Changes in a Reach of a Northern California Stream Following Wildfire

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ABSTRACT / One reach of a northern California stream, burned by intense wildfire in 1979, was studied to monitor changes and recovery from the fire. Benthic macroinvertebrates collected three weeks and one to four, six, eight, and 11 years following the wildfire were used to assess stream condition and compared to samples from a reach of a nearby unburned stream. Transportable sediment was measured 11 years following the fire. The fire was also used as a worst case example to compare results with a standard cumulative watershed effects assessment methodology.

Benthic invertebrate density and taxa richness of the burned reach were both low compared to the unburned reach three weeks after the fire. Mean density was significantly higher in the burned reach in the three years following the fire, while taxa richness was significantly lower in the burned reach over the same time period. Higher density and lower richness in the burned reaches persisted throughout the study period but were not significant after three years. Mean Shannon diversity of the burned reach was significantly lower than that of the unburned reach for each year of the study, although absolute differences diminished throughout the 11-year study period.

Transportable sediment was significantly higher in the burned reach than the unburned comparison. Pearson correlations between sediment and biological metrics were weak. Although the correlation between invertebrate diversity and a measure of watershed disturbance (equivalent roaded acres) was high (r = 0.95) for the burned watershed, the measure appeared to be a poor indicator of cumulative effects on stream condition. The measure (ERA) was poorly correlated with invertebrate diversity in the unburned reach and, while the ERA calculations indicated substantial recovery, biological and physical measures indicated recovery of the burned stream reach was incomplete.

Recovery of macroinvertebrate communities from both natural and anthropogenic disturbance has received increased attention (Yount and Niemi 1990, Wallace 1990). Both the ecological role of disturbance in stream systems (Vogl 1980) and the response of communities to watershed and channel alterations have been well documented. Generally, benthic communities are thought to recover slowly from activities that alter stream habitat, but understanding of recovery from habitat alteration is hampered by a lack of long-term studies (Likens 1984).

Numerous researchers have speculated that of all wildland disturbances, wildfires may pose the severest threat to stream systems of forested watersheds (e.g., Anderson and others 1976). Large areas of a watershed may be burned, with indiscriminate effects on sensitive lands, such as streamside areas and steep or unstable slopes. Many wildfires occur under fuel moisture and weather conditions that can result in extreme fire intensities, may remove protective groundcover, and cause hydrophobic soil conditions. Increased water yield from reduced transpiration and infiltration usually follows such fires. Intense wildfires that burn a high percentage of a watershed (typically low-order streams) present the greatest potential for long-term effects on stream systems (Minshall and Brock 1991).

A better understanding of stream response to wildfire effects is important for several reasons. Recovery of stream systems and the duration of effects is arguably more important than the magnitude of the initial change. If wildfire does represent a worse-case situation, then documentation of the long-term response to such events can serve as a valuable reference point from which to compare the response of streams to other disturbances.

Benthic macroinvertebrates have been used extensively to assess both the condition of aquatic systems and effects of different management activities on those systems (Rosenberg and Resh 1993). Benthic invertebrates offer the advantages of being abundant

