	22	46	75	365	983	1,348
		15%	68%	25%	29%	7%	6%	83%	6%	2%	3%	6%	27%	73%	100%
1966-1978	13	194	148	48	52	34	14	342	83	33	64	72	349	245	594
		33%	25%	8%	9%	6%	2%	58%	14%	6%	11%	12%	59%	41%	100%
1979-1988	10	131	295	81	149	31	34	426	56	13	74	49	236	382	618
		21%	48%	13%	24%	5%	6%	69%	9%	2%	12%	8%	38%	62%	100%
1989-2000	12	159	214	92	88	7	27	373	68	7	93	59	286	314	600
		27%	36%	15%	15%	1%	5%	62%	11%	1%	16%	10%	48%	52%	100%
	80	19%	60%	33%	19%	3%	5%	79%	8%	2%	5%	7%	33%	67%	100%
1921-2000	80	175	566	313	178	26	48	741	75	16	47	65	315	629	944

Source: GMA 2001

TABLE 27
BIG RIVER SEDIMENT SOURCE ANALYSIS
ROAD CONSTRUCTION HISTORY BY PLANNING WATERSHED AND AND SUB-WATERSHED

PLANNING WATERSHED		MILES OF ROAD CONSTRUCTED IN PERIOD						TOTAL BY PW OR SW	% TOTAL WATERSHED ROAD MILES
Sub-Watershed	Drainage Area	1921-1936	1937-1952	1953-1965	1966-1978	1979-1988	1989-2000	(mi)	(mi)
BIG RIVER HEADWATERS	**32.78**	**0.00**	**41.53**	**53.99**	**27.37**	**41.53**	**69.36**	**233.8**	**18.8%**
	% of PW Total	**0.0%**	**17.8%**	**23.1%**	**11.7%**	**17.8%**	**29.7%**		
Upper Mainstem Big River	12.55	-na-	16.05	26.70	5.33	12.56	24.13	84.8	6.82%
Martin Creek	9.28	-na-	16.28	15.05	8.33	4.66	22.47	66.8	5.38%
Lower Mainstem Big River	10.95	-na-	9.20	12.24	13.71	24.31	22.76	82.2	6.62%
NORTH FORK BIG RIVER	**43.49**	**3.60**	**82.15**	**71.11**	**68.16**	**19.70**	**44.27**	**289.0**	**23.3%**
	% of PW Total	**1.2%**	**28.4%**	**24.6%**	**23.6%**	**6.8%**	**15.3%**		
Upper North Fork Big River	8.46	-na-	13.31	18.59	7.79	3.41	16.76	59.9	4.82%
James Creek	6.96	-na-	12.29	17.70	4.20	7.01	10.30	51.5	4.15%
Chamberlain Creek	12.28	0.52	27.54	17.26	14.27	2.04	2.29	63.9	5.15%
East Branch North Fork Big	8.06	-na-	10.14	4.45	26.62	2.75	9.42	53.4	4.30%
Lower North Fork Big River	7.73	3.08	18.87	13.11	15.28	4.49	5.50	60.3	4.86%
MIDDLE BIG RIVER	**17.85**	**13.51**	**19.68**	**40.90**	**7.39**	**19.52**	**53.20**	**154.2**	**12.4%**
	% of PW Total	**8.8%**	**12.8%**	**26.5%**	**4.8%**	**12.7%**	**34.5%**		
Middle Big River	13.07	8.45	12.34	30.87	6.84	16.37	40.78	115.7	9.31%
Two Log Creek	4.78	5.06	7.34	10.03	0.55	3.15	12.42	38.6	3.10%
SOUTH FORK BIG RIVER	**54.46**	**1.83**	**82.57**	**72.32**	**69.08**	**22.86**	**67.94**	**316.6**	**25.5%**
	% of PW Total	**0.6%**	**26.1%**	**22.8%**	**21.8%**	**7.2%**	**21.5%**		
Upper South Fork Big River	8.32		17.59	8.77	1.60	1.56	4.76	34.3	2.76%
Middle South Fork Big River	11.17		20.85	17.90	5.94	1.22	11.81	57.7	4.65%
Daugherty Creek	16.65		33.83	26.38	18.53	5.06	25.17	109.0	8.77%
Lower South Fork Big River	18.32	1.83	10.30	19.27	43.01	15.02	26.20	115.6	9.31%
LOWER BIG RIVER	**32.47**	**48.83**	**30.35**	**22.55**	**36.19**	**38.81**	**71.77**	**248.5**	**20.0%**
	% of PW Total	**19.6%**	**12.2%**	**9.1%**	**14.6%**	**15.6%**	**28.9%**		
Lower Big River	7.69	20.88	3.65	2.21	18.85	4.43	13.28	63.3	5.10%
Little North Fork	12.49	12.03	18.62	11.21	4.31	13.89	24.79	84.8	6.83%
Laguna Creek	5.07	4.14	5.13	2.14	3.90	7.46	18.25	41.0	3.30%
Big River Estuary	7.22	11.78	3	6.99	9.13	13	15.45	59.3	4.78%
TOTAL BIG WATERSHED	**181.05**	**67.77**	**256.28**	**260.87**	**208.19**	**142.41**	**306.54**	**1242.06**	**100.0%**
% of Total Roads		**5.46%**	**20.63%**	**21.00%**	**16.76%**	**11.47%**	**24.68%**	**100.00%**	

Notes: Base road data from CDF, substantially added to and corrected by GMA.
Eastern portion of watershed not covered by 1936 aerial photographs.
Road segments not codified by year by CDF or mapped into specific period by John Coyle are all included in 2000 period.

Table 14: Summary of Sediment Inputs to the Noyo River Watershed as Derived from Data Presented by Matthews (1999)

Time period	Background Sediment Delivery (tons/mi^2/yr)			Management-related Sediment Delivery (tons/mi^2/yr)						Total (tons/mi^2/yr)
	Mass wasting	Surface erosion	Stream bank erosion*	Mass wasting--Railroad	Mass wasting--Harvest	Mass wasting--Roads	Surface erosion--Roads	Surface erosion--Skid trails	Fluvial erosion--Roads	
HAA (27.17 mi^2)										
1933-1957	24	75		0	2	0	41	18	unknown	160+
1958-1978	67	75		8	25	83	156	46	unknown	460+
1979-1999	140	75		9	8	106	162	17	unknown	517+
NFAA (25.07 mi^2)										
1933-1957	6	75		0	0	6	11	1	unknown	99+
1958-1978	145	75		0	35	78	142	55	unknown	530+
1979-1999	157	75		0	5	106	182	21	unknown	546+
SFAA (27.46 mi^2)										
1933-1957	305	75		13	9	14	74	3	unknown	493+
1958-1978	94	75		7	2	19	132	5	unknown	334+
1979-1999	98	75		0	5	18	148	13	unknown	357+
MAA (33.3 mi^2)										
1933-1957	46	75		157	0	2	17	2	unknown	299+
1958-1978	40	75		100	0	37	118	4	unknown	374+
1979-1999	22	75		12	53	76	201	13	unknown	452+
NOYO RIVER WATERSHED										
1933-1957	95	75	200	49	3	5	35	6	unknown	468+
1958-1978	83	75	200	33	14	53	136	26	unknown	620+
1979-1999	99	75	200	6	20	76	175	16	unknown	667+
1933-1999	91	75	200	31	12	42	111	15	unknown	577+

*Stream bank erosion was estimated by applying a regional figure to all but about 30% of Noyo River watershed stream miles. The 30% excluded from the calculation represent the Noyo River itself that from limited observation appears to have relatively stable banks. The calculation was not broken down by assessment area. As such, the total sediment delivery for each assessment area does not include streambank erosion and is therefore underestimated. The total estimates of sediment delivery per assessment area and for the whole watershed are also lacking figures for fluvial erosion from roads. For this reason, too, the calculation results are underestimates.

** Any discrepancies between Table 13 and Table 14 are the result of rounding numbers up and down.

Table 1. Preliminary sediment inputs and road density by planning watershed for Mendocino Redwood Company timberlands in the Big and Noyo River basins (data provided by Mr. Chris Surfleet, Hydrologist, Mendocino Redwood Company, Ft. Bragg,).

Calwater Planning Watershed	Watershed Analysis Unit	Total Sediment Inputs (tons/mi^2/yr^2)	Total Sediment Inputs (yd^3/mi^2/yr)	Non-Road Mass Wasting Sediment Input (%)	Road Mass Wasting Sediment Input (%)	Road Surface and Point Source Sediment Input (%)	Skid Trail Sediment Inputs (%)	Road Density (mi/mi^2)
Dark Gulch	Big River	230	180	48%	9%	ND	43%	ND
East Branch NF Big River	Big River	940	720	15%	11%	43%	32%	ND
Laguna Creek	Big River	ND	ND	ND	ND	ND	ND	ND
Lower North Fork Big River	Big River	870	670	28%	24%	31%	17%	ND
Martin Creek	Big River	ND	ND	ND	ND	ND	ND	ND
Mettick Creek	Big River	1050	810	18%	30%	19%	32%	ND
Rice Creek	Big River	740	570	37%	11%	49%	3%	ND
Russell Brook	Big River	910	700	10%	22%	48%	20%	ND
South Daugherty Cr	Big River	990	760	15%	27%	39%	18%	ND
Two Log Creek	Big River	1400	1080	20%	23%	21%	36%	ND
Hayworth Creek	Noyo River	690	530	50%	3%	14%	33%	6.2
McMullen Creek	Noyo River	490	380	53%	20%	11%	16%	6.8
Middle Fork Noyo River	Noyo River	440	340	37%	2%	32%	29%	7.5
North Fork Noyo River	Noyo River	370	280	28%	8%	31%	33%	8.1
Olds Creek	Noyo River	380	290	32%	27%	36%	5%	7.4
Redwood Creek	Noyo River	210	160	21%	14%	43%	22%	7.7
Upper Noyo River	Noyo River	730	560	26%	59%	15%	ND	14.7

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WATER RESOURCE ISSUES AND SOLUTIONS FOR FOREST ROADS IN CALIFORNIA

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ABSTRACT

Unpaved forest roads are recognized as being the primary source of anthropogenic sediment that is delivered to stream channels in managed forested watersheds in the western United States. Research studies, sediment budgets, and agency monitoring projects in California have shown that road-stream crossings and road segments that drain to crossings are high-risk sites for sediment input. In particular, older "legacy" roads and crossings that pre-date current forest practice rules can be sources of chronic and episodic sediment delivery. Because many California streams provide habitat for listed anadromous fish species, there has been increased emphasis on reducing road-related sediment production. Numerous efforts are made by resource agencies, landowners, and watershed groups to address road-related sedimentation. Most owners of large forest parcels have road management plans for inventorying and prioritizing high-risk road sites for remedial treatments or decommissioning. State and federal grant and cost-sharing programs are used extensively to support road improvement projects. A recent review of state-funded road decommissioning work showed that accepted protocols are generally effective for preventing erosion and sediment delivery, but that they are not uniformly implemented at road-stream crossings. Training workshops and guidebooks are employed to improve stakeholder knowledge regarding proper design and installation of crossing structures and treatment of roads to reduce sediment delivery.

Key words: sediment delivery, forest roads, erosion, California

INTRODUCTION

Unpaved roads are a major anthropogenic cause of sedimentation in many forested watersheds of the western United States (Megahan and Kidd 1972, Reid and Dunne 1984, Luce and Black 1999, MacDonald et al. 2004, Sugden and Woods 2007). Road-related sediment sources include surface erosion, gullying and mass wasting (Reid and Dunne 1984, Wemple et al. 2001). Sediment may be delivered to streams either episodically when roads or road-stream crossings catastrophically fail, or chronically due to incremental surface erosion. Excessive sedimentation associated with roads is of concern because of potential impacts on stream habitat and water quality (Cederholm et al. 1981, Murphy 1995, Waters 1995, Spence et al. 1996).

Forest roads provide access for timber operations and fire suppression. In California alone, it is estimated that there are at least 60,000 kilometers of unpaved road on approximately 22,000 square kilometers of private commercial timberland zoned for timber production (CDF 2003). Public lands, such as National Forests, have thousands of kilometers of additional unpaved roads. Many of these roads were built decades ago to very low design standards, often in environmentally sensitive locations such as on unstable slopes and near streams. A significant number of the older roads are part of the current road network, while others were neglected and abandoned with no consideration of ongoing erosional impacts. These "legacy" roads are particularly susceptible to catastrophic failure during high magnitude, low frequency storm events, such as the one in 1997 that caused extensive flooding throughout a large part of northern California (Furniss et al. 1998, Madej 2001).

In this paper, we review results from field, modeling and monitoring studies of forest road effects on erosion and sediment delivery in California. We then describe the regulatory and voluntary efforts that are being used by agencies and landowners to reduce adverse impacts of forest roads. Although many steps are being taken to address these problems, there are still lingering issues. New regulatory, monitoring and educational initiatives are briefly summarized.

LITERATURE REVIEW OF THE EFFECTS OF FOREST ROADS ON EROSION AND SEDIMENTATION IN CALIFORNIA

Regional variations in climate, geology, topography, and road construction practices drive the interactions between geomorphic processes and road networks (Montgomery 1999, Jones et al. 2000, Wemple et al. 2001). Consequently, road study methods and results have varied widely as a function of California's diverse geomorphology (Harden 1997, CGS 2002).

In response to this variability, we stratify studies into the following geomorphic regions: the northern Coast Range (i.e., North Coast), the Cascade Range, the Sierra Nevada, and the Klamath Mountains. The North Coast is dominated by the landslide-topography of the highly erodible Franciscan Complex (Jennings 2000, Harden 1997) and has some of the highest unit area peak runoff rates in the United States (O'Connor and Costa 2004). The Cascade Range is located in northeastern California and terminates at Lassen Peak. It is composed of relatively resistant volcanic rocks, gently sloping tablelands, and low overall stream density (Harden 1997, Benda et al. 2003). The Sierra Nevada's western slope is gently sloping with deeply incised river canyons, while the eastern escarpment has steeper slopes. Its geology consists primarily of granitic, metamorphic and volcanic rocks with varying erosion potentials. The Klamath province is located between the Coast and Cascade Ranges in northern California and is geologically similar to the Sierra Nevada, but with highly dissected topography and a climatic regime similar to the North Coast.

To compare data and to provide a watershed context, we have converted erosion rates into consistent units that reflect the annual mass of road erosion in a watershed (i.e., Mg km^{-2} yr^{-1}). The originally reported erosion rates are shown in Table 1. We assign a bulk density of 1.6 g cm^{-3} to convert volume estimates to mass, as has been done elsewhere in California (Best et al. 1995, Pitlick 1995, Coe 2006). Rates are normalized by watershed area by dividing the mass of erosion by the study area, or by multiplying mass per unit length of road (i.e., Mg km^{-1}) by the total length of road in the study area, or by the average road density of the study area. While we recognize that some erosion processes are episodic in nature, rates are normalized by number of years in the study period for comparison. These normalized erosion rates are to provide readers the ability to evaluate the relative differences in erosion rates between different road erosion processes (i.e., mass wasting vs. gully erosion vs. surface erosion) and different geomorphic regions.

Research Studies

Most of the road erosion studies in California have been conducted in the Coast Ranges. This includes a suite of studies that documented the effects of logging on sedimentation in the Redwood Creek watershed in Humboldt County. Most of these studies evaluated the integrated impact of past logging practices from the 1950's to 1970's and did not explicitly evaluate the impacts of logging after the passage of the California Forest Practice Act of 1973. These studies found that inadequately designed road-stream crossings were responsible for stream diversions and very high rates of gully erosion in tributaries to Redwood Creek at Redwood National Park (Hagans et al. 1986, Weaver et al. 1987, Hagans and Weaver 1987, Weaver et al. 1995, Best et al. 1995). Episodic gully erosion from diverted road-stream crossings in five Redwood Creek tributaries produced 8600 Mg km^{-2} over 33 years, or an average of 260 Mg km^{-2} yr^{-1} (Hagans et al. 1986). Sediment production from discrete road induced landslides averaged from 155-176 Mg km^{-2} yr^{-1} over approximately a 25 to 28 year time span (i.e., mid-1950's to early-1980's) (Best et al. 1995, Pitlick 1995). Surface erosion from roads and skid trails was less important, accounting for only 4 percent of measured sediment in the lower Redwood Creek basin or an average yearly input of 34 Mg km^{-2} yr^{-1} (Weaver et al. 1987, Hagans et al. 1986, Hagans and Weaver 1987). Road related erosion comprised more than 30 percent of the total sediment budget over a 25-year time span, or 356 Mg km^{-2} yr^{-1}, for the 10.8 km^2 Garret Creek watershed (Best et al. 1995).

Rice (1999) assessed road erosion on private lands in a portion of the Redwood Creek basin and found that erosion averaged 43 Mg km^{-2} yr^{-1} over a 17-year time span from 1980 to 1997. This rate was 12 percent of the value reported for the Garret Creek watershed (Best et al. 1995). Rice (1999) cautioned about direct comparisons of studies with different objectives and methods, but concluded that the lower rates of erosion were due to improved road requirements mandated by the Forest Practice Rules.

In the Caspar Creek watershed located in Mendocino County, erosion and sediment yield associated with timber harvesting and roads have been studied since 1962 (Lewis et al. 2001). As in Redwood Creek, high levels of erosion have been attributed to roads that were constructed prior to 1973 (Rice et al. 2004). Approximately 70 percent of landslides that occurred after logging in the South Fork of Caspar Creek were associated with roads and landings (Cafferata and Spittler 1998). Over a four-year period, erosion due to roads in the South Fork of Caspar Creek was estimated at 130 Mg km^{-2} yr^{-1} (Krammes and Burns 1973). In the North Fork, where timber operations were conducted under modern forest practice rules, road-related erosion was less than half that measured in the South Fork, and the erosion was rarely delivered to the channel network due to road location (Rice et al. 2004). In Caspar Creek and three nearby watersheds, landslides in clearcut units that delivered sediment to stream channels were mostly associated with old roads that were constructed decades ago (Bawcom 2003).

The findings of studies at Redwood and Caspar Creeks are corroborated by other studies in the North Coast region. McCashion and Rice (1983) evaluated 553 kilometers of logging roads in the Six Rivers National Forest of Humboldt County and found that the road network produced 60 percent of the management-related erosion. Approximately 95 percent of the road-related erosion was from mass wasting. A mass wasting study on non-federal timberlands encompassing the entire North Coast and North Interior parts of the state found that 76 percent of management-related erosion was related to roads and landings (Durgin et al. 1989; Lewis and Rice 1989, Rice and Lewis 1991). Weaver and Hagans (1999) documented that road-related sediment delivery over a 50 year period in three Humboldt County watersheds ranged from 37 to 185 Mg km^{-2} yr^{-1}, with the majority of sediment coming from stream crossing diversions and road fill failures.

There has been less research on forest road erosion in the Klamath, Sierra Nevada, and Cascade regions of California. Road impacts in areas with soils derived from highly erodible decomposed granitic parent material have been studied in some locations. In the Scott River basin of the Klamath Mountains, Sommarstrom et al. (1990) found that road cuts in decomposed granitic soils produced 64 percent of road-related erosion. Erosion rates from roads (33 kg m^{-2} yr^{-1}) were comparable to those reported for other California watersheds with decomposed granitic soils. Their study represented the condition of roads pre-1990, before most road improvement efforts began on private and public lands in this area.

There have been recent studies of road surface erosion in the central and southern Sierra Nevada, where mass wasting is infrequent (Rice and Lewis, 1991, MacDonald et al. 2004). Coe (2006) documented sediment production and delivery in the central Sierra Nevada on both National Forest and private forest lands. Data were collected from sites with soils derived from weathered grandodiorite, andesitic lahar deposits, and granitic glacial deposits. There was a 16-fold difference in median sediment production rates between rocked and un-rocked road segments, and roads that had been recently graded produced more than twice the sediment per unit area as un-graded roads. The highest rates of erosion came from road segments with unusually high rates of subsurface stormflow interception by road cutslopes. Spatially, stream crossings were the main linkage for sediment delivery to the channel network, although road-induced gullies delivered almost as much sediment to the channel network as chronic surface erosion. Assuming a road density of 3.1 km km^{-2} (i.e., 5 mi mi^{-2}), Coe (2006) estimated that approximately 6 Mg km^{-2} yr^{-1} of road sediment would be delivered to the channel network. Sediment delivery problems were mostly related to older roads. Newer roads or roads upgraded to current Forest Service and State Forest Practice Rule standards performed better than older roads.

In the southern Sierra Nevada, Korte and MacDonald (2007) measured road and hillslope erosion over three winters during the calibration period of the Kings River Experimental Watershed Study (KREW) (Hunsaker and Eagan 2003). The dominant lithology in the KREW study area is granite. Their results are consistent with the results from earlier work in the central Sierra Nevada (MacDonald et al. 2004, Coe 2006). High variability in sediment production rates was found between years, between different types of road surfaces, and between individual road plots. Native and mixed surface roads produced approximately three times the sediment as gravel surfaced roads. The estimated sediment delivery from roads was compared to the measured sediment yields in three of the lower-elevation watersheds, and in two of these watersheds forest roads were estimated to contribute less than 10 percent of the measured sediment yield. In the third basin roads were estimated to contribute 25-50 percent of the total sediment yield, and nearly all of the road-related sediment came from a single mixed surface road that crossed the stream a short distance above the weir pond (A. Korte, Colorado State University, Fort Collins, CO, personal communication).

Sediment Budgets and Road-Related Sediment

Numerous sediment budgets have been developed for watersheds in the Coast Ranges, most of which have relied on aerial photograph analysis of landslide and surface erosion features, GIS digital terrain models, and limited field investigation. Rapid sediment budgets can produce estimates within a factor of two of actual sediment yield (Reid and Dunne 1996). Most of this work has been done as part of Total Maximum Daily Load (TMDL) watershed assessments. TMDLs are pollution control plans produced for watersheds listed as impaired by U.S. EPA under the Federal Clean Water Act. In general, these sediment budgets reveal that road surface erosion and road-related landslides are the dominant sources of sediment from current land management activities. Erosion from timber harvesting units is usually less significant. In a review of TMDL documents for nine North Coast watersheds, Kramer et al. (2001) reported that, on average, roads and skid trails contributed approximately two-thirds of all management-related sediment loading. This concurs with our analysis of 19 sediment budgets (16 produced for Coast Range basins), which shows that road-related sediment is responsible for approximately two-thirds of management-related sediment, or roughly one-third of total sediment production (Figure 1). Total natural and management-related sediment loads were estimated to be approximately equal. Kramer et al. (2001) were not able to discern to what degree older roads or roads constructed under current Forest Practice Rules were responsible for sediment production.

Forest roads do not dominate management-related sediment production in all watersheds where sediment budgets have been estimated. Roads were relatively smaller sources of total estimated sediment yield in the highly unstable Van Duzen watershed (U.S. EPA 1999) and in the Scott River basin, where stream bank failures were estimated to produce a significant proportion of total sediment (NCRWQCB 2005) (Figure 2).[1] In the Bear Creek and Jordan Creek watersheds of Humboldt County, PWA (1998a, 1999) estimated that roads produced only 8 percent and 21 percent of total sediment delivery, respectively. Hillslope and streamside landslide erosion are the dominant sediment sources in these two small, highly unstable basins.

As with road erosion studies, there have been few sediment budgets developed for the Klamath Mountains and the Sierra-Cascade region. Sommarstrom et al. (1990) produced a sediment budget for granitic tributaries of the Scott River watershed. They estimated that roads contributed 82 percent of the management-related sediment in these drainages. Of the total erosion, road cuts represented 40 percent of the soil loss, road fill 21 percent, and road surfaces 2 percent. Other sources included streambanks (23 percent), skid trails (13 percent), and landslides (<1 percent). Benda et al. (2003) produced a sediment budget for the Judd Creek watershed, a small tributary to Antelope Creek, which drains into the Sacramento River below Redding in the southern Cascade Range. No landslides were observed in this basin, where the average hillslope gradient is only 15 percent (Benda et al. 2003). Road-related erosion was estimated to produce only 3.5 percent of total estimated sediment yield, or an annual input of 9.4 Mg km^{-2} yr^{-1}, while post-fire erosion (68 percent) and bank erosion/soil creep (28 percent) dominated long-term sediment production. Considering just management-related erosion, road surface erosion and harvest unit erosion were estimated to produce 87 percent and 13 percent of the total, respectively. Benda et al. (2003) suggested upgrading or decommissioning logging roads within 60 meters of stream channels that have a high probability of sediment delivery to reduce forestry impacts on water quality.

Reid and Dunne (1996) produced a sediment budget for the higher elevation area of the KREW project in the southern Sierra. They estimated that bank erosion was causing half of current sediment delivery. Road surface erosion was the only disturbance category listed and was estimated to contribute less than one-quarter of the total sediment yield, or approximately 3 Mg km^{-2} yr^{-1}. Long-term post-fire erosion was not estimated. Euphrat (1992) created a sediment budget for the upper Middle Fork of the Mokelumne River in the central Sierra Nevada, which is heavily used for commercial timber harvesting. He estimated that road surfaces produced approximately 40 percent of the annual erosion. Nolan and Hill (1991) developed sediment budgets for four Lake Tahoe tributaries on the east slope of the Sierra Nevada and found that nearly all mobilized sediment was derived from stream channels (either stream banks or streambed). Hillslope erosion was a minor component of these sediment budgets, ranging from less than 5 percent to 11 percent.

Agency Monitoring Programs Related to Road Erosion

Monitoring programs conducted by state and federal (U.S. Forest Service) agencies have documented the frequency and causes of road-related erosion problems on private and public timberlands in California. This monitoring has been conducted in conjunction with evaluation of state Forest Practice Rule and Forest Service Best Management Practice (BMP) effectiveness. The earliest monitoring project was a qualitative assessment of 100 state-issued Timber Harvesting Plans (THPs) conducted in 1986 by a team of four resource professionals on non-federal timberlands. The team concluded that the Forest Practice Rules were generally effective when implemented on terrain that was not overly sensitive (i.e., erodible soils, high mass wasting potential), and that poor rule implementation was the most common cause of water quality impacts. Poor road location, construction, drainage and/or abandonment were noted as common reasons for significant adverse impacts (CSWRCB 1987).

[1] The Scott River TMDL sediment budget listed roads as one category, but they were a large part of an additional category denoted as "Effects of Multiple Interacting Human Activities" (EMIHA), since road runoff affecting streambank stability was included. Therefore, it is difficult to determine the total percentage of road-related sediment for this basin from the TMDL document (S. Sommarstrom, Etna, CA, personal communication).

The qualitative assessment completed in 1986 was not considered sufficient evidence for certifying the state Forest Practice Rules as adequate best management practices. Consequently, two state-sponsored THP monitoring programs were conducted from 1996 through 2004. The Hillslope Monitoring Program (HMP) analyzed data collected by private contractors from 1996 through 2001 on 300 randomly selected THPs and Non-industrial Timber Management Plans (NTMPs) located throughout the state on non-federal timberlands (Cafferata and Munn 2002, Ice et al. 2004). The objective of the program was to evaluate the implementation and effectiveness of the California Forest Practice Rules in protecting water quality. On-site data were collected from randomly located 305 meter (1000 foot) road segments and at road-stream crossings, along with other high risk locations (i.e., skid trails, landings, etc.). The HMP found that the implementation of rules governing roads averaged 93 percent and that the required practices appeared to have been effective in preventing erosion when they were properly implemented. For this study, road erosion features were almost always associated with improperly implemented Forest Practice Rules. Overall, 5.5 percent of the road drainage structures were inadequately designed, constructed, or maintained on the 167 kilometers of road segments monitored, and approximately 15 percent of the 1,132 inventoried road erosion features (i.e., rills, gullies, mass failures, cutslope/fillslope sloughing) delivered sediment to stream channels.

In a similar study, California Department of Forestry and Fire Protection Forest Practice Inspectors collected onsite monitoring data from 2001 through 2004 as part of the Modified Completion Report (MCR) monitoring program (Brandow et al. 2006). A random draw of 12.5 percent of all completed THPs was evaluated (281 THPs), and high risk and highly sensitive parts of each plan (roads, crossings, and stream buffers) were randomly sampled and evaluated. Nearly all the identified road rule implementation departures were related to drainage (e.g., water-break spacing). Five percent of inventoried road-related features had improper implementation of rule requirements, and approximately 8 percent of road erosion features delivered sediment to stream channels, nearly always when road rules were improperly implemented. Road-stream crossing effectiveness ratings were generally similar to HMP results and showed that diversion potential, culvert plugging, and road drainage structure function near crossings were common problem areas. Approximately 20 percent of the stream crossings in both the MCR and HMP studies had significant implementation and/or effectiveness problems.

On federal lands in California, the U.S. Forest Service collected data from 1992 through 2002 on over 3,100 randomly located sites to evaluate the implementation and effectiveness of its water quality BMPs (USFS 2004). The BMP Evaluation Program used 29 different onsite monitoring protocols to evaluate BMP implementation and effectiveness, with the majority related to timber and engineering practices. Results showed that while some improvements to current practices were necessary, the program performed reasonably well in protecting water quality on National Forest lands (approximately 8 million hectares or one-fifth of the state). BMP implementation and effectiveness were relatively high for most activities (including timber and engineering) and impacts on water quality were relatively rare, particularly in recent years. Significant water quality impacts were typically caused by lack of or inadequate BMP implementation and mostly related to engineering practices (nearly 60 percent). Roads, and in particular stream crossings, created the greatest number of problems. Fifty-four percent of the sites where elevated water quality impacts were observed were associated with roads.

Summary of Literature Review

Research studies, sediment budgets and monitoring throughout California's forestlands have all identified road-related erosion as a prominent factor affecting sediment yields in watersheds managed for timber production. In the relatively unstable North Coast region, road-related mass wasting has been well-documented, but there has been less emphasis on road surface erosion. In the Sierra-Cascade region, hillslope instability is less common (Durgin et al. 1989), and the few studies that exist focus on road surface erosion. Causal mechanisms for road-related erosion have been identified through state and federal monitoring programs. Sediment budgets reveal that roads often produce at least two-thirds of management-related erosion in forested watersheds. Older "legacy" roads that pre-date current state Forest Practice Rules and U.S. Forest Service BMPs are commonly identified as a major source of sediment. Usually a small proportion of the total road system produces most of the sediment and there are certain site factors that predispose a road segment or road-stream crossing to erosion. In particular, native surface road segments located within 60 meters of streams that are connected to the channel by inboard ditches are particularly high-risk sites for fine sediment delivery.

To place road erosion rates into proper perspective it is important to compare rates of road-induced erosion to long-term background erosion rates. In recent years, background erosion rates have been derived over millennial time scales using cosmogenic radionuclide dating (Riebe et al. 2000, Kirchner et al. 2001, Ferrier et al. 2005). When compared to millennial averaged sediment production rates derived from cosmogenic radionuclide (CRN) dating (Riebe et al. 2001, Ferrier et al. 2005), road erosion rates are approximately equal to long-term rates in the North Coast and are an order of magnitude lower in the Sierra Nevada (Figure 3). Given that road erosion rates and background erosion rates for the North Coast are almost equal in magnitude, we conclude that roads have increased the risk and size of catastrophic erosion events in the North Coast region. While data are limited from the Sierra Nevada, we hypothesize that long-term erosion in the Sierra Nevada is dominated by infrequent, catastrophic disturbance and is similar to the erosional regime reported for the Idaho Batholith, where incremental erosion occurs most of the time but is a small fraction of long-term sediment yield (Kirchner et al. 2001). The disparity between road erosion rates and long-term background erosion rates in the Sierra Nevada (Figure 4) indicate that impacts from road erosion may be more an issue of timing rather than magnitude, as chronic erosion has the potential to disrupt aquatic ecosystems that are adapted to infrequent, catastrophic erosion (Kirchner et al. 2001).

APPROACHES FOR MITIGATING EFFECTS OF FOREST ROADS

Landowners and agencies address erosion and sedimentation caused by forest roads in California with both voluntary and regulatory approaches. The state Forest Practice Rules have been recently modified to address road erosion issues based on agency monitoring results and because of state and federal listings of anadromous fish species. One Regional Water Quality Control Board is requiring Erosion Control Plans for projects involving timber operations. Landowners have undertaken inventories to identify high-risk road segments and stream crossings for repair or removal. These inventories are commonly done in conjunction with the preparation of road management plans. Finally, there have been many road and stream crossing upgrading and decommissioning projects implemented using both private and public funds.

Regulatory Approaches

California's Forest Practice Rules contain extensive requirements designed to limit road erosion and sediment delivery associated with timber harvesting on non-federal timberlands (CDF 2007). To mitigate impacts from past poor road location and construction practices, the Forest Practice Rules require that existing road erosion sources must be repaired as part of an approved THP to lessen or avoid significant adverse water quality impacts. Numerous specific rules are enforced, such as the requirement that road-stream crossings be constructed and maintained in a manner that prevents diversion of streamflow down roads, which has been mandated since 1990. Rules added in 2000 require special road design and construction practices in watersheds with state or federally listed threatened and endangered fish species.[2] Among other things, these rules specify that roads should be out-sloped where feasible and drained with rolling dips or water-breaks, to reduce hydrologic connectivity and sediment delivery to streams. Additionally, to reduce the potential impacts of road-stream crossings, since 2000 the Forest Practice Rules have specified that all constructed or reconstructed permanent road-stream crossings must accommodate the estimated 100-year flow, including debris and sediment. To ensure that the Forest Practice Rules and additional THP requirements have been properly prescribed and implemented, California Department of Forestry and Fire Protection Forest Practice Inspectors conduct field inspections before, during and after harvesting.

