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MONITORING THE IMPACTS AND PERSISTENCE OF FINE SEDIMENT IN THE PRAIRIE CREEK WATERSHED

Results for Water Year 1993

and a summary of study years

1989 - 1993

- 4 study years

with recommendations based on data from

1981 - 1994

Final Report to the

California Department of Transportation

by

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ACKNOWLEDGMENTS

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EXECUTIVE SUMMARY

The impacts and persistence of fine material deposited in Prairie Creek that came off a highway construction alignment during an October, 1989, storm were studied for four years. The first year's results are documented in Redwood National Park (1991) and and the second two years are in Meyer et al. (1994). This report documents results from the fourth year, Water Year 1993 (WY93), summarizes biological study results for the four years, and reviews all available fishery study results obtained before and after the October storm to make conclusions and recommendations for future fisheries management. A summary of the physical results for five years and conclusions on the persistence of the fines will be provided in a separate report.

Precipitation, water discharge, and sediment discharge were monitored to determine persistence of fines in WY93. Salmon egg survival and factors affecting survival were studied to a much more limited extent than previous years to determine impacts to the fishery from the bypass-derived sediment. In WY91, results were compared between the impacted reach on Prairie Creek below Brown Creek and a less impacted reach (called a control) above Brown Creek. Lost Man Creek was used as a second control reach only when fry emergence from natural redds was compared. In WY92 and WY93, both Lost Man Creek and Prairie above Brown Creek were used as controls in the egg survival studies.

1. **Hydrologic season:** Rainfall was above average in WY93 (95 inches, 125% of normal), higher than the previous years of study, but the peak flood on Prairie Creek was still below a 2-year recurrence interval flood. Storms were frequent; the largest occurred on March 24.

2. **Suspended sediment:** Total suspended sediment loads were much higher in WY93 than in previous study years.

3. **Salmon egg survival:** Only 5 artificial redds were placed in the streams in WY93 compared to the 22, 20, and 18 redds placed in the streams in prior years. Mean egg survival-to-hatching rate in artificial redds in WY93 was similar to WY92 (40.7% in the impacted reach and 40.1% in the control reach vs. 43.5% and 32%, respectively in WY92). However, sac-fry mortality was higher than previous years. Egg survival was very low below Boyes Creek (< 2%), where worms infested the redd. The worm infestation rate in the impacted reach (1 of 3 redds) was lower than WY91, but higher than WY92. No worms were found in the control reaches.
Only 1 of 4 natural redds capped on Prairie Creek below Brown Creek produced fry. Over all years, none of the natural redds produced fry below Boyes Creek.

Due to the higher flows, 67% of the natural redds on Prairie Creek were found above Boyes Creek, where coho salmon egg survival averaged 53%. Chinook salmon, in particular were more abundant above Boyes Creek and likely had better egg survival in WY93 than previous years. Coho salmon and steelhead trout that spawned on the mainstem spawned lower in the system and probably had lower reproductive success in WY93 than WY92. Many fish of the latter two species spawn in the tributaries, some of which were not affected by the bypass and some (May, Boyes, and Brown) were impacted more than the mainstem, so reproductive success for these species overall in the Prairie Creek watershed is unknown.

Of all the study years, WY91 had the poorest egg survival. WY90 artificial redds, which were exposed to minor storms, showed good steelhead egg survival, but salmon survival (which was not studied that year) may have been lower because their earlier redds would have been exposed to storms of similar size to WY91. Egg survival improved in WY92 and WY93, although the reach below Boyes Creek appeared to still be a problem during higher flow years.

4. **Juvenile salmonid production:** Analysis of pre- and post-October 1989 storm downstream migrant data collected by Redwood National Park and the Pacific Coast Fish, Wildlife, and Wetlands Association indicated that WY90 and WY91 were poor juvenile production years for chinook salmon and coho salmon. Steelhead trout, which spawn later in the season, appeared to have better production those years. By WY94, salmonid juvenile production had improved to pre-1989 levels. WY94 was a low flow year; results from that year may not be indicative of production for future years having higher flows.

5. **Fine sediment, permeability, dissolved oxygen, and worms in redds:** Fines infiltrating into artificial redds were about 3 times higher in WY93 (6.5% in affected reach, 6.1% in control reaches by weight) than WY91, the year with very poor egg survival. Over all years, fines levels were strongly correlated to peak stream discharge during the incubation period ($r^2 = 0.77$), and differences between impacted and control reaches were minor relative to the effect of this variable. In WY93, permeability decreased in both reaches, but the decrease was greater in the affected reach. Dissolved oxygen remained high all years and was not correlated with egg survival. Fines and permeability were not strongly correlated to egg survival over the three years; the strongest correlation was with worm infestation rate (Haplotaxia ichthyophagous) by reach, accounting for 63% of the variance in egg survival. The combination of fines < 0.5 mm mixed with worm mucus may have coated and suffocated eggs.

Conclusions:

Impacts of bypass sediment: Surface silt increased on affected tributaries and the mainstem on Prairie Creek but not on the control stream, Lost Man Creek, following the October 1989 storm. Subsurface fines tended to increase on the mainstem following the storm, also, particularly below Boyes Creek and above Brown Creek. A major portion of the impacts on and below Boyes Creek occurred in 1988 before the October 1989 storm. May Creek was substantially impacted during the storm, but impacts for this creek had already been mitigated at the estuary.

The bypass sediment inputs in WY90 may have led to an increase in oligochaete worms a year later, which had an adverse effect on egg survival, particularly in WY91. If worms are cyclically prolific naturally, the bypass may not be responsible, unless the combination of the bypass fine sediment and worms or their mucous is detrimental. Salmon juveniles appeared to be few in number in WY90, even though worms were rarely found in redds. It is unknown whether this was due to the bypass, although other reasons are not obvious. Other studies showed amphibians were adversely affected in WY90, but macroinvertebrates examined at the order level did not appear to be affected or recovered quickly. Further analysis of invertebrate samples to lower taxonomic units would be needed to verify this. Egg survival and juvenile salmon production seemed to improve in WY92 and WY93. Fish production reached pre-bypass levels in WY94, a low flow year. Limited data suggests steelhead trout appeared to continue to be productive during the study years and were likely less affected by the bypass.

Recommendations: The life history of the oligochaete worm and its direct or indirect effect on salmonid eggs in relation to sediment need to be studied to determine the extent of the role the bypass plays in the low egg survival found in WY91 and below Boyes Creek. Juvenile production should be monitored following a high flow year (> 2-year recurrence interval flood), and if production is as high as WY94, the California Department of Transportation's obligations for the October, 1989, storm should be suspended (as long as another sedimentation event does not occur). In the interim, the hatchbox rearing program should be discontinued until the higher flow event occurs. If the high flow event implicates sediment as responsible for low juvenile production (fines infiltration should continue to be monitored to determine this), the need for the hatchbox program should be reconsidered as possibly necessary for high flow years until the streambed is scoured clean of the bypass sediment. Recommendations are described in more detail at the end of this report.

I. INTRODUCTION

A. Background

Construction of the U.S. Highway 101 Bypass project through the eastern boundary of Prairie Creek Redwoods State Park and portions of Redwood National Park (RNP) began in 1984 and was completed in 1992. In October, 1989, a relatively small rainfall event generated numerous mudflows on the long, steep fillslopes which drain into the tributaries. Several hundred tons of sediment eroded from the project and flowed into the Prairie Creek stream system. In anticipation of damage to fish and other aquatic resources, Caltrans proposed monitoring to study the impacts and persistence of this fine sediment. This project was funded by Caltrans and implemented by RNP and the U.S. Forest Service Pacific Southwest Range and Experiment Station (PSW). Three years of monitoring were completed and results evaluated in RNP (1991) and Meyer et al. (1994). This report provides results of studies conducted in the fourth year during the winter of 1993-94 referred to as Water Year (WY) 93. Because WY93 was the last year of egg survival studies, this report summarizes and compares the results from WY90 to WY93. Lastly, fisheries data were analyzed from 1981 to 1994 to develop and present recommendations on whether mitigation in the form of fish supplementation should continue.

B. Study Objectives

The overall objectives of the study were as follows:

1. To determine persistence of fine sediment derived from the U.S. Highway 101 Bypass Project in Brown and Prairie Creeks.

2. To determine how sediment inputs from the Project affected salmonid egg survival in Prairie Creek.

3. To review fisheries data for Prairie Creek and develop mitigation recommendations.

Specific objectives of the persistence of fine sediment study in WY93 were as follows:

1. To determine precipitation, water discharge, and sediment discharge for Prairie and Brown Creeks and a control reach and compare WY93 values to previous years.

Specific objectives of the egg survival study in WY93 were as follows:

1. To compare survival of coho salmon eggs in WY93 to previous years using select artificial redds located in the same sites as in WY92.

2. To compare fine sediment, worm infestation rates, permeability, and dissolve

oxygen in the select redds to values from previous years.

3. To detect differences in WY90 to WY93 egg survival and other redd parameters between the heavily bypass-impacted and less impacted or unimpacted reaches in the Prairie Creek watershed.

4. To test if sediment has had a direct effect on egg survival over the four years of the study.

The egg survival study in WY93 was substantially reduced in scale from the previous three years of study.

C. Study Area

The Prairie Creek watershed lies within the coast redwood region near the town of Orick in northern California. Prairie Creek, tributary to Redwood Creek, is 14 miles long, drains an area of 30 mi^2, has a gradient of 0.01, and flows along U.S. Highway 101 through portions of private agricultural land and old-growth forest in Prairie Creek Redwoods State Park and Redwood National Park. The stream reaches studied included five miles of Prairie Creek from just above the mouth of May Creek north to 1.5 miles upstream of the mouth of Brown Creek, and two Prairie Creek tributaries, Brown Creek and Lost Man Creek. The reaches of Prairie Creek downstream of Brown Creek and within Brown, Boyes, and May Creek tributaries were impacted by the October 1989 storm. The reach above Brown Creek was only slightly impacted by sediment inputs from Ten Tapo Creek. This reach and Lost Man Creek were used as controls in the studies (Figure 1). Except for Little Lost Man Creek (located just south of Lost Man Creek), substantial parts of the tributary basins from Boyes Creek southward were logged.

II. SEDIMENT TRANSPORT

A. Methods

Park geologists monitored precipitation, water discharge, and suspended sediment discharge using methods described in Meyer et al. (1994). Streambed mobility was not sampled with scour chains in WY93. Comparison and discussion of all sediment transport data will be completed in a separate physical monitoring report that includes data from WY94.

B. Results

Precipitation, water discharge, and suspended sediment discharge were higher than the previous study years. Results are presented in Figures 2, 3a-3f, and 4a-4c. The WY93 winter was the end of six years of drought in the West with almost 125% of normal rainfall in Humboldt County. The spring was wet, also, prolonging high flows in Prairie Creek through early May (Farro 1993). Storms occurred throughout the season. The largest storm occurred on March 24, but was still less than a 2-year recurrence interval storm.