KEY WORDS: Stream; Wildfire; Macroinvertebrates; Diversity

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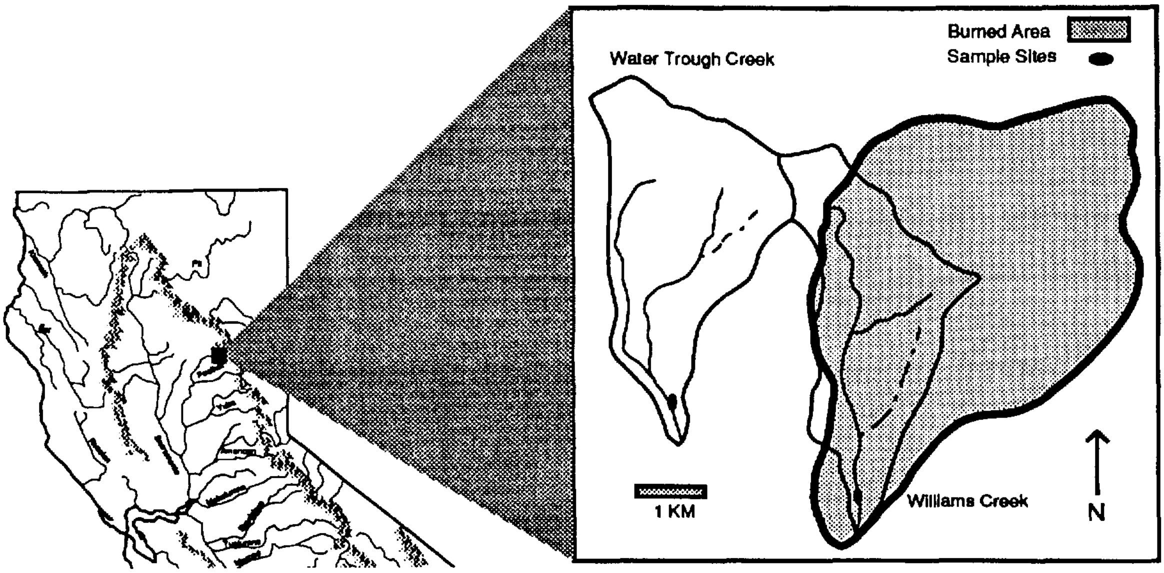

Figure 1. Location of Williams and Water Trough Creek Watersheds, approximate sampling sites and Williams wildfire boundary.

in most streams, having relatively long aquatic life stages and being sensitive to change (Gaufin 1973).

The objectives of this study were to use the macroinvertebrate community of a burned reach to assess changes over time, using a nearby unburned stream reach to supply reference conditions. Because differences in the macroinvertebrate samples from the two reaches persisted over time, transportable sediment was collected in both reaches as a measure of in-channel sediment 11 years following the fire. A final objective was to compare macroinvertebrate response to watershed recovery as predicted by a cumulative-effects model commonly employed by the US Forest Service in forests of California (USFS 1990).

History and Setting

Williams Creek drains a 825-ha watershed, ranging in elevations from 1100 to 1800 m at the northern extent of the Sierra Nevada mountain range, just north of the town of Greenville, California in the Plumas National Forest (Figure 1). Parent geologic material is paleozoic metavolcanic, from which Kinkle and Deadwood soils are derived. Erosion rates of these soils are moderate or high, depending on the slope, which ranges from 20% to 70%. The watershed lies in the transient snow zone, and averages 100 cm of precipitation annually. Prefire vegetation was typical west-side Sierra Nevada mixed coniferous forest (Kimmens 1987). At the headwaters, Williams Creek is a high-gradient (~5%) bedrock-dominated channel. It has a transitional moderate-gradient (~3%) section with lower elevation stream reaches in a lesser gradient (~2%) channel. The reach selected for sampling was in the lower-gradient section of the stream, where possible changes in sediment and flow from the wildfire might be most evident.

In September of 1979, a human-caused wildfire burned 95% of the drainage in less than 6 h. Firefighters estimated the rate of spread at 2000 m/h. Fire intensity was rated as high on two thirds of the burn. Areas on the fringe of the fire had moderate to low fire intensities. The stream channel in the lower two thirds of the watershed was intensely burned. Almost all the streamside vegetation was killed by the fire, which burned hot enough to consume much of the large woody debris in alluvial channel reaches.

Most of the intensely burned areas of the watershed were salvage logged during the year following the fire. Roughly 10% of the area was harvested using tractor systems, 30% with helicopters, and 50% with suspended cable logging. Prior to logging, grass was seeded on approximately 168 ha and dead poles felled on about 42 ha.

Subsequent monitoring documented vegetative response and channel enlargement. Postfire groundcover monitoring of 24 transects throughout the burn averaged 53% bare ground immediately after the fire. Regrowth of vegetation steadily reduced the amount of bare ground on the transects; groundcover averaged 85% when last measured in 1985. Channel cross-sectional area increased by about 20% in the four years following the fire, but had returned to immediate postfire (prerunoff event) cross-sectional areas six years following the fire (Roby 1988).