In 2003, the Regional Water Quality Control Boards (RWQCBs) in California began requiring a permit for discharge of sediment from timber harvesting on non-federal timberlands. The North Coast Regional Water Quality Control Board's (NCRWQCB) permit program requires prevention/minimization of new sediment sources and mitigation of existing sediment sources through an Erosion Control Plan (ECP). The ECP outlines how a landowner will identify areas of sediment delivery, identify areas at risk of sediment delivery, and control sediment delivery associated with past and present land management activities (NCRWQCB 2001). The ECP requirement has forced landowners on the North Coast to make improvements on roads over and above those mandated by the Forest Practice Rules. In addition, the Central Valley Regional Water Quality Control Board (CVRWQCB) requires that landowners perform qualitative hillslope monitoring for most THPs, with an emphasis on checking road segments and stream crossings that pose a high risk to water quality (CVRWQCB 2007).

Voluntary Approaches—Road Inventory Work and Modeling

Road inventories and assessments are used to determine which roads on an ownership have the potential to deliver large amounts of sediment to streams, to establish priorities for road improvement (or upgrading) projects, and to develop road decommissioning schedules (Weaver and Hagans 1999, CDFG 2006). Although these inventories have been conducted on numerous California properties, industrial forest managers have taken particular interest and initiative in completing them to reduce potential restrictions on future operations. The main steps for road inventory work include: (1) identifying all permanent, seasonal, temporary, and historic or "legacy" roads, usually with aerial photographs, (2) conducting a field inventory to document erosion sources that can deliver sediment to the stream channel, (3) inputting the estimates of potential sediment delivery in a database and summarizing the data, and (4) developing a plan to remedy identified problem sites

[2] Ligon et al. (1999) made recommendations for changes in forestry regulations related to protection of salmonid habitat, including suggested changes for rules related to road construction/maintenance and stream crossings. Some of the recommendations were included in the Threatened or Impaired Watersheds Rule Package passed by the State Board of Forestry and Fire Protection in July 2000.

(CDFG 2006). Estimates of potential sediment delivery are calculated and categorized by source (i.e., number of cubic meters of sediment due to perched fill, inadequate crossing design, etc.) (CDFG 2006). All stream crossings are included along with a rating of their ability to pass a designated flood flow (e.g., 100-year recurrence interval flood), ability to allow fish passage (if applicable), diversion potential (should the crossing fail), mechanical damage or repair needs, etc.

Predictive models are sometimes used to identify roads with the highest potential for sediment delivery. These include the physically-based Disturbed WEPP:Road and the empirical SEDMODL/SEDMODL2. A field test of these models in the Oregon Coast Range found that SEDMODL2 produced erosion estimates much closer to measured sediment data than WEPP:Road, but neither model consistently ranked the sediment production rates from measured road segments (Amman 2005). For this reason, road erosion models are best used to estimate the relative magnitude of road erosion at the watershed scale (Raines et al. 2005). Models can also be used as conceptual tools that provide a sound basis for field data collection (Raines et al. 2005).

Through road inventory work, assisted by modeling efforts in some cases, landowners have developed databases with lists of road segments/sites rated as high, moderate, or low priority for improvement work. Options for improvement work include road decommissioning, upgrading, and no-action. Cost-effectiveness for treating individual sites and road segments is determined by dividing the cost of accessing and treating a site by the volume of sediment prevented from delivery to stream channels (Weaver and Hagans 1999). Treatment priorities are then established, with the road sites and road segments having the most cost-effective treatments implemented first. High priority treatment sites typically have at least 19 m^3 (25 yd^3) of sediment delivery potential to a stream channel, a high or moderate treatment rating, and a predicted cost-effectiveness of approximately \$7 to \$20/m^3 or less (PWA 1998b). The largest road sediment sources for the North Coast region are usually: (1) stream crossing fills, (2) fill slope failures, and (3) road surface erosion (W. Weaver, PWA, Arcata, personal communication). For the Sierra Nevada, where there is a much lower landslide frequency, road surface erosion and gully erosion are usually the largest sediment sources.

Road inventory and modeling work on both private and public landowners are often part of a more comprehensive road management plan (RMP). These are long-term plans for the ownership's transportation system and are often included as part of larger scale land management plans or landscape-level documents. RMPs usually include a section for scheduling/prioritization of sites requiring work based on completed or anticipated road inventory work. They also commonly have sections for specifying road design and construction standards, road use restrictions, and a road inspection/maintenance program.

Completed Road Improvement Work

The identification and treatment of problematic road sites and road segments has become big business on the North Coast of California. Millions of dollars of public and private money are expended yearly on upgrading or "decommissioning" (i.e., closing or removing) problem roads and stream crossings. Road improvement work is funded by a variety of sources, including state and federal funding for public ownerships, private funds for company timberlands, and grants provided by state and federal programs for non-federal timberland owners. State and federal grant and cost-sharing programs are used extensively to support road improvement projects.

Most publicly funded road improvement projects are catalogued in the California Habitat Restoration Project Database (CHRPD) maintained by the California Department of Fish and Game (DFG) and the Pacific Marine Fisheries Management Council. The CHRPD was queried to provide summary data on state funded road projects. Altogether, on all types of ownerships, nearly 1,600 kilometers of roads have been treated or are under contract to be treated utilizing funding provided by DFG. Forty percent of these road segments are or will be decommissioned and 60 percent upgraded. Approximately 4,000 road-stream crossings have been removed or replaced over the past 10 years in coastal California using state and federal grant funding (L. Williams, DFG, Sacramento, CA, personal communication).

Road improvement work has occurred on industrial timberland in California using a combination of private and public funding. The Pacific Lumber Company (PALCO) has "storm-proofed" (i.e., added surface rock, upgraded crossings, hydrologically disconnected) 845 kilometers of sub-standard road at a cost of \$22,000-25,000 per kilometer over the past seven years and has upgraded (i.e., improved crossings, hydrologically disconnected) an additional 438 kilometers of road (K. Sullivan, PALCO, personal communication). The company had also decommissioned 61 kilometers of road through 2004 (J. Barrett, PALCO, personal communication). All of this work has been done at company expense. Other major timber companies in California also have road upgrading and decommissioning programs. As of 2004, roughly 129 kilometers of road had been decommissioned on land owned by Green Diamond Resource Company (formerly Simpson), Hawthorne Timber Company and Mendocino Redwood Company, using matching funds from the DFG (W. Weaver, Pacific Watershed

Associates, personal communication). Mendocino Redwood Company alone has decommissioned in excess of 62 kilometers of road from 1998 through 2006 (K. Vodopals, Mendocino Redwood Company, personal communication). Campbell Timberland Management (CTM, managers for Hawthorne Timber Company lands) partners with cooperators such as Trout Unlimited to seek restoration grant funding through state and federal programs. These programs have facilitated road inventorying and road restoration projects at the planning watershed (1,200 to 4,000 ha) scale. Restoration grant projects are not linked to Timber Harvesting Plan permits, but rather occur as part of an ongoing environmental enhancement program. CTM prioritizes watershed restoration efforts based on two criteria: (1) maintenance or improvement in watersheds with the highest quality habitat and the most robust populations of listed anadromous fish, and (2) watersheds targeted by state or federal planning documents as key watersheds for recovery (P. Ribar, Campbell Timberland Management, personal communication).

In the Klamath Mountains, local watershed organizations have been active in bringing together stakeholders to complete watershed-scale forest road inventories and road improvement work (S. Farber, Timber Products Company, Yreka, personal communication). One notable success story has been the French Creek Watershed Advisory Group, which includes the U.S. Forest Service, Fruit Growers Supply Company, Timber Products Company, Roseburg Resources Company, and the Siskiyou County Road Department. In this watershed composed of decomposed granitic soils, over 61 kilometers of road was re-contoured and rocked, 6 kilometers of road were decommissioned, and many kilometers of road were closed to wet season use. Improvements in aquatic habitat conditions in French Creek have been documented with instream monitoring following the completion of the road improvement work (S. Sommarstrom, Etna, personal communication). An unknown but potentially substantial amount of road upgrading and decommissioning has been done at private expense on industrial timberland in the Sierra-Cascade region (Harris and Cafferata 2005).

Road upgrading and decommissioning is also occurring on public land. The U.S. Forest Service, National Park Service, Bureau of Land Management (BLM) and California State Parks have all been working to reduce sediment production from their road systems, primarily by decommissioning old logging roads. The Forest Service decommissioned 2,500 kilometers of road in California from 1994 through 2005 and reconstructed an additional 11,916 kilometers (J. TenPas, USFS, Vallejo, personal communication). Since 1978, 370 kilometers of former logging road, including 990 stream crossings, have been decommissioned in Redwood National Park (Harris and Cafferata 2005). The BLM decommissioning program has focused on lands in the Sinkyone Wilderness, the Mattole River watershed, the South Fork Eel River basin, and the Headwaters Forest Reserve, all of which are in Mendocino and Humboldt Counties. Since 1995, the BLM has decommissioned approximately 56 kilometers of former logging roads (D. Averill, Bureau of Land Management, personal communication).

In rural subdivisions in the Coast Ranges, at least 322 kilometers of road have been upgraded and 16 kilometers of road decommissioned using DFG funds (W. Weaver, PWA, personal communication). Also, the Five County Salmon Conservation Program, which covers Del Norte, Humboldt, Mendocino, Trinity and Siskiyou Counties, has led efforts to improve the quality of county-maintained road systems (M. Lancaster, Trinity County Planning Department, personal communication). This has included complete assessments of future sediment delivery, the development of prioritized plans for thousands of sediment reduction projects, and the identification and elimination of barriers to fish passage (Lancaster and Pérez 2001). Some coastal counties have increased their regulatory controls over rural road construction, primarily through the development and enforcement of grading ordinances.

Effectiveness of Current Road Improvement Approaches

Several assessment and monitoring projects have been undertaken to determine how effective road decommissioning and upgrading work has been in reducing road impacts. PWA (2005a) evaluated 82 kilometers of road decommissioning completed between 1998 and 2003 with funding from DFG in northwestern California, including 275 stream crossings and 111 landslides. The purpose of this assessment was to ascertain whether DFG's standard protocols for road decommissioning were successful in achieving their objectives. These procedures include complete removal of crossing fill material, excavation of unstable fill from the road prism and landings, and disconnecting road drainage from stream channels.

Overall, 57 percent of the crossing sites evaluated did not meet one or more of the accepted DFG decommissioning protocols. Crossings were found to account for 85 percent of the documented post-decommissioning sediment delivery. The average post-project sediment delivery at crossings was approximately 5 percent of the pre-treatment fill volume and the estimated average erosion for all 275 decommissioned crossings was approximately 26 m^3 following one to six over-wintering periods. The average delivery volume for stream crossings meeting DFG protocols was 18 m^3/site, while the average delivery volume for crossings that did not meet one or more of the DFG decommissioning standards was 32 m^3/site. PWA (2005a) concluded that erosion and sediment delivery from decommissioned stream crossings is unavoidable at all but the smallest crossings. The most common problem at decommissioned stream crossing sites was unexcavated fill. Sixty percent of sediment delivery at decommissioned stream crossings was due to natural or relatively unavoidable causes and 40 percent was attributed to operator or supervision problems. Approximately 70 percent of the avoidable operator-caused erosion was due to unexcavated fill left in the crossing. PWA (2005a) further concluded that: (1) the DFG decommissioning protocols for stream crossings are effective but are not being uniformly implemented at all sites, (2) the DFG decommissioning protocols for landslides are effective and are being followed, and (3) the DFG decommissioning protocols for road drainage are effective and are being employed correctly.

Short-term sediment impacts due to channel adjustments following crossing removal have been documented in several other smaller-scale northwest California studies (Klein 1987, Madej 2001, Klein 2003, PWA 2005b, and Keppeler et al. 2007). Post-project sediment delivery at stream crossings for these studies has ranged from approximately 12 m^3 to 50 m^3 (Figure 4). In general, these studies have shown that road treatments can reduce the long-term sediment production from decommissioned roads. Excavated crossings are the major short-term source of sediment input to stream channels following road decommissioning. Post-treatment sediment delivery from decommissioned crossings will likely be approximately 5 percent of pre-treatment sediment delivery potential, but can range up to roughly 20 percent. Most of the sediment input at excavated crossings can be expected to occur during the first few winters following treatment.

Only one study conducted to date has documented sediment delivery associated with stream crossing upgrade work. Harris et al. (in press) studied 30 crossings in small headwater streams located in northwestern California before and after new culverts were installed as part of THPs. They found that 11 sites showed no measurable erosion or sediment delivery. Five sites produced 3.8 m^3 or more of sediment, mainly due to channel incision. The maximum sediment production was approximately 7.6 m^3 at two sites, and average erosion for all sites was roughly 1.5 m^3 (Figure 4). In general, the extensive erosion control measures implemented at most of these study sites were effective in preventing construction-related adjustments after one winter.

The environmental benefits of upgrading forest roads with rock surfacing to minimize turbid runoff during wet weather road use in northwestern California was recently evaluated by Toman and Skaugset (2007). In this study, suspended sediment concentration was lowest from treated segments where ruts were not produced in truck wheel paths, and formation of ruts was found to be a function of aggregate depth. Toman and Skaugset (2007) suggest designing the aggregate surface to resist rutting to minimize sediment delivery. In the Sierra Nevada, Coe (2006) reported that rock surfacing generally reduces road sediment production by an order of magnitude or more relative to unsurfaced roads.

NEW REGULATORY, MONITORING AND EDUCATIONAL APPROACHES

Although it is certain that today's road design, construction and management practices are a vast improvement over the practices used decades ago, there are still lingering issues regarding forest roads in California:

- Monitoring indicates that stream crossing installations on Timber Harvesting Plans are sometimes improperly implemented.
- Although road erosion can be effectively controlled by surfacing, not all high risk road segments at or near streams have been treated to reduce surface erosion.
- Research on road surface erosion has been focused in the Sierra Nevada. There are few data currently available for the Coast Ranges.

There are other important issues that are not discussed in this paper. For example, the implications of a general shift in California from even-aged to uneven-aged management due to wildlife and fisheries concerns have not been evaluated. Road density is generally higher on ownerships harvested repetitively with ground-based harvesting systems (Harris and Cafferata 2005).

Uneven-aged management can result in more intensive road use and a higher potential for surface erosion compared to even-aged management. Also, while this paper focuses on issues related to forest management, it is likely that there are other road problems that are more important in the long term than timber-related road issues. In particular, our observations reveal that there are often greater problems with roads associated with lands that are not subject to state regulatory controls, such as are found in rural subdivisions (Harris and Cafferata 2004, 2005).

To address these lingering issues, new regulatory, monitoring and educational approaches are being used. A Road Rules Committee of the State Board of Forestry and Fire Protection (BOF) is currently working on ways to improve, revise and reorganize the California Forest Practice Rules related to roads. A draft version of this major revision is expected to be available by May 2007. This effort has the potential to improve the effectiveness of road-related Forest Practice Rules by upgrading practices, as well as making them easier to implement and enforce (Brandow et al. 2006). Additionally, the BOF is currently considering adoption of rules for a voluntary Road Management Plan.

To arrive at a common understanding of forestry related impacts on water quality, an Interagency Mitigation Monitoring Program (IMMP) was established in 2005 with members from state agencies involved in timber harvesting plan review. The initial focus has been on developing field methods for determining the implementation and effectiveness of practices at higher risk (non-random) road-stream crossing sites and road segments that drain to crossings. Pilot project work completed in 2006 showed that improper installation of crossings and drainage structures near crossings is often the major cause of water quality problems. Preliminary conclusions from the pilot work are that improved implementation of practices can be accomplished with additional timber operator education and more frequent multi-agency crossing inspections, both during logging operations and immediately following completion of harvesting. It is anticipated that this program will improve the effectiveness of practices applied at higher risk sites associated with crossings.

Several manuals and visual aids on road management and restoration have been developed for agency personnel, industry foresters, consulting foresters, and other resource professionals (Weaver and Hagens 1994, Flanagan et al. 1998, Keller and Sherar 2003, Cafferata et al. 2004, MCRCD 2004, CDFG 2006, Kocher et al. in press). To reduce erosion from county roads, road maintenance manuals have been produced for five northwestern California counties (Sommarstrom 2002) and six central coast counties (FishNet 4C et al. 2004). In addition, numerous workshops sponsored by state agencies, University of California Cooperative Extension and professional forestry organizations have been held in the recent past to educate stakeholders on how to address legacy road problems, how to design and construct road-stream crossings, and how to monitor water quality in forested watersheds. Points that have been stressed at workshops include: (1) requiring adequate long-term road maintenance and winter maintenance work for crossings, (2) improving construction/maintenance of road drainage structures—particularly those built near crossings, (3) disconnecting existing roads from the stream system by removing the inside ditch and out-sloping roads with rolling dips where possible, (4) surfacing roads located near streams, and particularly at stream crossings, (5) decreasing spacing between road drainage structures, (6) placing new roads on or near ridges, away from streams and reducing the number of crossings, (7) improving crossing decommissioning techniques through training with experienced individuals, and (8) improving the design of road-stream crossings for wood, sediment, and 100-year stream flow passage.

It is highly likely that these new initiatives will incrementally improve the conditions and performance of forest roads in California. Perhaps the greatest uncertainty is whether or not forest management will continue to be the dominant land use in large portions of the state with commercial timberlands.

SUMMARY AND CONCLUSIONS

Watershed research and monitoring work in California has shown that roads and road stream crossings are usually the dominant source of management-related erosion in forested environments. Past work has shown that a relatively small proportion of the total road length produces most of the road-related sediment delivered to streams. Detailed field surveys are the main tool available to identify the road segments of greatest concern (Korte and MacDonald 2007, MacDonald et al. 2007). Public and private landowners are actively inventorying their road networks, prioritizing road segments requiring road improvement or decommissioning work, and completing projects. A considerable amount of road upgrade work has been completed to date with both public and private financing. While there are short-term impacts associated with road decommissioning, particularly at road-stream crossings, road treatments will reduce the long-term sediment production from older roads (Madej 2001). Improved operator practices are required for road-stream crossing installation at high-risk sites and at decommissioned crossing sites. Guidebooks and training workshops have, and will continue to be, used to improve stakeholder knowledge regarding road and crossing practices.

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FIGURES AND TABLES

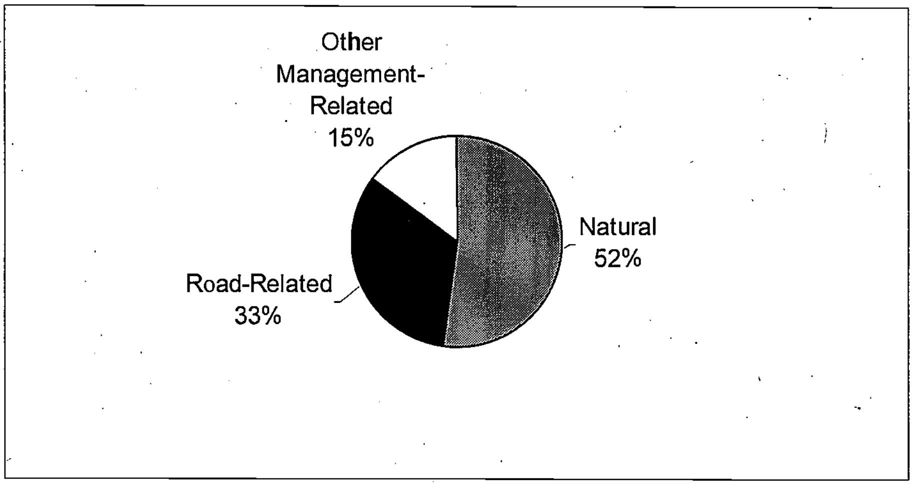

Figure 1. Distribution of natural and management-related sediment from 19 sediment budgets located in California.

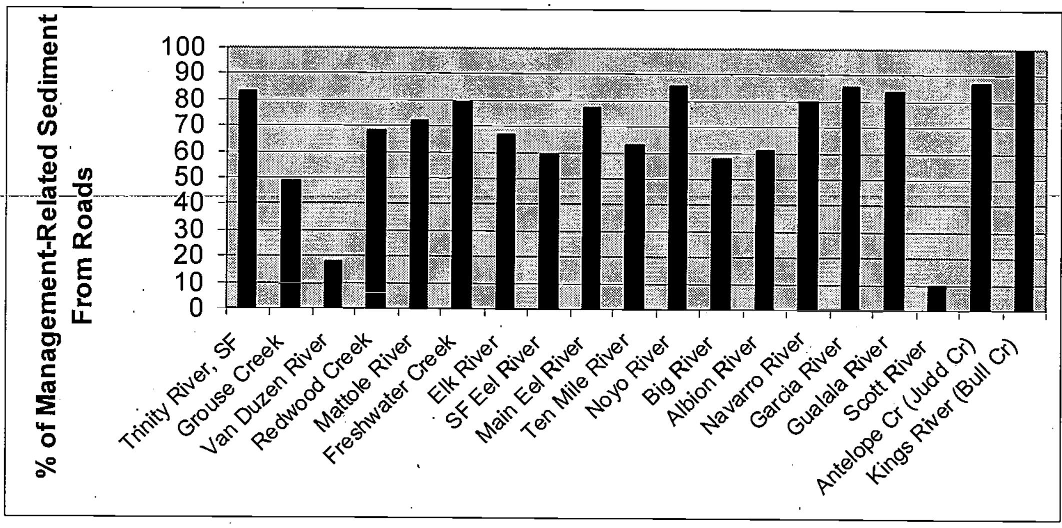

Figure 2. California sediment budgets from the northern Coast Ranges (16), Klamath Mountains (1), Cascade Range (1) and Sierra Nevada (1), showing the percent of management-related sediment estimated to derive from forest roads. The Redwood Creek sediment budget includes roads and skid trails; 68 percent of sediment for the Antelope Creek (Judd Cr.) sediment budget is estimated to result from long-term post-fire surface erosion; 14 of the 19 sediment budgets displayed were produced as part of TMDL documents for watersheds listed as impaired by the U.S. EPA.

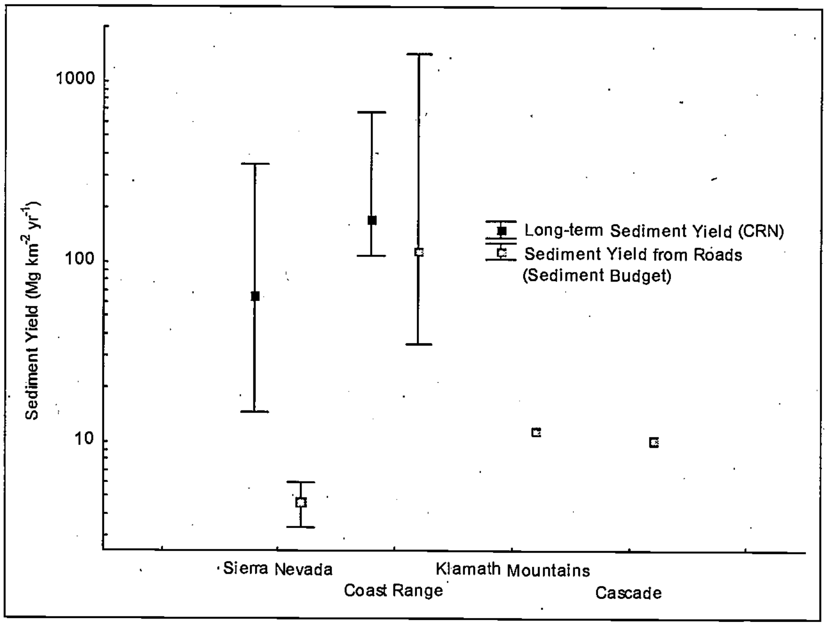

Figure 3. Comparison of millennial averaged sediment production rates derived from cosmogenic radio nuclide dating (CRN) to short-term sediment yields derived from sediment budgets produced for California watersheds. The boxes represent the median value and the error bars represent the range of values. CRN data is not available for the Klamath Mountains and the Cascade Range.

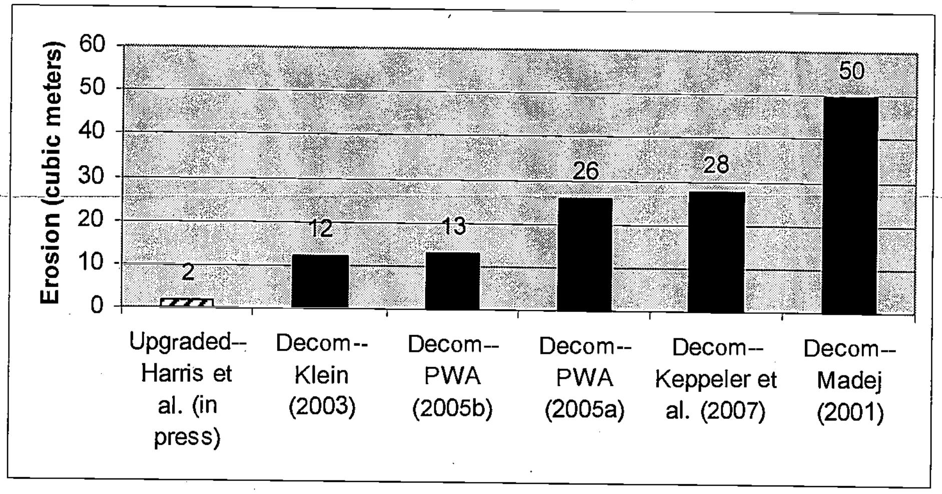

Figure 4. Comparison of post-treatment erosion volumes from decommissioned versus upgraded watercourse crossings.

Watershed	Geomorphic Province	Estimated Road-Related Erosion	Study Conducted	Reference
Kings River	Sierra Nevada	0.7 kg $m^{-2} yr^{-1}$	2004-2006	Korte and MacDonald 2007
American River	Sierra Nevada	0.9 kg $m^{-2} yr^{-1}$	1999	Coe 2006, MacDonald et al. 2004
Scott River, granitic tribs.	Klamath Mountains	33 kg $m^{-2} yr^{-1}$	1988-1989	Sommarstrom et al. 1990
Six Rivers National Forest	Coast Ranges	190 $m^3 km^{-1}$	1976	McCashion and Rice 1983
Redwood Creek--Cooper Cr	Coast Ranges	5200 $m^3 km^{-1}$	1954-1980	Weaver et al. 1995
Redwood Creek--Garret Cr	Coast Ranges	4730 $m^3 km^{-1}$	1956-1980	Best et al. 1995
Redwood Creek	Coast Ranges	180 $m^3 km^{-1}$	1997	Rice 1999
North Coast Region	Coast Ranges	250 m^3 km-1	1985-1986	Rice and Lewis 1991
Jordan Cr, Bear Cr, Elk River	Coast Ranges	720 $m^3 km^{-1}$	1998-1999	Weaver and Hagans 1999
South Fork Caspar Creek	Coast Ranges	200 $m^3 km^{-1}$	1967-1971	Krammes and Burns 1973
North Fork Caspar Creek	Coast Ranges	90 $m^3 km^{-1}$	1985-2000	Rice et al. 2004

Table 1. Road erosion rates from research studies conducted in California.

THESIS

SEDIMENT PRODUCTION AND DELIVERY FROM FOREST ROADS IN THE SIERRA NEVADA, CALIFORNIA

Submitted by

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Department of Forest, Rangeland, and Watershed Stewardship

In partial fulfillment of the requirements

For the Degree of Master of Science

Colorado State University

Fort Collins, Colorado

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COLORADO STATE UNIVERSITY

April 3, 2006

WE HEREBY RECOMMEND THAT THE THESIS PREPARED UNDER OUR SUPERVISION BY DREW BAYLEY ROGERS COE ENTITLED "SEDIMENT PRODUCTION AND DELIVERY FROM FOREST ROADS IN THE SIERRA NEVADA, CALIFORNIA" BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF MASTER OF SCIENCE.

Committee on Graduate Work

Adviser

Department Head

ABSTRACT OF THESIS

SEDIMENT PRODUCTION AND DELIVERY FROM FOREST ROADS IN THE SIERRA NEVADA, CALIFORNIA

Sediment production and sediment delivery from unpaved forest roads was assessed in the Sierra Nevada of California from 1999 to 2002. Sediment production was measured on 27-65 road segments over 3 years in a mixed rain-snow regime. Sediment delivery was evaluated by conducting a detailed survey of 20 km of unpaved roads with 285 distinct road segments.

Sediment production rates varied greatly between years and between road segments. Sediment production rates from native surface roads were 12-25 times greater than from rocked roads. On average, recently-graded roads produced twice as much sediment per unit of storm erosivity as roads that had not been recently-graded. Unit area erosion rates were 3-4 times higher in the first wet season than in either of the following two wet seasons, as the first wet season had near normal precipitation and a higher proportion of rainfall. An empirical model using the product of road segment area and slope (A*S), annual erosivity, and the product of road segment area and a binary variable for grading (A*G) explained 56% of the variability in sediment production. Road sediment production is best mitigated by rocking native surface roads, decreasing sediment transport capacity by improving and maintaining drainage, and avoiding sites where unusual soil characteristics increase road surface or ditch runoff.

Twenty-five percent of the surveyed road length was connected to the channel network. Stream crossings accounted for 59% of the connected road segments, and gullying accounted for another 35% of the connected road segments. The travel distance of sediment below road drainage outlets was controlled by the presence or absence of gullies, soil erodibility, traffic level, and road segment length. The amount of sediment delivered from episodic gully erosion below road segments (0.6 Mg km^{-1} yr^{-1}) is comparable to the amount of sediment being delivered from the road surface (1.4 Mg km^{-1} yr^{-1}).

An analysis of the data from this and other studies shows that road-stream connectivity is strongly controlled by mean annual precipitation and the presence or absence of engineered drainage structures (R^2=0.92; p<0.0001). Road sediment delivery can be minimized primarily by reducing the number of stream crossings, rocking the approaches to stream crossings, reducing the length of roads draining to stream crossings, and minimizing gully formation below drainage outlets.

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1.0. INTRODUCTION

Sediment is one of the most common causes of water quality impairment for streams and rivers in the U.S. (http://oaspub.epa.gov/waters/national_rept.control#TOP_IMP). Unpaved roads are the dominant source of surface erosion in many forested landscapes (Megahan and Kidd, 1972; Reid and Dunne, 1984; Bilby et al. 1989; Luce and Black, 1999). Road-derived sediment has been shown to increase turbidity and suspended sediment concentrations, alter channel substrate and morphology, and adversely affect water quality (Cederholm and Reid, 1981; Bilby et al., 1989; Waters, 1995). Data on road erosion and sediment delivery rates are critical for assessing road impacts on aquatic resources, and a sound understanding of road erosion processes is needed to minimize road sediment production and delivery.

Since 1999 researchers from Colorado State University have attempted to quantify hillslope erosion rates in the Sierra Nevada of California. Sediment fences (Robichaud and Brown, 2002) were used to measure sediment production rates from roads, timber harvest, wildfires, prescribed fires, and recreational off-highway vehicle use. The initial data showed median sediment production rates from roads were nearly an order of magnitude higher than any other source except a recent high-severity wildfire (MacDonald et al., 2004) (Figure 1.1). Given that unpaved forest roads are a ubiquitous feature in the Sierra Nevada landscape, the goal of this study was to quantify sediment production and sediment delivery from unpaved forest roads.

There is a paucity of data on road sediment production and delivery in the Sierra Nevada of California. Regional knowledge on the magnitude and controls of these processes is important for site-scale mitigation of road erosion and sediment delivery. Data on road erosion rates and sediment delivery are vital for assessing and predicting cumulative watershed effects.

In this thesis Chapter 2 examines sediment production from unpaved forest roads, and Chapter 3 examines the delivery of sediment from unpaved forest roads to the channel network. The overall objectives were to: (1) measure sediment production rates from unpaved roads over three wet seasons; (2) identify the dominant controls on road sediment production and develop predictive models; (3) document and quantify the hydrologic and sediment pathways that control the delivery of sediment from unpaved roads to the channel network; and (4) compare connectivity results from the Sierra Nevada with data from other studies.

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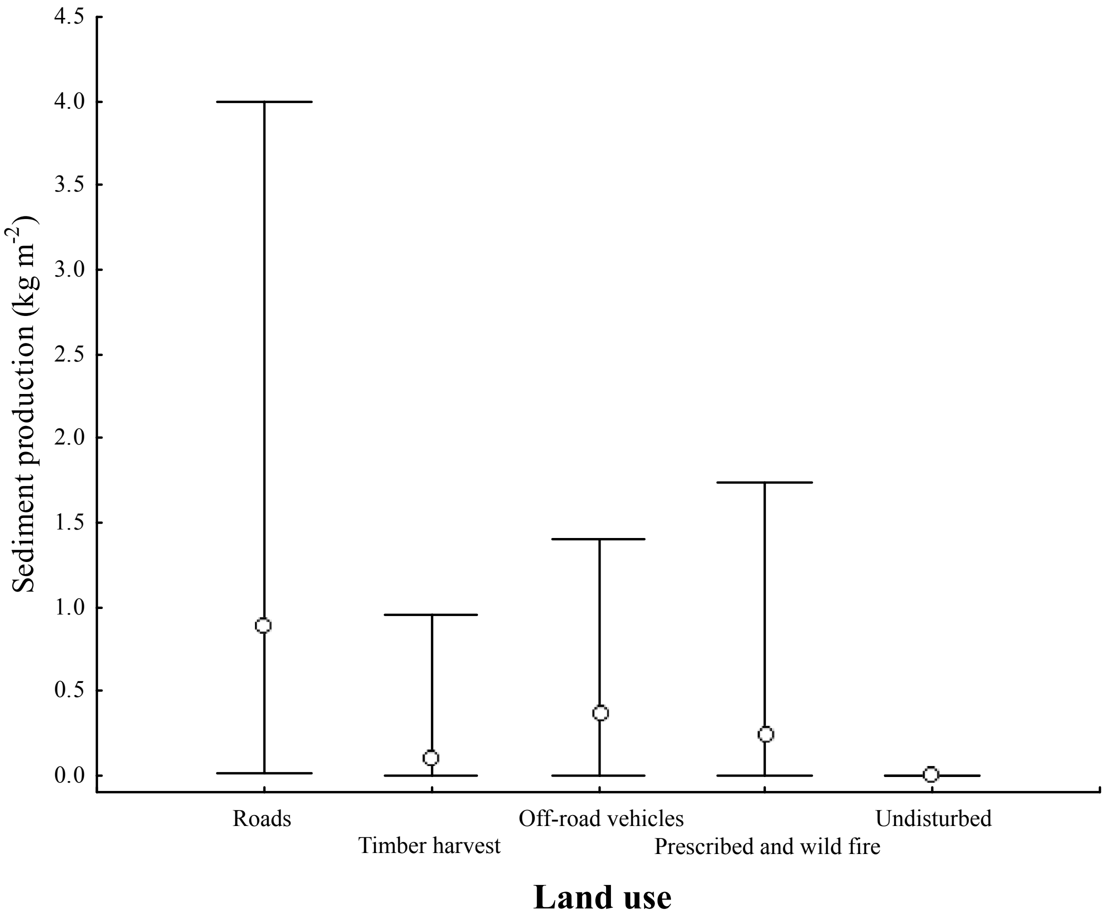

Figure 1.1. Mean and range of sediment production rates by type of land use. Circles represent the mean and bars indicate the range of measured values (from MacDonald et al., 2004).