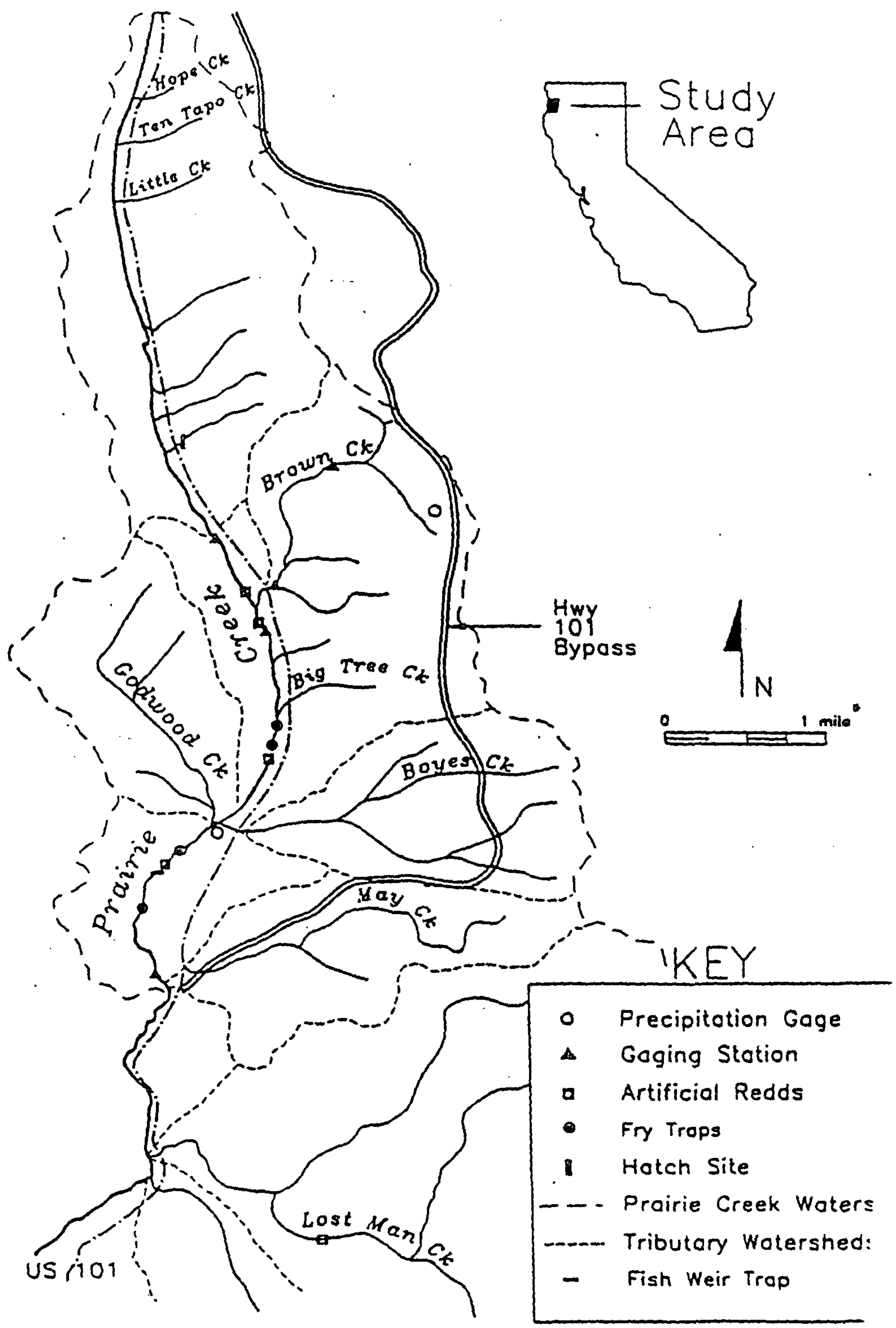


Figure 1. Location map of study area and WY93 sample areas.

Figure 2. CUMULATIVE RAINFALL AT PRAIRIE CREEK HWY 101 BYPASS, 1992-93 WATER YEAR

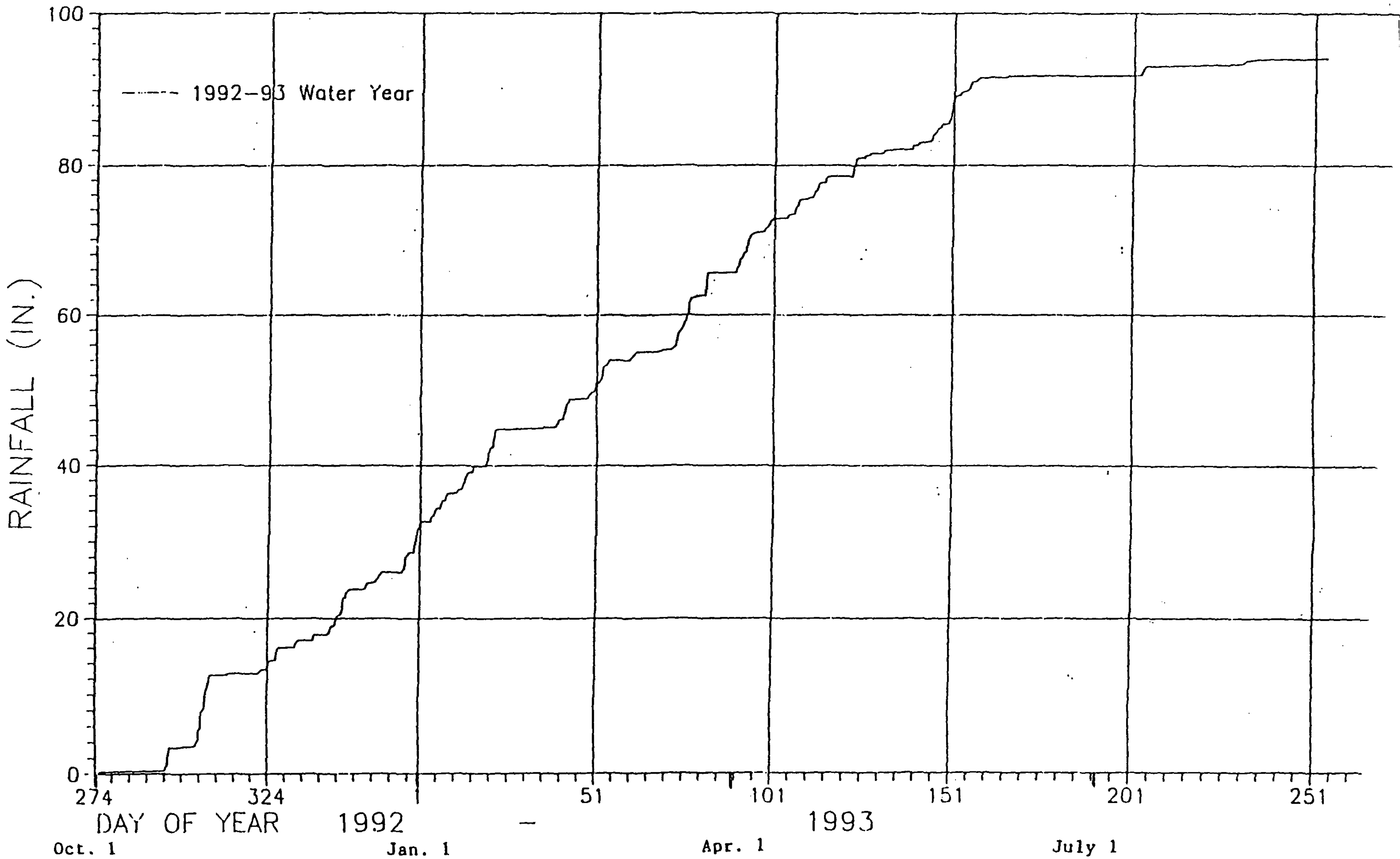


Figure 3a. REDWOOD NATIONAL PARK Prairie Cr. Bypass Study Hydrograph 10/01/92 – 12/31/92

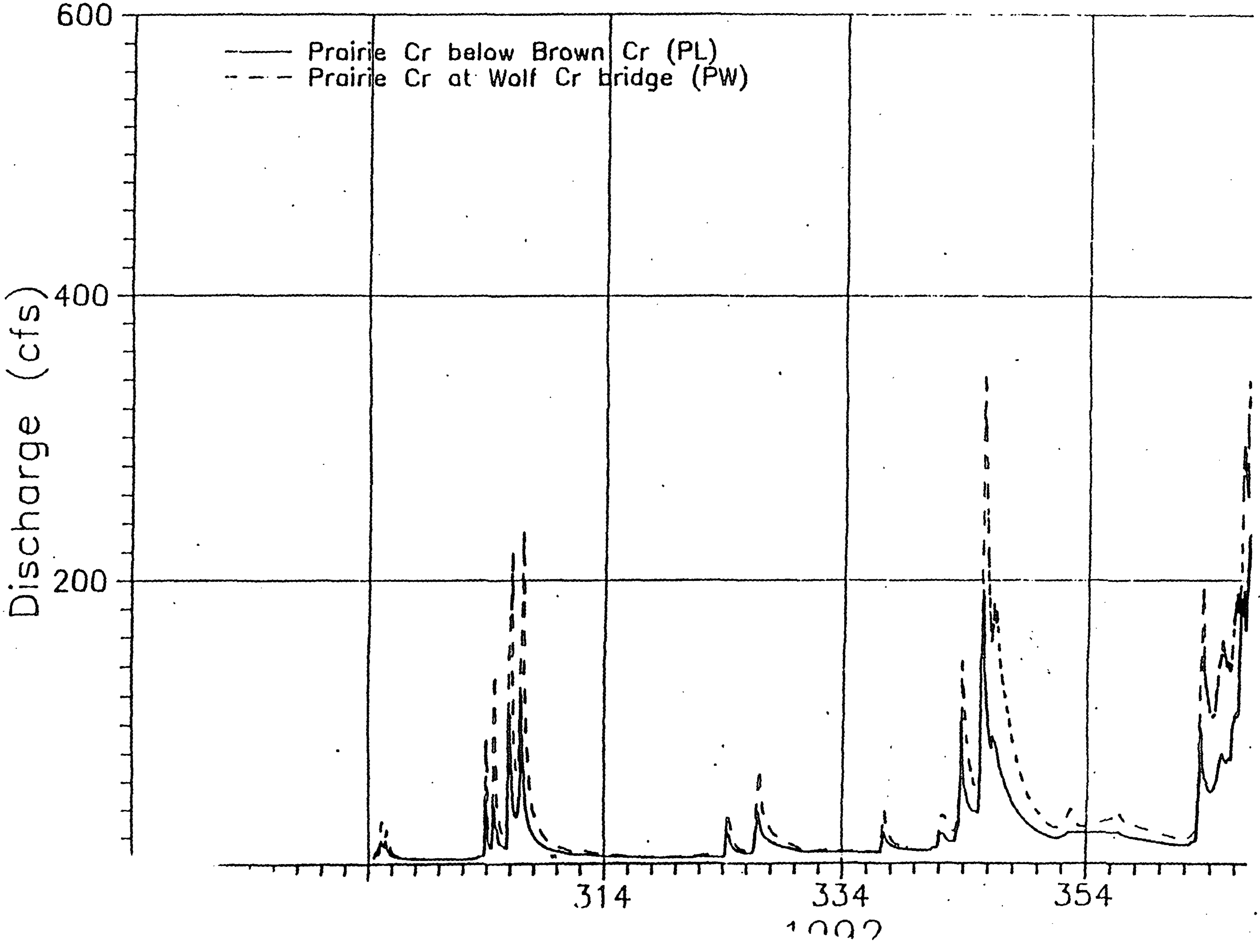


Figure 3b. REDWOOD NATIONAL PARK Prairie Cr. Bypass Study Hydrograph 01/01/93 – 03/31/93