Rationale and Methods

Responses of the Williams Creek stream reach were compared with those from a stream reach in Water Trough Creek, which drains a 520-ha watershed. Water Trough Creek is the nearest perennial stream to Williams Creek and lies about 3 km to the west. Water Trough, like Williams Creek, drains portions of Keddie Ridge, has similar elevations and climatic characteristics, is composed of the same geologic material and soils, and supports the same vegetation types. Channel gradient at the sampled reach is 2%.

Benthic invertebrate samples were collected in the early fall using a Surber sampler ($0.09 m^2$, $\times$1.0-mm mesh) from both Williams and Water Trough creeks annually from 1979 to 1983 and in 1985, 1987, and 1991. Four samples were taken at each site in 1979, 1983, 1985, and 1987. Six samples were collected in 1980, 1981, and 1982. To evaluate sample variation more completely, 16 samples were collected in 1991. Each year, samples were collected from the same two stream reaches. The channel was stratified into pool, run, and riffle units, and samples randomly selected from riffle habitat types met (by eye) the following criteria: substrate size (gravel–small cobble), water depth (4–10 cm) and velocity (>0.3 m/sec).

Aquatic insects were keyed to genus (Ephemeroptera, Odonata, Plecoptera, and Trichoptera) or family (Coleoptera, Diptera, Lepidoptera). Other invertebrates were generally keyed to order. Information was expressed in terms of density, number of taxa, number of Chrironomidae, dominant taxa, and Shannon diversity for each sample (except for 1983).

Differences between stream reaches and within years in diversity, density, and number of taxa were tested using a t test. Variance in diversity values was examined with a two-way fixed-effects ANOVA (Kleinbaum and Kupper 1978), using year and stream as factors. Within-stream diversity value differences were assessed using a modified Tukey's multiple-comparison procedure (Keselman and Rogan 1978). An index of similarity (Odum 1971) was used to compare taxa in the two streams. The relationship between biological metrics and transportable sediment was quantified using a Pearson correlation. A Spearman rank correlation was used to evaluate the relationship between a watershed disturbance index (ERA) and stream-reach Shannon diversity, as the index values could not be expected to have a normal distribution.

In 1990, transportable sediment (Erman and Mahoney 1983) was measured by catching sediment 30.5 cm, 61 cm, and 91.5 cm downstream of each invertebrate sample site. Material was collected in 10.2-cm-diameter metal cans placed on the stream bottom as each Surber sample was collected. After large rocks were cleaned and removed from the sampler frame, the substrate was disturbed to a depth of 8 cm for 1 min. Coarse-grained material caught in the Surber net was also collected and included as transportable sediment.

The method of cumulative watershed effects assessment prescribed for use in the Pacific Southwest Region of the US Forest Service (USFS 1990) was used to describe the degree of watershed disturbance in the study watersheds. The method accounts for all types of watershed disturbance in a common disturbance unit, the equivalent roaded acre (ERA). Coefficients are applied to different disturbance types to equate each disturbance to that of a road. In this study, a coefficient of 0.25 was used for tractor-logged areas, 0.10 for cable-logged areas, 0.05 for areas logged with helicopter yarding, and 0.25 for areas burned by wildfire. Watershed disturbance levels in ERA units were calculated for each year invertebrates were collected and expressed in ERA as a percent of watershed area. A 25-year linear recovery was assumed for all disturbances.

Results

Diversity

Diversity values from the Williams Creek stream reach samples increased over the 11-year study period. The lowest mean diversity value was sampled in 1980, one year following the wildfire. The correlation between year and diversity was significant ($P < 0.05$, $r = 0.866$). Mean diversity of samples from the Water Trough Creek stream reach was also positively correlated with year ($P < 0.05$, $r = 0.768$). Recovery of the Williams Creek reach, using the Water Trough reach as a reference standard appeared incomplete, as 11 years following the fire, sample diversities from Williams Creek remained significantly lower ($P < 0.05$) than those from Water Trough Creek. Diversities were significantly lower in the Williams Creek

Table 1. Number of samples, mean Shannon diversity, and 95% confidence interval for difference between means for Williams and Water Trough creeks, 1979–1990

Year	Samples (N)	Shannon diversity: Williams	Shannon diversity: Water Trough	95% CI Williams–Water Trough
1979	4	1.79	2.64	(−1.30, −0.40)
1980	6	1.65	2.45	(−1.06, −0.55)
1981	6	1.78	2.66	(−1.22, −0.54)
1982	6	2.25	2.46	(−0.58, −0.12)
1983	4	2.21	2.78	[a]
1985	4	2.22	2.70	(−0.77, −0.19)
1987	4	2.37	2.77	(−0.67, −0.14)
1990	16	2.45	2.92	(−0.46, −0.24)

[a]Individual sample diversities not available.

reach than the Water Trough Creek reach in every year that samples were collected (Table 1).