2.0. SEDIMENT PRODUCTION FROM FOREST ROADS IN THE SIERRA NEVADA

ABSTRACT:

This study used sediment fences to measure sediment production from 27-65 road segments over three wet seasons in the Sierra Nevada of California. The first wet season had near-normal precipitation and annual storm erosivity (EI_A). The second and third wet seasons had below normal precipitation, and EI_A was less than 50% of the long-term mean as most of the precipitation fell as snow rather than rain. The mean sediment production rate from native surface roads was 0.81 kg m^{-2} in the first wet season versus 0.22 and 0.23 kg m^{-2} in the second and third wet seasons, respectively. The median sediment production rate from ungraded native surface roads was 15 times greater than rocked roads. Comparisons among segments showed that recently-graded native surface roads produced twice as much sediment per unit storm energy as ungraded native surface roads. Sediment production on native surface roads was best predicted by the product of road area times road slope (A*S), annual erosivity, and the product of road area and a binary variable for grading (A*G) (R^2=0.56). Normalized sediment production rates on mid-slope roads increased with decreasing soil depth. This increase is attributed to the greater interception of subsurface stormflow and resulting increase in road surface runoff. Road sediment production can be reduced by rocking native surface roads, increasing the frequency of road drainage structures, avoiding locations that generate more road surface and ditch runoff, and minimizing grading and traffic. The study illustrates the difficulties of predicting road erosion rates, particularly in a mixed rain-snow climate.

2.1. INTRODUCTION

Unpaved roads are the dominant source of surface erosion in many forested landscapes (Megahan and Kidd, 1972; Reid and Dunne, 1984; Bilby et al. 1989; Luce and Black, 1999). Road-derived sediment has been shown to increase turbidity and suspended sediment concentrations, alter channel substrate and morphology, and adversely affect water quality (Cederholm and Reid, 1981; Bilby et al., 1989; Waters, 1995). Data on road erosion and sediment delivery rates are critical for assessing road impacts on aquatic resources, and a sound understanding of road erosion processes is needed to minimize road sediment production.

Several studies have identified unpaved roads as a major sediment source in the Sierra Nevada of California, but none of these studies directly measured road erosion rates. Forest roads were estimated to contribute 74% of the sediment produced from a 194 km^2 catchment in central Sierra (Euphrat, 1992), and 19% of the sediment yield for a 6.8 km^2 catchment in the southern Sierra (Reid and Dunne, 1996). Both of these studies used the Universal Soil Loss Equation (USLE) to estimate sediment production rates. Unpaved roads have the highest disturbance coefficient in the methodology used to assess cumulative watershed effects on national forest lands in California (Cobourn, 1989), but there are no data on either the relative or the absolute contribution of unpaved roads to landscape-scale sediment production rates in the Sierra Nevada.

The extrapolation of road erosion rates to the Sierra Nevada from either the Pacific Northwest (Reid and Dunne, 1984; Bilby et al., 1989; Luce and Black, 1999; Luce and Black, 2001a) or the Idaho batholith (Megahan and Kidd, 1972; Megahan, 1974; Burroughs and King, 1989) is uncertain given the mixed rain-and-snow regime and the relative lack of winter traffic. The freezing level of winter storms usually fluctuates between 1000 m and 2500 m (Kattelmann, 1996), and this causes a corresponding fluctuation in the depth and extent of snow cover. As a result, the erosive energy available for sediment detachment and sediment transport changes according to whether the precipitation falls as rain or snow (Cooley et al., 1988).

Given the lack of data on road erosion rates in the Sierra Nevada and the concern over anthropogenic sediment inputs (Millar, 1996), there is an urgent need to quantify road sediment production rates and road erosion processes. A better knowledge of the magnitude and controls of road erosion processes is important for site-scale mitigation of road erosion. Furthermore, data on road erosion is vital for assessing and predicting cumulative watershed effects. With these considerations in mind, the objectives of this study were to: (1) measure sediment production from ungraded native surface roads, recently-graded roads, and rocked roads in mid-elevation areas in the central Sierra Nevada; (2) determine the temporal variability in road sediment production rates within and between winter wet seasons; (3) identify the dominant controls on road sediment production; and (4) develop empirical models for predicting road sediment production.

2.2. BACKGROUND

Sediment production from unpaved roads is a function of the erosive energy applied to the road surface and the erodibility of the road surface (Luce and Black, 1999; Ziegler et al., 2000a; Luce and Black, 2001a). Erosion from road surfaces can be partitioned into rainsplash and hydraulic components (Ziegler et al., 2000a):

$$e = e_s + e_h \tag{2.1}$$

where e is the net erosion rate from the road surface, e_s is rainsplash erosion, and e_h is the hydraulic erosion from overland flow. Rainsplash erosion results from the force of falling raindrops and is a function of storm intensity, raindrop size, storm depth, and soil erodibility (Wischmeier and Smith, 1978; Brown and Foster, 1987; Renard et al., 1997).

Hydraulic erosion is a function of the sediment transport capacity of overland flow and can be expressed by:

$$e_h = k\,(\tau - \tau_c)^{\,n} \tag{2.2}$$

where k is an index of the erodibility of the soil, τ is the shear stress applied by overland flow, τ_c is the soil's critical hydraulic shear strength, and n is an exponent between 1 and 2 (Kirkby, 1980; Nearing et al., 1994). Shear stress is defined as:

$$\tau = \rho_w\, g\, d\, s \tag{2.3}$$

where ρ_w is the density of water, g is the acceleration due to gravity, d is the depth of overland flow, and s is the water surface slope (Wohl, 2000). Since the mean flow depth (d) is a function of discharge (Knighton, 1998), hydraulic erosion is proportional to the amount of road surface runoff.

Road surface runoff is typically generated by Horton overland flow (HOF) plus the interception of subsurface flow (ISSF) by road cutslopes (Megahan, 1972; Luce and

Cundy, 1994; Ziegler and Giambelluca, 1997; Ziegler, 2001c; Wemple and Jones, 2003). Hence, total road surface runoff (Q_t) can be described as:

$$Q_t = Q_{HOF} + Q_{ISSF} \tag{2.4}$$

where Q_{HOF} is the runoff due to HOF generation and Q_{ISSF} is the runoff due to ISSF. HOF from a road surface is calculated by:

$$Q_{HOF} = (P - I)\ A \tag{2.5}$$

where P is precipitation intensity, I is the infiltration rate of the road surface, and A is the road surface area.

The volume of Q_{ISSF} is related to upslope soil properties, including the saturated hydraulic conductivity (K_s), depth to bedrock, hillslope gradient, topographic or bedrock contributing area, antecedent moisture conditions, and storm precipitation (Freer et al., 1997; Sidle et al., 1995; Freer et al., 2002; McGlynn et al., 2002; Weiler and McDonnell, 2004). ISSF occurs when the depth of the road cut (D_R) exceeds the depth to the water table (D) (Wigmosta and Perkins, 2001; Wemple and Jones, 2003). Assuming that the soil overlies a relatively impermeable layer, D will be smaller for shallow soils than for deeper soils, and roads crossing shallow soils will have a higher likelihood of intercepting subsurface flow. Conversely, the runoff from roads on deeper soils is more likely to be dominated by Q_{HOF} (Ziegler et al., 2001c).

The dependence of road sediment production rates on the erodibility of the road surface has been well documented (Megahan, 1974; Ziegler et al., 2000; Ziegler et al., 2001a,b; Luce and Black, 2001a,b). Traffic and road maintenance each increase the erodibility (K) of unpaved road surfaces by increasing the abundance of easily detachable sediment (Reid and Dunne, 1984; Ziegler et al., 2000; Luce and Black, 2001b; Ziegler et al., 2001a,b; MacDonald et al., 2001; Ramos-Scharron and MacDonald, 2005). As the more erodible surface material is removed, the road surface coarsens and becomes more resistant to rainsplash and the shear force exerted by overland flow (Ziegler et al., 2000; MacDonald et al., 2001).

Since the unpaved roads in the Sierra Nevada vary widely in terms of traffic, grading, and soil depth, comparisons between years and segments can help elucidate the importance of these different factors and provide insights into the underlying processes. This information can be used to help minimize sediment production from existing roads, guide future road designs, and set priorities for road rehabilitation or road obliteration.

2.3. METHODS

2.3.1. Site Description

The study area lies on the west slope of the Sierra Nevada mountain range in California, and is bounded to the north by the Rubicon River drainage and to the south by the South Fork of the Cosumnes River (Figure 2.1). Elevations range from 910 to 2000 m. The primary forest type is mixed conifer, but this turns to red fir with increasing elevation (SAF, 1980). The Mediterranean-type climate means that nearly all of the precipitation falls between 1 October and 1 June (USDA, 1985). Mean annual precipitation at the Pacific House rain gage at 1036 m is 1300 mm, but the standard deviation is 440 mm and the range over a 60-year period is from 450 mm to 2310 mm. The majority of the study area is from 1000 to 1800 m a.s.l., which is within the rain-on-snow climatic zone (Cobourn, 1989). Most of the study sites were on the Eldorado National Forest, although some sites were on interspersed Sierra Pacific Industries (SPI) property.

The dominant lithologies are weathered granitic batholith, granitic glacial deposits, andesitic lahar (Mehrten formation), and metasediments (USDA, 1985). The soils are typically coarse-textured loams, and contain up to 60% gravel by weight (USDA, 1985). Most of the soils are over a meter thick, but the range of soil depths is from 0.3 to 1.7 m. Soil erodibility (K) factors range from 0.013 to 0.042 t ha h ha^{-1} MJ^{-1} mm^{-1} (USDA, 1985).

2.3.2. Study Design

Sediment production was measured from road segments using sediment fences (Robichaud and Brown, 2002) over three wet seasons (1999-2000, 2000-2001, 2001-2002). Each study segment had a discrete drainage point (e.g., waterbar, rolling dip, or a relief culvert) so that all of the sediment produced from that segment could be captured by one or more sediment fences. Twenty-seven segments were monitored during the first wet season, 47 segments in the second wet season, and 65 segments in the third wet season (Table 2.1). The road segments were stratified into ungraded native surface roads, recently-graded native surface roads, and rocked roads. Ungraded native surface roads were defined as segments that had not been graded or used for timber hauling within the previous two years. Rocked roads were surfaced with approximately 10 cm of coarse gravel. One rocked road segment had its ditch graded prior to the first wet season, while the remaining rocked road segments (n=9) had no recent grading activity (Table 2.1).

Most of the study segments were designed to be outsloped, but repeated grading had formed a berm along the downslope edge of these segments. This berm held the surface runoff on the road segment until it reached a functioning waterbar or rolling dip. In areas with shallow soils and rock outcrops, the roads were generally insloped and had an inside ditch that was drained by a relief culvert. Most of the segments added in the second and third field seasons were on ridgetop roads in order to minimize cutslope erosion and the interception of subsurface stormflow. Traffic loads were not measured directly, but the recently-graded roads had more traffic because grading was generally a prerequisite to timber hauling.

2.3.3. Measurement Procedures

The sediment fences were constructed of geotextile fabric staked with reinforcing steel rods (rebar) 1.3 cm in diameter and 1.2-1.5 m long. Fences were constructed with Amoco 2130 fabric that had an opening size of 0.6 mm and a flow rate of 405 L min^{-1} m^{-2} (Robichaud and Brown, 2002). Multiple fences were constructed below selected road segments to increase storage capacity and sediment trapping efficiency. Fabric aprons were laid down in front of the sediment fences to facilitate the identification and removal of the deposited sediment.

The length and total width of the road segment draining to each fence was measured to the nearest decimeter. The measured width included the width of the road surface and ditch but did not include the width of the cutslope or fillslope. Road segment slope were measured with a clinometer and recorded as a decimal. The lithology and soil type was determined from the Eldorado National Forest Soil Survey (USDA, 1985) and field verified. The mean elevation of the study sites was 1424 m in 1999-2000, and as additional sites were added this gradually increased to 1510 m in 2001-2002. The elevation of individual sites ranged from 1015 m to 1829 m.

Sediment production was determined by excavating the sediment trapped by the sediment fences and weighing it to the nearest 0.1 kg. After weighing, the sediment was mixed and two samples were taken to determine soil moisture content (Gardner, 1986). The mean moisture content was used to convert the field-measured wet weights to a dry mass, and annual sediment production rates were calculated by dividing the mass of sediment by the contributing surface area of the road segment. Many sites were not accessible during the winter, so the primary data set consists of annual sediment production rates.

Hydrologic data were obtained at three locations (Figure 2.1). Precipitation was measured at Pacific House (PH) at 1036 m with a tipping bucket rain gage that had a resolution of 1.0 mm (http://cdec.water.ca.gov/cgiprogs/staMeta?station_id=PFH). The Pacific House gage is believed to be representative of the entire study area because wet season precipitation is derived from large frontal storms. Snowpack data were taken from the Robbs Powerhouse SNOTEL site (RP) at 1570 m (http://cdec.water.ca.gov/cgi-progs/staMeta?station_id=RBP) (Figure 2.1). Mean daily discharge data were taken from the Michigan Bar gaging station on the Cosumnes River (MB) (http://cdec.water.ca.gov/cgi-progs/staMeta?station_id=MHB), as this drains the southern half of the study area. Although this station is only at 51 m a.s.l., the Cosumnes is the only undammed river in or near the study area and the discharge data at Michigan Bar closely reflect both the magnitude and type of precipitation in the study area.

For each wet season the maximum storm erosivity and annual erosivity were calculated from the rainfall data at Pacific House. Individual storms were defined as precipitation events separated from each other by at least 6 hours (Mutchler et al., 1994). The erosivity (EI_{30}) for each storm was calculated by multiplying the total storm energy (E) by the maximum 30-minute rainfall intensity (I_{30}), (Renard et al., 1997). The total energy (E) for each storm was calculated by multiplying the rainfall energy (e_r) by total storm depth (P). The rainfall energy (e_r) for each storm was calculated by the equation developed for the western U.S. (Brown and Foster, 1987):

$$e_r = 0.29\ [1-0.72^{(-0.05i)}] \tag{2.6}$$

where i is average rainfall intensity of the storm in mm h^{-1}. The annual erosivity (EI_A) was calculated by summing the EI_{30} values for each wet season.

2.3.4. Statistical Analysis

The primary dependent variable was annual sediment production in kg yr^{-1}. To better assess the effect of the various independent variables, this was normalized by contributing road surface area, road slope, rainfall erosivity, or a combination of these variables (Table 2.2). The significance of each of the independent categorical variables (Table 2.2) was evaluated by post-hoc pairwise comparisons using Tukey's Honestly

Significantly Difference (HSD) (Ott, 1993; STATISTICA, 2003). Sediment production rates were log-transformed for pairwise comparisons when sediment production rates were log-normally distributed. The large sample size for native surface roads (n=109) meant that the sediment production for these segments could be related to each of the continuous independent variables in Table 2 by multiple regression using forward stepwise regression with a selection criteria of α=0.05. The presence or absence of grading was treated as a binary variable. Sources of model errors were explored through residual analyses.

2.4. RESULTS

2.4.1. Road Segment Characteristics

Sediment production was measured from native surface and rocked road segments with a wide range of road surface areas and road gradients. For the native surface road segments, road surface areas ranged from 30 to 2170 m^2 (i.e., 8 to 395 m in length) with a mean of 368 m^2. For rocked road segments the mean road surface area was 29% smaller at 261 m^2, and the range was from 107 to 1022 m^2. The mean road surface area for the recently-graded native surface road segments was 228 m^2 as compared to 561 m^2 for the ungraded native surface road segments. The three segments with the largest road surface area had drainage structures that were no longer functioning and therefore somewhat atypical. The gradients for native surface road segments ranged from 0.02 to 0.21 m m^{-1} with a mean of 0.09 m m^{-1}. Gradients for the rocked road segments were similar (0.05 to 0.20 m m^{-1} with a mean of 0.09 m m^{-1}).

The road segments used to measure road sediment production were typically outsloped and drained by waterbars and rolling dips. Only four of the native surface road segments and one of the rocked road segments (i.e., 15 data points over three wet season) were insloped and drained by inside ditches. Each of these five insloped road segments drained hillslopes with shallow soils less than 0.5 m in depth. The roads were generally under 30-40 years in age, and most had been reconstructed using current best management practices (BMPs) in recent years (D. Arrington, pers. comm., 2000).

2.4.2. Precipitation and Runoff

Annual precipitation in the first wet season was 1290 mm, which is very close to the long-term mean of 1300 mm. In the second and third wet seasons precipitation was only 68% and 82% of the long-term mean, respectively (Figure 2.2). In the first wet season approximately 50% of the annual precipitation fell between 11 January and 14 February, while precipitation in the second and third wet seasons was much more evenly distributed (Figure 2.2).

The total erosivity (EI_A) in the first wet season was 847 MJ mm ha^{-1} hr^{-1}. The EI_A values in the second and third wet seasons were respectively only 441 and 456 MJ mm ha^{-1} hr^{-1}, or less than 60% of the value from the first wet season. In the first wet season the maximum storm erosivity in the first season was 252 MJ mm ha^{-1} hr^{-1} from a 175-mm storm in late January. Since this storm increased the snow water equivalent (SWE) at Robbs Powerhouse by only 4 mm (Figure 2.3), precipitation below this elevation was mostly rain. In the second and third wet seasons the maximum storm erosivity was only 98 and 83 MJ mm ha^{-1} hr^{-1}, respectively.

The SWE data show that the snow cover was thinner and less frequent in the first wet season relative to the second and third wet seasons (Figure 2.3). In 1999-2000 the snowpack at Robbs Powerhouse didn't begin to accumulate until 7 December and meltout occurred by 31 March, resulting in 115 days with snow cover (Table 2.3). SWE was below 70 mm until mid-February, suggesting a lack of snow cover at the lower elevation sites. The peak SWE was 302 mm in the second week of March, which is less than half of the 30-year mean peak SWE of 656 mm.

In the second wet season the first storms were unusually cold and the snowpack began accumulating on 26 October (Figure 2.3). Most of the subsequent precipitation fell as snow, and the SWE steadily increased from mid-December until the peak SWE of 406 mm was reached in early March. Meltout occured on 24 April, indicating 167 days of snow cover (Figure 2.3).

Although some data are missing from the third wet season, by early December there were 150 mm of SWE, indicating that much of the early season precipitation had fallen as snow rather than rain (Table 2.3; Figure 2.3). As in 2000-2001, the snowpack persisted until late April. The greater duration of snow cover in the second and third wet seasons is confirmed by our field observations, as the road segments above 1400 m were generally accessible until mid-February in the first wet season, and largely inaccessible from early January to until late March in both the second and third wet seasons.

The daily discharge data confirm the preponderance of rain and much greater erosivities in the first wet season, as four storms each generated mean daily flows in the Cosumnes River of more than 150 $m^3 s^{-1}$ (Figure 2.4). The largest mean daily flow during the study period was 289 $m^3 s^{-1}$ on 14 February 2000, and this has an estimated recurrence interval of 2.4 years. This peak flow was due to 114 mm of precipitation in 48 hours as measured at the PH rain gage. Since this storm increased the SWE at RP by only 66 mm, almost half of the precipitation below 1570 m fell as rain. Many of the field sites that had been snow covered became accessible during and after this storm, indicating that the high flows were due to a combination of rain and snowmelt.

In the second wet season there were no obvious rain-on-snow events in the annual hydrograph, and the largest daily flow was just 28 $m^3 s^{-1}$ in late March (Figure 2.4). In the third wet season there were four small rain-on-snow events, but the largest daily flow was only 70 $m^3 s^{-1}$, or 24% of the maximum daily flow recorded during the first wet season (Figure 2.4).

2.4.3. Sediment Production Rates by Road Surface Type and Wet Season

The distribution of sediment production rates was highly skewed by a few segments with exceptionally high values (Figure 2.5). For native surface roads the mean annual sediment production rate was 0.32 kg m^{-2} yr^{-1} (Table 2.4), while the median value was only 0.14 kg m^{-2} yr^{-1}. Rates were highly variable as the range for native surface road segments was from 0.0002 kg m^{-2} yr^{-1} to 4.0 kg m^{-2} yr^{-1} (Figure 2.5).

The distribution of sediment production rates for rocked roads was even more skewed, as the overall mean of 0.12 kg m^{-2} yr^{-1} was 13 times the median value of 0.009 kg m^{-2} yr^{-1} (Table 2.4). The larger skew was due primarily to one segment that yielded 3.3 kg m^{-2} yr^{-1} in the first wet season. This is nearly 170 times the mean value of 0.02 kg m^{-2} yr^{-1} for the other 29 segment-years of data. The high sediment production rate from this segment was attributed to the fact that the inboard ditch had been graded during the previous summer, and the upslope area had very thin soils and scattered rock outcrops, resulting in visibly high rates of Q_{ISSF}.

The 2.5-fold difference in the overall mean sediment production rates between the native surface and the rocked roads was significant at $p<0.0001$. Given the large amount of skew in the data, the 15-fold difference in median sediment production rates is a more accurate indication of the effect of rocking on road sediment production.

Sediment production rates varied greatly between wet seasons (Figure 2.5). In the first wet season the mean sediment production rate from native surface roads was 0.81 kg m^{-2}, and this was approximately four times the mean values in the second and third wet seasons. The mean sediment production rate for rocked roads in the first wet season was 0.36 kg m^{-2} (Table 2.4). If the one segment with a recently-graded inside ditch is excluded, the mean sediment production rate for the rocked roads was only 0.03 kg m^{-2} in the first wet season. In the second and third wet seasons the mean sediment production rates for rocked roads was only 0.01 and 0.02 kg m^{-2}, respectively.

2.4.4. Other Controls on Road Sediment Production

For native surface roads the annual rainfall erosivity (EI_A) explained 15% of the variability in sediment production rates between years ($p<0.0001$). Maximum storm erosivity (EI_M) and total precipitation explained 14% and 10% of the variability, respectively. EI_A was not significantly related to sediment production rates for the entire data set of rocked roads, but if the extreme outlier in Figure 2.5 is excluded, EI_A explains 20% of the variability in sediment production rates between years ($p=0.02$). Similarly, total precipitation and EI_M each explained about 20% of the variability for rocked roads once the extreme data point in Figure 2.5 was excluded from the data set.

Several segment-scale variables were important controls on sediment production rates for both native surface and rocked roads. For native surface roads, road surface area explained 33% of the variability in sediment production per unit erosivity ($p<0.0001$) (Figure 2.6a). When treated as a continuous variable, road slope was significantly but weakly related to the normalized sediment production rate (kg m^{-2} EI_A^{-1}) for native surface roads ($R^2=0.04$; $p=0.04$). However, the mean sediment production rate for native surface road segments with slopes ≥7% was approximately 75% higher than segments with slopes less than 7% ($p=0.005$; Figure 2.7).

For the native surface road segments, the product of road surface area and road slope (A*S) explained 44% of the variability in sediment production per unit erosivity. Road surface area times slope (A*S) was more strongly correlated with normalized sediment production rates (kg yr^{-1} EI_A^{-1}) for the steeper roads segments ($R^2=0.56$; $p<0.0001$). Sediment production rates were not significantly related to A*S for the native surface road segments with slopes <7% ($p=0.60$).

For the rocked road segments, road surface area explained 32% of the variability in sediment production rates per unit erosivity. Removing the outlier in Figure 2.5 increased the R^2 for this relationship to 0.87 (Figure 2.6b). Road slope was not significantly related to normalized sediment production (kg m^{-2} EI_A^{-1}) ($p=0.73$). In contrast to the native surface roads, road surface area was more strongly related to the normalized sediment production rates than A*S ($R^2=0.48$; $p=0.01$).

The native surface road segments that had been recently graded produced about twice as much sediment per unit erosivity as the ungraded segments ($p=0.02$) (Figure 2.8). A pairwise comparison indicated that there was no evidence of a decline in sediment production rates between the first and second years after grading ($p=0.86$). Hence the term recently-graded refers to any segment that had been graded within the past two wet seasons.

A more detailed analysis shows that grading has a strong effect on sediment production rates at lower elevations, but not at higher elevations (Figure 2.9). For the native surface roads below 1400 m, the recently-graded segments produced approximately eight times more sediment than the ungraded segments when sediment production rates were normalized by A*S and EI_A ($p=0.0008$). In contrast, grading had no apparent effect on normalized sediment production rates for the native surface roads above 1400 m ($p=0.92$) (Figure 2.9). The recently-graded native surface roads below 1400 m also produced nearly 5 times more sediment than the recently-graded native surface roads above 1400 m, and this difference was highly significant ($p=0.0005$) (Figure 2.9). For the ungraded roads, there was no significant difference in normalized sediment production rates with elevation class ($p=0.14$).

Stepwise multiple regression shows that sediment production from native surface road segments is controlled by the product of road surface area and slope (A*S), annual storm erosivity (EI_A), and the product of road surface area and a binary variable for grading (A*G) that has a value of 1 if the segment has been recently graded and 0 if the segment has not been graded. The resultant model is:

$$SP_{ns} = -329 + 3.56\ (A*S) + 0.542\ EI_A + 0.389\ (A*G) \qquad (2.7)$$

where SP_{ns} is sediment production for native surface roads in kilograms per year (Table 2.5). The overall model R^2 is 0.56, the adjusted R^2 is 0.54, and the standard error is 142 kg.

2.5. DISCUSSION

2.5.1. Comparisons to Previous Studies

The mean annual sediment production rate for the native surface road segments ranged from 0.23 to 0.81 kg m^{-2} yr^{-1}, with a 3-year average of 0.32 kg m^{-2} yr^{-1} (Table 2.4). Assuming an average road width of 5.0 m, this converts to 1.6 Mg km^{-1} yr^{-1}. Road erosion rates for unpaved roads with moderate traffic in the Olympic Peninsula in the state of Washington were 41 Mg km^{-1} yr^{-1} (Reid and Dunne, 1984), or approximately 26 times higher than the 3-year mean reported here. The overall mean from the present study is 67% of the reported mean erosion rate of 0.48 kg m^{-2} for unpaved roads in the Idaho batholith (Megahan, 1974). The similarity in road erosion rates for the Sierra Nevada and the Idaho batholith might be attributed to the similarities in lithology and climate.

The mean sediment production rate from rocked roads ranged from 0.01 to 0.36 kg m^{-2} yr^{-1}, but the upper end of this range was due to one road segment that had a recently-graded ditch and exceptionally high runoff rates. If this segment is excluded, the mean sediment production rate from rocked roads was 0.02 kg m^{-2} yr^{-1}, and the maximum value for a single segment was 0.09 kg m^{-2} yr^{-1}. These values fall within the range of

0.01-0.21 kg m^{-2} yr^{-1} for rocked roads in the Idaho batholith (Burroughs and King, 1989), but the mean is much lower than the rate reported from the Olympic Peninsula (Reid and Dunne, 1984). Since there was no wet season traffic and five of the rocked road segments were behind locked gates, the lower sediment production rates for rocked roads in the Sierra may be attributed to the lack of wet season traffic and lower precipitation relative to the Olympic Peninsula. This rationale is consistent with data from the Oregon Coast Range, where rocked roads with no traffic and no recent grading produced less than 0.02 kg m^{-2} yr^{-1} (Luce and Black, 2001b).

2.5.2. Climatic Controls on Rainsplash and Hydraulic Erosion

The lower sediment production rates from the native surface roads in the second and third wet seasons is due to the difference in precipitation as well as the difference in the type of precipitation. The first wet season had larger and more intense rain events as well as more precipitation, and the annual rainfall erosivity in the first wet season was nearly double the value in the second and third wet seasons. Perhaps more importantly, the second and third wet seasons were colder so more of the precipitation fell as snow and there was constant snow cover on most of the sites. Snowfall has minimal erosive energy when it hits the soil surface (Cooley et al., 1988), and snow cover protects the road surface from rainsplash erosion during rain-on-snow events.

Previous research suggests that rainsplash erosion accounts for approximately 50% of the total erosion from unpaved roads (Ulman and Lopes, 1995; Ziegler et al., 2000), and that erosion rates are linearly related to rainfall erosivity (Renard et al., 1997). Since the EI_A in the second and third wet seasons was roughly 50% of the value from the first wet season, if road surface erosion is proportional to rainfall erosivity the sediment production rates in the second and third wet seasons should have been about half of the value from the first wet season. However, the sediment production rates from native surface roads in the second and third wet seasons were roughly one-quarter of the value from the first wet season, or about half of the expected value. This suggests that the more continuous snow cover during the second and third wet seasons may have reduced the amount of rainsplash erosion (e_s) and/or hydraulic erosion (e_h) by an additional 50 percent.

The reduction in e_s due to a shift from rain to snow is self evident, but the effect of this shift on e_h is more complex. Maximum snowmelt rates in the alpine Sierra are on the order of 30 mm d^{-1} (Kattelmann and Elder, 1991), while rainfall inputs can exceed 100 mm d^{-1}. The lower intensity of snowmelt inputs will reduce both the depth and velocity of overland flow and hence e_h. The presence of a snowpack on the road surface should also reduce the velocity of overland flow, but there are no data on this effect. The prediction of road erosion rates is further complicated by the observation that rills up to 10 cm wide can develop under the snowpack.

The amount of runoff on the road surface also will vary with the amount of Q_{ISSF} (Ziegler et al., 2001c; Wemple and Jones, 2003). For the 17 midslope road segments with data from all three seasons, the normalized sediment production rates (kg $A*S^{-1}$ EI_A^{-1}) decreased with increasing upslope soil depth (R^2=0.17; p=0.002). The relationship between upslope soil depth and normalized sediment production was stronger and slightly more non-linear for the rain-dominated first wet season (R^2=0.32) than the snow-dominated second and third wet seasons (R^2=0.15) (Figure 2.10).

The amount of subsurface stormflow (SSF) varies with upslope soil depth and antecedent soil moisture conditions (Sidle et al., 1995; Freer et al., 1997; Freer et al., 2002; Tromp-van Meerveld and McDonnell, 2006b). SSF is threshold driven, in that it requires subsurface saturation along flowpaths before it can occur (Tromp-van Meerveld and McDonnell, 2006a, 2006b). Subsurface saturation occurs first in shallow soils, and shallow soils can generate SSF during small to medium-size storms (Tromp-van Meerveld and McDonnell, 2006b). In the present study, the first wet season had more precipitation, higher rainfall intensities, and generally wetter soil conditions. I hypothesize that: (1) subsurface saturation occurred on hillslopes more often during the first wet season; and (2) the hillslopes with the shallowest soils produced the most SSF. The larger amount of intercepted SSF in the first wet season resulted in more hydraulic erosion and a stronger relationship between upslope soil depth and sediment production (Figure 2.10). The second and third wet seasons were drier and antecedent soil moisture conditions were presumably lower, resulting in less Q_{ISSF} and a weaker relationship between soil depth and normalized sediment production (Figure 2.10b).

2.5.3. Controls on Road Surface Erodibility and Sediment Supply

Rocking the road surface reduced median sediment production rates by at least an order of magnitude, and this can be attributed to the resulting decreases in e_s, e_h, and the supply of erodible sediment. The 5-20 mm gravel protects against e_s (Burroughs and King, 1989) and greatly increase τ_c (Eq. 2.2). Rocking also increases flow roughness, thereby reducing flow velocities and the erosion due to e_h. Rocking may not be effective if the inside ditch is not rocked, as the highest sediment yield for a single road segment

(3.4 Mg) came from a rocked road segment at 1450 m elevation in the first wet season. This 241 m long, midslope segment intercepted SSF from a hillslope with shallow soils on top of relatively impermeable andesitic lahar deposits (USDA, 1985), and it had a recently-graded inside ditch. Large amounts of Q_{ISSF} were observed from the cutslope during moderate and large rainstorms, and field observations indicated that the amount of Q_{ISSF} changed quickly in response to changes in rainfall intensity. The resultant high flows in the ditch were able to transport cobble-sized clasts (>128 mm). Sediment yields from this segment in the second and third wet seasons were only 1-2% of the value from the first wet season, and this indicates that grading generated a large supply of erodible sediment. These results show that rocking can be a very effective means for reducing road erosion, but in some cases road design, maintenance activities, and local site conditions can negate the usual benefits of rocking the road surface.

The lower sediment production rates from ungraded native surface roads relative to recently-graded roads has been attributed to a more limited supply of easily erodible fine sediment (Ziegler et al., 2000; Ramos-Scharron and MacDonald, 2005). The A*G term in the model (Eq. 2.7) indicates that increase in road sediment production due to grading is proportional to the road surface area, and that a recently-graded road segment produces an additional 0.39 kg per square meter of road surface area than an ungraded road segment.