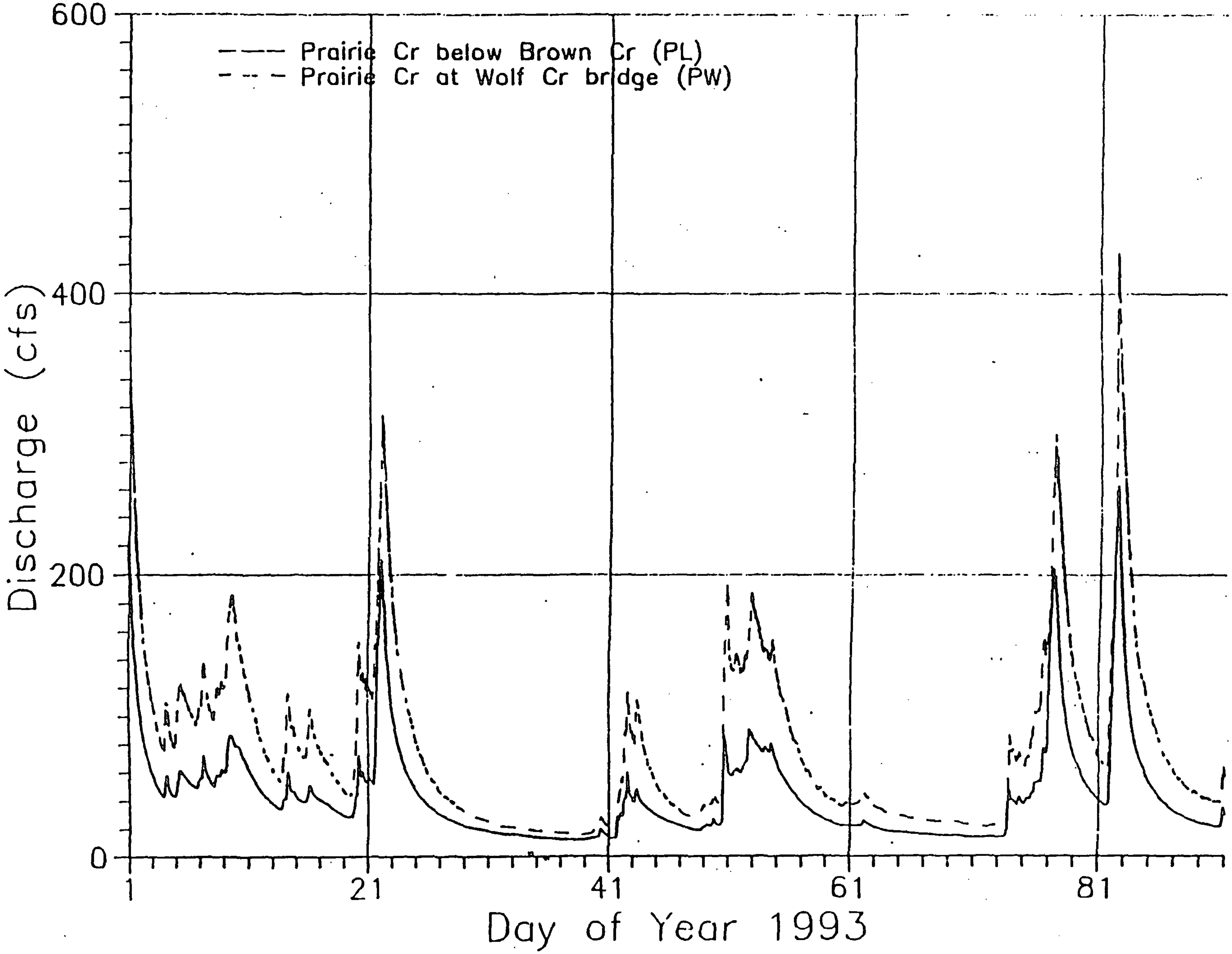


Figure 3c. REDWOOD NATIONAL PARK Prairie Cr. Bypass Study Hydrograph 04/01/93 – 06/30/93

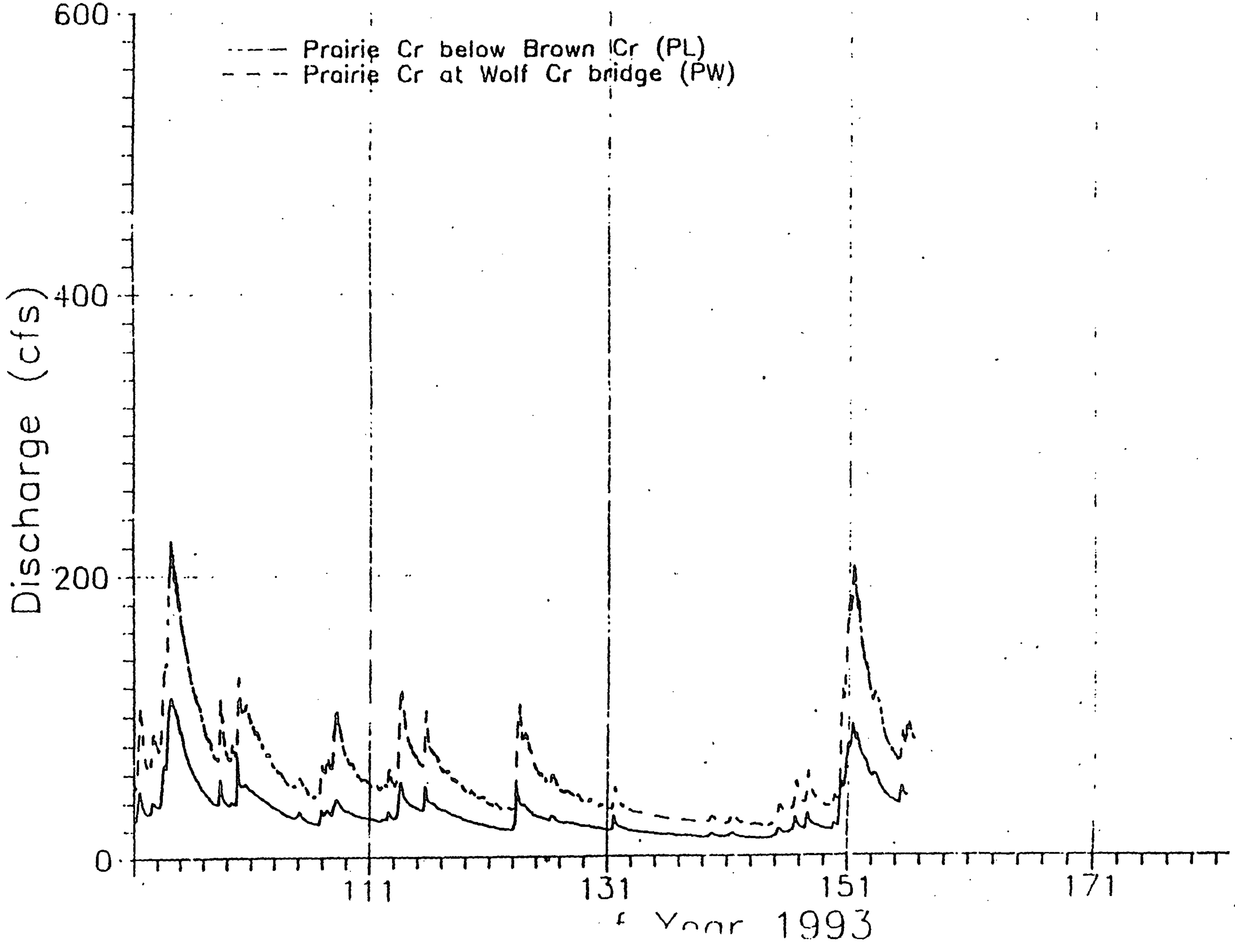


Figure 3d. REDWOOD NATIONAL PARK Prairie Cr. Bypass Study Hydrograph 10/01/92 - 12/31/92

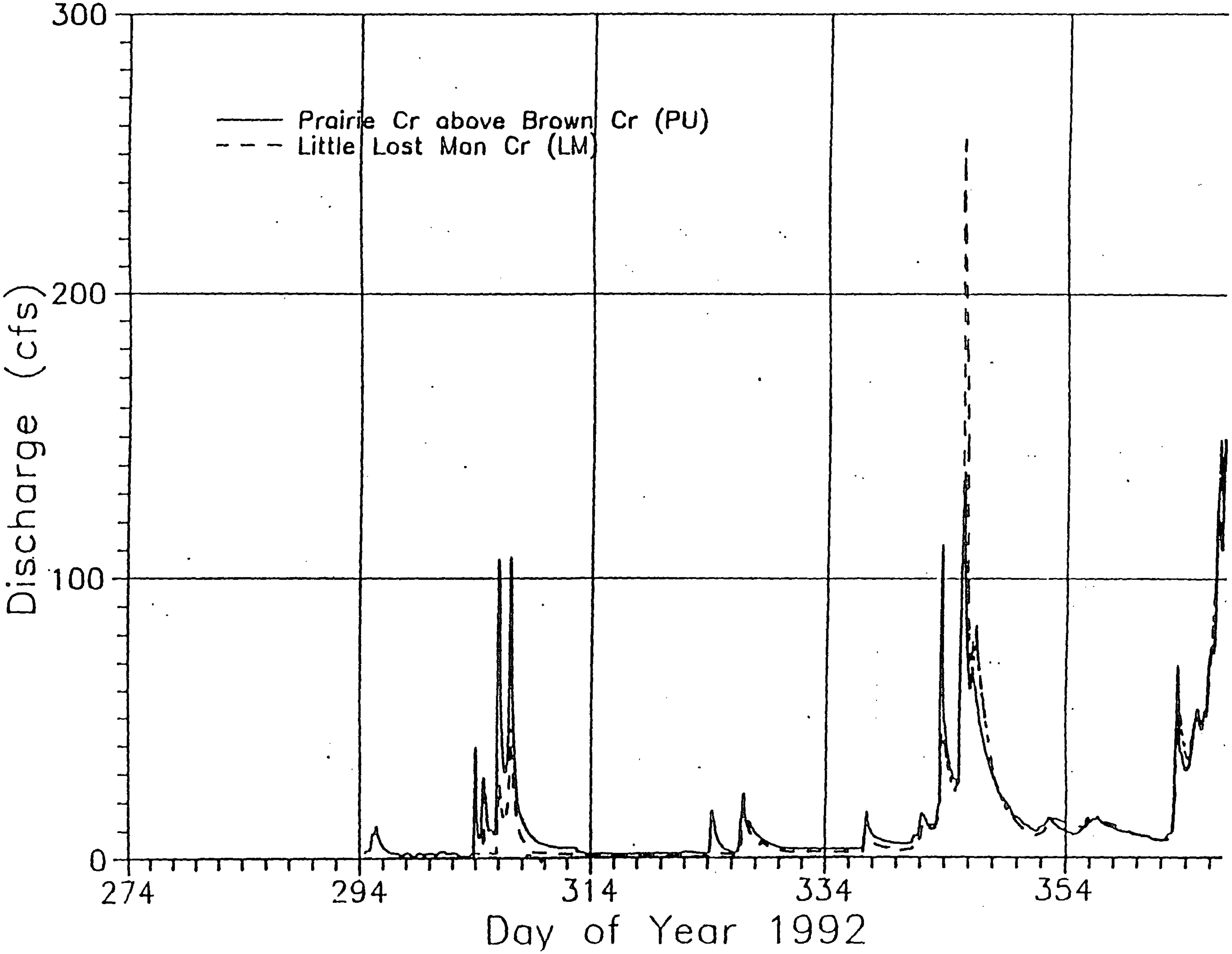


Figure 3e. REDWOOD NATIONAL PARK Prairie Cr. Bypass Study Hydrograph 01/01/93 – 03/31/93