The samples collected from Williams Creek in 1979 had the lowest density (mean 24 organisms/m^2) and number of taxa collected (Tables 2 and 3; mean 9, total 15) of any year for either stream. Diversity at this time, and one year following the fire, was 33% lower in the Williams Creek reach than the Water Trough reach. By the end of the study, mean diversity of Williams reach samples was 16% lower than those from the Water Trough reach (Figure 2).

Diversity values from both reaches fluctuated over time. In Williams Creek, diversities in 1980 were significantly lower ($P < 0.05$) than diversity in 1982–1990. The multiple-comparison procedure found 1987 and 1990 as being significantly different from 1981; 1982 and 1985 were significantly lower than 1990. In Water Trough Creek, diversity was significantly lower in 1980 and 1982 than 1990 (modified Tukey's test $P < 0.05$). Within-reach between-years variability was greater in the burned reach than the unburned Water Trough reach.

Density

Mean density of organisms increased from 1979 to 1982 in the Williams Creek reach and then declined through the remainder of the study (Table 2). Although the calculated value of mean density was greater in the burned reach each sampling year except 1979 (Figure 2), the difference was significantly higher ($P < 0.05$) only in 1980–1982.

Mean density values for individual years showed more variation in the Williams Creek reach than in the Water Trough Reach. The multiple-comparison procedure indicated that the mean density in 1980 was less than 1981 in Water Trough Creek. Mean density was less in 1979, 1980, 1985, and 1990 than in 1982 in Williams Creek. Mean density was also found to be greater in 1981 than in 1979 in Williams Creek.

Taxa

More taxa were found in the Water Trough Creek reach in every year sampled except 1980, when the Williams Creek reach had 28 and Water Trough reach 24 taxa (Table 3). The mean number of taxa per sample was significantly lower ($P < 0.05$) in the Williams Creek reach in only 1979 and 1981. The lower diversity values in the Williams Creek reach are due to higher density of a few taxa (Chironomidae, Baetidae, Hydropsychidae) and a slightly lower number of taxa.

Mean number of taxa was more variable in the Williams Creek reach than in the Water Trough reach. The multiple-comparison procedure found the mean for 1981 and 1990 to be different than 1980 and 1982 in Water Trough Creek. The mean number of taxa was different in 1979 than all other years except 1980 in Williams Creek. The mean also differed between 1980 and both 1982 and 1990 in Williams Creek.

Correlation of Biological Metrics with Sediment and ERA

Transportable sediment (collected only in 1990) was significantly higher in the Williams Creek reach (mean 439 g/sample) than the Water Trough Creek reach (262 g/sample). There were no strong correlations between the amount of transportable sediment collected and individual sample diversity, density, taxa richness, or number of Chironomidae for the Williams Creek reach. The correlation between the number of Chironomidae and transportable sediment in the Water Trough Creek reach (0.69) was the strongest found (Table 4).

Mean diversity for each year was negatively correlated with corresponding ERA watershed disturbance values for Williams Creek (−0.893 Spearman value). The correlation between Water Trough diversity and ERA values was much weaker (Spearman value of −0.071).