For some of the more easily-accessible segments, sediment production was measured several times within a wet season. The data from four recently-graded road segments show that sediment production rates per unit precipitation were much higher in the early portion of the wet season (Figure 2.11). The high initial sediment pulse can be attributed to the rapid removal of the thick, fine dust layer that had formed on the road surface as a result of grading and timber hauling activities. The subsequent decline in sediment production per unit rainfall suggests that the recently-graded roads rapidly become supply limited as the road surface becomes armored and more resistant to sediment detachment and transport processes. On the other hand, there was no apparent decline in sediment production rates per unit erosivity between the first and second years after grading. The lack of a decline may be due to continuing high traffic loads on many of recently-graded roads, as the combination of grading and harvesting increased the amount of traffic from firewood cutters and recreationists, and the high traffic levels increase the amount of readily-erodible sediment (Ziegler et al., 2001a.). Wheel ruts also began to appear on many of these roads, and the concentrated flow in these ruts also can increase sediment production rates (Foltz and Burroughs, 1990).

Figure 2.9 shows that grading had no effect on sediment production on road segments above 1400 m in elevation. The lack of a grading effect above 1400 m can be attributed to the fact that most of the precipitation falls as snow and there is more continuous snow cover. This shields the erodible dust layer from e_s and e_h, and this apparently minimizes the effects of grading on sediment production.

The effects of lithology and soil erodibility on road sediment production were difficult to discern given the interacting and confounding effects of the other controlling factors. The mean normalized sediment production from road segments on metasediments was four times greater than segments on other lithologies (p=0.0001). However, there were only four data points for road segments on metasediments, and each of these road segments had been recently graded. Soil erodibility was positively correlated with normalized sediment production (kg A*S^{-1} EI_A^{-1}) for recently graded native surface roads (R^2=0.19; p=0.0004) (Figure 2.12), but not for ungraded native surface roads or rocked roads. These results suggest that erodibility indices such as lithology and soil erodibility tend to have a secondary influence compared to other variables such as A*S, rainfall erosivity, and grading. Lithology and soil erodibility were only significant when the road surface has been recently disturbed by grading and sediment production rates are relatively high. Lithology and soil erodibility are less likely to be good predictors of sediment production once the road surface is armored.

2.5.4. Model Performance and Implications for Long-term Road Erosion Rates

The empirical model presented in equation 2.7 accounts for 56% of the variability in sediment production rates from native surface roads (Figure 2.13). The model is much better at predicting sediment production rates for road segments with a slope ≥7% (R^2=0.62; p<0.0001) than for segments with slopes <7% (R^2=0.21; p=0.01). The greater predictability for the steeper segments can be partly attributed to the significant relationship between A*S and normalized sediment production (kg EI_A^{-1}) for the steeper segments (R^2=0.56; p<0.0001). In contrast, the normalized sediment production rates for road segments with slopes of less than 7% are not significantly related to A*S (R^2=0.01; p=0.60). The significant relationship for the steeper roads does not appear to be due to the greater spread in A*S data, as some of the flatter road segments also have relatively large A*S values. Other studies have suggested that an increase in road length does not necessarily lead to higher sediment production rates for flatter segments (Luce and Black, 1999; Ramos-Scharron and MacDonald, 2005).

The inclusion of A*S in equation 2.7 indicates that sediment production is a linear function of road surface area and slope. However, the normalized sediment production rates (kg m^{-2} EI_A^{-1}) for ungraded road segments are most strongly related to segment slope raised to the 1.9 power (R^2=0.23; p=0.0007). An exponent of 1.9 is close to the values of 1.5-2.0 reported in other studies (Luce and Black, 1999; Ramos-Scharron and MacDonald, 2005). However, sediment production for the entire dataset is best predicted by a linear function of A*S rather than a non-linear function of A*S.

The empirical model in equation 2.7 doesn't include all of the factors that appear to affect road erosion rates. For example, upslope soil depth was not significant in the overall model, and this may be partly due to the fact that 84% of the data came from ridgetop roads where sediment transport capacity is controlled by Q_{HOF} rather than Q_{ISSF}.

The empirical model also doesn't include a factor for elevation, even though road erosion rates significantly decline with increasing elevation for the recently-graded road segments. This decline is due to the shift from rain to snow and the corresponding increase in the frequency of snow cover. The overall model R^2 increased from 0.41 to 0.54 when EI_A was included, as this accounted for much of the difference in sediment production rates between years. However, EI_A was only measured in one location so it could not account for the spatial variability in rainfall erosivity and snow cover. Since the model doesn't include an elevation term it will tend to underpredict sediment production rates from the road segments at lower elevations. Including site-specific EI_A data could potentially improve the performance of the model.

The empirical model in equation 2.7 provides a useful first estimate of road erosion rates for native surface roads in the northern Sierra, but the measured and predicted road erosion rates are probably low relative to the long-term average. Road erosion studies in other areas have shown that the largest storm events generate most of the erosion (Luce and Black, 2001a; Ramos-Scharron and MacDonald, 2005). In the study area the long-term mean EI_A is between 1020 and 1360 MJ mm ha^{-1} hr^{-1} (Renard et al., 1997), or approximately 20-60% more than the EI_A in the first wet season and 220-310% more than the EI_A in the second and third wet seasons. According to equation 2.7, an ungraded native surface road segment with an average road surface area of 368 m^2 and an average slope of 0.09 m m^{-1} would generate 526 kg of sediment in a year with an EI_A of 1360 MJ mm ha^{-1} hr^{-1}, but only 248 kg in the first wet season when the EI_A was 847 MJ mm ha^{-1} hr^{-1}.

The potential underprediction of road erosion rates may be even greater for the midslope roads, as the record peak flow at Michigan Bar in January 1997 was more than eight times the largest instantaneous peak flow recorded during the study period. The magnitude of SSF can increase by a factor of 75 once hillslope hydrologic connectivity is achieved (Tromp-van Meerveld and McDonnell, 2006b). Given that normalized road erosion showed a non-linear relationship with upslope soil depth in the first wet season, this non-linear relationship is likely to be even more pronounced during wetter years. As a result, one would expect a large increase in erosion due to Q_{ISSF} during wetter years, particularly on the road segments that have a cutbank draining shallow soils.

2.5.5. Implications for Management

This study shows that sediment production rates are at least an order of magnitude lower from rocked roads than native surface roads. Rocking decreases rainsplash erosion

(Eq. 2.1), increases the critical shear stress necessary for erosion (Eq. 2.2), and reduces the supply of easily erodible sediment.

The empirical model (Eq. 2.7) indicates that the product of road surface area and road gradient is an important control on road erosion. However, the model also suggests that sediment production is a linear function of A*S, and that frequent road drainage does not necessarily reduce unit area road erosion. Logic still suggests that sediment production rates can be decreased by reducing road contributing area, as this is consistent with erosion theory and other research (Luce and Black, 1999; Luce and Black, 2001a; Ramos-Scharron and MacDonald, 2005). Frequent road drainage also can reduce the likelihood of sediment delivery to the channel network (Wemple et al., 1996; Croke and Mockler, 2001).

Road surface area can be decreased by increasing the frequency of drainage structures such as waterbars or cross-relief culverts, or by outsloping the road surface. In the study area the periodic grading of outsloped roads often has created berms along the downslope edge of the road segment. By keeping the overland flow on the road surface, these berms effectively increase A*S and hence the sediment production rate. Both road drainage structures and outsloping must be maintained if one wishes to minimize surface runoff and reduce road sediment production.

Rocking and drainage are particularly critical for road segments on hillslopes with shallow soils and rock outcrops, as these site characteristics tend to increase the proportion of rainfall and snowmelt that becomes surface runoff. The resulting increase in runoff will increase erosion from cutslopes, inside ditches if present, and the road surface. Soil depth data are generally available from soil surveys, and these data can help land managers identify the soil types and sites that are most susceptible to Q_{ISSF} and high road surface erosion rates.

The recently-graded roads produced more sediment than ungraded roads. A reduction in the frequency of grading will decrease the supply of easily erodible sediment, and this is particularly important for the lower-elevation roads where the easily erodible surface layer is subjected to more rainfall and higher surface runoff rates. The effects of grading did not appear to diminish over a two year period, but recovery may have been masked by the confounding effect of increased traffic after grading.

2.5.6. Future Research

This study showed that road sediment production rates are a complex response to climate, site, and management factors. A more rigorous and quantitative assessment of these factors will require more controlled, process-based studies. Runoff and erosion rates from the road surface need to be measured on segments with varying upslope soil depths under different antecedent conditions for rain, snowmelt, and rain-on-snow events, respectively. Hillslope piezometers above the road segments would help corroborate the discharge data and determine the relative importance of subsurface stormflow as a function of slope position, upslope drainage area, cutslope height, and soil depth. Storm-by-storm measurements of runoff and sediment production would help indicate the relative importance of Q_{HOF} and Q_{ISSF} on road surface runoff and sediment production rates.

The range and complexity of the interactions between local site conditions (e.g., soil depth, erodibility), road segment properties (e.g., A*S, road maintenance), and climate (e.g., rain vs. snow) have important implications for the use and reliability of spatially-distributed, physically-based models such as WEPP (Water Erosion Prediction Project) (Elliot et al., 1995) and DHSVM (Distributed Hydrologic Soil Vegetation Model) (Bowling and Lettenmaier, 2001; Wigmosta et al., 1994). The accuracy of the model outputs depends upon the representation of the underlying processes. Additional research is needed to help refine the numerical representation of HOF, ISSF, sediment detachment, and sediment transport processes and to help verify these models across a range of climatic and environmental conditions.

2.6. CONCLUSIONS

Sediment production was measured from 139 road segments over 3 years in a mixed rain-snow regime in the Sierra Nevada of California. Sediment production rates varied greatly between years and between road segments. The mean sediment production rate from native surface roads was 0.81 kg m^{-2} in the first wet season as compared to 0.22 and 0.23 kg m^{-2} in the second and third wet seasons, respectively. Sediment production rates from native surface roads were 12-25 times greater than from rocked roads. On average, recently-graded roads produced twice as much sediment per unit of storm erosivity than ungraded native surface roads. An empirical model using the product of road area and road slope, annual erosivity, and the product of road area and a binary variable for grading explained 56% of the variability in sediment production. On midslope roads, normalized sediment production increased with decreasing soil depth.

Most of the interannual variability in sediment production rates can be attributed to differences in the magnitude and type of precipitation, and the resulting effect on rainsplash and hydraulic erosion. The first wet season had near-normal precipitation and much of the precipitation in the lower portions of the study area fell as rain rather than snow. In the second and third wet seasons precipitation was below normal and tended to fall as snow. Unit area erosion rates were 3-4 times higher in the first wet season than the second and third wet seasons due to the higher rainfall erosivity, a less persistent snow cover that helps shield the road surface against rainsplash erosion, and reduced road runoff rates.

Road sediment production is best mitigated by rocking native surface roads, decreasing sediment transport capacity by improving and maintaining drainage, and avoiding sites with soil characteristics that increase road surface and ditch runoff. Grading road surfaces and ditches should be kept to a minimum as this increases sediment production rates. Additional process-based studies are needed to quantify the sources of road and ditch runoff, and to measure the effect of runoff rates on sediment detachment and transport. These data are needed to develop and test spatially-distributed, physically-based road erosion models. Accurate road erosion models are needed to help design effective BMPs and provide guidance for land managers.

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2.7. TABLES AND FIGURES

Wet season	Native surface roads		Rocked roads		
	Ungraded	Recently-graded	Ungraded	Recently-graded	Totals
1999-2000	15	2	9	1	27
2000-2001	15	22	9	1	47
2001-2002	15	40	10	0	65
Totals	45	64	28	2	139

Table 2.1. Number of road segments monitored by wet season and road surface type.

Dependent variables	Independent variables
Sediment production = kg	Road segment slope (S)
Sediment production rate = kg m^{-2}	Road surface area (A)
Normalized sediment production = kg EI_A^{-1}	Road area x slope (A*S)
Normalized sediment production rate = kg m^{-2} EI_A^{-1}	Road area x $slope^2$ ($A*S^2$)
Normalized sediment production rate = kg $A*S^{-1}$ EI_A^{-1}	Elevation
	Road grading (categorical)
	Road surface type
	Annual precipitation (P)
	Annual storm erosivity (EI_A)
	Maximum storm erosivity (EI_M)
	Soil series
	Lithology
	Soil depth
	Soil erodibility (K factor)
	Soil texture

Table 2.2. List of dependent and independent variables.

Wet season	Start of snowpack	End of snowpack	Number of days with snowpack	Maximum SWE (mm)
1999-2000	7 Dec	31 March	115	302
2000-2001	26 Oct	24 April	167	406
2001-2002	na*	21 April	na	353

* SWE was 150 mm on 6 December 2001.

Table 2.3. Duration of the snowpack and maximum SWE for each of the three wet seasons. na indicates not available.

Wet season	Native surface roads				Rocked roads			
	Mean ($kg\ m^{-2}$)	St. dev. ($kg\ m^{-2}$)	CV (%)	n	Mean ($kg\ m^{-2}$)	St. dev. ($kg\ m^{-2}$)	CV (%)	n
1999-2000	0.81	1.2	148	17	0.36*	1.00	278	10
2000-2001	0.22	0.3	136	37	0.01	0.01	100	10
2001-2002	0.23	0.28	122	55	0.02	0.02	100	10
Mean or total	0.32	0.56	175	109	0.13*	0.6	462	30

* Removing the one segment with the graded inboard ditch reduces the 1999-2000 mean to 0.03 $kg\ m^{-2}$ and the overall mean to 0.02 $kg\ m^{-2}$.

Table 2.4. Mean, standard deviation, and coefficient of variation (CV) of the sediment production rates for each wet season for native surface and rocked road segments.

Variable	Coefficient	Standard error of coefficient estimate	p-value
Intercept	-329	58.1	<0.0001
A*S (m^2)	3.56	0.380	<0.0001
EI_A (MJ mm ha^{-1} hr^{-1})	0.542	0.100	<0.0001
A*G (m^2)	0.389	0.100	0.0018

Table 2.5. Model parameters for predicting annual sediment (kg) from native surface road segments in the study area. The model R^2 is 0.56, the adjusted R^2 is 0.54, and the standard error is 142 kg.

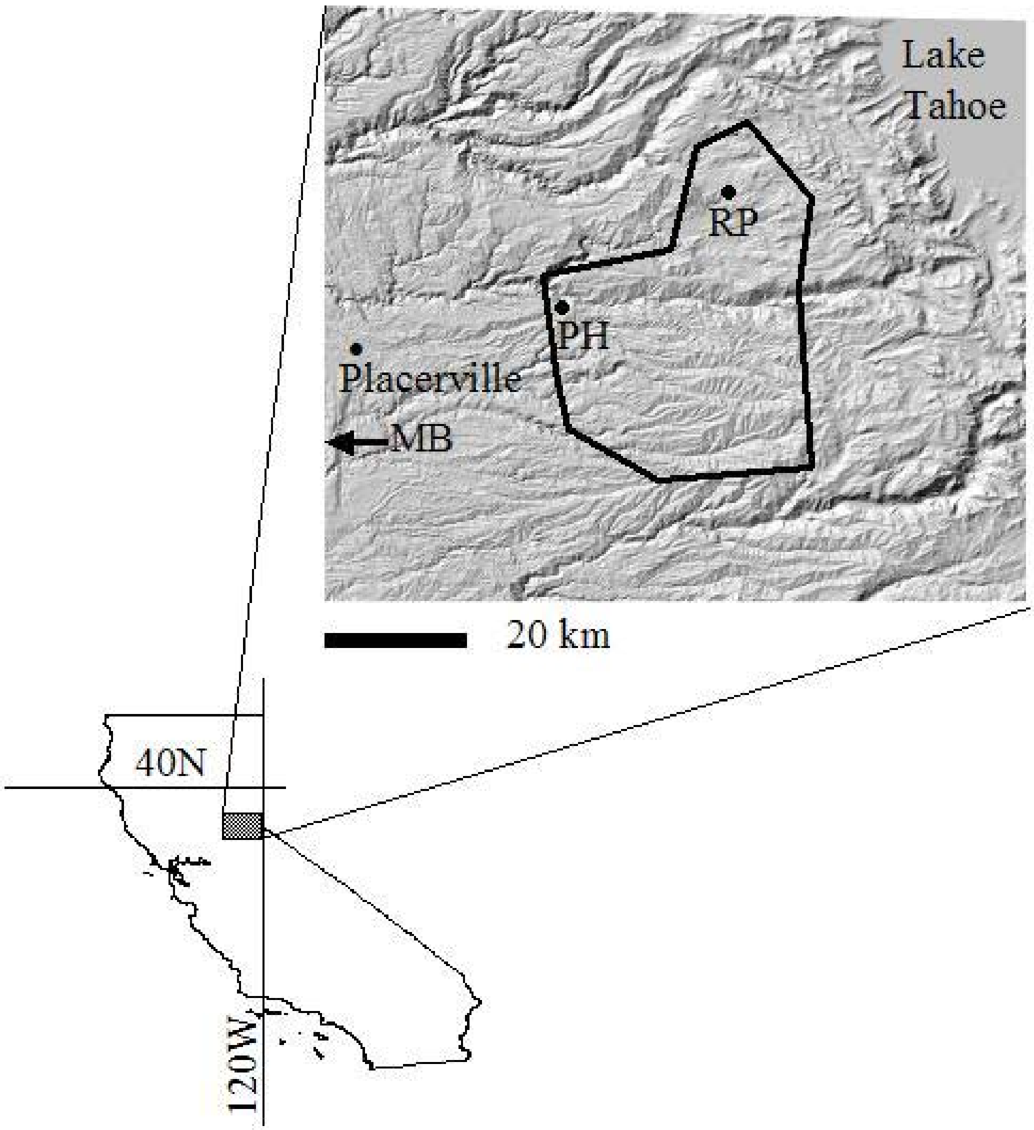

Figure 2.1. Map of the study area. PH is the Pacific House rain gage, RP is the Robbs Powerhouse SNOTEL site, and MB is the Michigan Bar gaging station on the Cosumnes River.

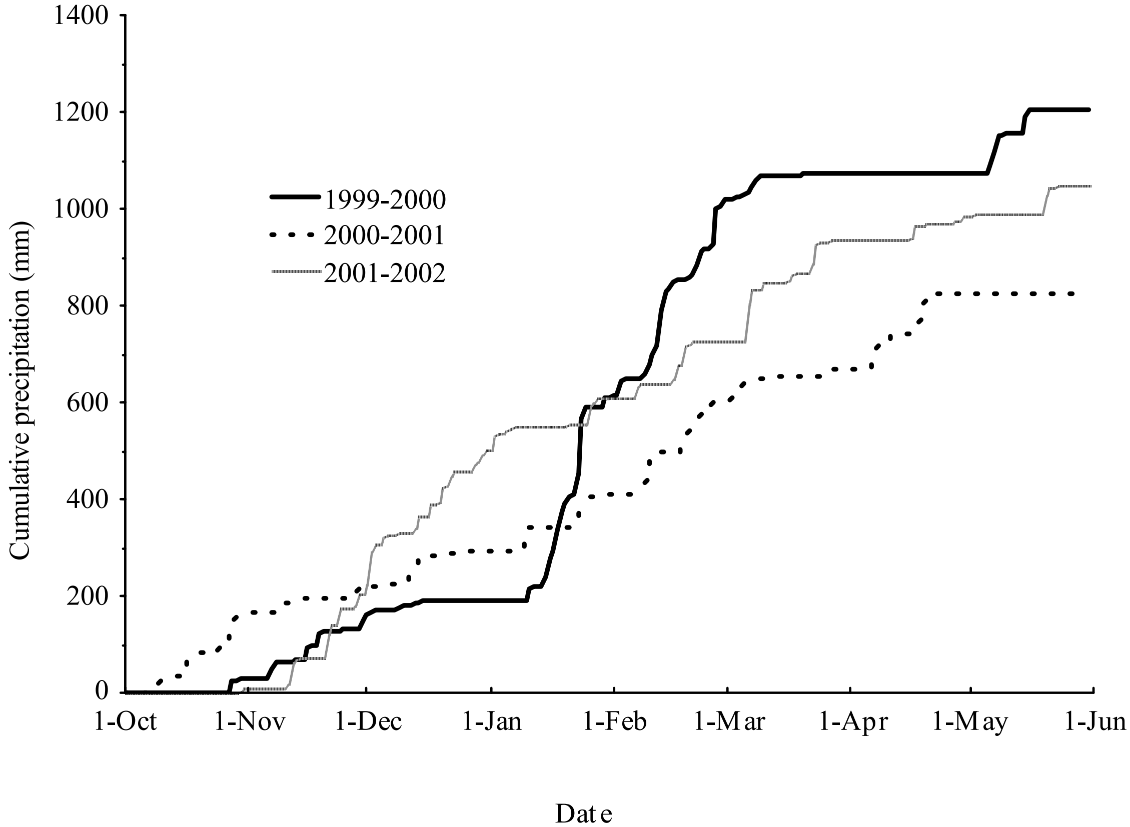

Figure 2.2. Cumulative precipitation at Pacific House from 1 October to 1 June for each of the three wet seasons.

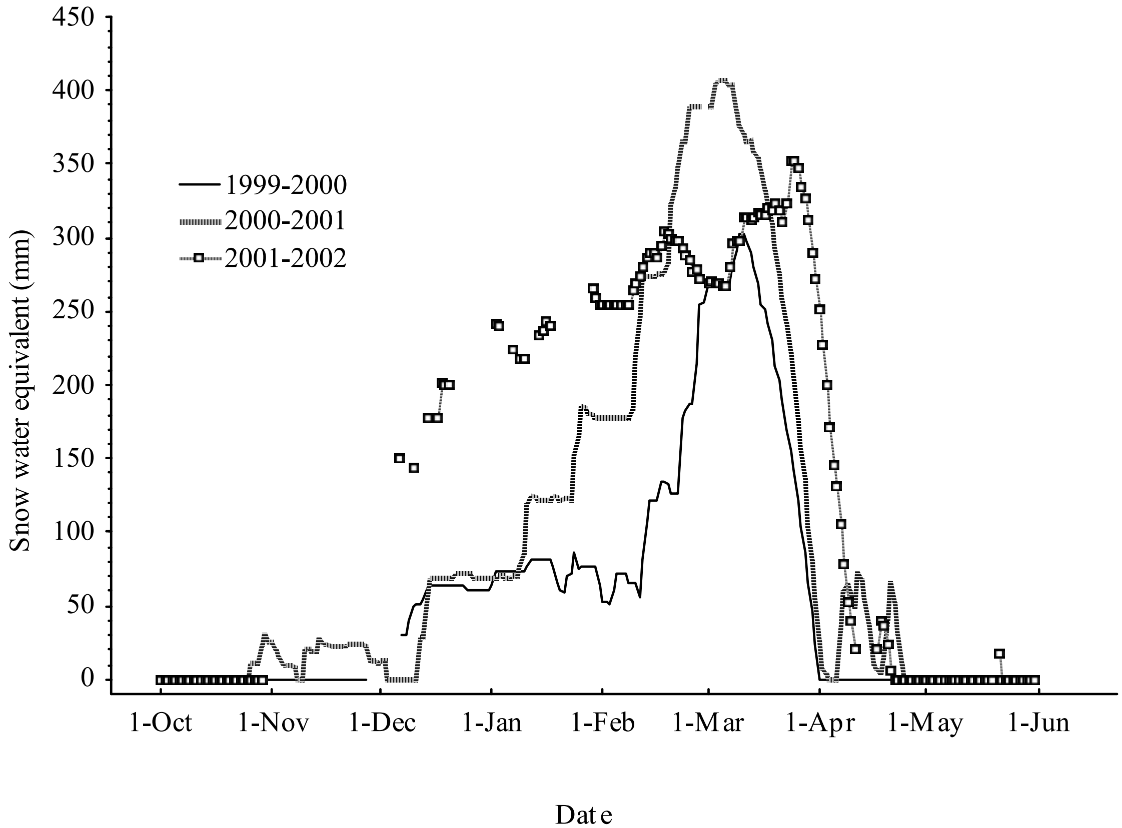

Figure 2.3. Snow water equivalent at Robbs Powerhouse for each of the three wet seasons. Data for 2001-2002 are incomplete.

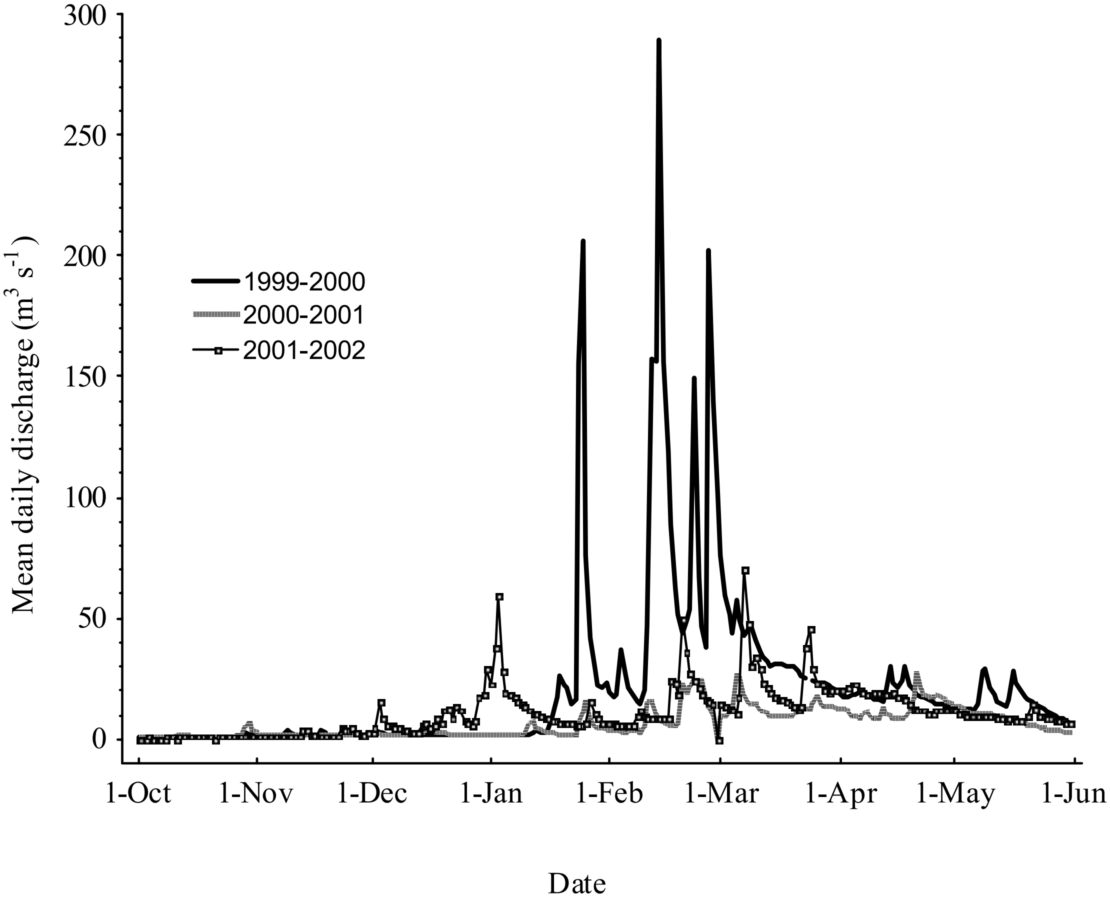

Figure 2.4. Mean daily discharge of the Cosumnes River at Michigan Bar for each of the three wet seasons.

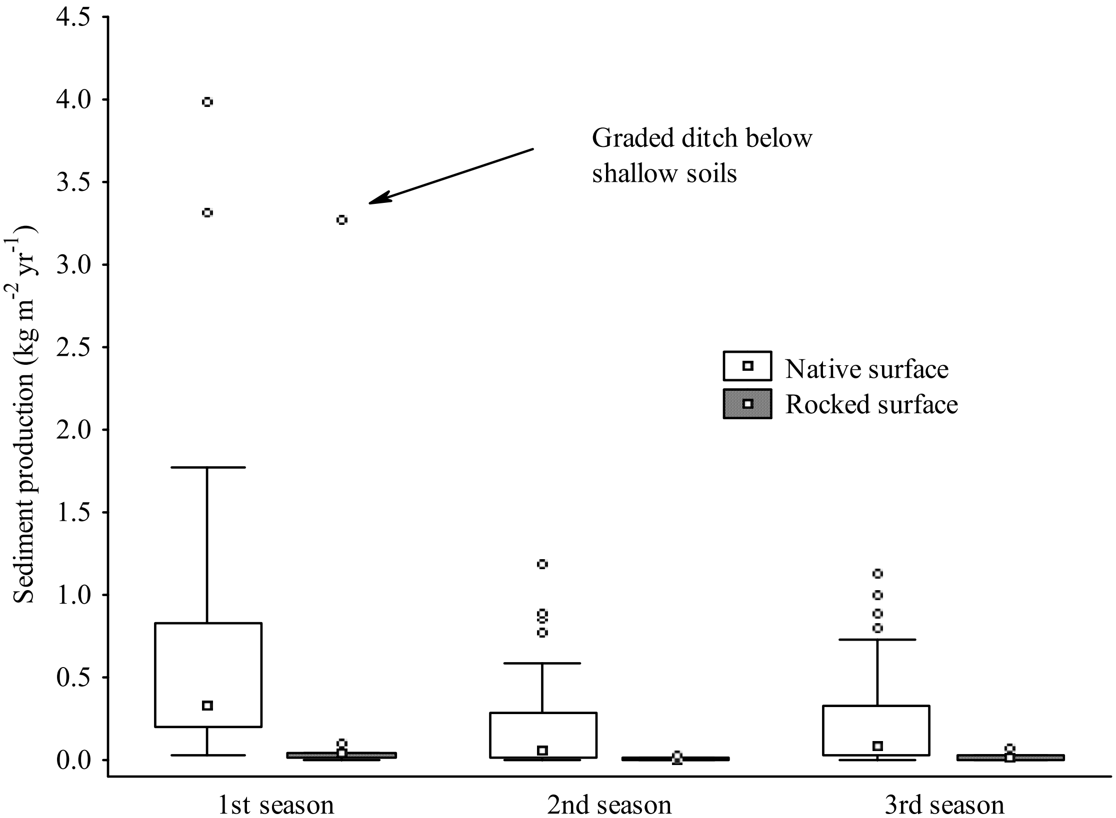

Figure 2.5. Annual sediment production rates for native surface and rocked road segments by wet season. Boxes represent the 25th to 75th quartiles, and the small boxes represent the median value. Circles represent outliers.

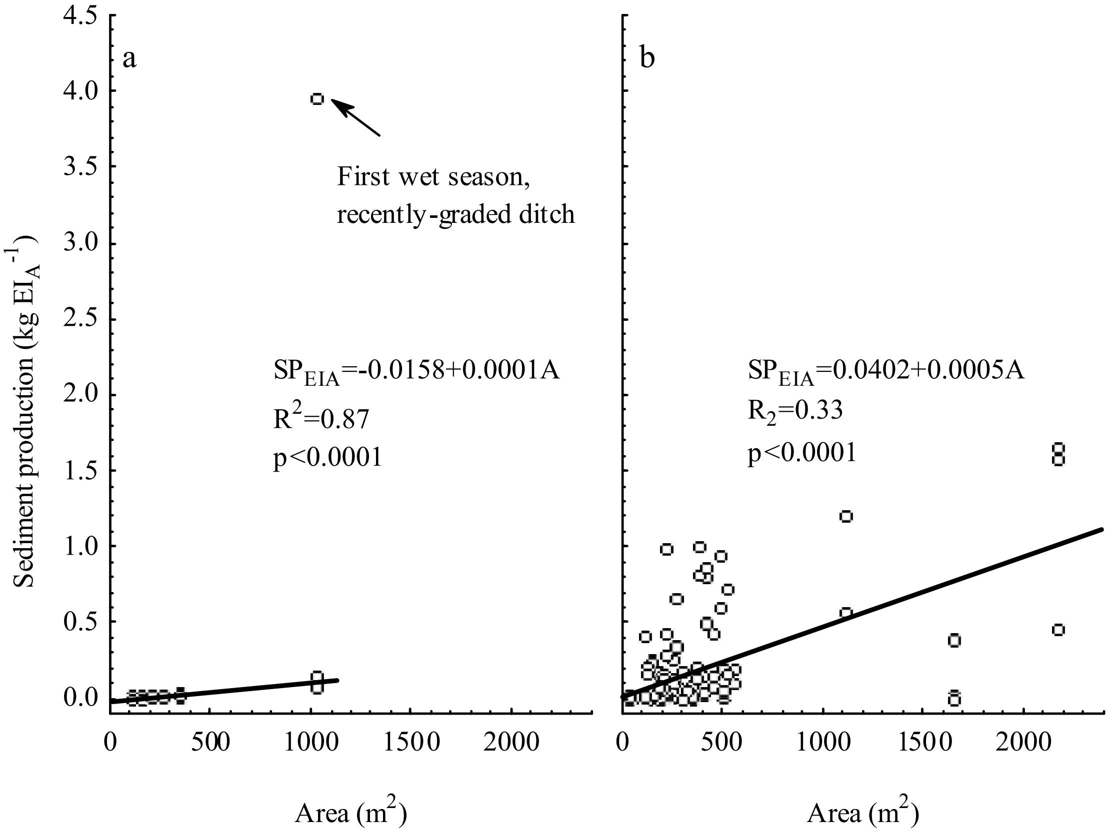

Figure 2.6. Road surface area versus normalized sediment production for: (a) rocked roads, and (b) native surface roads. The data point for the rocked road segment with the graded ditchline is shown, but this point was not included in the regression equation.