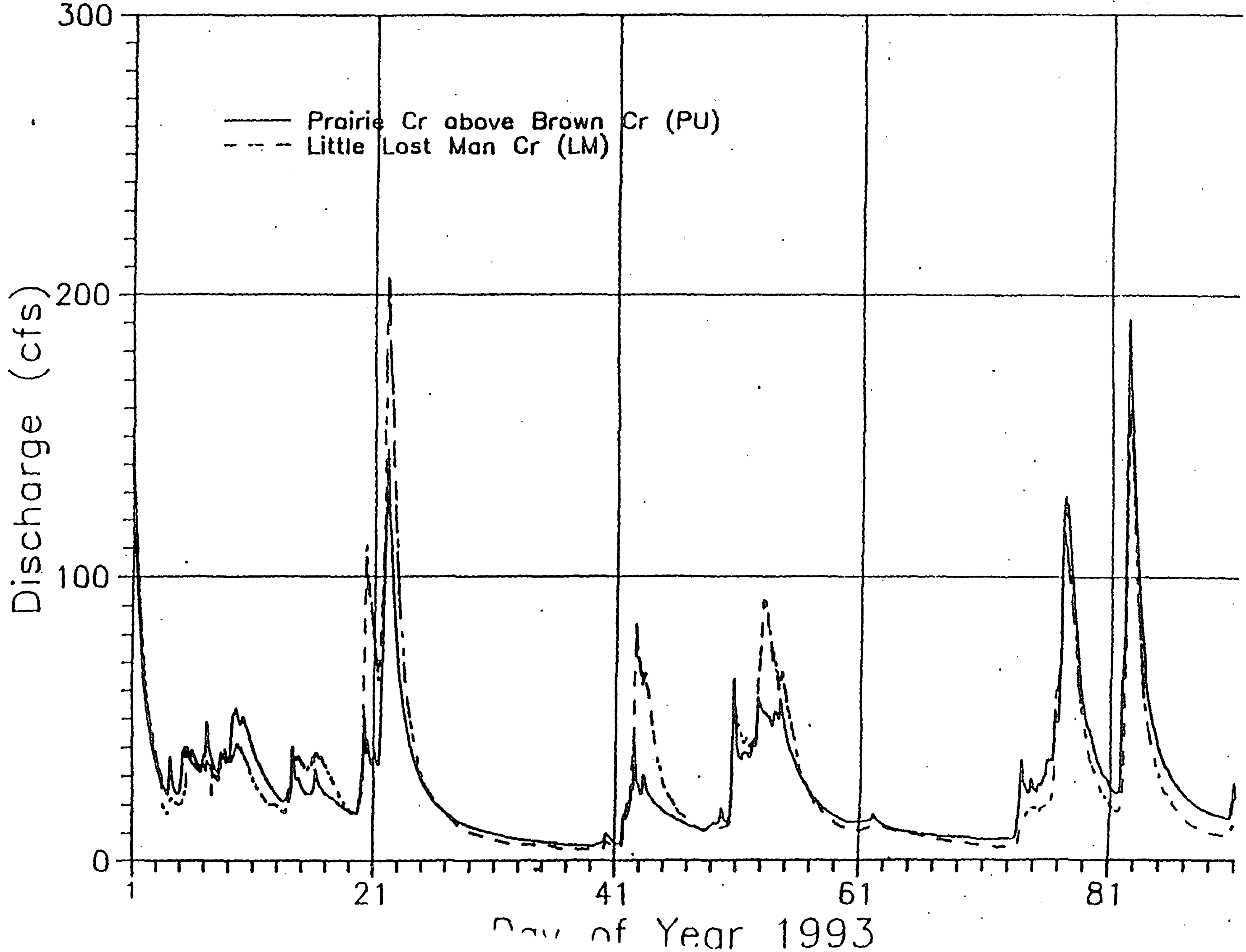


Figure 3f. REDWOOD NATIONAL PARK Prairie Cr. Bypass Study Hydrograph 04/01/93 - 06/30/93

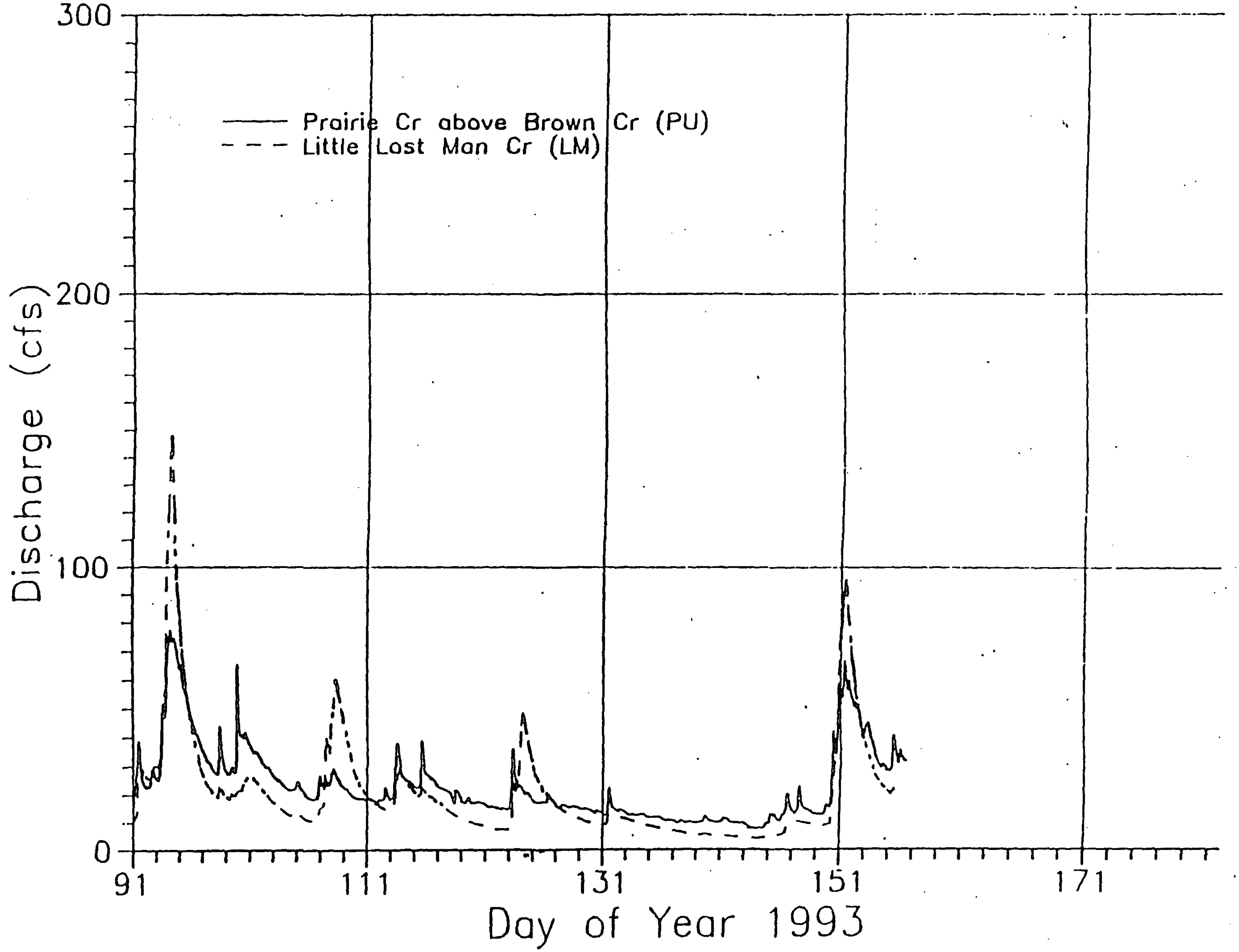


Figure 4a. Prairie Cr Drainage Cumulative Sediment Flux WY93

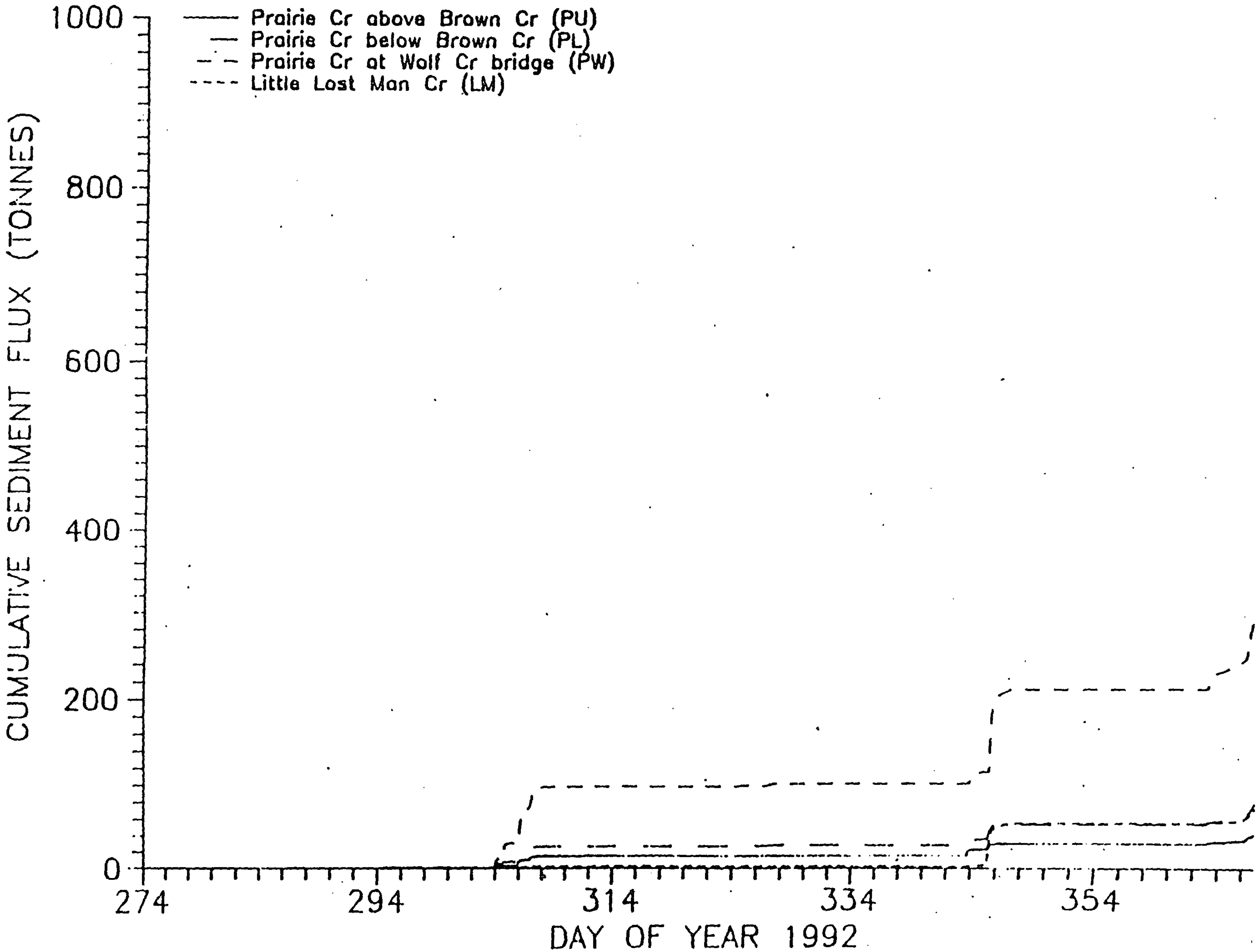


Figure 4b.