Similarity

The similarity of taxa lists from the Williams and Water Trough Creek stream reaches is high (Table 3), with taxa unique to either stream representing a small percentage of their populations after 1979. The last three sample years had the highest similarity values, but a linear regression (Kleinbaum and Kupper 1978)

Table 2. Indices of invertebrate samples from Williams (Will) and Water Trough (WT) Creeks, 1979–1990

Year	Mean density (N/0.1 m^2)		Mean taxa (N/0.1 m^2)		Dominant taxon (% of population)	
	Will	WT	Will	WT	Will	WT
1979	24	64	9[b]	23[b]	*Cinygmula* (37)	Chironomidae (13)
1980	120[a]	40[a]	14	17	Chironomidae (42)	Hydropsychidae (18)
1981	216[a]	96[a]	18[b]	26[b]	Chironomidae (32)	Chironomidae (16)
1982	328[a]	48[a]	19	24	Chironomidae (34)	Chironomidae (17)
1983[c]	240	64			Chironomidae (27)	Hydropsychidae (15)
1985	88	56	17	22	Chironomidae (33)	Chironomidae (12)
1987	136	80	20	24	Chironomidae (21)	Hydropsychidae (16)
1990	72	56	20	23	Chironomidae (21)	*Cinygmula* (13)

[a]Significant difference in density between streams ($P < 0.05$).
[b]Significant difference in # of taxa between streams ($P < 0.05$).
[c]Individual sample counts not available.

Table 3. Index of similarity, *S* (Odum 1971), for Taxa[a] and ERA values (as percent of watershed area) from Williams and Water Trough creeks, 1979–1990

Year	Taxa (N)		Taxa in common (N)	*S*	ERA (%)	
	Williams	Water Trough			Williams	Water Trough
1979	15	31	15	0.65	18.2	3.8
1980	28	24	19	0.73	18.0	3.6
1981	31	37	18	0.60	17.4	3.5
1982	32	34	21	0.64	16.8	3.4
1983[b]					16.2	3.3
1985	31	34	28	0.86	15.0	3.1
1987	30	34	24	0.75	13.8	2.8
1990[c]	41	50	38	0.80	11.9	4.2

[a]Total taxa from all samples (differs slightly from Table 1 mean taxa/sample).
[b]Individual invertebrate sample data not available for 1983.
[c]Sample size = 16.

Table 4. Correlation of transportable sediment with invertebrate community indices, Williams and Water Trough creeks, 1990 (Pearson coefficients)

	Williams	Water Trough
Diversity	−0.415	−0.371
Density (N/m^2)	−0.012	0.568[a]
Chironomidae (N/m^2)	0.343	0.697[a]
Taxa (N)	0.082	0.270

[a]Significantly different than zero.

indicated that the trend for increasing number of common taxa over time was not significant ($P <$ 0.095).

Discussion

Effects of Wildfire

Due to the unplanned nature of their occurrence, the effects of wildfires on watersheds and stream systems are difficult to study, although several researchers have documented effects of intense wildfire on forested watersheds that appear similar to Williams Creek. We found significantly more transportable sediment in the Williams Creek reach than the Water Trough reach. Significant increases in sediment production following wildfire have been reported by Helvey (1980), Campbell and others (1977), Rich (1962), and Tiedemann and Klock (1974). Much of the sediment increase was attributed to channel sources by Rich (1962) and Helvey (1980), who also documented increases in stream discharge following wildfire.

We found significant differences in the macroinvertebrate communities collected from burned and unburned stream reaches that persisted throughout the study period. Impacts of wildfire on stream biota have not been studied as extensively as the physical changes discussed above. Lotspeich and others (1970)

Table 5. Shannon diversity values (*H*) and number in dominant taxa as a percent of total number of individuals from other streams in Plumas National Forest

Stream	Year sampled	Dominant taxa (% of total)	Samples (*N*)	*H*	Reference
Lights Creek	1975	25.0	4[a]	2.51	Erman and others (1977)
Lights Creek	1982	17.0	4[a]	3.00	Erman and Mahoney (1983)
Lights Creek	1990	31.1	4[a]	2.72	Fong (1990)
Upper Taylor Creek	1975	22.9	4[a]	2.78	Erman and others (1977)
Soda Creek	1987	12.8	6[b]	2.67	Roby (1988)
Middle Mill Creek	1983	28.6	4[c]	2.40	Schultz (unpublished)
Middle Mill Creek	1984	22.1	4[c]	2.26	Schultz (unpublished)
Middle Mill Creek	1985	29.3	4[c]	2.59	Schultz (unpublished)

[a]Modified surber 0.1 m^2.
[b]Double surber 0.18 m^2.
[c]Surber 0.09 m^2.

found little change in the invertebrate community following an Alaskan wildfire. Albin (1979) compared burned and unburned tributaries to Yellowstone Lake and found higher diversity, higher number of Chironomidae, and lower density of organisms in the unburned stream. In both cases, sampling stations were some distance downstream of the burns. Richards and Minshall (1992) compared the invertebrate communities of ten streams tributary to the Middle Fork of the Salmon River, five which had been burned by wildfire. They found significant differences between the burned and unburned streams in terms of species richness and taxa present.