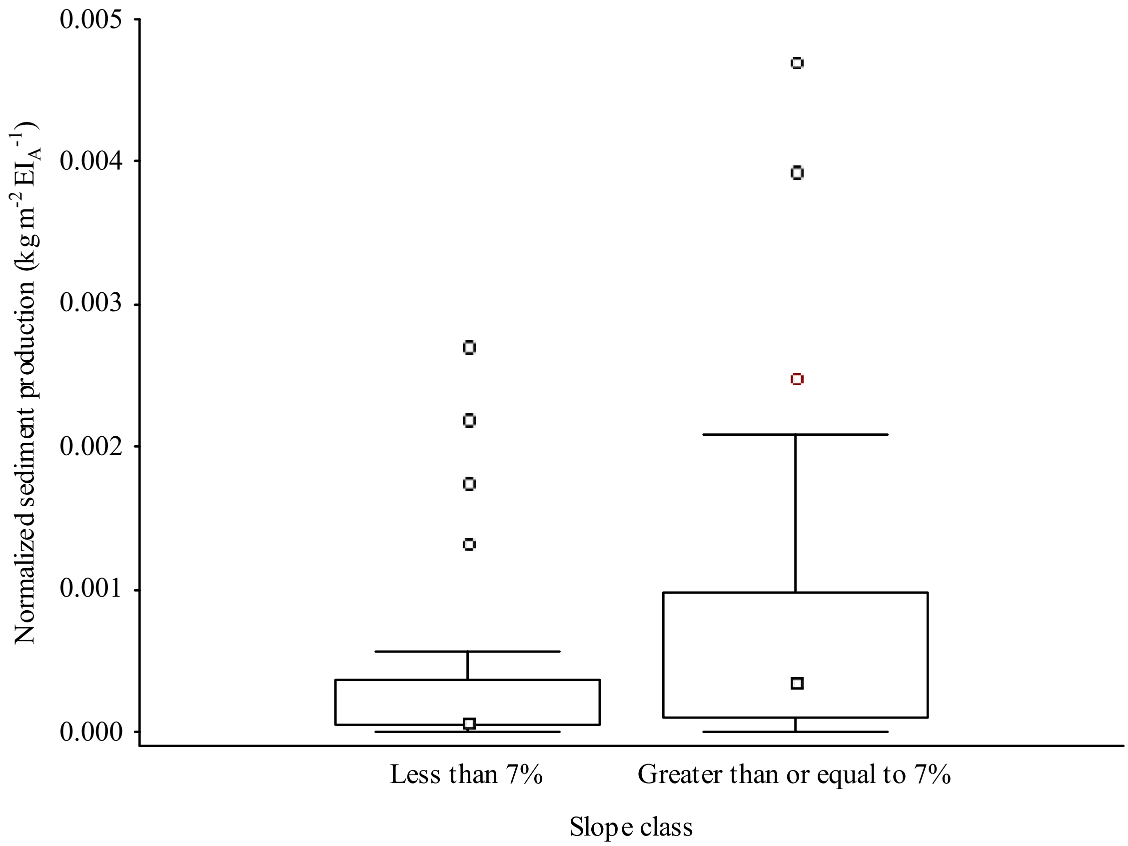

Figure 2.7. Normalized annual sediment production rate for native surface road segments by slope class.

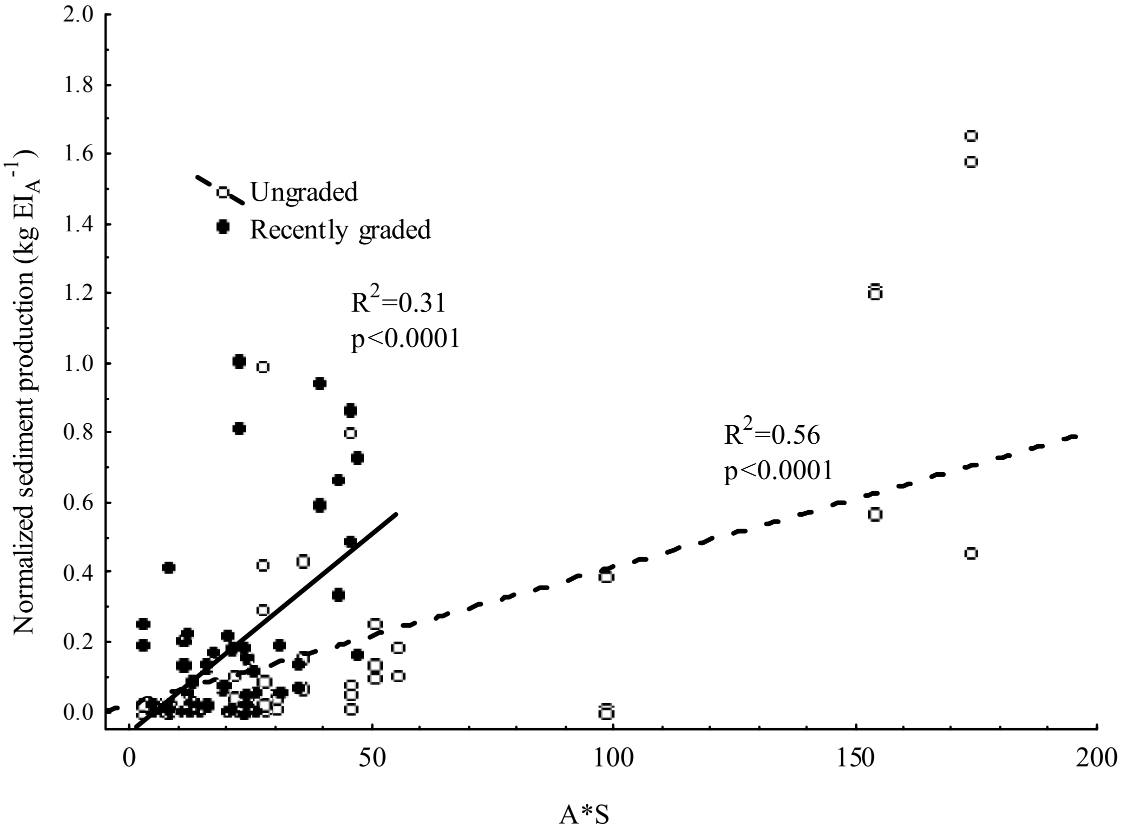

Figure 2.8. Sediment production normalized by EI_A versus road segment area times slope (A*S) for ungraded and recently-graded road segments. Recently-graded roads produce significantly more sediment than ungraded roads when using A*S as a covariate (p=0.02).

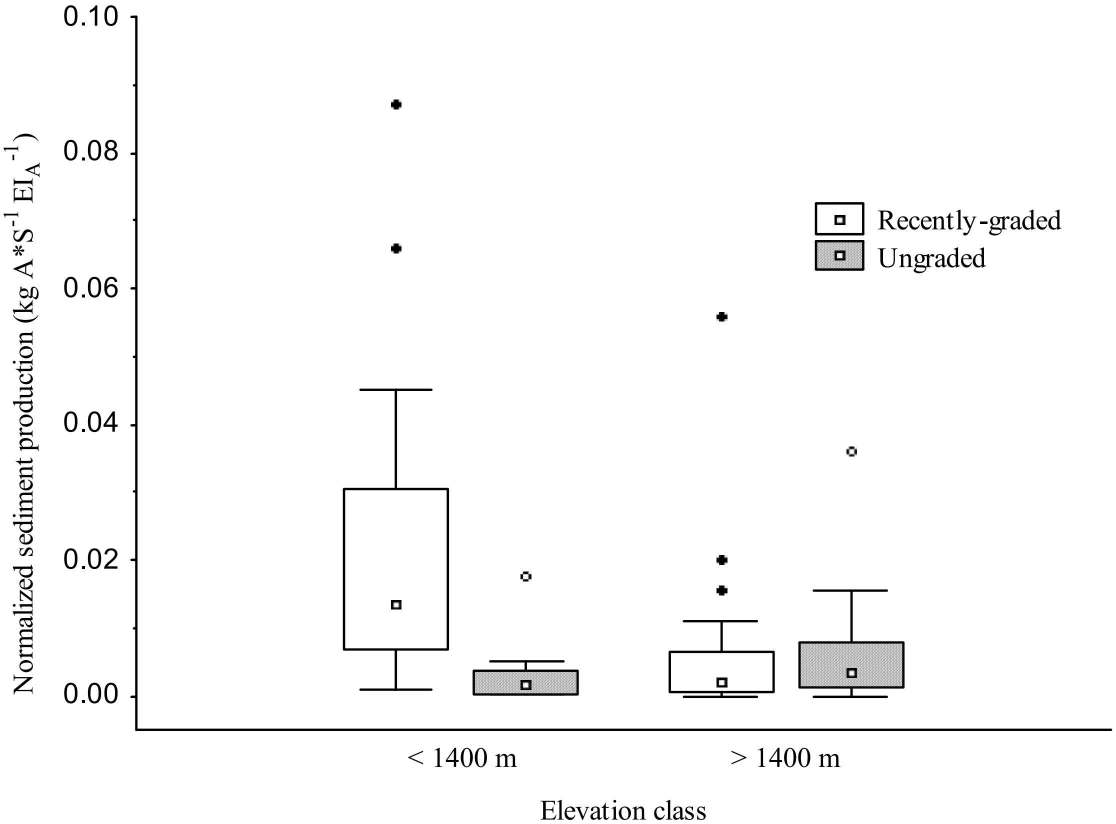

Figure 2.9. Sediment production rates normalized by A*S and EI_A for ungraded and recently-graded road segments by elevation class.

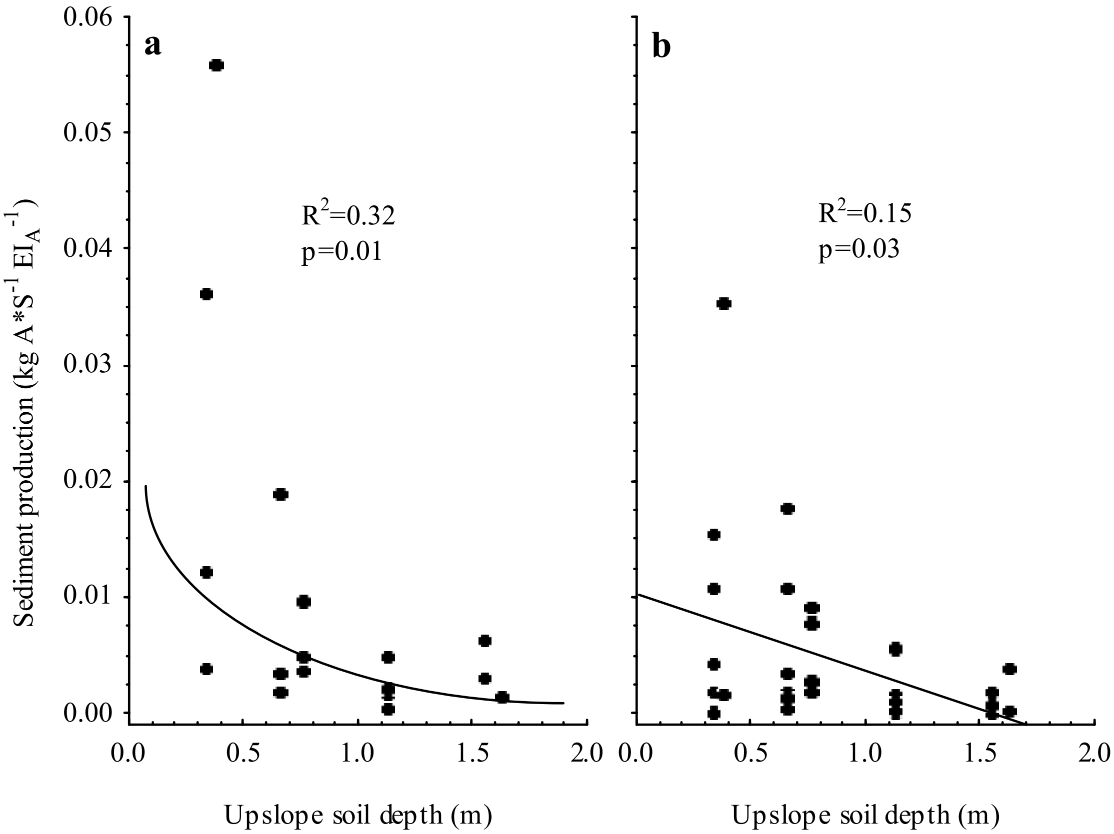

Figure 2.10. Sediment production normalized by A*S and EI_A versus upslope soil depth for midslope road segments in: (a) the first wet season, and (b) the second and third wet seasons.

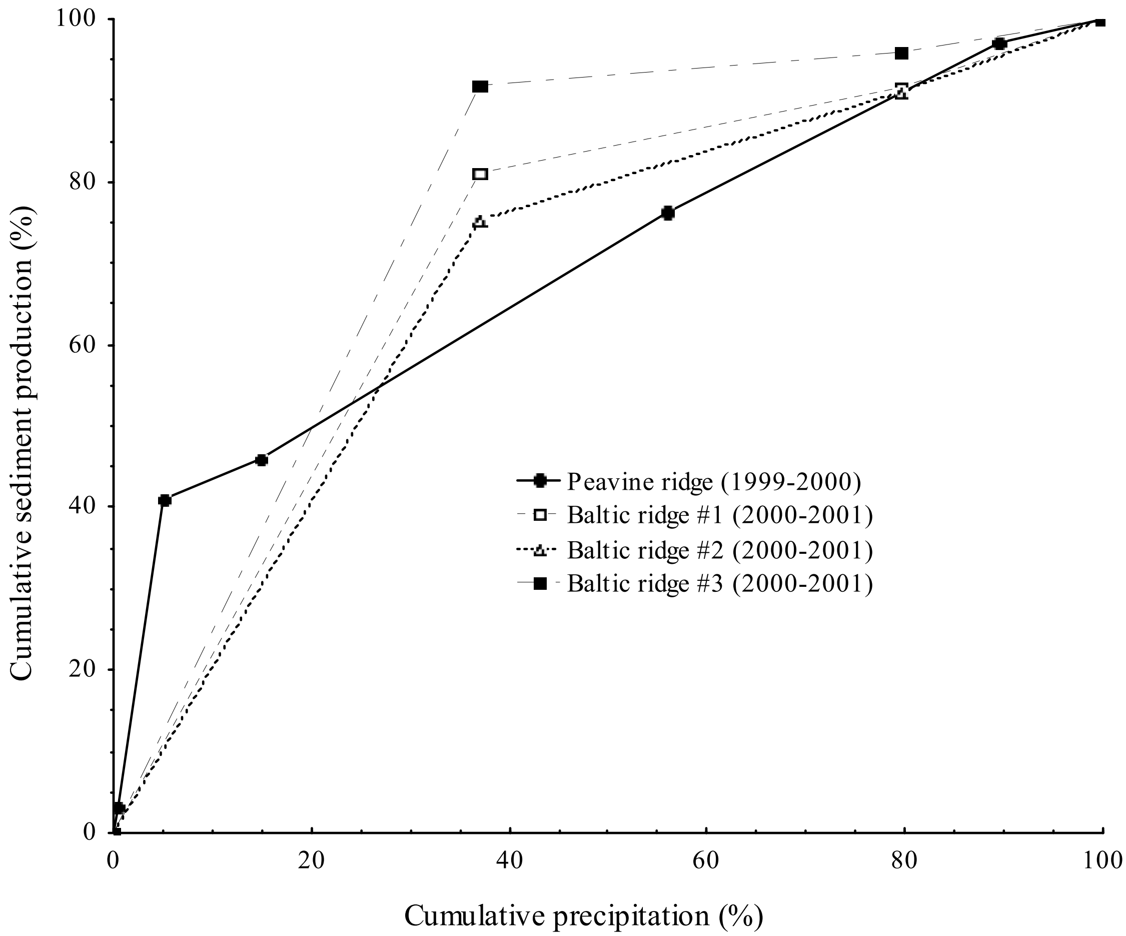

Figure 2.11. Cumulative precipitation versus cumulative sediment production for four recently-graded native surface road segments.

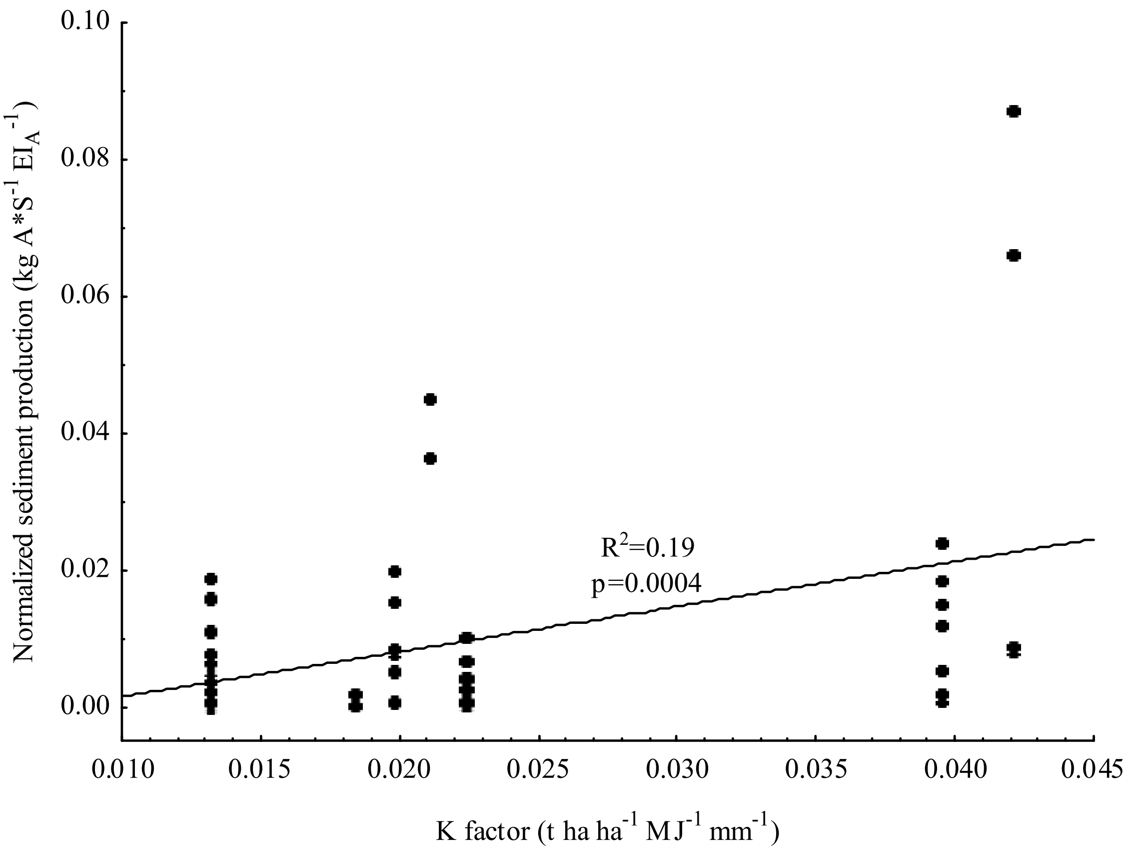

Figure 2.12. Sediment production normalized by A*S and EI_A for recently-graded native surface roads versus the published soil erodibility or K factor.

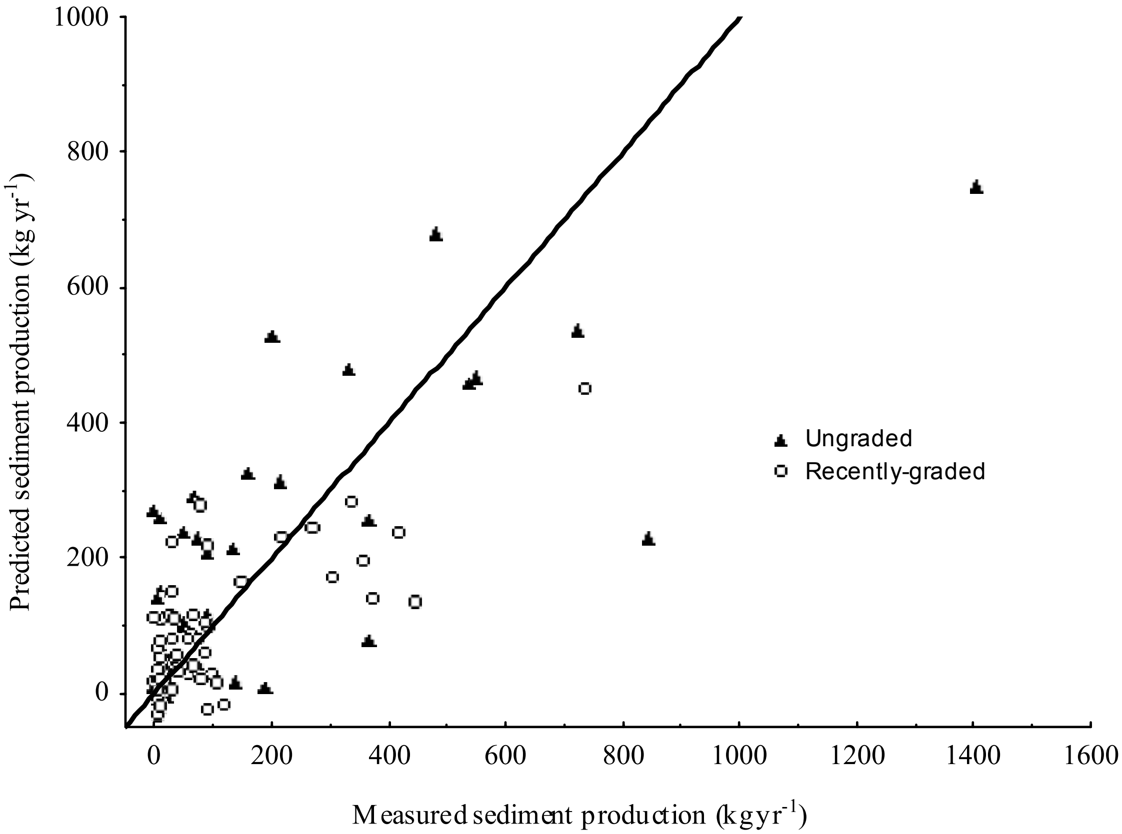

Figure 2.13. Measured versus predicted sediment production for the native surface road segments.

3.0. SEDIMENT DELIVERY FROM FOREST ROADS IN THE SIERRA NEVADA

ABSTRACT:

Sediment delivery was assessed by an intensive survey of 285 road segments along 20 km of roads in the Sierra Nevada Mountains of California. Overall, 16% of the 285 road segments and 25% of the road length were connected to the channel network. Fifty-nine percent of the connected road segments were due to stream crossings, while 35% of the connected segments resulted from road-induced gullies. Six percent of the segments were connected via sediment plumes. Sediment traveled less than 42 m below the drainage outlet for 95% of the road segments. The mean length of road-induced gullies was three times the mean length of road-induced sediment plumes. Thirty-nine percent of the variability in sediment travel distance was explained by the presence or absence of a gully below the drainage outlet, soil erodibility, estimated road traffic class, and road segment length. Gully initiation increased with road segment length, sideslope gradient, road designs that concentrated road runoff, and factors that affected the roughness and infiltration capacity below the drainage outlet. The presence or absence of gullying below a road segment was predicted with 90% accuracy by a logistic regression model. Road-induced gully volume was significantly related to the product of road length and hillslope gradient, soil erodibility, and road drainage type (R^2=0.60). The magnitude of sediment delivery from episodic gully erosion is 0.6 Mg km^{-1} yr^{-1}, compared to 1.4 Mg km^{-1} yr^{-1} of sediment delivered from road surfaces. Road sediment

delivery can be minimized by reducing the number of stream crossings in new road construction, disconnecting road drainage from stream crossings, frequently draining road segments on steep or erodible soils, and outsloping roads. An analysis of data from this and other studies shows that the proportion of road length that is connected to the stream channel network is strongly correlated with mean annual precipitation and the presence or absence of engineered drainage structures (R^2=0.92).

3.1. INTRODUCTION

Unpaved roads are chronic sediment sources in many parts of the western United States (Megahan and Kidd, 1972; Reid and Dunne, 1984; Luce and Black, 1999). Erosion from forest roads can exceed natural erosion rates by one or more orders of magnitude (Megahan and Kidd, 1972; Reid and Dunne, 1984; MacDonald et al., 2001; Ramos-Scharron and MacDonald, 2005). The resulting sediment can adversely impact aquatic resources if it is delivered to the channel network (Cederholm et al., 1981; Waters, 1995; Nelson and Booth, 2002; Suttle et al., 2004). Therefore, it is important to quantify the amount of road sediment that reaches the channel network and understand the causal mechanisms for road sediment delivery.

Several recent studies have assessed road-to-stream connectivity to help predict the hydrologic effects of roads (Wemple et al., 1996; La Marche and Lettenmaier, 2001; Bowling and Lettenmaier, 2001), and the potential for road-related sediment to be delivered to the channel network (Croke and Mockler, 2001). The most obvious road-to-stream connection occurs at stream crossings (Wemple et al., 1996; Croke and Mockler, 2001). Connectivity also occurs when road-generated Horton overland flow (Q_{HOF}) and intercepted subsurface stormflow (Q_{ISSF}) induce gullies that extend to the stream network (Montgomery, 1994; Wemple et al., 1996; Croke and Mockler, 2001; La Marche and Lettenmaier, 2001; Bowling and Lettenmaier, 2001). Road-related sediment also may travel downslope as sediment plumes, and some of this sediment can be delivered to the channel network (Haupt, 1959; Megahan and Ketcheson, 1996; Brake et al., 1997).

Studies in the Pacific Northwest (Montgomery, 1994; Wemple and Jones, 1996; La Marche and Lettenmaier, 2001) and southeastern Australia (Croke and Mockler, 2001) have shown that road sediment delivery is controlled by factors such as road segment length, road drainage type, hillslope gradient, hillslope curvature, and distance to the stream. However, little is known about the controlling factors for road sediment delivery in the mixed rain-snow climate in the California Sierra Nevada. The one study on road-stream connectivity in the Sierra Nevada focused on paved road networks (Montgomery, 1994), and data from different areas are needed to better understand the site-specific controls and variations in road-to-stream connectivity.

Along with high-severity wildfires, unpaved roads in the Sierra Nevada have the highest surface erosion rates in the Sierra Nevada (MacDonald et al., 2004). Data on road-to-stream connectivity are needed to predict and model the delivery of sediment from forest roads, and for assessing cumulative watershed effects. The resulting information can be used by land managers to help disconnect road sediment sources from the channel network and prioritize road maintenance and restoration efforts.

The specific objectives of this study were to: (1) characterize and quantify the pathways that control the delivery of runoff and sediment from unpaved forest roads to the channel network; (2) quantify the effect of the different site-scale factors on road-stream connectivity; (3) develop empirical models to predict road-stream connectivity; and (4) compare connectivity results from the Sierra Nevada with data from other studies.

3.2. BACKGROUND

The connectivity between roads and stream channels depends on a variety of factors. Conceptually, road-stream connectivity should increase with an increase in road and stream density due to the resultant increase in the number of stream crossings (Jones et al., 2000). In the western Cascades of Oregon, road-stream crossings accounted for almost 60% of all connected road segments (Wemple et al., 1996). The magnitude and importance of road connectivity at stream crossings will depend on the road design (e.g., outsloping), the proximity of road drainage structures on either side of the stream crossing, and all of the other factors that affect road runoff and erosion.

For the road segments that do not intersect that channel network, the travel distance of road-derived sediment depends on the amount of road-derived runoff and the factors that control the sediment transport capacity of runoff below the road drainage outlet (Megahan and Ketcheson, 1996). For roads dominated by Horton overland flow (Q_{HOF}), road length and road surface area are surrogates for the amount of runoff from a given road segment (Montgomery, 1994; Luce and Black, 1999; Chapter 2). However, for roads dominated by the interception of subsurface stormflow (Q_{ISSF}), the amount of road runoff will vary with other factors, such as the upslope drainage area and the ratio of cutslope height to soil depth (Montgomery, 1994; Wigmosta and Perkins, 2001; Wemple and Jones, 2003).

The sediment travel distance below the road segment also depends on the hillslope gradient, hillslope roughness, road drainage type, and time since construction (Haupt, 1959; Packer, 1967; Burroughs and King, 1989; Megahan and Ketcheson, 1996; Brake et al., 1997). Research in Idaho has shown that road sediment travel distance is controlled by hillslope gradient, obstructions on the hillslopes below the road drainage outlets, and road drainage type (Burroughs and King, 1989; Megahan and Ketcheson, 1996). In the Oregon Coast Range newly-constructed roads have longer sediment travel distances than older roads (Brake et al., 1997).

Several studies have evaluated the role of gullying on road sediment delivery. In western Oregon, 23% of the road drainage outlets were connected to the channel network via gullying (Wemple et al., 1996). In southeastern Australia 18% of the road sgements were connected to the stream network by gullying (Croke and Mocker, 2001). Road-induced gullies can be both a pathway for delivering road surface runoff and sediment to the channel network (Wemple et al., 1996; Croke and Mockler, 2001; LaMarche and Lettenmaier, 2001), and a source of sediment to the channel network as they develop and enlarge over time.

A gully is more likely to develop below a road drainage outlet as segment length increases (Montgomery, 1994; Wemple et al., 1996; Croke and Mockler, 2001) and hillslope gradient increases (Wemple et al., 1996). Quantitatively, the following relationship has been proposed for gully initiation:

$$L = L_t / \sin \theta \qquad (3.1)$$

where L is the critical contributing length of road necessary to initiate gullying (m), θ is the hillslope angle in degrees, and L_t is an empirical constant that represents the threshold road length (m) (Montgomery, 1994; Croke and Mockler, 2001). Gullies initiate when the product of road length and hillslope gradient exceed the L_t value.

3.3. METHODS

3.3.1. *Site Description*

The study area lies on the west slope of the Sierra Nevada mountain range in California (Figure 3.1). To the north it is bounded by the Rubicon River drainage, and to the south by the South Fork of the Cosumnes River. The primary forest type is mixed conifer, but this turns to red fir with increasing elevation (SAF, 1980). The Mediterranean-type climate means that most of the precipitation falls between November and April (USDA, 1985). Elevations range from 910 to 2000 m, and the mean annual precipitation at 1036 m is 1300 mm. The majority of the study area corresponds with the rain-on-snow climatic zone (Cobourn, 1989). Most of the road surveys were on the Eldorado National Forest, although some sites were on interspersed Sierra Pacific Industries (SPI) property.

The dominant lithologies are weathered granitic batholith, granitic glacial deposits, and volcanic (i.e., Mehrten formation) (USDA, 1985). The soils are typically coarse-textured loams. Most of the soils are over a meter thick, but the range is from 0.3 m to 1.5 m.

3.3.2. *Survey Procedures*

Twenty 1-km road transects were randomly selected and were surveyed in the summer of 2001. Each road transect was identified by randomly selecting one of the 1:24,000 USGS topographic maps in the study area, randomly selecting a section on the selected map, numbering each road in the selected section, and then randomly selecting a road using a random number generator. The roads were broken into subunits at road intersections, and one road intersection was randomly chosen as the starting point for the survey.

Each 1-km road transect was broken into road segments as defined by drainage outlets such as waterbars, rolling dips, or ditch-relief culverts, or a change in drainage direction due to ridges or stream crossings. The length of each segment was measured to the nearest decimeter with a flexible tape. The road gradient was measured at each break in slope with a clinometer, and a distance-weighted mean gradient was calculated for each segment. The width of the road tread was measured at several points and used to determine a mean width. Road segment length times the mean width yielded the road surface area for each segment.

The road segments were classified into three main drainage types: 1) outsloped segments; 2) outsloped and bermed segments; and 3) insloped segments drained by cross-relief culverts. By definition, the outsloped segments had diffuse drainage to the outside edge of the road and onto the hillslope. The outsloped and bermed roads were designed to be outsloped, but the combination of traffic and grading resulted in ruts or a berm along the outside edge that prevented runoff from leaving the road surface; drainage from these segments only occurred at a rolling dip, waterbar, or stream crossing. Segments drained by inside ditches were typically insloped, and were constructed using a cut-and-fill design with periodic relief culverts. If a segment was crowned and had an inside ditch, the road surface was divided into an outsloped and insloped portion and was counted as two road segments. In general, the outsloped roads had been more recently constructed and represented current road construction and maintenance standards, whereas the older roads were more typically insloped.

For each road segment the traffic level was qualitatively assessed as high, medium, or low. High traffic segments had evidence of recent timber hauling and typically had a thick layer of fine sediment on much of the road surface. Moderate traffic segments had evidence of frequent use by recreational traffic but no evidence of recent timber hauling. Low traffic segments had dense brush cover that prevented the use of the road by most vehicles.

Lithology, soil type, and soil depth were determined from soil survey data (USDA, 1985); lithology was field verified. The cutslope height was measured at varying intervals along the road segment length and averaged for each segment. The mean cutslope height to soil depth ratio was calculated for each segment. Hillslope gradients (m m^{-1}) below the drainage outlet and above the cutslope were measured with a clinometer. These values were averaged to obtain a mean hillslope gradient.

Each drainage outlet was assessed for signs of sediment delivery to the channel network using four connectivity classes (CC) (Wemple et al., 1996; Croke and Mockler, 2001) (Table 3.1). Road segments classified as CC1 had no signs of gullying or sediment transport below the drainage outlet, and have a very low potential for sediment delivery. Road segments classified as CC2 had gullies or sediment plumes that extended for no more than 20 m from the drainage outlet, and are considered to have a low to moderate potential for sediment delivery. Road segments identified as CC3 had gullies or sediment plumes that were at least 20 m in length, but ended more than 10 m away from the bankfull width of the nearest stream channel; these were considered to have a moderate to high potential for sediment delivery. Segments classified as CC4 intersected stream channels at stream crossings or had gullies or sediment plumes that extended to within 10 m of the bankfull edge of a stream channel. CC4 segments were classified as connected and have the highest potential for delivering sediment to the channel network (Table 3.1).

If present, the geomorphic feature below each drainage outlet that was used to indicate the sediment transport distance was categorized as either a sediment plume or a gully. Sediment plumes were defined by the presence of diffuse sediment and the absence of an actively incising channel. Gullies were defined by signs of channelized flow and incision. The length of each sediment plume and gully was measured. The top width and maximum depth of each gully was measured at 5-m intervals, and the cross-sectional area was calculated by assuming the gully had a triangular cross-section (i.e., cross-sectional area=1/2 * width * maximum depth). This area was multiplied by the length represented by each cross-section (typically 5 m) to yield a volume, and the sum of these volumes yielded the total volume for each gully.