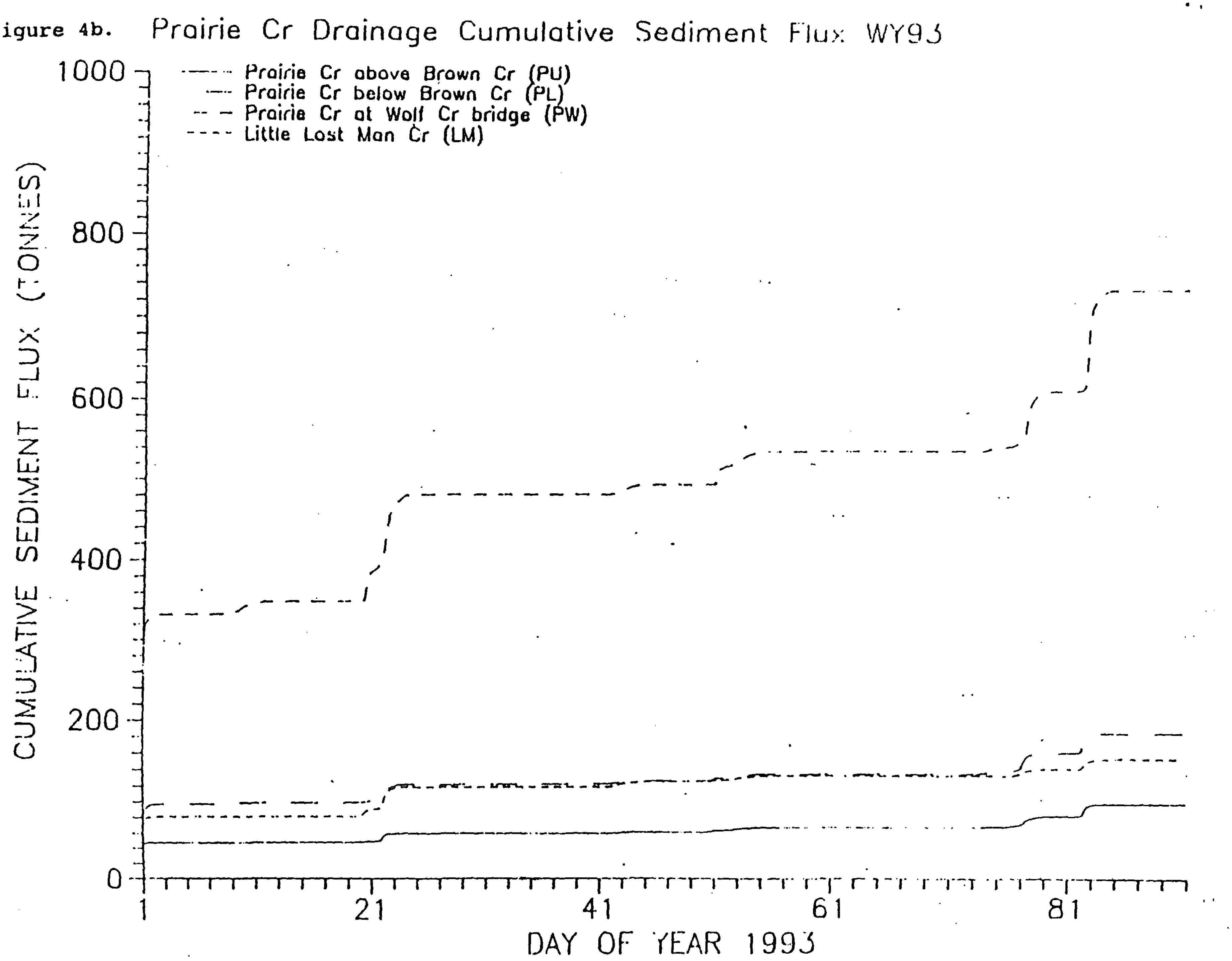
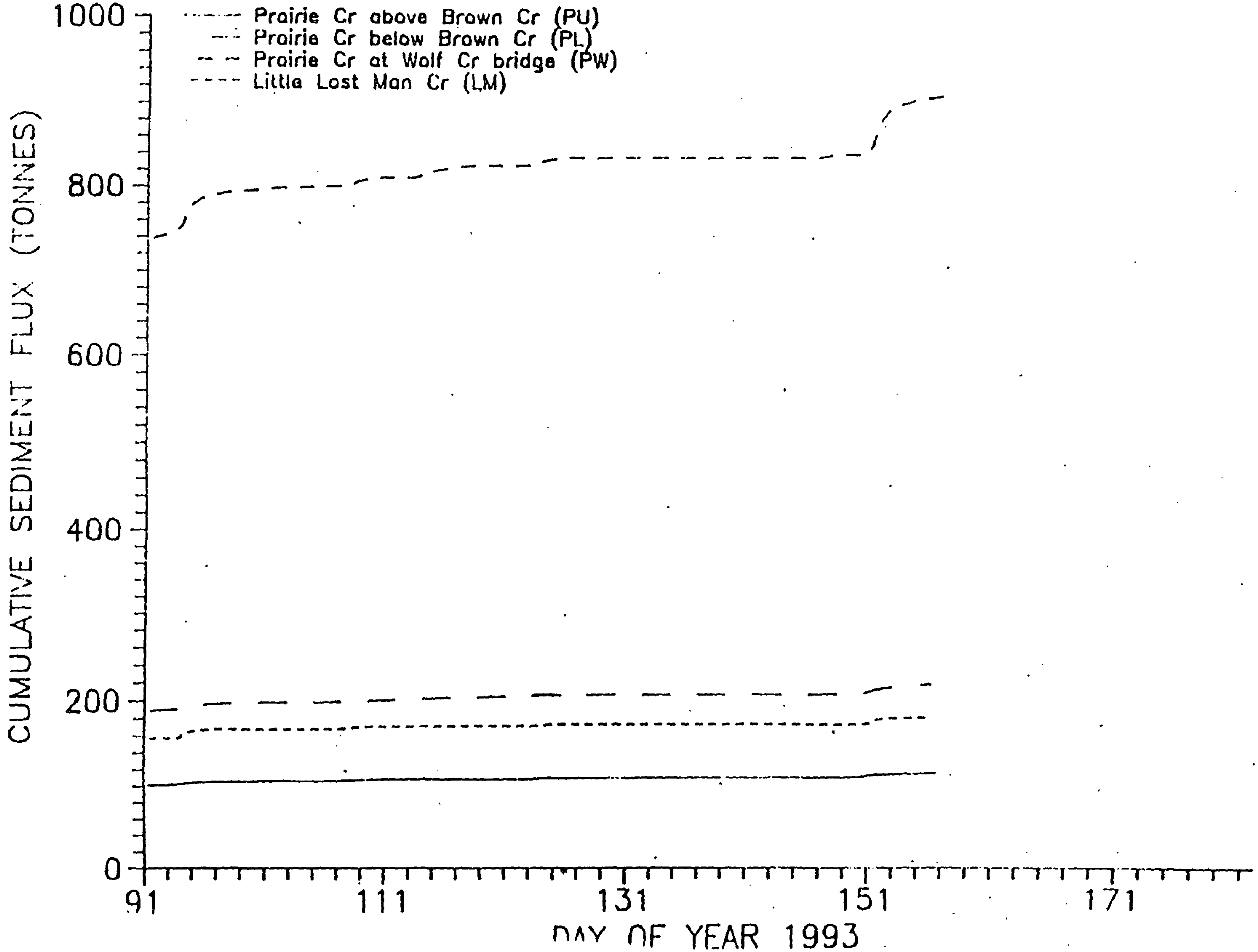

Prairie Cr Drainage Cumulative Sediment Flux WY93
Prairie Cr above Brown Cr (PU)
Prairie Cr below Brown Cr (PL)
Prairie Cr at Wolf Cr bridge (PW)
Little Lost Man Cr (LM)
CUMULATIVE SEDIMENT FLUX (TONNES)
1000
800
600
400
200
0
1
21
41
61
81
DAY OF YEAR 1993

Figure 4c. Prairie Cr Drainage Cumulative Sediment Flux WY93



III. EFFECTS ON REDD FINES, PERMEABILITY, DISSOLVED OXYGEN, AND COHO SALMON EGG SURVIVAL

A. Introduction

Results from the first three years of this study have shown that streamflows have been insufficient for flushing fine sediment deposited during the 1989 storm from the Prairie Creek streambed. However, small storms in WY91 were capable of moving enough sediment into redds to apparently affect egg survival. This portion of the study was designed to assess the effects of the bypass on egg survival and physical parameters in the redds should larger stormflows occur during the winter season.

B. Methods

Methods followed those used in WY92 (Meyer et al. 1994) except for the following changes: Only five artificial redds were constructed in WY93. Three were constructed in the affected reach, one in the control reach above Brown Creek, and one in Lost Man Creek. The sites selected were at the exact same location of an artificial redd constructed the previous year. This facilitated comparison between WY93 and WY92, despite the low sample size. Redds from the previous year that had close to average egg survival values were selected (site numbers 1,8,11,13,20). In the affected reach, one redd was placed in each of the following three strata: below Boyes Creek, below Big Tree Creek, and below Brown Creek (Figure 1). Baskets were placed in the redds on January 13, 1993 and removed from March 22 to March 25, 1993. Because they were redundant of the permeability pipes in the baskets, Mill Creek pipes were deleted from the artificial redd setup in WY93.

To determine egg mortality due to handling, two gravel-filled baskets containing eggs from the female salmon used in the study were placed in a trough receiving streamflow from an unimpacted tributary of Prairie Creek. These eggs were incubated until they hatched. Survival to hatching was averaged between the two baskets.

Approximately 700 eggs (500 eggs for artificial redds and 200 for handling mortality estimation) were removed from one female coho salmon on January 13. The remaining eggs were incubated in hatch troughs at the Pacific Coast Fish, Wildlife, and Wetlands Restoration Association (PCFWWRA--formerly called PCFFA) rearing site (Figure 1). Hatching success, number of days and temperature units to hatching were recorded for these eggs.

Because of WY93's higher, turbid spring flows, placement of fry traps on natural redds was delayed. We were only able to place four emergence fry traps in Prairie Creek in WY93, and all were in the affected reach. Two were below Boyes Creek and two were below Big Tree Creek. From downstream to upstream, the redds were first found on January 28, February 1, February 17, and January 8, 1993. Traps were placed on the redds on April 14.