Watershed and Stream Recovery

The recovery of lotic macroinvertebrate communities from different forms of environmental stress has been extensively documented (e.g., Cairns and Dickson 1977, Hynes 1960) and recently summarized by Wallace (1990). In general these studies indicate that stream organisms are well adapted to quick recovery, and a long (>2 year) duration of effects was limited to cases of habitat alteration. In a comprehensive review of stream recovery case studies, Niemi and others (1991) found 23 instances in which recovery of macroinvertebrate communities had taken longer than 18 years. These disturbances included mining (where toxics were present), stream channelization, and timber harvest. In only two cases was there documentation of recovery from timber harvest impacts taking more than five years.

Fewer studies have addressed the recovery of burned forested watersheds. Lyon (1976, 1984) reported that native vegetation had increased 12 years following a western Montana fire and had been replaced by young pine stands 21 years after the fire. Tiedemann and others (1979) reported erosion rates did not return to prefire levels until ten years following wildfire, while Helvey (1980) noted a decrease in sediment production ten years following wildfire, associated with natural channel stabilization and increased upslope infiltration. Helvey (1980) expected it would be 40–50 years before the runoff regimen returned to prefire conditions. As watershed condition strongly influences channel condition (Hynes 1975), a long recovery period from wildfire might be expected for the streams of burned watersheds.

The Will Fire subjected Williams Creek to both short- and long-term changes as described by Minshall and others (1989). Short-term changes included increases in stream temperature, flow, and nutrients, which have been documented by many researchers, notably Schindler and others (1980). Long-term changes included increased sediment production, increases in both large and small organic debris, and a shift in the type and amount of leaf litter as described by Minshall and others (1989).

The response of the invertebrate community of the Williams Creek reach (Figure 2) appears to reflect both these short- and long-term changes. Initial changes were fairly dramatic, with community similarity, density, taxa richness, and Shannon diversity all significantly different between the burned and unburned reaches. Taxa richness for the burned reach was significantly lower in 1979 than in all subsequent sample years except 1980. These changes may be due in part to lethal temperatures caused by the fire. Partial recovery of the invertebrate community appears to have occurred quickly, with no difference between the burned and unburned reaches after one year in