The condition of the hillslope immediately below the drainage outlet was qualitatively assessed for the factors that may affect gully or sediment plume length. If a road segment discharged onto forest litter, the hillslope condition was categorized as "litter". If a road segment discharged runoff onto dense vegetation (e.g., brush) or large woody debris (LWD), then the hillslope condition was categorized as "energy dissipator". If a road segment discharged runoff onto compacted or disturbed soil, the hillslope condition was categorized as "disturbed".

3.3.3. Statistical Analysis

A variety of statistical methods were used to evaluate the effect of the different categorical and continuous variables on connectivity class, length of sediment plumes and gullies, gully presence or absence, and gully volume (Table 3.2). The mean values of the independent variables were compared across the discrete dependent variables, such as connectivity class or geomorphic feature, using Tukey Honestly Significant Difference (HSD) (Ott, 1993; STATISTICA, 2003). Log-normally distributed data were transformed before the Tukey HSD analysis to meet the assumptions of normality. A value of 0.1 was substituted for zero values for gully volumes, gully lengths, and sediment plume lengths in order to facilitate log transformation. Stepwise multiple regression with a selection criteria of $p<0.05$ was used to develop predictive models for gully and sediment plume lengths. Categorical variables were represented as binary variables in the model selection process. Forward stepwise logistic regression with a selection criteria of $p<0.05$ was used to predict the presence and absence of gullies below the drainage outlet. Additional logistic regression models were explored using Akaike Information Criterion (AIC) best subset model selection process (STATISTICA, 2003). All of the segments at stream crossings were excluded from the datasets used in the multiple and logistic regression analyses since the sediment plume lengths, gully lengths, and gully volumes for these segments were zero. Some gullies and sediment plumes from CC4 road segments were truncated by the stream channel, but they were left in the analysis to increase the sample size.

3.4. Results

3.4.1. Road Connectivity

The road survey covered 20 km of native surface roads and delineated 285 road segments. The mean segment length was 81 m, but lengths were highly variable as the standard deviation was 64 m and the range was from 7 m to 401 m (Table 3.3). The mean road gradient was 6%, and the range was from 0% to 17%. Hillslope gradients averaged 26% and ranged from 0% to 57%. The mean cutslope height for all road segments was 1.9 m, and values ranged up to 8.0 m. Cutslope height was significantly correlated with hillslope gradient (R^2=0.31, $p<0.0001$).

Seventy-seven percent of the road segments were outsloped but also were drained by waterbars or rolling dips. Fourteen percent were outsloped but had berms that kept the water on the road surface; these also were drained by waterbars or rolling dips. The remaining 9% of the road segments were insloped and drained by relief culverts.

Sixty-four percent of the road segments were on volcanic lithology, and the other 36% were either on weathered granitic (14%) or glacial granitic lithologies (22%). Thirty-one percent of the road segments were classified as having a high level of traffic, 48% had a moderate level of traffic, and 21% were classified as low traffic.

Sixteen percent of the road segments were connected to the stream network (Table 3.4), but these represented 25% of the total road length. Forty-nine percent of the road segments, or 38% of the total length, were categorized as CC1, meaning that there was no indication of gullying or sediment transport below the drainage outlet. Another 28% of the road segments were classified as CC2, indicating that sediment plumes and gullies extended for less than 20 m. Only 7% of the road segments had rills or sediment plumes extending more than 20 m (CC3).

Stream crossings were the dominant causal mechanism for sediment delivery to the channel network, as these accounted for 59% of the connected road segments. Another 35% of the road segments classified as CC4 were connected to the channel network by gullies. Only 6% of the road segments classified as CC4 were connected to the channel network via sediment plumes (Figure 3.2).

Connectivity class tended to increase with longer segment lengths (Figure 3.3). The mean length for the segments classified as CC1 was 63 m versus 109 m for the segments classified as CC4. The road segments classified as CC3 and CC4 were significantly longer than the segments classified as CC1 and CC2 ($p<0.0001$; Figure 3.3).

Connectivity class was strongly related to the type of road design, as approximately 90% of segments that were insloped and drained by relief culverts were classified as CC3 or CC4. In contrast, only 16% of the road segments that were drained by waterbars or rolling dips were classified as CC3 or CC4 (Figure 3.4).

3.4.2. Gully and Sediment Plume Lengths

Sediment travel distances depended on whether the geomorphic feature below the drainage outlet was a sediment plume or a gully (Figure 3.5). If the 25 segments draining directly to a stream crossing are excluded, sediment plumes were present below 29% of the road segments and the mean length was 11.8 m. The longest plume was 183 m, and this was due to road runoff being routed onto and down a skid trail. Gullies were found below just 13% of the road segments, but the mean length was nearly 37 m, or more than three times the mean sediment plume length (p=0.0001) (Figure 3.5). Ninety-five percent of the road segments had sediment plumes or gullies that were less than 42 m in length. Sediment plumes accounted for 89% of the geomorphic features present below the CC2 road segments, while gullies accounted for 67% of the geomorphic features below CC3 road segments and 83% of the geomorphic features below CC4 road segments.

The lengths of the sediment plumes increased with traffic class (Figure 3.6). The mean sediment plume length below segments with low levels of traffic was only 3.7 m, or 28% of the mean sediment plume length for roads with high or moderate levels of traffic (p=0.001).

Gully length was a power function of the soil K factor (R^2=0.27; p=0.001), indicating that gully length increased for more erodible soils. Gully length was not significantly correlated with either road segment length (p=0.07) or hillslope gradient (p=0.76).

Multivariate models could predict only 39% of the variability in gully and sediment plume lengths for the 260 road segments that were not associated with stream crossings. The best model is:

$$\begin{aligned}\mathrm{Log}_{10}(D) = 0.965 + 1.278(\log_{10} K) + 0.409(\log_{10} L) \\ + 1.431G + 0.420T\end{aligned} \tag{3.2}$$

where D is the length (m) of the geomorphic feature, K is soil erodibility (t ha h ha^{-1} MJ^{-1} mm^{-1}) ($p=0.004$), L is road length (m) ($p=0.04$), G is a binary variable where 0 represents the absence of a gully and 1 indicates that a gully is present ($p<0.0001$), and T is a binary variable where 0 represents a low level of traffic and 1 represents a moderate to high level of traffic ($p=0.001$) (Figure 3.7). The adjusted R^2 for the model is 0.37, and the standard error is only 3.0 m because so many segments have either a very short or no sediment plume or gully.

3.4.3. Controls on Gully Initiation

Gullies were more likely to be present below the longer road segments, segments with relief culverts, and where the ratio of cutslope height to soil depth was greater than 1.0. The mean length of the 36 road segments with gullies was 118 m versus 64 m for the 224 segments without gullies ($p<0.0001$) (Figure 3.8). Approximately half of the 36 segments with gullies were insloped with relief culverts. The mean ratio of cutslope height to soil depth was 3.1 for segments with gullies; segments without gullies had a significantly lower mean ratio of 2.2 ($p=0.001$; Figure 3.9). A higher ratio indicates a greater likelihood of intercepting subsurface stormflow and a corresponding increase in surface runoff. Only one of the 36 road segments with a gully below the outlet had a cutslope height that was less than the soil depth.

Gully initiation was not significantly related to hillslope gradient ($p=0.14$), and there was not a distinct road segment area*slope or length*slope threshold (i.e., L_t) for gully initiation. However, for a given hillslope gradient a gully was more likely to occur below the longer segments (Figure 3.10). No gullies were present for road segments less than 35 m long or hillslope gradients less than 16%.

The presence or absence of gullies below road segments is best predicted by a logistic regression equation:

$$P_G = 1 / 1 + \exp [4.08 - 0.0574(L*S_H) - 3.30C + H_C] \quad (3.3)$$

where P_G is the probability of gullying; $L*S_H$ is the product of road segment length (m) and hillslope gradient (m m^{-1}); C is a binary variable with 0 representing an outsloped or bermed road segment drained by a waterbar or rolling dip and 1 representing an insloped road segment with a relief culvert; and H_C is a variable representing the condition of the hillslope 1 m below the drainage outlet. H_C is equal to zero if the drainage discharges onto forest litter, 7.1 if obstructions are present 1 m below the drainage outlet, and –2.5 if the drainage outlet discharges onto compacted soil (e.g., a skid trail or landing). If the threshold for gullying is $P_G>0.50$, the model has a 49% success rate in predicting the presence of gullies and a 96% success rate in predicting the absence of gullies, resulting in an overall model performance of 90%. If the threshold for gullying is set at $P_G>0.30$, then the model correctly predicts 63% of the gullied segments and 93% of the non-gullied segments for an overall model performance of 89%.

3.4.4. Gully Volumes

Within the study area gullies are important because they are the most common feature connecting roads to streams, and because they also can be an important source of sediment. The mean gully volume for the 36 road segments with gullies was 10.3 m^3, but the distribution was highly skewed as the median gully volume was only 3.9 m^3 and the range was from 0.01 to 153 m^3. The largest gullies are of most interest because these tended to be longer and hence more likely to reach a stream channel. In general, the cross-sectional area of gullies tended to decline as gullies progressed downslope. However, two gullies reached the inner gorge of stream channels and apparently triggered small, shallow landslides. The volume of these two slides (89.2 m^3 and 153 m^3, respectively) accounted for 54% of the total volume of sediment from gullying.

Sixty percent of the variability in gully volumes can be predicted from the following equation:

$$\text{Log10 } V = 1.88(\log10\ K) + 1.32(\log10\ L{*}S_H) + 0.515C + 1.503 \qquad (3.4)$$

where V is gully volume (m^3), K is soil erodibility (t ha h ha^{-1} MJ^{-1} mm^{-1}) (p=0.04), L is road length (m), S_H is hillslope gradient (m m^{-1}) ($L{*}S_H$; p=0.0004), and C is a binary variable with 0 representing the presence of a waterbar or rolling dip and 1 representing the presence of a relief culvert (p=0.04). The adjusted R^2 for the model was 0.57, and the standard error of prediction was 3.8 m^3 (Figure 3.11).

3.5. Discussion

3.5.1. *Gully and Sediment Plume Lengths*

The gully and sediment plume lengths from this study are generally less than or similar to other reported values. For newly constructed roads in the Idaho batholith, the mean length of sediment plumes was 53 m for segments with relief culverts and 12 m for segments with rock drains (Megahan and Ketcheson, 1996). The comparable mean sediment transport lengths for the mixed lithologies in this study were 29 m for segments with relief culverts and 6 m for segments drained by waterbars and rolling dips. However, the mean sediment transport lengths on weathered granitic batholith sites were 37 m for segments with relief culverts and 12 m for segments drained by waterbars and rolling dips. These latter values are very similar to the values from granitic sites in the Idaho Batholith. In central Idaho, the mean gully and sediment plume lengths below relief culverts were 20% shorter on metasedimentary lithologies than volcanic and granitic lithologies (Burroughs and King, 1989). The overall mean sediment travel distance of 8.7 m in this study is very similar to the mean sediment transport distances on sandstone lithology in the Oregon Coast Range of 5.1 m for old roads and 9.3 m for new roads (Brake et al., 1997).

The empirical model developed to predict gully and sediment plume length uses four variables (Eq. 3.2), and each of these variables has a physical basis. Gully or sediment plume length increases with increasing road segment length because the latter is a surrogate for the amount of road surface runoff. An increase in runoff will increase both the amount of eroded sediment and the downslope transport capacity (Luce and Black, 1999). The binary variable for the presence or absence of a gully implicitly recognizes that gullies have more concentrated runoff and a greater travel distance than the more diffuse flow associated with sediment plumes. The greater length with an increase in the K factor reflects the increase in soil erodibility with decreasing particle size and decreasing soil permeability (Lal and Elliot, 1994). Silts and fine sands are more easily detached and transported than larger particles, and a lower permeability will reduce downslope infiltration and thereby increase the travel distance.

Higher traffic levels were associated with an increase in sediment plume length but not an increase in gully length. An increase in traffic on unpaved roads increases the supply of erodible sediment that can be transported below the drainage outlet (Ziegler et al., 2001a; Ziegler et al., 2001b). In this study sediment plume lengths were significantly shorter for roads that were partly overgrown and characterized as having a low level of traffic. The vegetation on these low traffic segments is presumably reducing the amount of both runoff and erosion, and the mean plume length of 3.7 m for the low traffic segments is consistent with this explanation.

3.5.2. Gully Initiation

Gully initiation was more likely with longer road lengths, steeper hillslope gradients, insloped roads, and smoother hillslopes (Eq. 3.3). It has already been shown that longer road segment lengths are a surrogate for increased runoff and flow depths (Luce and Black, 1999). An increase in runoff and hillslope gradient will increase shear stress, and gully initiation is more likely as shear stress increases (Montgomery, 1994). The inclusion of $L*S_H$ in equation 3.3 is consistent with results from the western Cascades in Oregon, where $L*S_H$ was a significant variable in a logistic regression model developed to predict gully initiation below road drainage outlets (Wemple et al., 1996).

The type of road drainage is an important control on gully initiation, as much shorter segment lengths are needed to initiate gullies on insloped roads drained by relief culverts than for outsloped or bermed roads drained by waterbars or rolling dips. Using Equation 3.3 and assuming the mean segment length of 81 m and the mean hillslope gradient of 26%, the probability for gullying increases from 0.05 to 0.61 when a road segment is insloped and drained by a relief culvert as opposed to outsloped and waterbarred. The higher likelihood of gullying can be attributed to the more highly concentrated flow at the outlet of the relief culvert. In southeastern Australia the majority of gullies also were also associated with relief culverts as compared to other types of drainage outlets (Croke and Mockler, 2001). Figures 3.12a and 3.12b show the critical road segment length needed to have a 50% probability of gully initiation for a given hillslope gradient and hillslope condition for two drainage types.

The condition of the hillslope below the drainage outlet is important because this controls other factors, such as surface roughness and infiltration capacity, that directly affect the likelihood of gullying. Gully initiation was least likely when natural energy dissipating obstructions such as brush or LWD were present 1 m below the drainage outlet (Figure 3.12). Gully initiation was most likely when road runoff was discharged onto compacted or disturbed soils, such as skid trails. According to equation 3.3, an outsloped road with a mean length of 81 m and the mean hillslope gradient of 26% has a zero probability of gullying when an energy dissipating obstruction is below the drainage outlet, a 5% probability when the segment discharges onto forest litter, and a 42% probability of gullying if the segment discharges onto compacted soil. The corresponding probabilities for a comparable insloped road are zero, 61%, and 95%, respectively. This indicates that gully initiation below insloped roads with relief culverts is particularly sensitive to the condition of the hillslope below the drainage outlet (Figure 3.12b), and that the placement of energy dissipators below relief culverts are an effective best management practice to prevent gully erosion.

Upslope soil depth was not included in the model to predict gully initiation because it had a p-value of 0.11, but in some situations soil depth can be an important factor in gully initiation. For midslope roads, gullying is more likely when the cutslope height exceeds soil depth, as this will increase the amount of Q_{ISSF} (Wigmosta and Perkins, 2001; Ziegler et al., 2001c; Wemple and Jones, 2003). Soil depth was included when the Akaike Information Criterion (AIC) model selection process was used instead of stepwise regression. If soil depth is added to the predictive model, the success rate of predicting the presence of gullies increased from 48% to 54% when using a P_G of 0.50. Soil depth is much less likely to be important for ridgetop roads or valley bottom roads with small cutslopes, and this is probably why soil depth was not included in the overall model.

3.5.3. Gully Volumes

Gully volumes increased with longer road segment lengths, steeper hillslopes, higher K factors, and the presence of relief culverts (Eq. 3.4). As noted earlier, longer segments increase the amount of road runoff and steeper hillslope gradients increase shear stress and gully erosion (Mongtomery, 1994). Road drainage type determines whether the runoff is partially dispersed or concentrated at the drainage outlet, and the flow velocity. The logistic regression equation used to predict the presence or absence of gullies also explains 29% of the variability in log-transformed gully volumes ($p=0.0007$). This shows that the road segments with the highest probability for gullying also should have the highest gully volumes.

The connectivity data and the predictive equations can be used to calculate the amount of sediment being delivered from road-induced gullying versus the amount of sediment being delivered from road surfaces. The total volume of sediment delivered to the channel network by gully erosion was 355 m^3, or 18 m^3 per km of road. If a bulk density of 1.6 Mg m^3 is assumed, the sediment delivery rate from road-induced gullies is 29 Mg per kilometer of road length. In the western Cascades of Oregon road-induced gullies were associated with flood events with a 30- to 100-year recurrence interval (Wemple et al., 2001). If gullies are assumed to form in response to storms with a recurrence interval of 50 years, the mean annual sediment delivery rate from gullies would be 0.6 Mg km^{-1} yr^{-1}.

This value can be compared to the amount of sediment being produced and delivered from the road surface. The prediction equation for road surface erosion from native surface roads is:

$$SP_{ns} = -329 + 3.56\ (A*S) + 0.542\ EI_A + 0.389\ (A*G) \qquad (2.7)$$

where SP_{ns} is sediment production in kilograms per year, A*S is the product of road area and road slope (m^2), EI_A is annual erosivity (MJ mm ha^{-1} hr^{-1}), and A*G is the product of road area and a binary variable (G) with 1 representing a recently-graded road and 0 representing an ungraded road (Chapter 2). This equation was used to predict the amount of sediment being produced from each road segment that was connected by a stream crossing, gully or sediment plume. The calculations assumed a mean annual erosivity of 1360 MJ mm ha^{-1} hr^{-1} (Renard et al., 1997), that none of the roads had been recently graded, and that all of the sediment from a connected road segment was reaching the stream channel. The resulting sediment delivery rate for road surface erosion was 1.4 Mg km^{-1} yr^{-1}, or 2.3 times the estimated gully erosion rate of 0.6 Mg km^{-1} yr^{-1}.

The validity of this comparison depends on the assumptions regarding the storm recurrence interval for gully formation, the mean annual erosivity, the frequency of road maintenance activities, the percent of sediment delivered from the connected segment and the gully, and the accuracy of the sediment prediction model. Road-induced gully erosion may be a larger contributor of sediment to the channel network if gullies form during storms with a shorter recurrence interval. For example, the amount of sediment from gullies would double if gully erosion results from storms with a recurrence interval of 25 years rather than 50 years. The amount of sediment from road surfaces is sensitive to the annual erosivity and the presence or absence of grading. For example, assuming an EI_A of 2000 MJ mm ha^{-1} hr^{-1} would increase sediment delivery from road surfaces from 1.4 to 2.2 Mg km^{-1} yr^{-1}. If all roads are recently-graded, the sediment delivery from road surfaces would increase by 50% to 2.1 Mg km^{-1} yr^{-1}. The key point is that large amounts of sediment can be produced and delivered from road-induced gullies as well as road surface erosion.

3.5.4. *Connectivity*

The road survey showed that 16% percent of the road segments and 25% of the total road length was connected to the channel network. These values are low relative to most other studies. In southeastern Australia, 38% of the road length was connected to the streams in an area with similar Mediterranean climate (Croke and Mockler, 2001). In northwestern California 32% of the road segments were connected to the channel network (Raines, 1991). However, in the drier Front Range of Colorado, 18% of the total road length was connected to the channel network (Libohova, 2003).

An analysis of the data from these and other studies suggests that the percentage of unpaved roads that are connected to the stream network increases with mean annual precipitation and decreases with the presence of engineered road drainage structures such as waterbars, rolling dips, and relief culverts (Reid and Dunne, 1984; Raines, 1991; Wemple et al., 1996; Bowling and Lettenmaier, 2001; Croke and Mockler, 2001; Ziegler et al., 2000; Libohova, 2004; Sidle et al., 2004; A. Ziegler, personal comm., 2003). An empirical prediction equation using these two factors can explain 92% of the variability in road connectivity:

$$C = 12.9 + 0.016\ P + 39.5\ M \qquad (3.5)$$

where C is either the percent of road length or percent of road segments that are connected to the channel network, P is the mean annual precipitation (mm), and M is a binary variable with 0 representing roads with engineered drainage structures, and 1 representing roads without engineered drainage structures ($p<0.0001$) (Figure 3.13). Mean annual precipitation explains 41% of the variability in connectivity ($p=0.03$) for the entire dataset, and 84% of the variability in connectivity for roads with engineered drainage structures ($p=0.001$). The standard error of the estimate is 8.2%. To develop this equation it was assumed that the percent of connected segments was equivalent to the percent of the connected road length. Although this assumption is not strictly true because the longer segments are more likely to be connected, it was necessary in order to pool the data collected using each approach.

There are several reasons why mean annual precipitation is the dominant control on road-stream connectivity. Increasing precipitation tends to increase drainage density (Gregory, 1976; Montgomery and Dietrich, 1988), and an increase in drainage density will increase the number of stream crossings. An increase in precipitation also will increase the amount of road runoff, which will increase the number and length of road-induced gullies (Montgomery, 1994; Luce and Black, 1999; Croke and Mockler, 2001).

The binary variable reflects the ability of road drainage structures to disconnect road segments from the channel network. Frequent drainage structures reduce the amount of runoff available for gully initiation and the downslope transport of road-related sediment (Montgomery, 1994; Croke and Mockler, 2001). The careful placement of drainage structures also can help reduce the amount of road drainage that reaches the stream at stream crossings. The coefficient for the dummy variable in Eq. 3.5 indicates that engineered drainage structures will decrease the connectivity by about 40% relative to roads without engineered drainage structures.

3.5.5. Management Implications

The data in Figure 3.13 indicate that road connectivity is lower in the study area than in wetter areas such as the Pacific Northwest, but that sediment is being delivered to the streams from 25% of the road network. A study of 28 pool-riffle reaches in the study area found a positive correlation between estimated road sediment production and residual pool infilling (R^2=0.14; p=0.02) (MacDonald et al., 2003). Relatively small increases in fine sediment can adversely affect fish by decreasing the growth and survival of juvenile fish, and decreasing the availability of invertebrate prey species (Suttle et al., 2004). The response of juvenile fish and invertebrates to fine sediment loading is linear, suggesting that any increase in fine sediment will have a detrimental effect (Suttle et al., 2004).

The results of this study have important management implications for reducing road sediment delivery. First, most roads are connected at stream crossings, so the number of stream crossings should be minimized when designing and constructing unpaved roads. Second, the production and delivery of road sediment to stream crossings can be reduced by rocking the approaches to stream crossings (Chapter 2) and minimizing the length of the road segments that drain directly to the crossing (Eq. 2.7).

Third, the size and length of sediment plumes and gullies can be minimized by reducing road runoff and reducing traffic. This will reduce the amount of sediment that is delivered and the amount of sediment that is generated by gully erosion. The amount of runoff from a road segment can be reduced by shortening the road segment length, outsloping the road surface, and minimizing cutslope heights on shallow soils. Gully initiation below road segments can be minimized by avoiding sensitive sites as identified by hillslope gradient, soil depth, and hillslope condition. Gully initiation also can be minimized by improved road designs in terms of decreasing the spacing of drainage structures, changing road drainage type, and minimizing cutslope height. The road drainage guidelines in Figure 3.12 can be used to minimize the risk of gullying below a road drainage outlet.

Fourth, sediment delivery from gully erosion can be minimized by improved road drainage. Gully volumes and travel distance can be reduced by shortening segment lengths and outsloping the road surface. Managers should avoid insloping road segments on erosive soils and steeper hillslopes. Finally, 95% of road segments transported sediment less than 42 m from the drainage outlet. If roads can be placed or relocated at least 40 m from stream channels, sediment delivery via sediment plumes and gullies should be minimized.

3.6. Conclusions

This study measured the extent to which unpaved forest roads in the Sierra Nevada of California are connected to the stream channel network. A detailed survey along 20 km of unpaved roads identified 285 road segments. Sixteen percent of the 285 road segments and 25% of the road network length were connected to the channel network. Fifty-nine percent of the connected road segments were due to stream crossings, while 35% were connected by road-induced gullies. Only 6% of road segments were connected via sediment plumes.

The mean gully length was 37 m. or roughly 3 times larger than the mean sediment plume length, and the longest gully was 95 m. Multivariate analysis indicated that the length of sediment plumes and gullies below road drainage outlets was controlled by the presence or absence of gullies, soil erodibility, traffic level, and road segment length (R^2=0.39; p<0.0001). Road-induced gullies were more frequent on insloped roads drained by relief culverts, longer road segments on steeper slopes, and drainage outlets discharging onto hillslopes with relatively low surface roughness or low infiltration due to compaction. A logistic regression model using these factors had a 90% success rate in distinguishing between gullied and ungullied segments. Gully volume was significantly related to the product of road segment length and hillslope gradient, soil erodibility, and road drainage type (R^2=0.60; p<0.0001). Gully volumes were significantly higher below relief culverts than for waterbars or rolling dips. The amount of sediment delivered from road-induced gully erosion was 43% of the amount of sediment delivered from road surfaces. Road sediment delivery can be minimized by reducing the number of stream crossings, outsloping and frequently draining roads on erosive soils and steep hillslopes, and placing new roads further from stream channels.

An analysis of data from 10 studies shows that road-stream connectivity is strongly controlled by mean annual precipitation and the presence or absence of engineered drainage structures (R^2=0.92; p<0.0001). The absence of engineered drainage structures will increase connectivity by approximately 40%. The findings of this and other studies indicate that maintaining and improving road drainage is an effective means to reduce road sediment delivery.

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3.7. TABLES AND FIGURES

Connectivity class	Geomorphic criteria	Potential for sediment delivery
1	No signs of gullying or sediment transport below drainage outlet	Low
2	Gullies or sediment plumes <20 m in length	Low/moderate
3	Gullies or sediment plumes >20 m in length, but more than 10 m from stream channel	Moderate/high
4	Gullies or sediment plumes to within 10 m of a stream channel	High

Table 3.1. Road connectivity classes and their estimated potential for sediment delivery.

Dependent variables	Independent variables
Connectivity class (CC)	Road segment gradient (S)
Geomorphic feature (gully or sediment plume)	Road surface area (A)
Sediment travel distance below outlet (m)	Road length (L)
Gully presence or absence	Hillslope gradient (SH)
Gully volume	Cutslope height
	Soil series
	Lithology
	Soil depth
	Soil erodibility (K factor)
	Road drainage type (outsloped, bermed, or insloped with relief culvert)
	Geomorphic feature (gully or sediment plume)
	Hillslope condition

Table 3.2. List of dependent and independent variables used in pairwise comparisons, multiple regression, and logistic regression.

Variable	Mean	Range Minimum	Range Maximum	Std. dev.
Segment length (m)	76	7	401	64
Segment area (m^2)	563	43	5260	587
Segment gradient ($m\ m^{-1}$)	0.06	0	0.17	0.03
Cutslope height (m)	1.9	0.2	8.0	1.1
Hillslope gradient ($m\ m^{-1}$)	0.26	0.01	0.57	0.11
K factor ($t\ ha\ h\ ha^{-1}\ MJ^{-1}\ mm^{-1}$)	0.017	0.013	0.032	0.017
Soil depth (m)	1.0	0.30	1.6	0.40

Table 3.3. Mean, range, and standard deviation of the independent variables used to characterize each segment.

Connectivity class	Number of segments	Percent of total segments	Road length (km)	Percent of total length
1	138	48.4	8.11	37.7
2	81	28.4	5.62	26.1
3	20	7.0	2.25	10.5
4	46	16.2	5.55	25.7
Total:	285	100	21.53	100

Table 3.4. Number of road segments and road length by connectivity class.

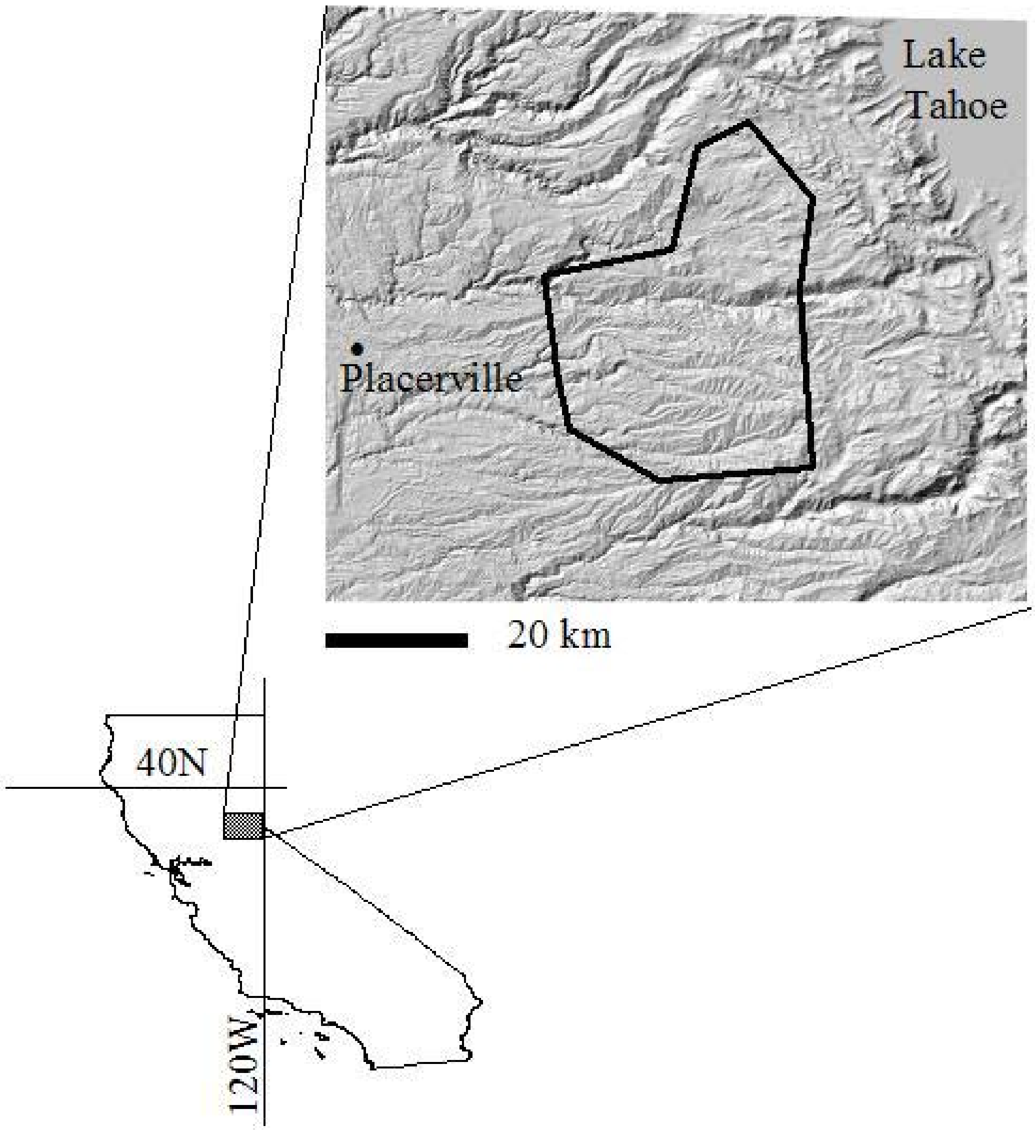

Figure 3.1. Map of the study area.

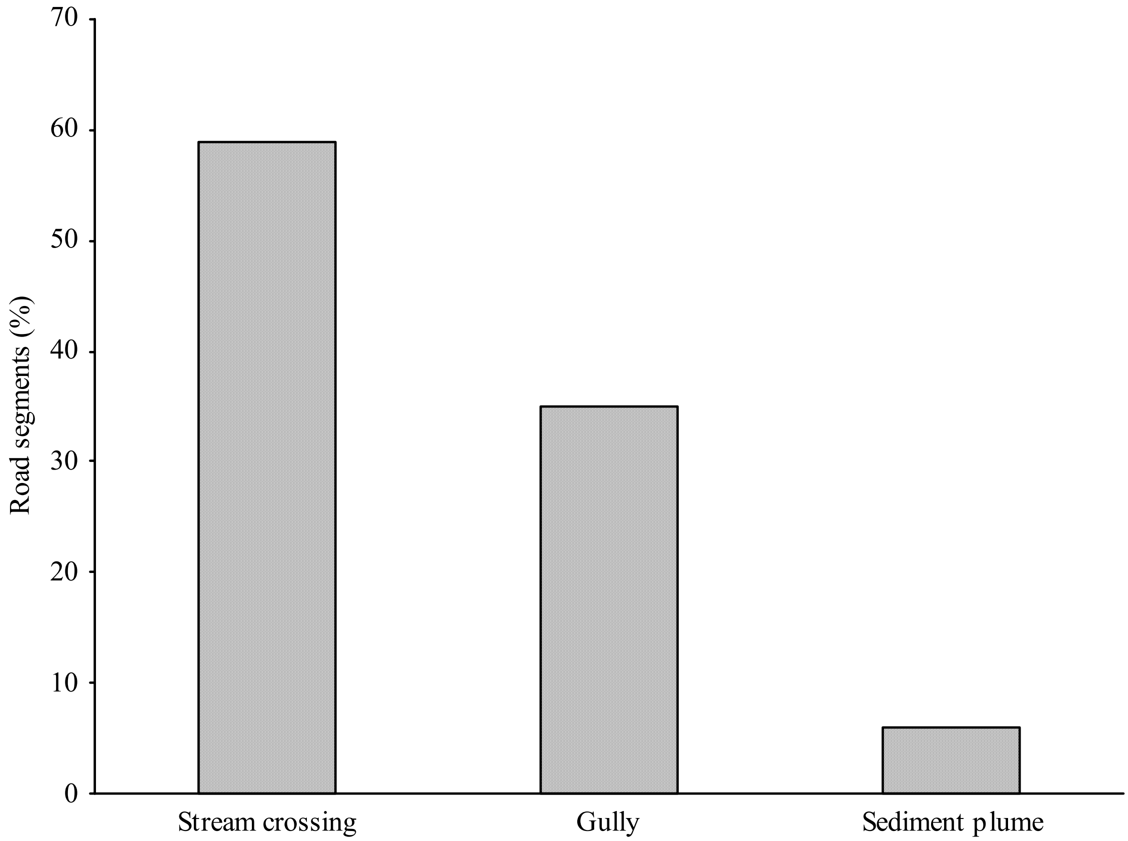

Figure 3.2. Percent of road segments connected to the channel network by causal mechanism (n=46).