Because sample sizes were low, statistical tests were not conducted to compare reaches in WY93. Instead, trends were compared to previous years. Data from WY91, WY92, and WY93 were used together to test for correlations, significant single and multiple stepwise regressions, and two-way analysis of variance (ANOVA) among years and affected vs. control reaches. WY90 data were not included because eyed rather than fertilized "green" eggs were used and permeability was not measured in the egg baskets. As in previous years, a difference or value was considered significant when $P < 0.05$.

C. Results and Comparison to Previous Years

Egg Survival. Mean egg survival-to-hatching in the handling mortality control baskets under clean-flowing water was 65%; on the control tray in the hatch trough, it was 63%. As in previous years, this indicated handling mortality was very low. We used the latter figure to adjust egg survival for fertility losses. Eggs hatched in the troughs on March 4, after 50 days and 689 temperature units. Eggs were incubated in each stream reach in slightly higher temperatures than the hatch troughs, ranging between 7° and 10° C. Eggs buried in the stream hatched after an average of approximately 68 days following fertilization.

In all three years, eggs hatched later in Prairie Creek than in hatch troughs. The latter were fed by a small tributary that had similar or slightly cooler temperatures than Prairie Creek. In WY91 and WY93, the difference was 15 (198 degree days) and 18 days, respectively. In WY92, the difference was much greater, 31 days (544 degree days). However, hatching date was based on when baskets were opened. Fry appeared to be two to three weeks old in the egg baskets by the time they were opened in WY92. In WY91, they had barely hatched, and in WY93, they appeared no more than a week old. Because they had just hatched, the WY91 difference of 15 days is the most accurate estimate of the difference in timing.

In WY93 egg survival ranged from 1.6% to 87.3% in the affected reach and from 20.6% to 59.5% in the control reaches. Mean egg survival was similar between the affected and control reaches and similar to WY92 values (Table 1). Survival was much lower in the redd below Boyes Creek in WY93 than WY92, but much higher for the redd below Brown Creek (Figure 5). In each redd, zero to 1/3 of the dead eggs observed had eyed, but between 50 and 75% of the dead eggs were missing.

Dead sac-fry were found in the baskets in WY93. They appeared to have died a few days earlier. No dead sac-fry were found in the control baskets. In the redd below Brown Creek, 10% of the hatched fry were dead; in the redd below Big Tree Creek, 36% were dead. Only two sac-fry survived to hatch in the redd below Boyes Creek, and both were found alive. Dead sac-fry were very rare in other years.

Comparison of egg survival values for impacted individual strata from WY90 to WY93 shows the reach below Boyes Creek had the lowest values and the reach below Brown Creek had the highest values. The control reaches often had lower survival than the affected reach (Figure 6). When only values for fertilized "green" eggs were included (only two redds below

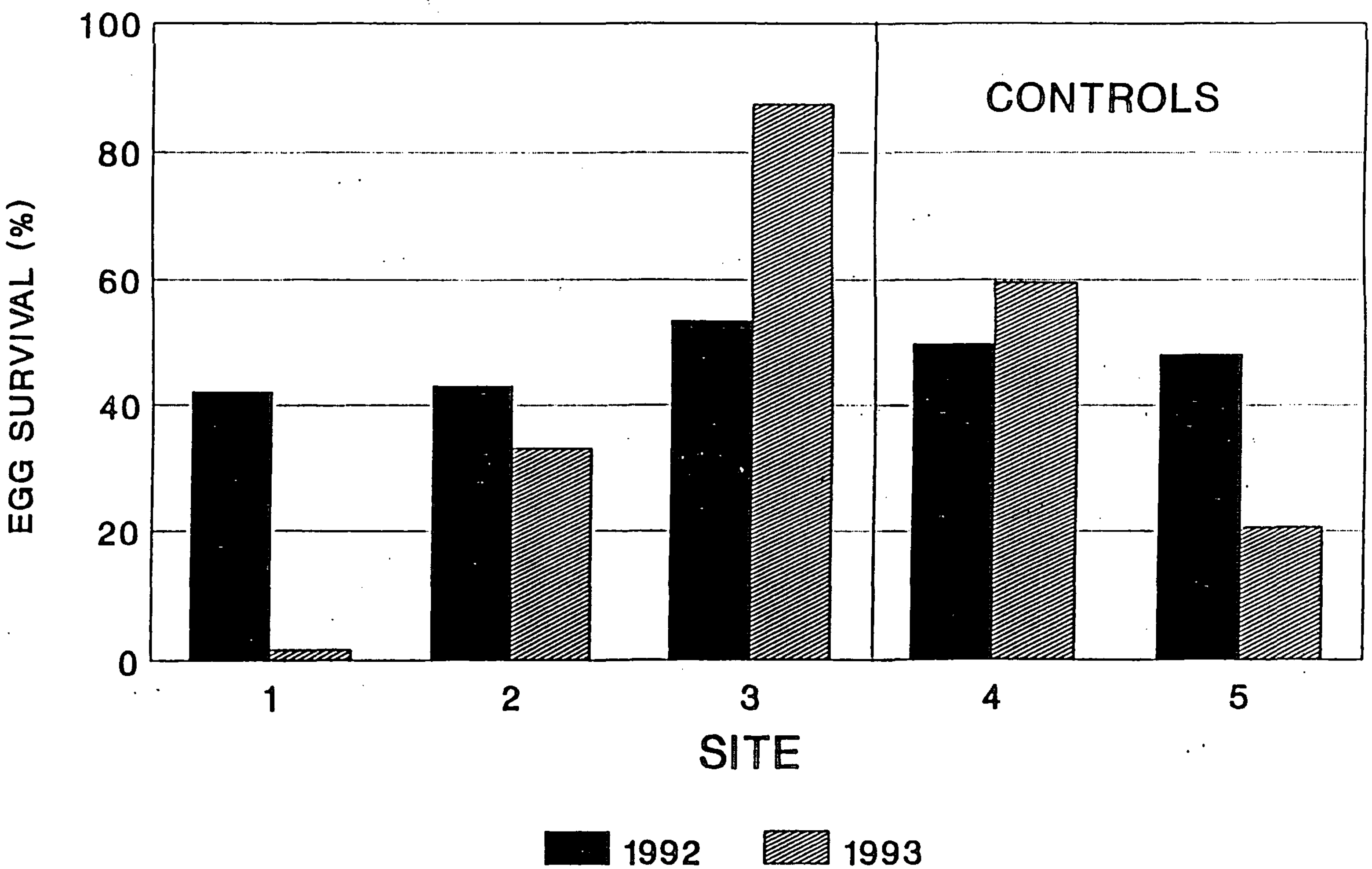


Figure 5. Comparison between WY92 and WY93 of coho salmon egg survival in 5 artificial redds created at the same location both years. Sites 1 - 4 were on Prairie Creek: 1 was below Boyes Creek, 2 was below Big Tree, 3 was below Brown, and 4 was above Brown. Site 5 was on Lost Man Creek control stream.

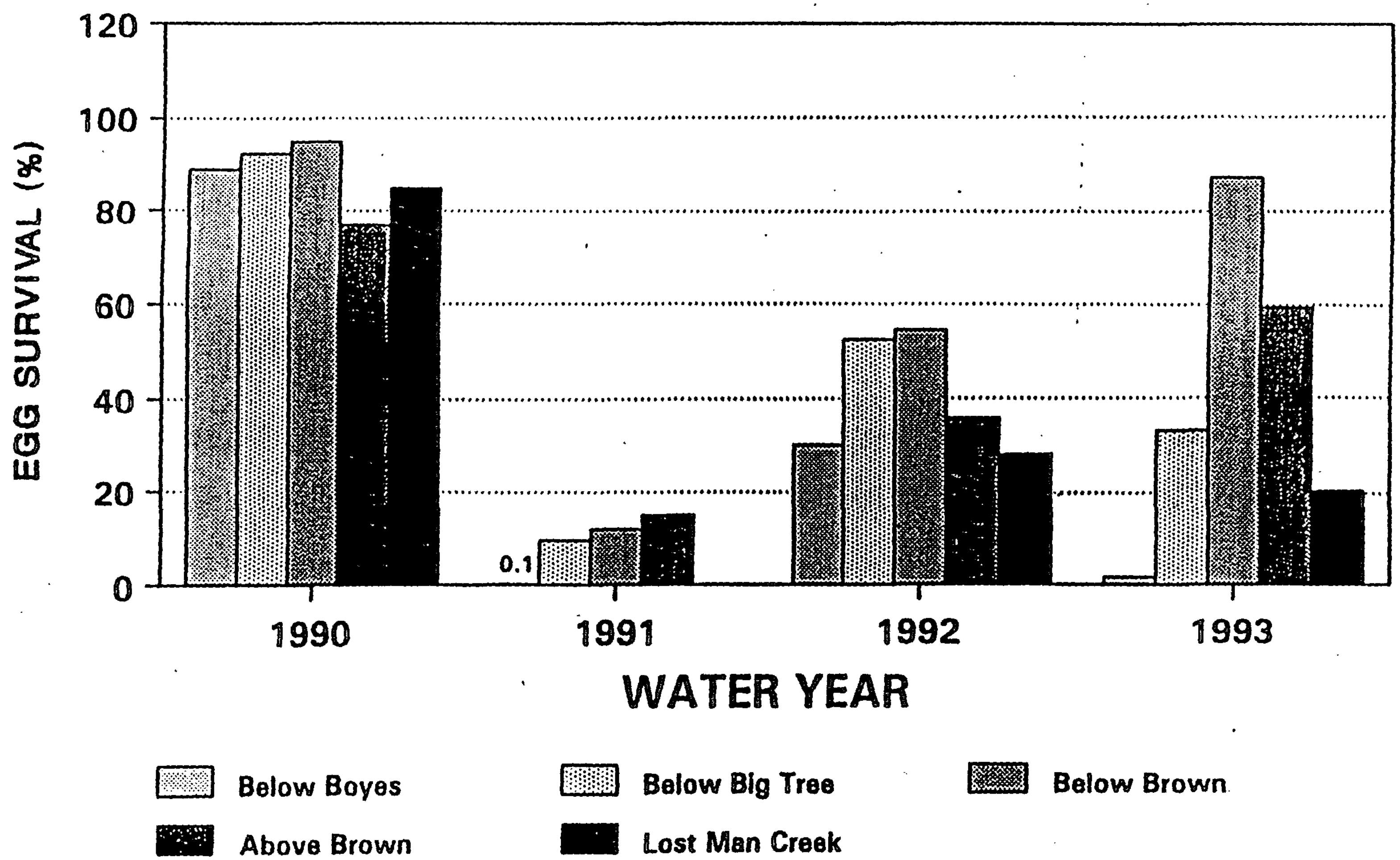


Figure 6. Mean survival of salmonid eggs in reaches on Prairie and Lost Man Creeks, WY90-WY93. Prairie above Brown Creek and Lost Man Creek were the control reaches. 1990 eggs were eyed steelhead eggs. Other years were green cohō salmon eggs. No Lost Man data exists