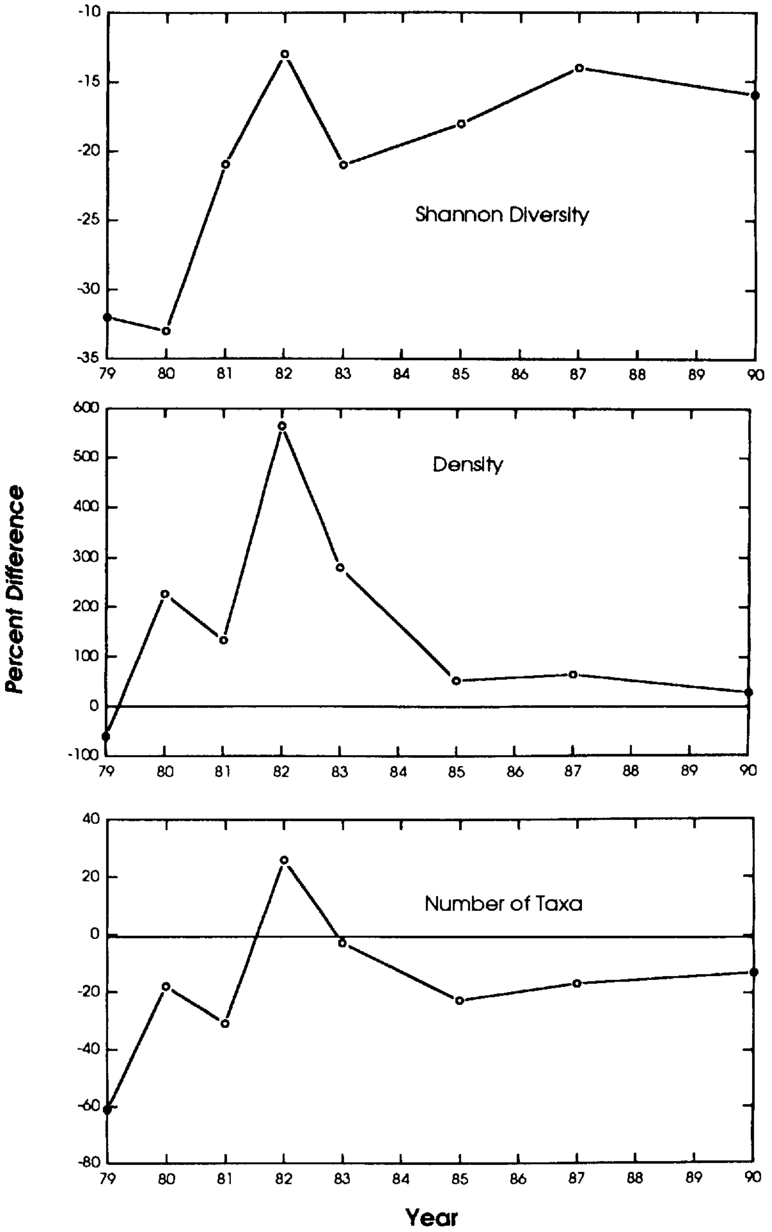

Figure 2. Difference in mean values between Williams and Water Trough Creeks as percent of Water Trough for three biotic indices through time.

terms of community similarity, and after three years in terms of density and taxa richness. More subtle long-term changes are indicated by the significant differences in diversity between the burned and unburned reaches that persisted throughout the 11-year study period. The diversity is a function of higher density and lower richness in the burned reach relative to the control. Enrichment from elevated temperatures and nutrient supply appears the likely mechanism responsible for these differences.

This response is generally consistent with that anticipated by Minshall and Brock (1991), who speculated stream recovery from the Yellowstone Fires would be broken into rapid and then slower stages following initial suppression of biotic index values following wildfire. Studies of biotic stream recovery from disturbances other than wildfire show a similar trend. Erman and others (1977) found macroinvertebrate diversities of streams logged without protective buffer strips were about 25% lower than those of comparable unlogged streams one to three years following harvest. Erman and Mahoney (1983) studied the same streams (seven to ten years after logging) and found diversities about 10% below comparable unlogged streams. Fong (1990) revisited the same streams (15–17 years after logging) and found no difference between the logged and unlogged streams. Anderson (1992) followed the recovery of streams following the eruption of Mt. Saint Helens. He found that repeated sediment flows retarded recovery in Ape Canyon. In Clearwater Creek, rapid recovery of species richness occurred for the first four years following the eruption, with a slower recovery rate thereafter. Murphy and Hall (1981) found macroinvertebrate communities in logged second-growth streams similar to those of old-growth forests 12–35 years following harvest.

Very little comparative information on recovery of stream biota from wildfire effects is available. Richards and Minshall (1992) concluded the burned streams of the Middle Fork Salmon River had not recovered five years following wildfire.

This study gauged response of a reach of Williams Creek against conditions in a reach of Water Trough Creek and assumed that differences between the two reaches was due to the wildfire and subsequent salvage operations and not simply natural variation. Despite the lack of prefire data, we make this assumption due to the similarity of the basins in terms of their natural characteristics including basin size, precipitation, geology, soils, aspect and the findings of other researchers. The biotic indicators used to assess recovery did not remain constant in the Water Trough Creek reach. Such variation in benthic communities has been shown in many studies (McElravy and others 1989, Lamberti 1992) and is to be expected in dynamic systems subject to natural events such as drought and flood. No prefire samples were taken from either creek; 1979 sampling was conducted a few weeks following the fire. Sampling of other Plumas National Forest Streams with similar characteristics (Erman and others 1977, Erman and Mahoney 1983, Fong 1990, Roby 1988, Schultz unpublished) (Table 5) show that macroinvertebrate communities of similar undisturbed streams generally have Shannon diversities in the 2.5–3.0 range, supporting the contention that lower diversities in Williams Creek are the result of wildfire and not natural variation. It should also be noted that in November 1979, heavy rain the day after a road-oiling project in the Water Trough Creek watershed resulted in contamination of the creek. This event may explain reduced density, richness, and diversity of the 1980 sampling from the Water Trough Creek reach.