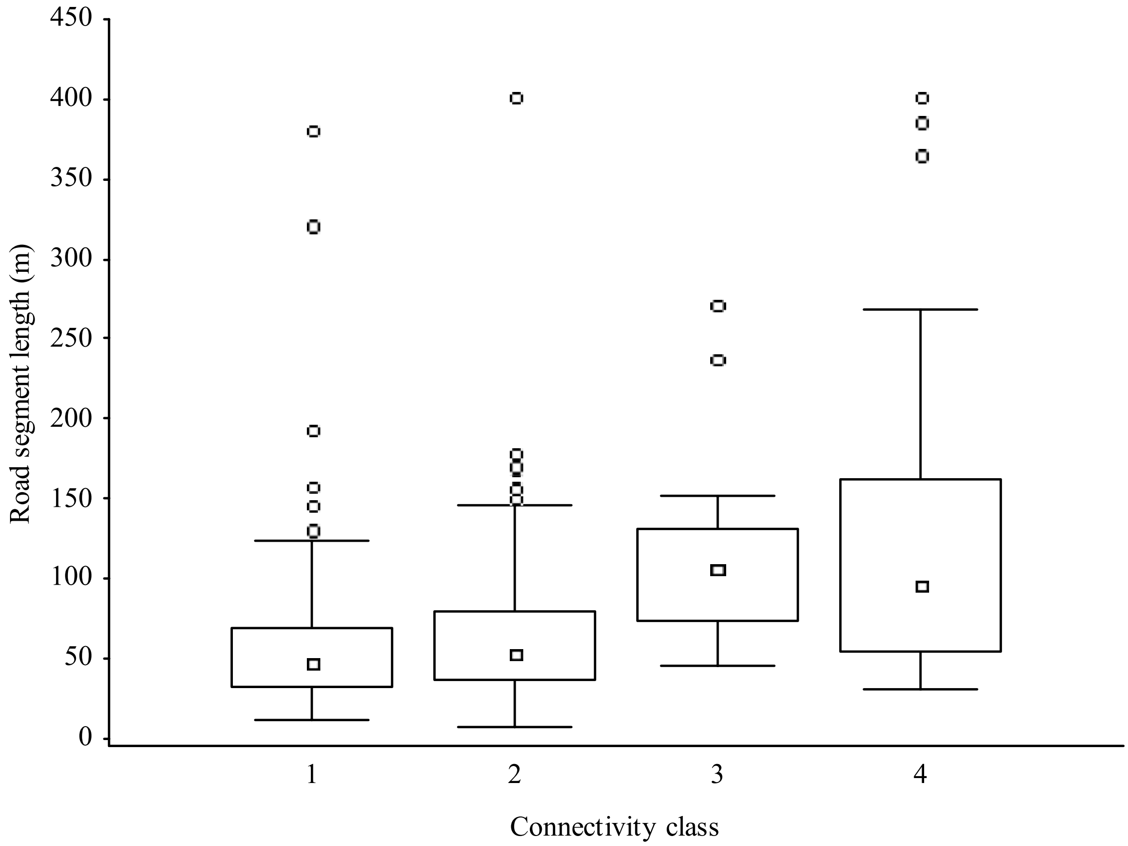

Figure 3.3. Road segment length by connectivity class. The small squares are the median segment length, the boxes indicate the 25th and 75th percentiles, the bars show the 95% confidence interval, and the open circles represent outliers.

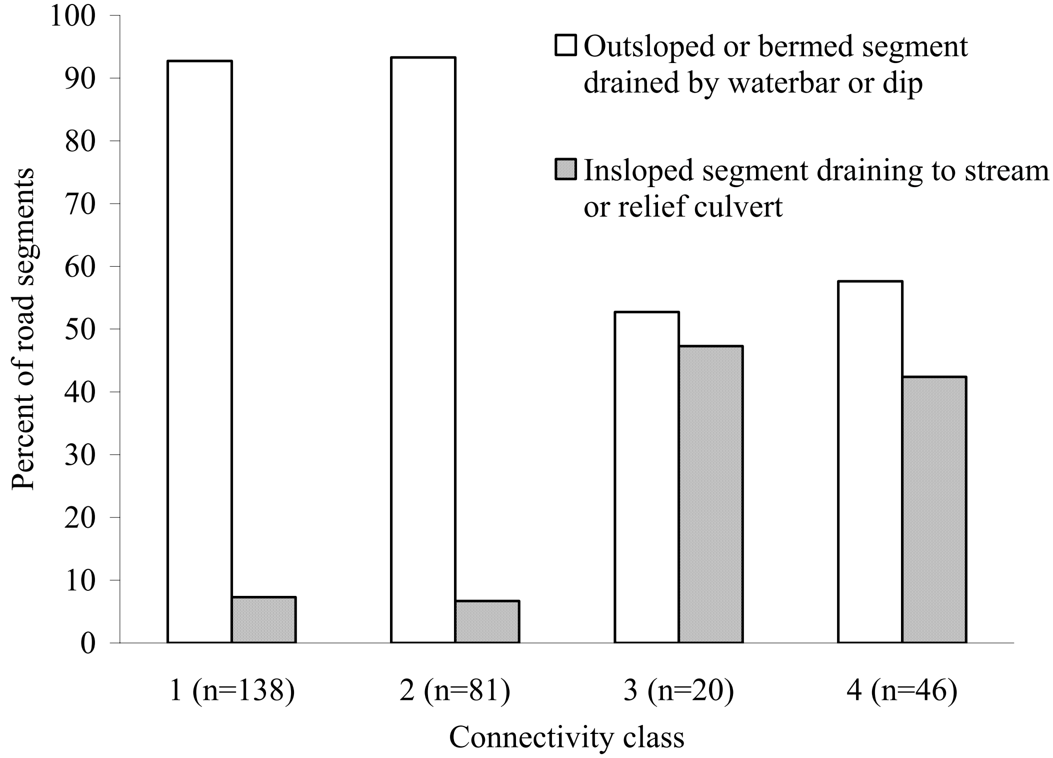

Figure 3.4. Percent of road segments by road drainage type for each connectivity class.

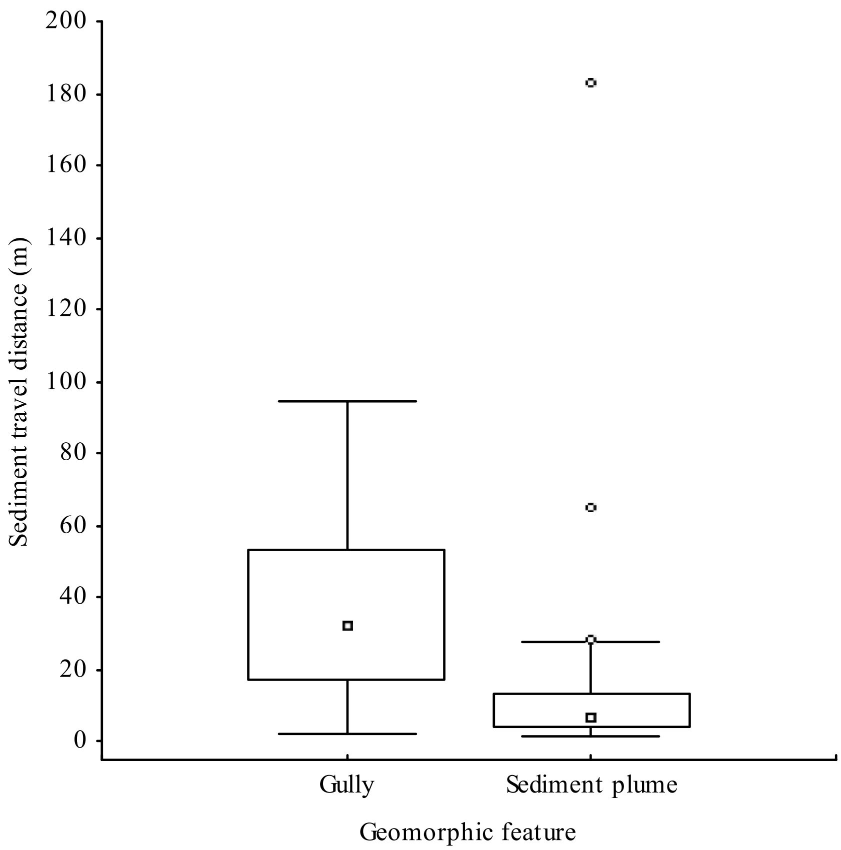

Figure 3.5. Lengths of gullies and sediment plumes for the segments classified as CC2, CC3, and CC4. The small squares are the median length, the boxes indicate the 25th and 75th percentiles, the bars show the 95% confidence interval, and the open circles represent outliers.

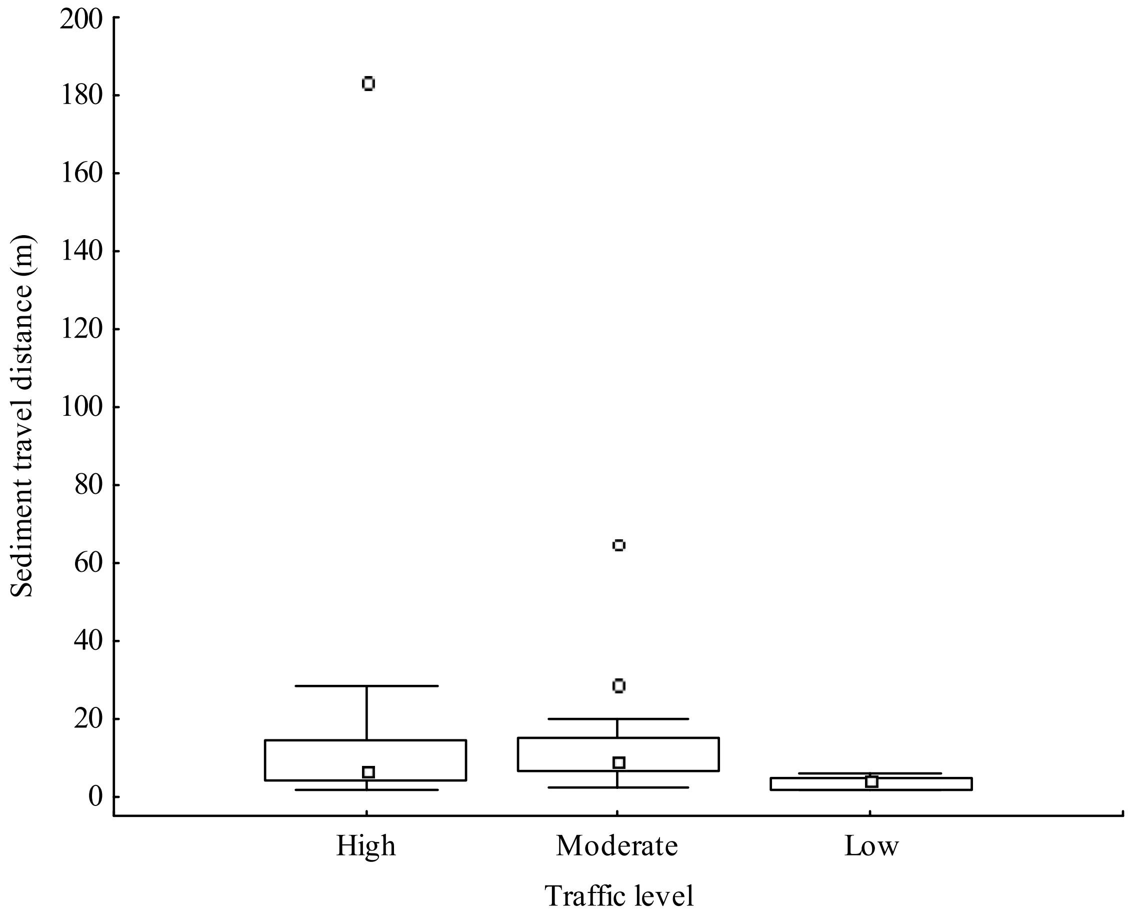

Figure 3.6. Lengths of sediment plumes by traffic level. The small squares are the median segment length, the boxes indicate the 25th and 75th percentiles, the bars show the 95% confidence interval, and the open circles represent outliers.

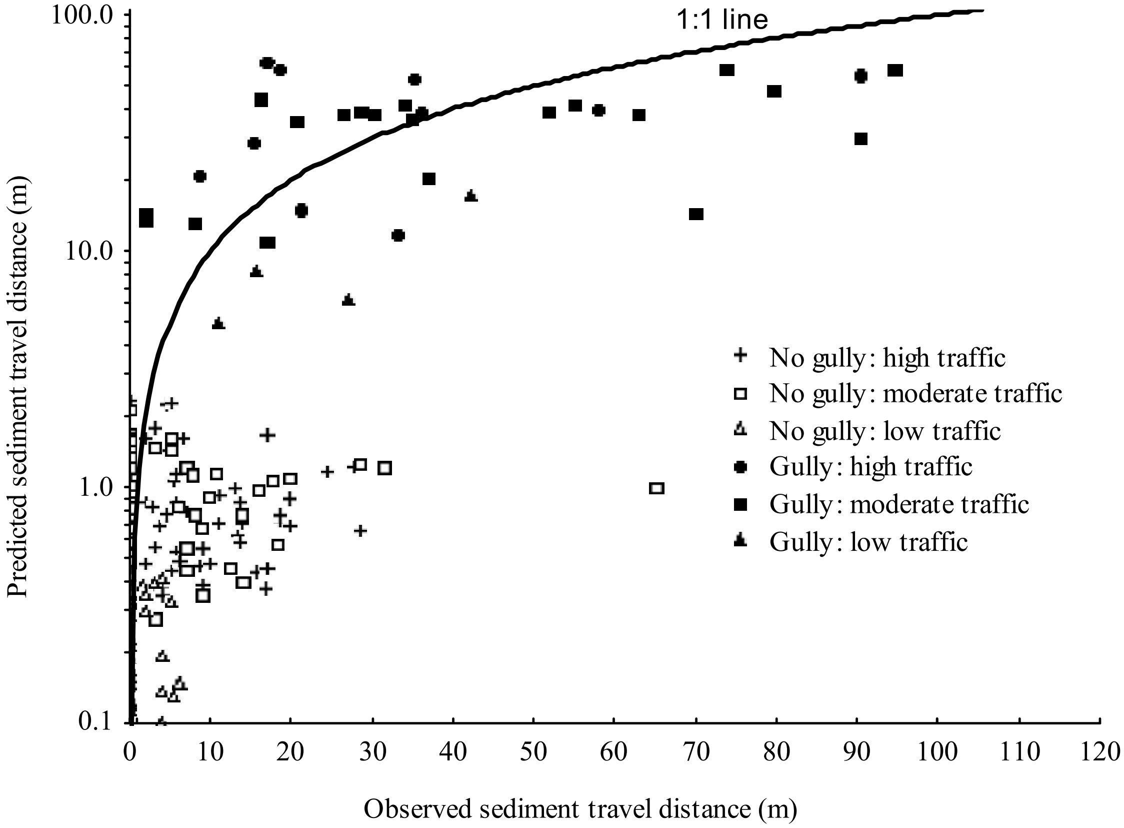

Figure 3.7. Predicted gully and plume lengths versus observed values by geomorphic feature and traffic class.

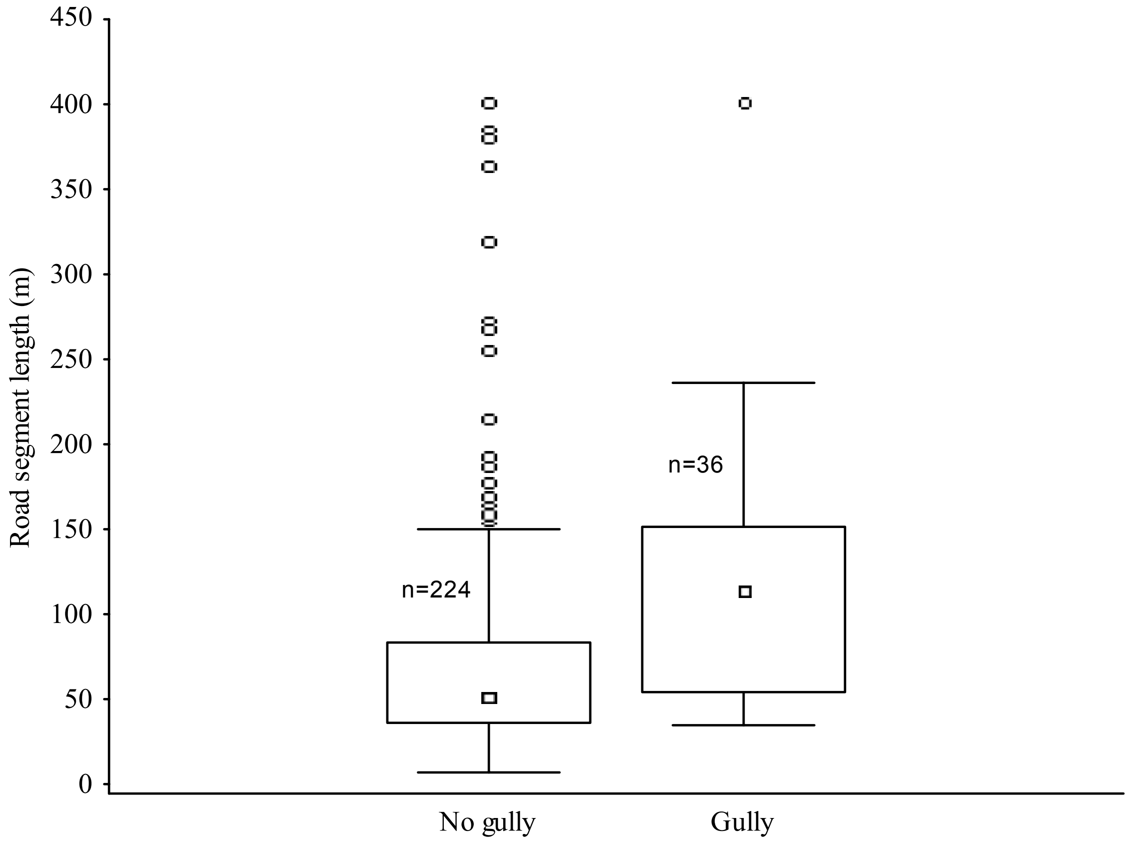

Figure 3.8. Road segment length for outlets with and without gullies. The small squares represent the median road segment length, the boxes indicate the 25th and 75th percentiles, error bars represent the 95% confidence intervals, and the open circles represent outliers.

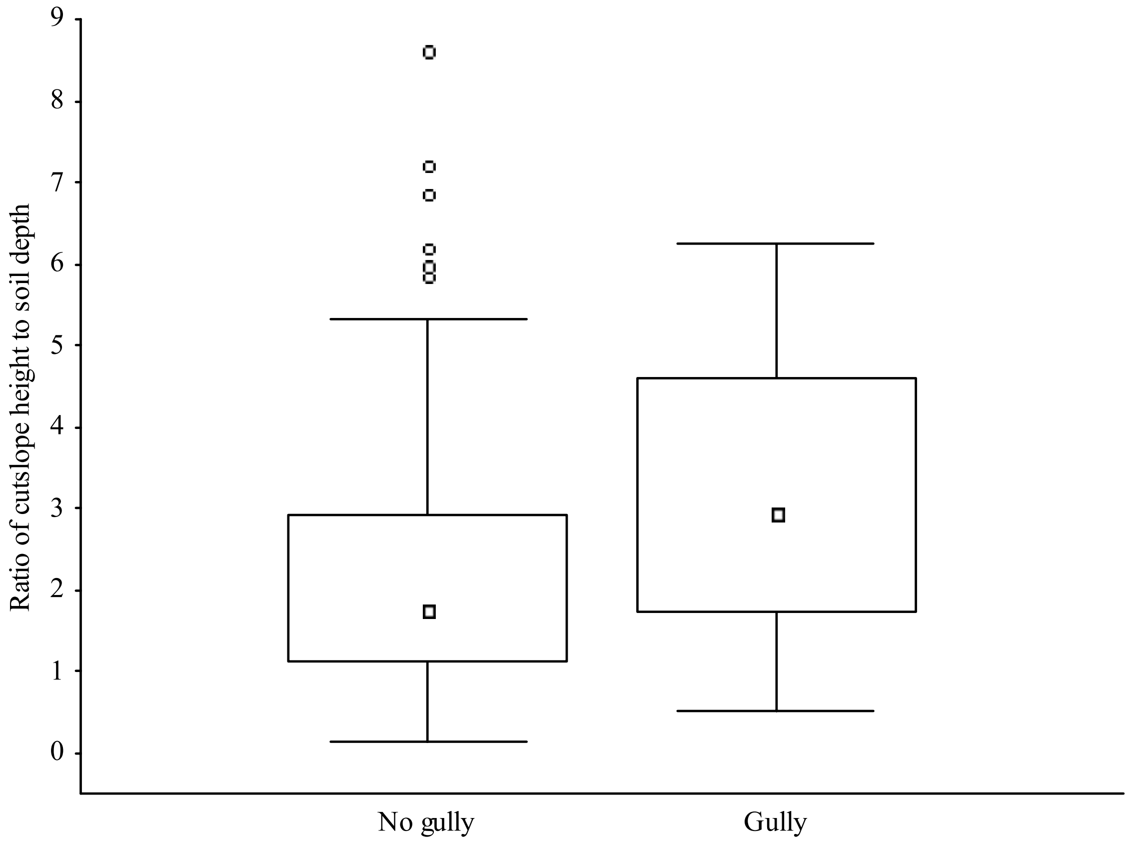

Figure 3.9. Ratio of cutslope height to soil depth for segments with and without gullies below the drainage outlet. The small squares represent the median ratio, the boxes indicate the 25th and 75th percentiles, error bars represent the 95% confidence intervals, and the open circles represent outliers.

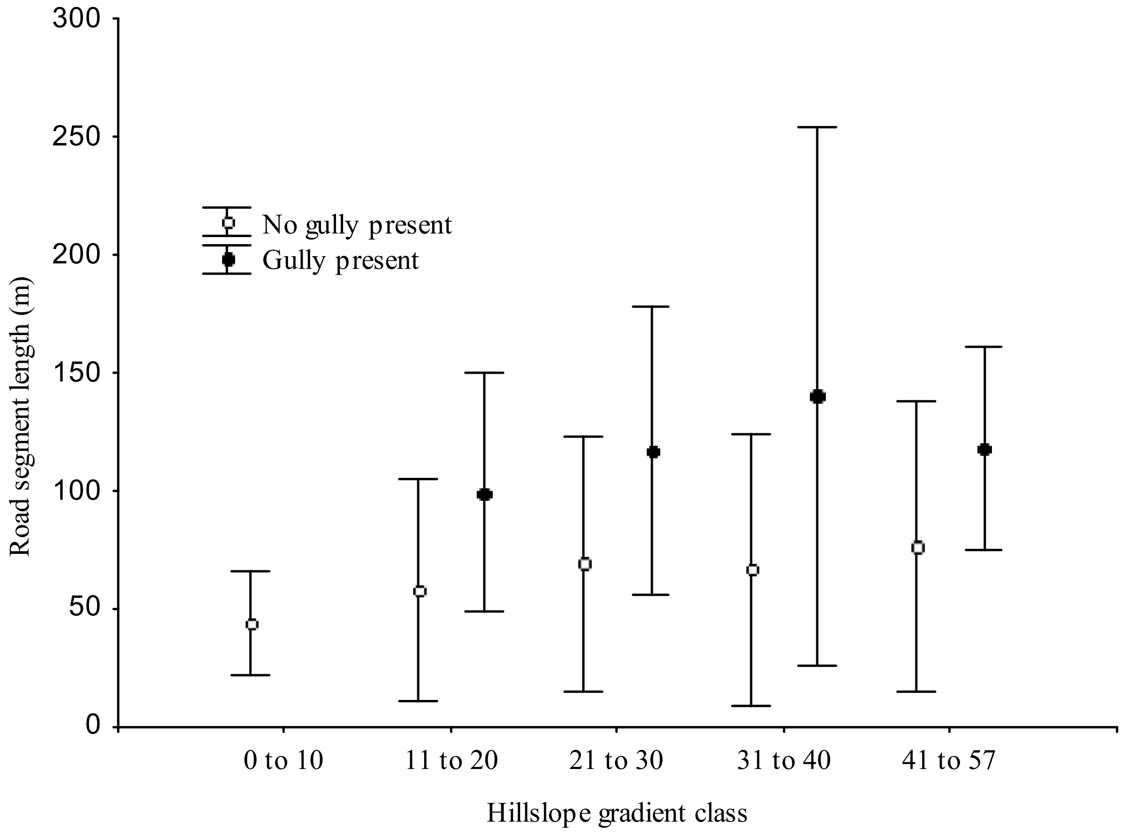

Figure 3.10. Mean road segment length for gullied and ungullied road segments by hillslope gradient class. Bars represent one standard deviation.

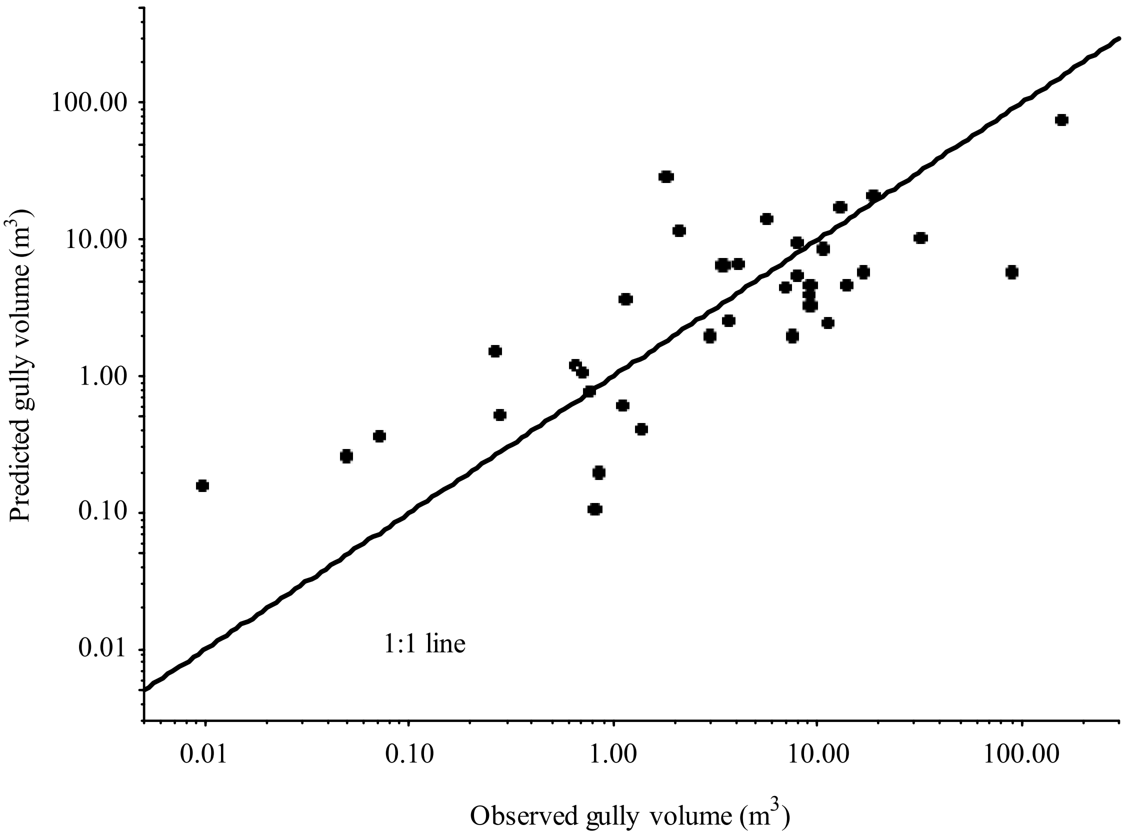

Figure 3.11. Predicted versus observed gully volumes.

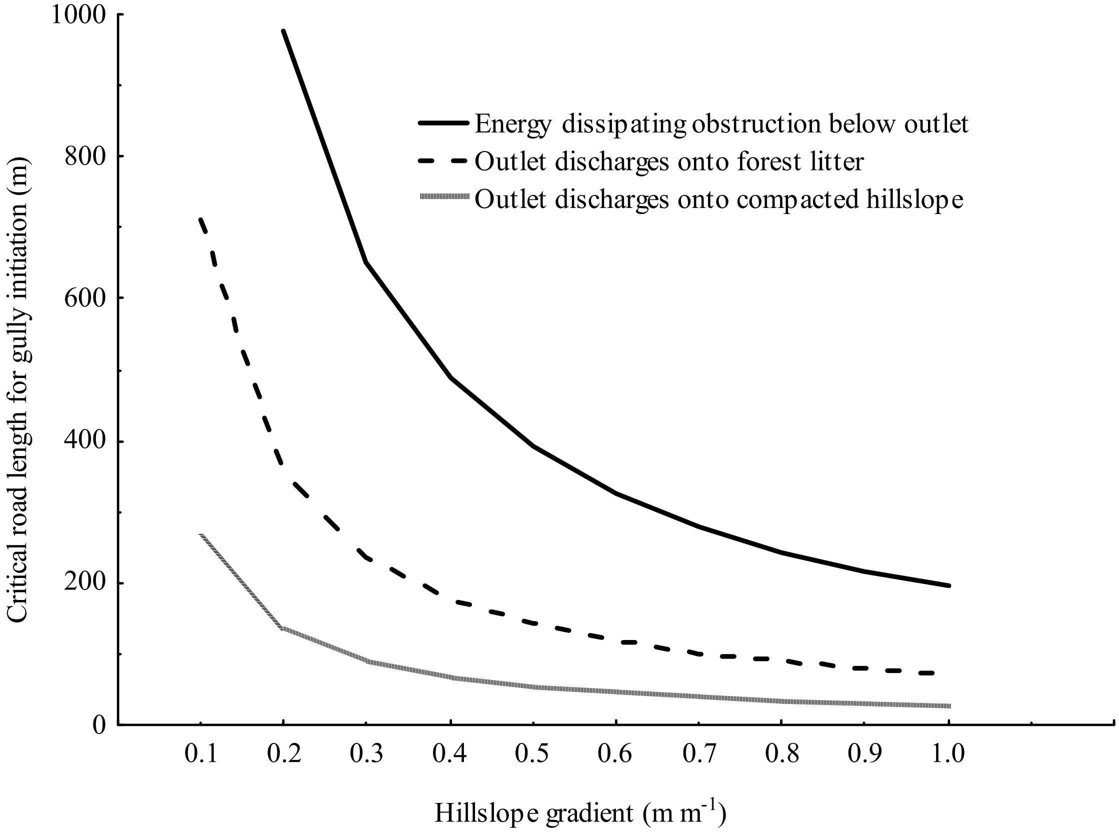

Figure 3.12a. Predicted road segment length thresholds (L_t) for avoiding gully initiation below outsloped roads drained by waterbars and rolling dips. Each curve represents a 50% probability of gullying for a different hillslope condition across a range of hillslope gradients.

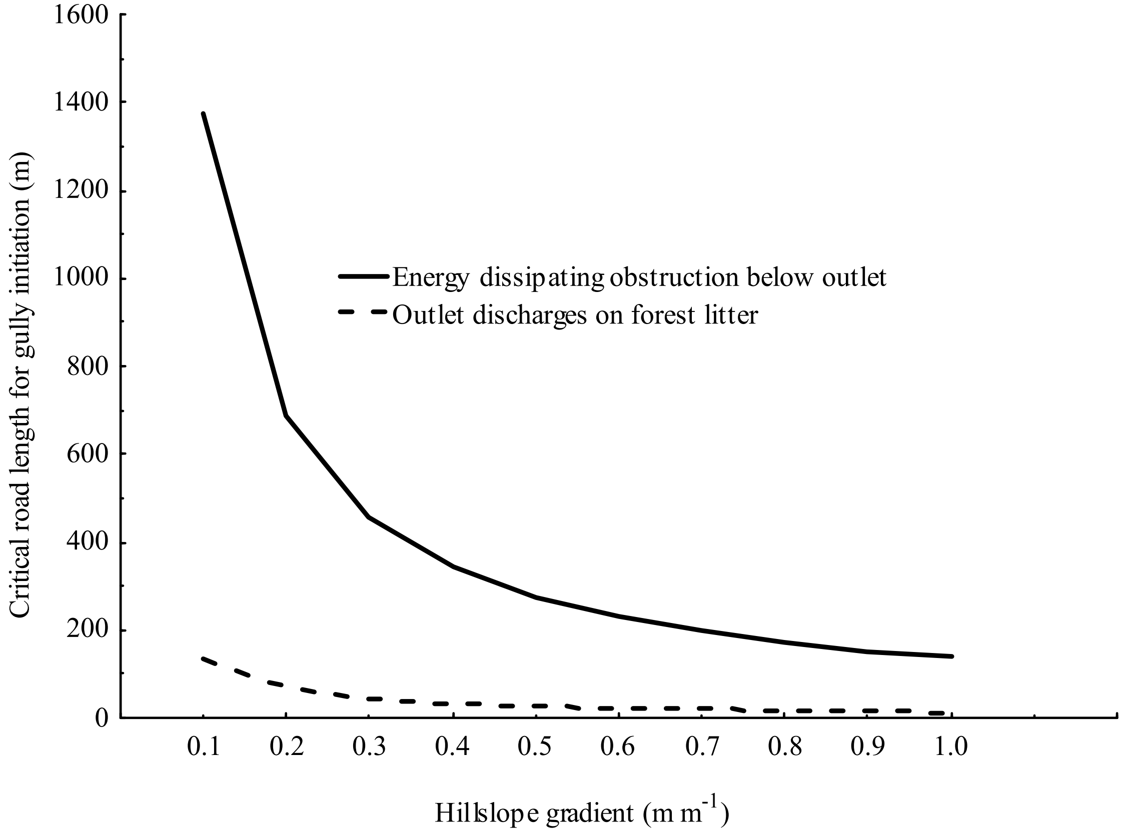

Figure 3.12b. Predicted road segment length thresholds (L_t) for avoiding gully initiation below insloped roads drained by relief culverts. The two curves represent a 50% probability of gullying for two different hillslope conditions across a range of hillslope gradients. No curve is shown for compacted hillslopes as all relief culverts that discharge onto compacted hillslopes are predicted to have gullies.