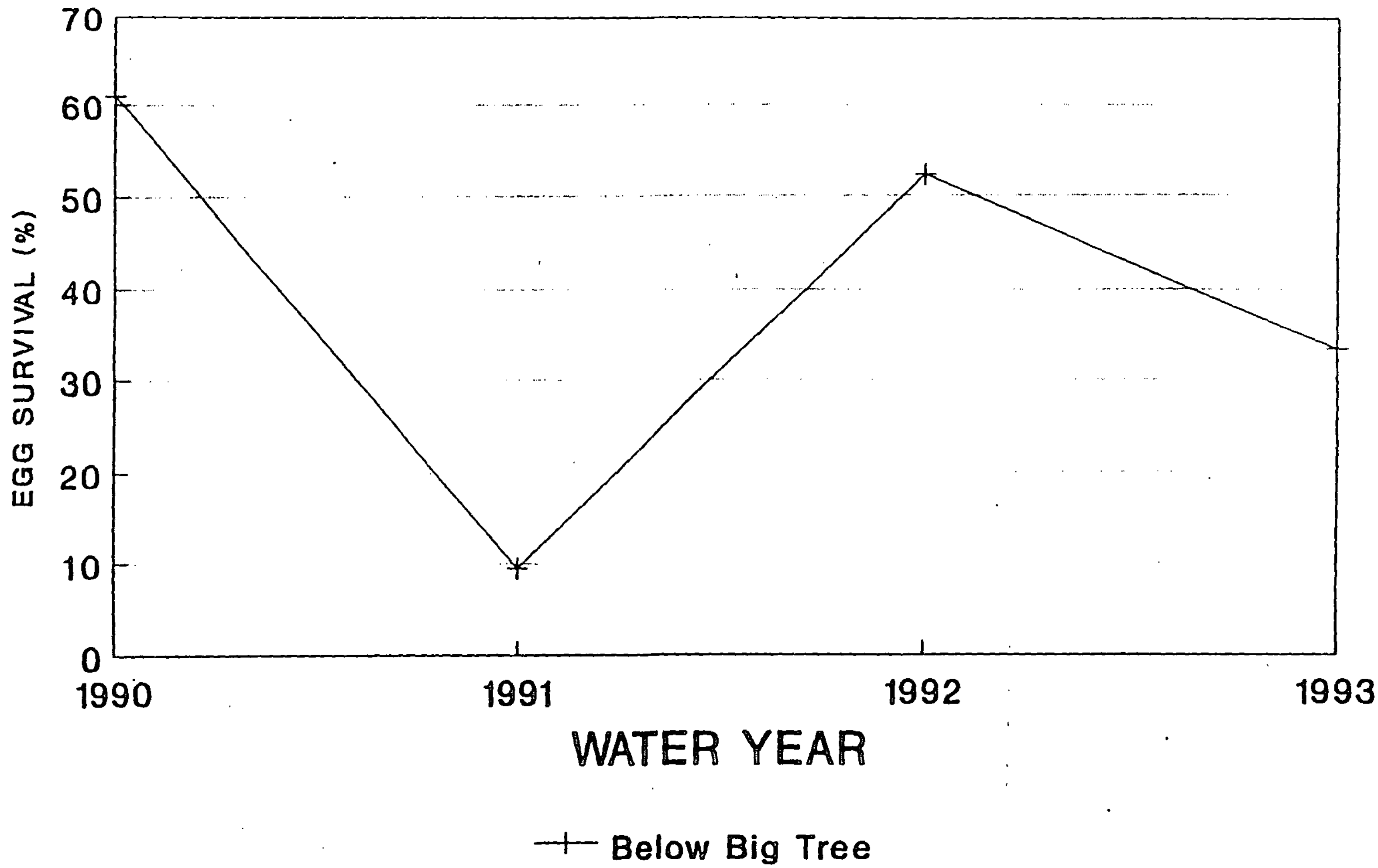


Figure 7. Comparison of egg survival between years on Prairie Creek below Big Tree Creek. All eggs were in place just after fertilization. 1990 eggs were steelhead; 1991 to 1993 eggs were coho salmon.

Big Tree Creek had green eggs in WY90), WY90 values below Big Tree Creek were similar to WY92 values in that reach (Figure 7). Egg survival was not determined for Lost Man Creek in WY91. Differences among years were greater than differences between control and affected reaches (ANOVA, $P < 0.001$ by year, $P = 0.696$ by effect, insignificant interaction $P = 0.083$). WY91 had lower survival than the other years (Duncans multiple range test, $P < 0.05$). Survival was significantly lower in the affected reach than the control only in WY91.

Table 1. Comparison between WY92 and WY93 of egg survival means in artificial redds on affected and control reaches of Prairie and Lost Man Creeks.

	Mean (%)	S.E.[a]	Mean (%)	S.E.
	Affected Reach		Control Reach	
WY92 (all sites)	43.5	4.8	32.2	4.9
WY92 (WY93 sites only)[b]	46.2	3.6	49.1	0.8
WY93	40.7	25.0	40.1	19.4

[a] Standard error.
[b] Includes only the 5 WY92 redds located at the same site as the 5 WY93 redds.

Worms. Only the artificial redd located below Boyes Creek contained worms in WY93. This redd was infested with large masses of worms in both egg baskets (132 g total), and egg survival was correspondingly very low (< 2%). In this most downstream reach, worms were found in one of five artificial redds in WY92, in all artificial redds in WY91, and in no redds in WY90. Other reaches in the study area on Prairie Creek (above May Creek) had no worms in any of the years except WY91, when infestations were high (Figure 8). Lost Man Creek had a small number of worms in one redd in WY92 and a few worms in a redd in WY90. On Prairie Creek below May Creek, worm infestations were high (2 out of 2 redds infested) in WY90, the only year redds were placed in that reach.

Percent Fines and Gravel Framework. Levels of fine sediment accumulated in the redds over the incubation period were much higher in WY93 than the WY92 redds in the same location (Figures 9-11). Means for affected and control reaches were higher than all previous water years (Figure 12). In WY93, mean fine sediment levels were not that different between reaches (Table 2), but the proportion of fine sediment composed of < 0.5 mm was much greater below Boyes Creek than below Brown Creek (Figure 11). Differences in fines were greater among years than between affected vs. control reaches (< 0.5 mm by weight, ANOVA, $P < 0.001$ by year, $P = 0.604$ by effect, significant interaction $P = 0.022$) (Figure 12). Fines significantly differed each year (Duncans test, WY90 not included). Fines were significantly higher in the affected reach than the control reach only in WY91. In WY91 this difference was greater after the effect of local variations in the gravel framework was removed (Meyer et al. 1994).

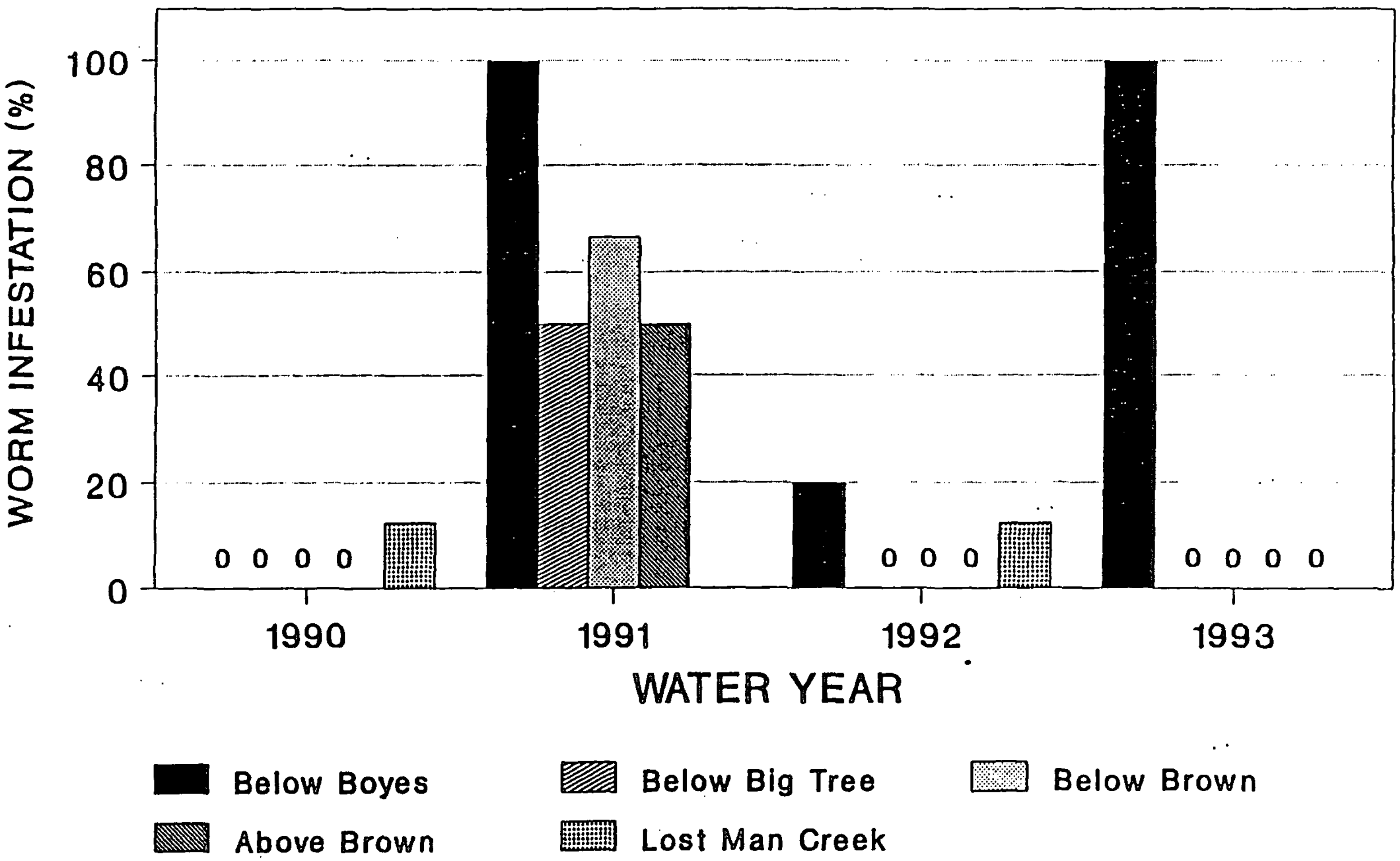


Figure 8. Comparison of worm infestation rates among years and stream reaches on Prairie and Lost Man Creeks. Prairie Creek above Brown and Lost Man Creek were the control reaches. The reach below Boyes starts above May Creek. No data exists for Lost Man Creek WY91.