Minshall and Brock (1991) postulated that the extent of wildfire effects on the benthic community is dependent upon watershed size (effects are diluted at large scales) and the intensity of the fire. As Williams Creek is a low (second) -order stream that had a high percentage of its watershed burned intensely, severe disruptions of the benthic community of the sampled reach might be expected. Certainly, the impacts to Williams Creek appear more severe and certainly extend temporally much longer than changes in the seven logged reaches (relative to controls) studied over a similar time scale by Erman and others (1977), Erman and Mahoney (1983), and Fong (1990).

It seems likely the incomplete recovery of the invertebrate community is a result of incomplete stream channel and watershed recovery and that the difference in response between the Williams Creek reach and the more complete recoveries referenced for silvicultural activities is due to the severity of the fire and its resultant impacts to both the watershed and the reach. The difference in transportable sediment 11 years following wildfire supports this contention, as does the literature on watershed response to wildfire quoted earlier. Incomplete watershed recovery and transport of increased sediment, water, nutrients, and organic material to the channel should be reflected in the invertebrate community.

Many of the physical stream system components (canopy cover, channel stability, substrate embeddeness) of Williams Creek, including the sampled reach, appear to have recovered more rapidly than the invertebrate community. Channel cross sections had returned to immediate postfire cross-sectional area by 1985 (Roby 1988), but other physical parameters were not measured. Obviously, determination of the cause of delayed recovery can best be made by continued monitoring of the stream community along with expanded measurement of physical parameters.

Cumulative Watershed Assessment

The Pacific Southwest Region of the Forest Service has used a disturbance accounting system based on equivalent roaded acres (ERA) as a key methodology in analysis of cumulative watershed effects. This procedure has been used extensively on national forests throughout California. A basic version of this methodology assumes incremental recovery over a 25-year period. We used this method to compare theoretical Williams Creek watershed recovery with reach response as indicated by Shannon diversity of the invertebrate community.

Shannon diversity and ERA were closely correlated for Williams Creek, but not for Water Trough Creek. The strength of this relationship is questionable, however. The diversity of the sampled reach invertebrate communities of both Williams and especially Water Trough Creek fluctuated over the course of the study, trending generally upward over time, perhaps due to drought-related stable, low-flow conditions over the last two sampling periods. When this "natural" variation was accounted for by comparing samples from the creeks in terms of percent difference (rather than absolute values) (Figure 2), the Williams Creek reach showed little if any recovery over the last seven years of the sampling period.

The 1990 ERA value for the Williams Creek watershed was just under 12%. Typical "threshold" values, used to prescribe a "safe" level of watershed disturbance, are often set at about this level (threshold values vary with watershed sensitivity) and would be close to 12% for a watershed with Williams Creek's geologic, climatic, and topographic characteristics. While this ERA value might classify a watershed as being in "acceptable" condition, actual instream conditions, as indicated by both transportable sediment and the invertebrate community of the sampled reach, appear to be less than desired, with recovery incomplete.

There are numerous reasons for the discrepancy between the predictive and in-channel indicators. Use of an ERA-based accounting of watershed disturbance in this case assumes that the primary changes from the wildfire will be to sediment and water delivered to the channel. Changes to other processes, namely stream productivity and biological interactions, may be dominant, thus obscuring flow and sediment effects. Wildfire effects may not be adequately reflected by the timber-based disturbance coefficients used in the analysis. ERA is an instantaneous indication of past and present watershed activities (Reid 1993) and does not reflect effects that may be delayed over time. Further, the assumption of linear recovery made in the ERA analysis appears to be a major shortcoming. There is a need for additional studies that relate channel condition to watershed disturbance models to continually check assumptions of cumulative watershed assessment methodologies.

Acknowledgments

We thank Don Erman, Chuck Hawkins, Jeff Kershner, and Liz Bergey for valuable reviews of earlier drafts.

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