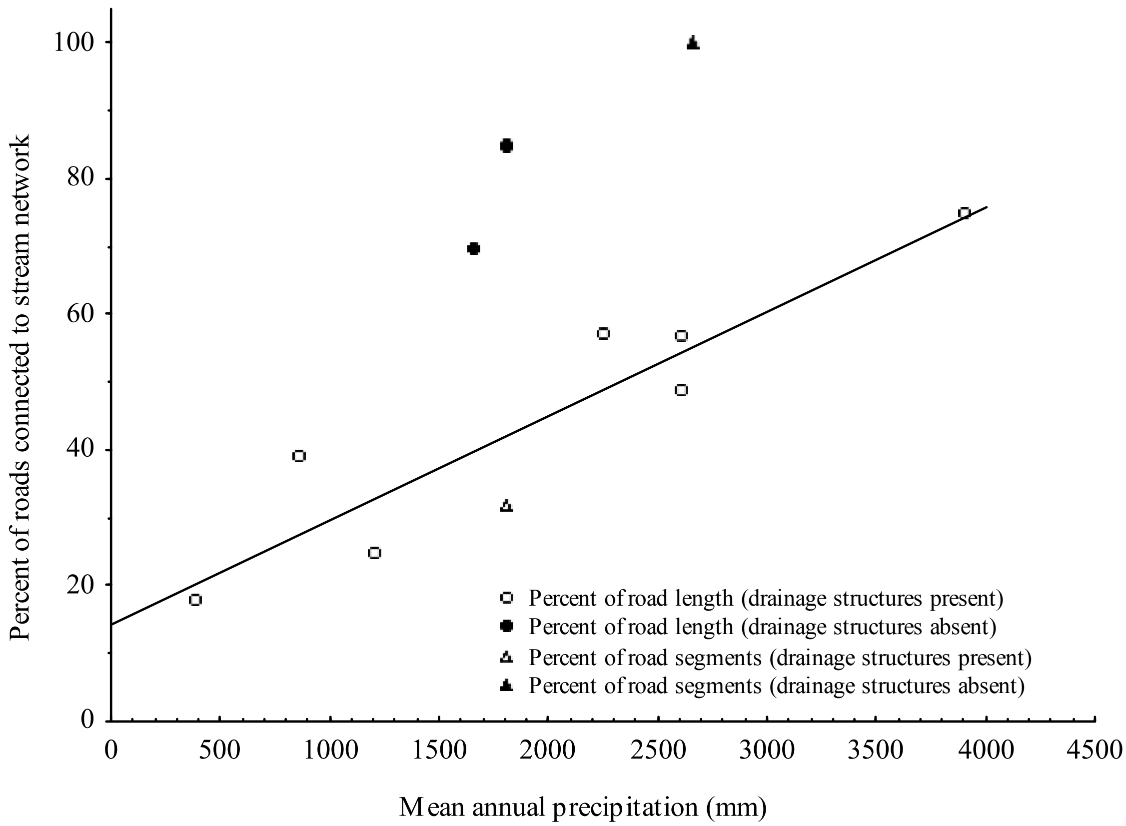

Figure 3.13. Percent of roads connected to the stream network versus mean annual precipitation for roads with and without engineered drainage structures. Regression line is for roads with engineered drainage structures.

4.0. Conclusions

The two studies provide a unique and quantitative understanding of sediment production and sediment delivery from unpaved roads in the Sierra Nevada of California. Sediment production rates varied greatly between years and between road segments. Most of the interannual variability in sediment production rates can be attributed to differences in the magnitude and type of precipitation, and their resulting effect on rainsplash and hydraulic erosion. The first wet season had near-normal precipitation and much of the precipitation in the lower portions of the study area fell as rain rather than snow. In the second and third wet seasons precipitation was below normal and tended to fall as snow. The resultant differences in rainfall erosivity, persistence of snow cover, and road runoff rates meant that unit area erosion rates were 3-4 times higher in the first wet season than in either of the two following wet seasons. On midslope roads with cutslopes, normalized sediment production increased as upslope soil depth decreased, and this is attributed to the increase in intercepted subsurface stormflow (ISSF).

Twenty-five percent of the surveyed road length was connected to the channel network. Stream crossings accounted for 59% of the connected road segments, and road-induced gullying accounted for another 35% of the connected road segments. The travel distance of sediment below road drainage outlets was controlled by soil erodibility, road segment length, traffic level, and the presence or absence of gullies (R^2=0.39). The likelihood of a gully below a road segment increased with longer road segment lengths on steeper slopes, with shallower soils, and road drainage designs that concentrate rather than disperse runoff. A logistic regression model using these factors had a 90% success rate in distinguishing between gullied and ungullied segments. Gully volume was significantly related to the product of road segment length and hillslope gradient, soil erodibility, and road drainage type (R^2=0.60). Gully volumes were significantly higher below relief culverts than below waterbars or rolling dips.

Both studies show that road sediment production and some aspects of sediment delivery are strongly controlled by road area (A) or road length (L), and the interaction of A or L with road gradient (S) or hillslope gradient (S_H). A*S is a surrogate for the sediment transport capacity of runoff on the road surface, and $L*S_H$ is a surrogate for the sediment transport capacity of road runoff below a drainage outlet. Higher $L*S_H$ values increase the likelihood that a gully will form below a drainage outlet and deliver sediment to the channel network. Frequent road drainage serves to reduce both A*S and $L*S_H$. An analysis of existing data on road-to-stream connectivity suggests that the absence of engineered road drainage structures increases road-stream connectivity by 40%.

Both studies indicate that the interception of subsurface stormflow (ISSF) can increase both road sediment production and sediment delivery. Variables such as soil depth and the ratio of cutslope height to soil depth have the potential to explain some of the variability in road sediment production rates and gully initiation. However, the role of ISSF is difficult to include in empirical predictive equations because of the tremendous spatial and temporal variability in the amount and interception of subsurface stormflow.

Overall, these studies show that road sediment production is best mitigated by rocking native surface roads, decreasing sediment transport capacity by improving and maintaining drainage, and avoiding unusual soil features that increase road surface and ditch runoff. Road sediment delivery can be minimized primarily through reducing the number of stream crossings, reducing the length of road segments that drain to stream crossings, rocking the approaches to stream crossings, preventing gully formation below road drainage outlets, and placing new roads further from stream channels. The results of these studies can help managers reduce road sediment production and delivery, and thereby reduce the adverse impacts of unpaved forest roads on aquatic resources.

Erosion and Channel Adjustments Following Forest Road Decommissioning, Six Rivers National Forest

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Although watershed restoration through road decommissioning has become common in the last decade, few studies have attempted to investigate the success or failure of such projects. This study examines 117 km (73 miles) of decommissioned roads, including 262 stream crossings, on the Six Rivers National Forest, northwestern California, to quantify erosion and identify failure mechanisms and potential areas for improvement. Although most crossings had experienced some adjustment, erosion was generally minor. The average amount of erosion for stream crossings was 21 m^3 (28 yd^3), which represents 4.5% of the amount of fill excavated. Of this volume, 40% of the erosion was due to channel adjustment and 60% was due to bank failures. Erosion from the roadbed between crossings was very small and was observed only in areas of highly unstable geology. The amount of erosion appears well correlated with the timing and intensity of storm events. Large storm events occurring the first winter after decommissioning produced elevated erosion levels. After several dry winters, erosion was very minor, even from large storm events.

Keywords: *roads, decommissioning, obliteration, erosion*

Introduction

It has long been recognized that roads, particularly roads in steep, mountainous terrain, can have significant impacts on water quality and aquatic ecosystems by: accelerating erosion and sediment loading, altering channel morphology, and changing the runoff characteristics of watersheds (Furniss et al. 1991). Where forest roads are located in steep terrain, mass soil movement is a common mechanism of erosion and sediment delivery (Lyons and Beschta 1983). Also common are road-stream crossing failures that occur when culverts fail to pass wood, sediment or storm discharge. The plugging of culverts may result in the loss of the roadbed at the stream crossing or the diversion of the stream offsite, both of which may generate large erosional features and sedimentation of adjacent water bodies. Road cuts can also intercept groundwater and reroute subsurface water into streams. This increase in stream discharge may result in channel enlargement including downcutting and bank erosion.

On Six Rivers National Forest, northwestern California, roads are the leading source of management-related sediment inputs, predominantly associated with mass wasting features such as shallow debris slides and debris torrents. The majority of road-related erosion and sediment delivery are associated with large storm events that trigger culvert failures, stream diversions, and mass wasting such as debris slides and smaller slumps within the roadbed. With declining road maintenance funding, the risk of road failures and elevated sediment delivery is increasing, particularly in the event of large storms.

In an effort to reduce erosion and sediment delivery associated with forest roads, a road-decommissioning program was initiated in the early 1990s on Six Rivers National Forest. Over the past decade, the forest has decommissioned approximately 341 km (212 mi) of forest roads. Road decommissioning efforts target abandoned and low-use roads with high erosion and sedimentation risks. The focus of road decommissioning efforts has been on improving water quality and aquatic ecosystems through reducing sediment introduced to streams, and on reducing the risk of future sediment delivery from roads in the event of a large storm.

The primary road decommissioning treatments on the Six Rivers has been the removal of culverts and the associated fill in stream crossings, and recontouring the stream crossing to as close to the original channel morphology as possible or practical. Roads were not completely obliterated (i.e., fully recontoured) as is typically done in the Redwood National Park restoration program (Madej 2001), but rather were placed in a free-draining condition. Where areas of instability were evident, road fill between stream crossings was recontoured, but in areas where there were no signs of instability, the roadbed was

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.

left in place. Generally speaking, the upper to mid slope location of the majority of forest roads on Six Rivers National Forest and the geology in many of the areas was such that leaving the majority of the road prism intact was seen as a low risk and a means to cost-effectively decommission more miles of road. In addition to culvert removal and occasional outsloping of the roadbed, installation of water bars or rolling dips was also employed to make the road free-draining. In some cases, the roadbed was ripped or sub-soiled, or trees were planted to speed up revegetation. In all cases, the ultimate objective of the road decommissioning treatment was to make the road free-draining, maintenance-free, and hydrologically benign relative to future erosion and sedimentation risk.

Because of the relatively new emphasis on road decommissioning in recent years on national forest lands and on private timber lands, little attention has yet been given to understanding the extent of erosion and sedimentation that occurs after a road has been decommissioned. There is a general recognition that there will be short-term effects associated with road decommissioning. These short-term effects are considered small given the long-term gain in reducing the larger sedimentation risk if more roads were to fail during large storm events. However, few studies have quantified the sediment lost due to post-decommissioning erosion and channel adjustments. A recent post-treatment decommissioning study was conducted in Redwood National Park (Madej 2001). Madej found that on stream crossing sites, post-treatment sedimentation was small and the majority of the post-treatment erosion and sedimentation were attributable to treated roadbeds. Regardless of treatment, post-project erosion and sedimentation were low when compared to untreated sites. For the period 1980 to 1997, an average of 50 m^3 (66 yd^3) of sediment delivery per stream crossing occurred (Madej 2001). Klein (2003) conducted a post-treatment erosion and turbidity monitoring study on decommissioned roads in the Mattole River watershed in northern California. Klein reported an average of 11 m^3 (15 yd^3) of sediment delivery associated with restored stream crossings. During the first winter after treatment, erosion and elevated turbidity within the restored stream crossings was common but the erosional responses diminished considerably over the winter sampling period. Dunkley et al. (2004) assessed the effectiveness of road deactivation (‘decommissioning’) techniques in reducing the incidence of landslide initiations but did not assess the volumes of erosion and sedimentation associated with landslides. On the Clearwater National Forest in Idaho, a study was conducted to assess the short-term total suspended-solid concentrations resulting from stream crossing obliteration (Brown 2002). The Clearwater study determined that turbidity and suspended sediments increased during stream crossing restoration and that total sedimentation could be reduced if sediment traps were installed. No analysis was conducted to determine how long the turbidity and suspended solids lasted after the restoration work ceased.

Project Objectives

In light of the extensive road decommissioning that had occurred over the past ten years on the Six Rivers National Forest and the high likelihood that more roads will continue to be decommissioned in the near future, assessing road decommissioning projects was identified as a critical step that would quantify the extent of post-treatment road decommissioning erosion and sedimentation, and improve future road decommissioning projects through incorporating lessons learned. The objectives of the post-treatment road decommissioning assessment were to:

1. Quantify the amount of fill removed and the amount, types and locations of post-project erosion. Assess the effectiveness of treatments in reducing sediment inputs
2. Identify successful or unsuccessful treatment techniques (e.g., ripping, leaving road prism intact, and so on)
3. Identify criteria that would facilitate predicting when future projects may need special treatments (e.g., stream power, slope gradient, slope position)
4. Identify any limiting factors contributing to less than fully successful road decommissioning

These objectives are aimed at assessing the relative risk of implementing road-decommissioning projects and quantifying the nature of the short-term post-treatment erosion risk. This information is particularly relevant given the extent of threatened and endangered anadromous fisheries within the Six Rivers National Forest and the potential for road decommissioning projects to result in adverse effects on these fisheries and water quality.

Project Area Description

Six Rivers National Forest is located in northern California within the Klamath Mountains and California Coast Ranges Geomorphic Provinces. The Klamath Mountains Province has been uplifted relatively rapidly and is deeply dissected, contributing to the ruggedness of the terrain. Both rugged and gentle terrain is found in the Coast Ranges, and drainage systems are generally smaller in area. Topography throughout the Forest is typically steep (slopes ranging from 30 to 80 percent), well dissected, and forested, although there are extensive areas of grassland in the south and barren areas of ultramafic rock in the north. The large number of older, deep-seated landslides and younger, shallow landslides is due to rapid downcutting by streams through weak bedrock and overlying surficial materials in response to the rapid uplift, as well as the active tectonic environment. Over 90 percent of the high and extreme landslide hazard areas occur adjacent to stream channels on slopes steeper than 65 percent which typically occur in inner gorges of stream channels, recently active landslides, toe zones of deep-seated landslides, and fault zones of weakened bedrock. In general, roads located on slopes steeper than 50% have a greater risk of failure and sediment delivery (USDA 1999, Appendix 2). Roads located within these areas pose a higher risk of erosion and sedimentation during large storm events.

The geology of Six Rivers National Forest is complex and can be categorized in four groups: (1) sedimentary and metasedimentary rocks; (2) metaigneous [or metavolcanic] rocks; (3) ultramafic intrusive rocks; and (4) igneous rocks. The sediments and metasediments include graywacke, shale, schist, and chert of the Franciscan Complex, slate, phyllite and sandstone of the Galice Formation, and pre-Cretaceous metavolcaniclastic rocks of the Rattlesnake Creek and Hayfork Terranes. The metavolcanic rocks are mainly associated with the Galice and Rogue formations or part of the pre-Cretaceous terranes, or occur as isolated blocks in the Franciscan Complex. The ultramafic rocks are predominantly serpentinite, dunite and serpentinized peridotite. The igneous rocks range from diorite and quartz diorite to gabbro, and most occur as relatively small intrusive bodies or inclusions in mélange.

Six Rivers National Forest is characterized by a Mediterranean climate with cool, moist winters and warm dry summers. Precipitation is moderately heavy over most of the forest. It ranges from around 1,270 mm (50 inches) on the middle to southern portion of the forest and up to 3,048 mm (120 inches) on the more northerly portions of the forest. Roughly 90 percent of the total precipitation falls in the six-month period between November and April. Stream runoff is mainly from rainfall, and snowmelt makes up only a minor part of total runoff. The largest storm runoff is usually associated with rain-on-snow events. The relatively high rainfall in combination with steep terrain poses challenges for maintaining forest roads, particularly during large storm events.

To examine the effect of rainfall on decommissioned roads, monthly rainfall from each ranger district was analyzed. Information about individual storm intensities would be more relevant, but this data is not available. Instead, the highest monthly rainfall of the year was used to create an annual maximum series. Rainfall with a return period of 5 years or greater was used to designate that year as a “wet” year. Rainfall return periods were similar to flood discharge return periods for the few stream gages that exist nearby.

Since the decommissioning program began, three large storm events have occurred. Intensity varied from district to district (Figure 1), but most decommissioning sites have experienced at least an 18-year storm. The largest event was the “1997 New Year’s Day Flood” which had a return period between 22 and 52 years (Figure 1). Such heavy rainfall should reveal decommissioned sites that are erosion prone or inadequately restored.

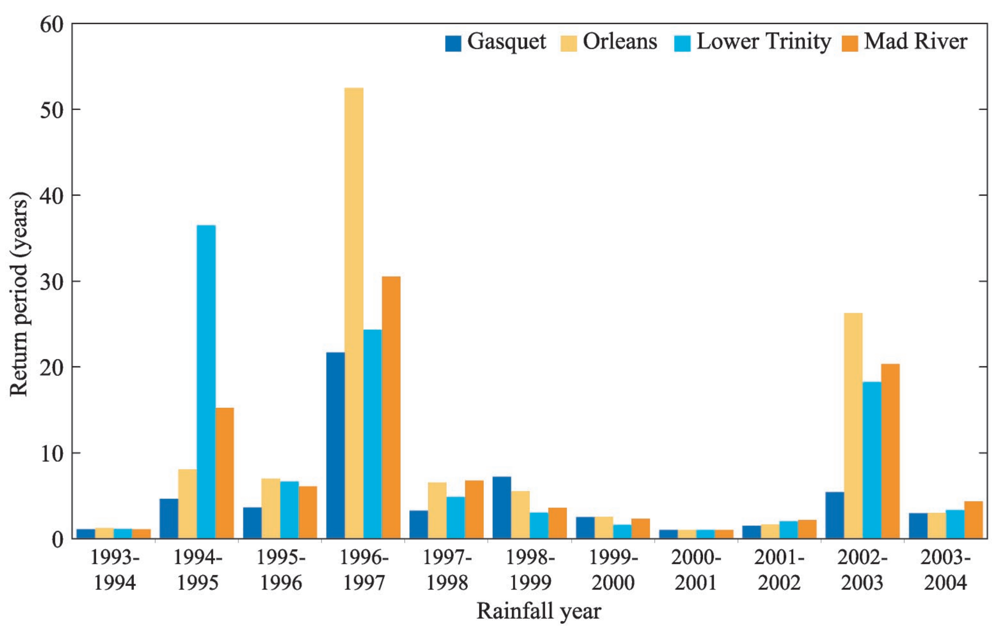


Figure 1: Recent monthly rainfall return periods for the Gasquet, Orleans, Lower Trinity, and Mad River Ranger Districts, coastal northern California ,from 1993 to 2004.

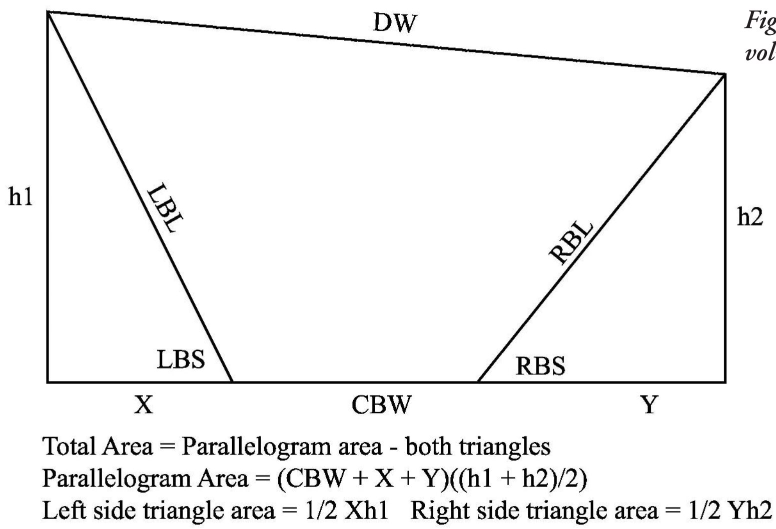


Figure 2. Method for estimating excavated volume: cross-sectional area.

Methods

A total of 117 km (73 mi) of previously decommissioned roads were assessed, or 34% of the total miles of decommissioned road (341 km or 212 mi of road decommissioned). Decommissioned roads were haphazardly selected based on age since treatment; however, attempts were made to get a cross section of different geologic terrains as well as a variety of stream crossing treatment sizes (i.e., large fill volumes versus small fill volumes). The intent of the post-treatment road decommissioning monitoring was to sample as many roads as possible that had weathered large storm events and thereby had treatments that were "tested" by winter storms. Eighty-eight percent of the sampled decommissioned roads had experienced at least three winters of post-treatment adjustment, including storm events with at least an 18-year recurrence interval.

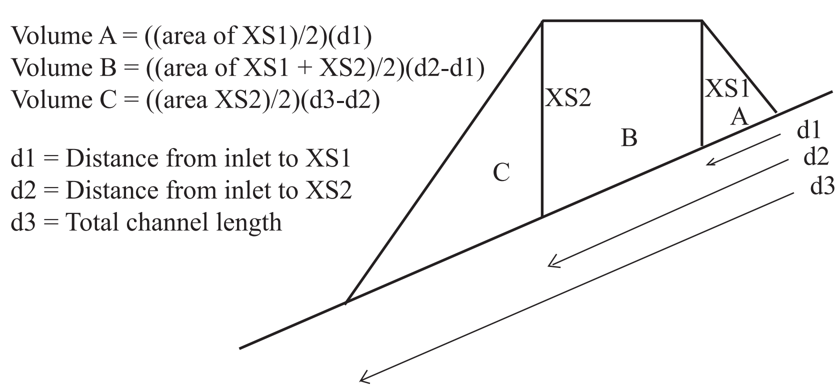
Decommissioned roads were assessed to determine the effectiveness of the treatments at stream crossings as well as the roadbed between the stream crossings. Methods and questions assessed are discussed below.

Estimating Excavated Stream Crossing Fill Volume

The post-treatment road decommissioning monitoring protocol required that the amount of material excavated from stream crossings be measured along with the volume of any post-treatment erosional features. A simplified model of the excavated crossings was used to facilitate rapid measurement of fill volume (see Figure 2 and Figure 3). In using the model, two cross-sections were measured (using a tape and clinometer), along with channel length and slope. The cross-sections were modeled as parallelograms. Total area of the cross-section was calculated by subtracting the area of the two triangles on the sides from the parallelogram area.

This method allowed estimating excavated fill volume for stream crossings that had a significant road gradient going through the channel, yet kept the measurements simple. When measuring the channel bottom width, due to post-treatment adjustments, estimates of the original channel bottom configuration were made to estimate the total fill volume excavated. Total volume for the excavated stream crossing was calculated by scaling the cross-sectional areas by the appropriate distance along the channel (Figure

Figure 3. Profile measurements for estimating fill volume.



3). Cross-sectional area was assumed to be zero at the upstream and downstream end of the crossing.

Most crossings were adequately measured with two cross-sections. In the case of very small crossings, only one cross-section was needed. In such instances, the profile was viewed as a rectangle with sides equal to the cross-section and bottom equal to the total channel bottom length. The upstream cross-section was located where the inboard edge of the old road meets the excavated crossing. The downstream cross-section was placed at the widest point of the excavation. Where significant road curvature exists, excavated fill volume may be slightly over-estimated.

Measuring Post-Treatment Erosional Features

Post-treatment erosional features in both excavated stream crossings and on the roadbed between stream crossings were measured. All visible erosional features were measured using average width, average length and average depth, which yielded estimates of total volume of material eroded. The type of feature was also noted (e.g., channel incisement, slump, gully, rill, debris slide, and so on). Sheet erosion was not measured. When possible, the cause of the erosion was noted and discussed. An estimate of percent delivery of sediment to watercourses from all erosional features was made. For the purposes of this study, all reported erosion volumes within stream crossings were considered 100% delivered sediment.

In addition to measuring the excavated and eroded volume, post-treatment assessments were made to determine if the erosional features had stabilized or were still susceptible to chronic future adjustment and sedimentation. For sites with large post-treatment erosion, evidence as to whether or not poor contract design or implementation was a factor was also assessed.

Post-treatment erosion was compared to the following independent variables: hillslope position, hillslope gradient, stream power, storm return interval, and size of excavation. The intent of the analysis was to identify interactions that might serve to predict when the intensity of road decommissioning design needs to be elevated because the risk of post-treatment adjustments are high. In other words, under what circumstances are the risks of post-treatment erosion and sedimentation high and how can they be reduced?

Results and Discussion

Approximately 117 km (73 mi) of decommissioned roads (or 34% of the total length of decommissioned roads) were monitored for post-treatment erosion (stream channel and sideslope erosion at the crossings, and erosion of the roadbed between stream crossings). The majority of these roads and stream crossings had experienced large storms (18-year or greater return period) during the winters of 1995, 1997 and 2003. High intensity storms were considered a good test of the type and magnitude of post-treatment erosion that occurs on typical road decommissioning projects in the Six Rivers National Forest.

Stream Crossings

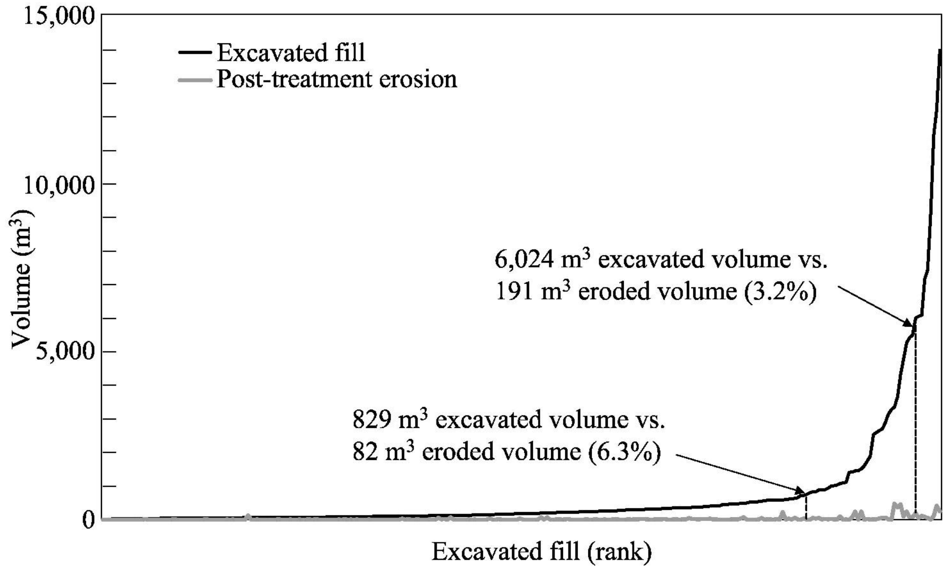
A total of 262 stream crossings were assessed for post-treatment channel erosion. These stream crossings were predominantly perennial streams but intermittent and ephemeral streams were also assessed. Post-treatment stream channel erosion was quite common, and was generally not large when compared to the amount of material excavated during the initial stream crossing restoration. On average, the amount of post-treatment erosion occurring within stream crossings was 4.5% of the total volume of fill excavated, or 21 m^3 (28 yd^3) (Table 1).

Post-treatment erosion on larger stream crossings (greater than 765 m^3 or 1000 yd^3 of fill) was more variable. The average post-treatment erosion on the larger stream crossings was approximately 3.2% of the total fill excavated but varied from 0 to 16%. Large post-treatment adjustments were rarely observed. In approximately 80% of these large sites, post-treatment erosion was less than 4.5% of the total fill excavated or under 111 m^3 (146 yd^3) (Table 1). The majority (64%) of the inventoried stream

Table 1: Stream crossing excavations and post-treatment erosion.

Excavated Stream Crossing Volume (yd^3)	Percent of Stream Crossings (%)	Volume Excavated (yd^3)	Average post-treatment erosion (yd^3)	Range of post-treatment erosion (yd^3)	80% of sites eroded less than # (yd^3)	Percent of excavated fill volume lost to post-treatment erosion (%)
0-400	64	153	6.7	0 to 156	5.7	5.1
400-1000	20	612	21	0 to 284	25.1	3.3
>1000	16	4692	124	0 to 621	146	3.2
All sites	100	967	28	0 to 621	20	4.5

Figure 4. Stream crossing excavation and erosion volumes.



crossings had excavated fill volumes under 305 m^3 (400 yd^3) and these stream crossings had very small amounts of post-treatment channel adjustments and erosion (average of 5 m^3 or 6.7 yd^3). Stream crossings larger than 305 m^3 (400 yd^3) experienced slightly more post-treatment erosion (average of 16 m^3 or 21 yd^3) but these quantities were small compared to the volume of material excavated from the stream crossings.

While the total volume of post-treatment erosion increased with the size of the excavation, the relative proportion of post-treatment erosion compared to fill volume excavated decreased from 5.1% for small stream crossings to 3.2% for large stream crossings (Table 1). Post-treatment channel erosion remained fairly constant regardless of stream crossing size (Figure 4). While occasional large stream crossing restoration sites have significant post-treatment channel erosion, on average the post-treatment channel erosion on these larger treatment sites is small considering the amount of fill removed.

Photo 1: Typical stream crossing erosion due to post-treatment channel incisement.

On Six Rivers National Forest, approximately 40% of measured post-treatment stream crossing erosion was attributable to channel downcutting or widening, and 60% was attributable to channel bank or sideslope failures. Post-treatment channel downcutting or widening was commonly found at most treatment sites (Photo 1) but excessive downcutting may indicate inadequate restoration design or contract implementation (Photo 2). Rills and gullies were only rarely observed on channel sideslopes and usually only associated with a nearby spring. The bulk of channel sideslope failures were attributable to oversteepened slopes and could be a result of inadequate restoration design or implementation (Photo 3). Depending on the size of the stream crossing restoration site, oversteepened slopes led to shallow slumps and small debris slides. Channel sideslope failures are an inherent risk in restoring stream channels, particularly in naturally steep and incised topography, regardless of the quality of contract design and implementation. Nevertheless, in some instances inadequate channel design, such as too narrow a stream channel configuration or too shallow a depth of excavation, resulted in channel incision, which led to oversteepened and unstable sideslopes (Photo 4). The data indicate that while the total post-treatment erosion was small, a greater emphasis on designing and excavating less steep channel sideslopes would likely reduce post-treatment erosion to

Photo 2: Atypical and excessive post-treatment channel incisement (note exposed root across channel indicative of depth of erosion).

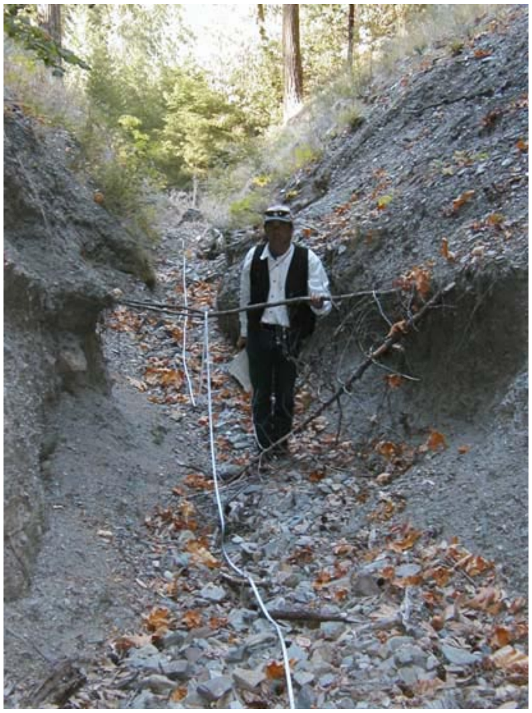


Photo 3: Incipient slope failure due to oversteepened channel side slope.

even smaller volumes, especially in stream crossings that are not topographically constrained by narrow, incised valleys.

Post-treatment erosion on excavated stream crossings is widely recognized as an inherent short-term effect that is offset by larger long-term gains in reducing the risk of major sedimentation resulting from culvert and fill

Photo 4: Channel bank failure due to oversteepened slope and inadequate channel width

failures. Fill failures and diversions of road stream crossings have been shown to be significant contributors of fluvial hillslope erosion (Best et al. 1995; Weaver et al. 1995). Furniss et al. (1998) assessed stream crossing failures on non-decommissioned forest roads in Washington, Oregon and Northern California and found that after the winter floods of 1995 and 1996, significant portions of road fill were lost due stream crossing failures. Figure 5 illustrates the proportion of stream crossing fill eroded where streamflow overtopped the road. The Furniss et al. data indicate that in approximately 35% of the culvert failures sampled, over 25% of the stream crossing fill eroded, and that 44% of the failures had between 1 and 25% of the stream crossing fill eroded. While the total percentage of storm-related stream crossing fill erosion on non-decommissioned roads varies, it is clear that the proportion lost due to post-treatment road decommissioning erosion is significantly smaller than the erosion that occurs during large storm events. Post-treatment road decommissioning erosion on the Six Rivers varied between 3.2 to 5.1% of the total stream crossing fill volume and was typically considerably less than the volume of erosion that occurs on untreated roads during large storm events.

A general assumption of stream crossing restoration is that the risk of post-treatment erosion and volume of sediment generated increased with size of the stream channel being restored. Data from this assessment indicate that while the total volume of erosion increased with larger stream crossings, the total volume of material was small compared to the total material excavated (typically 3 to 5%) (Figure 4). Photos 5 and 6 are examples of typical small and large stream restoration sites with very limited post-treatment channel erosion and sedimentation.