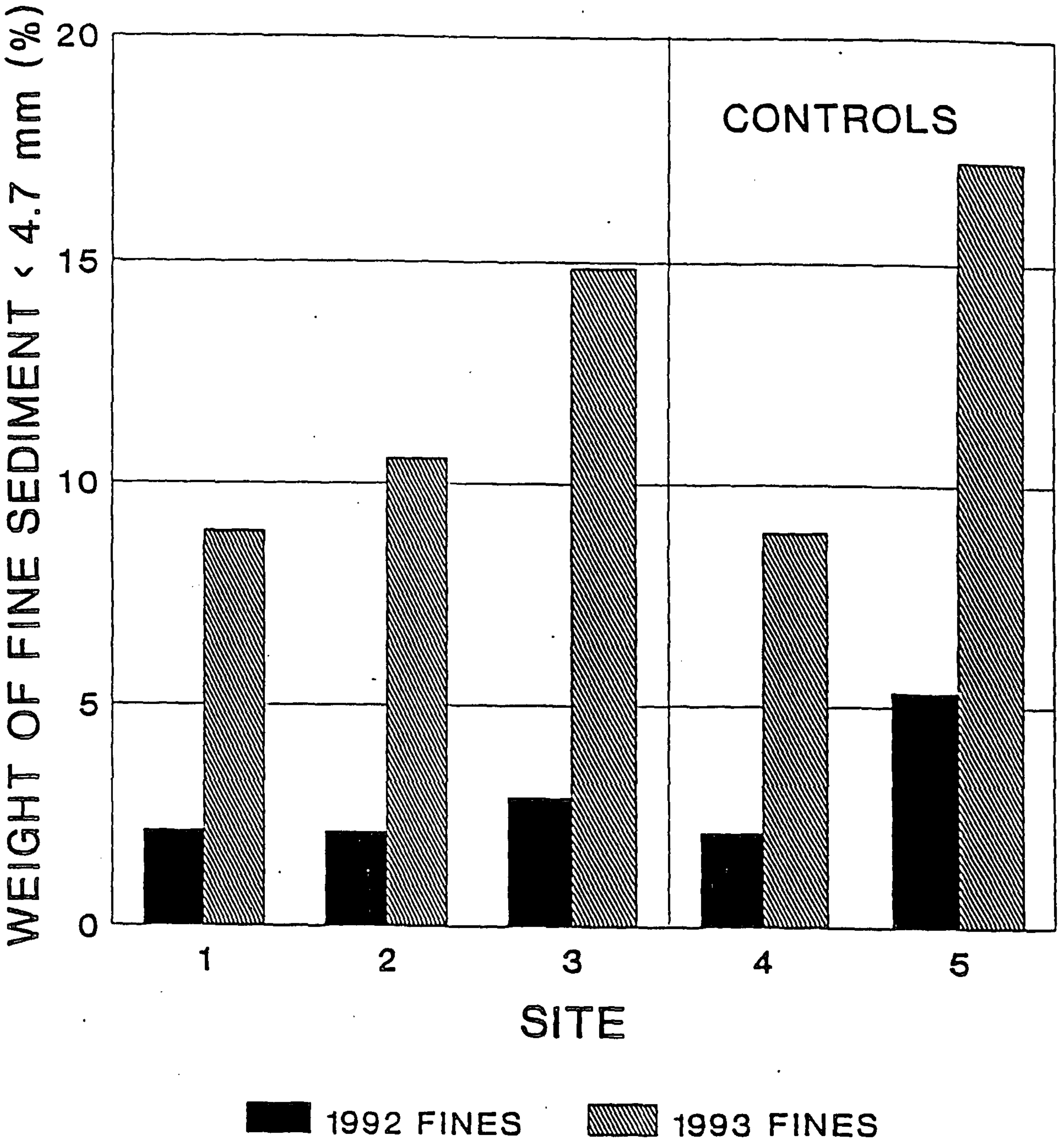


Figure 9. Comparison between WY92 and WY93 of fine sediment <4.7 mm in 5 artificial redds created at the same location both years. Sites 1-4 were on Prairie Creek; 1 was below Boyes Creek, 2 was below Big Tree, 3 was below Brown, and 4 was above Brown. Site 5 was on the Lost Man Creek control stream.

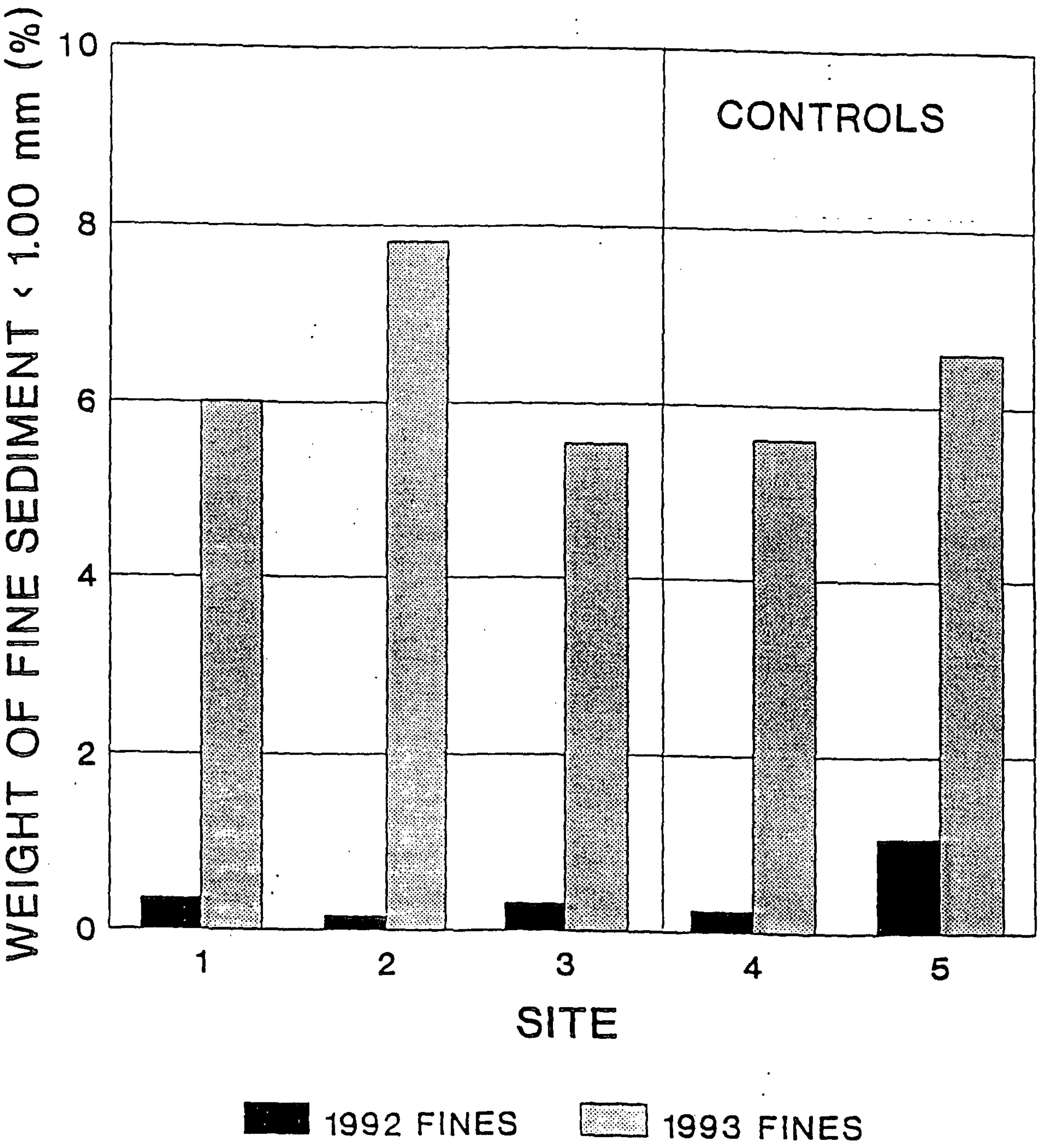


Figure 10. Comparison between WY92 and WY93 of fine sediment <1.00 mm in 5 artificial redds created at the same location both years. Sites 1-4 were on Prairie Creek; 1 was below Boyes Creek, 2 was below Big Tree, 3 was below Brown, and 4 was above Brown. Site 5 was on the Lost Man Creek control stream.

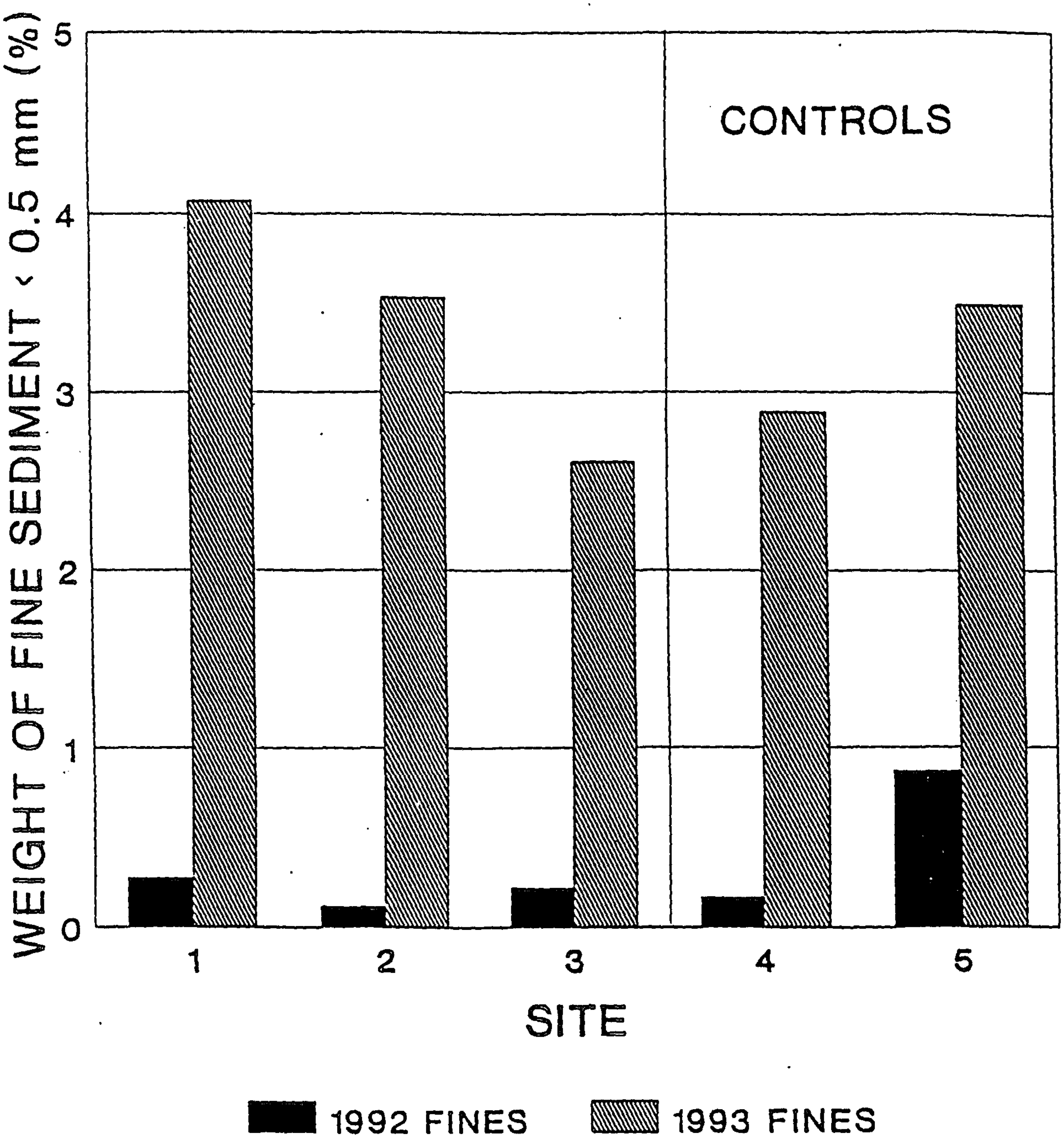


Figure 11. Comparison between WY92 and WY93 of fine sediment <0.5 mm in 5 artificial redds created at the same location both years. Sites 1-4 were on Prairie Creek; 1 was below Boyes Creek, 2 was below Big Tree, 3 was below Brown, and 4 was above Brown. Site 5 was on the Lost Man Creek control stream.

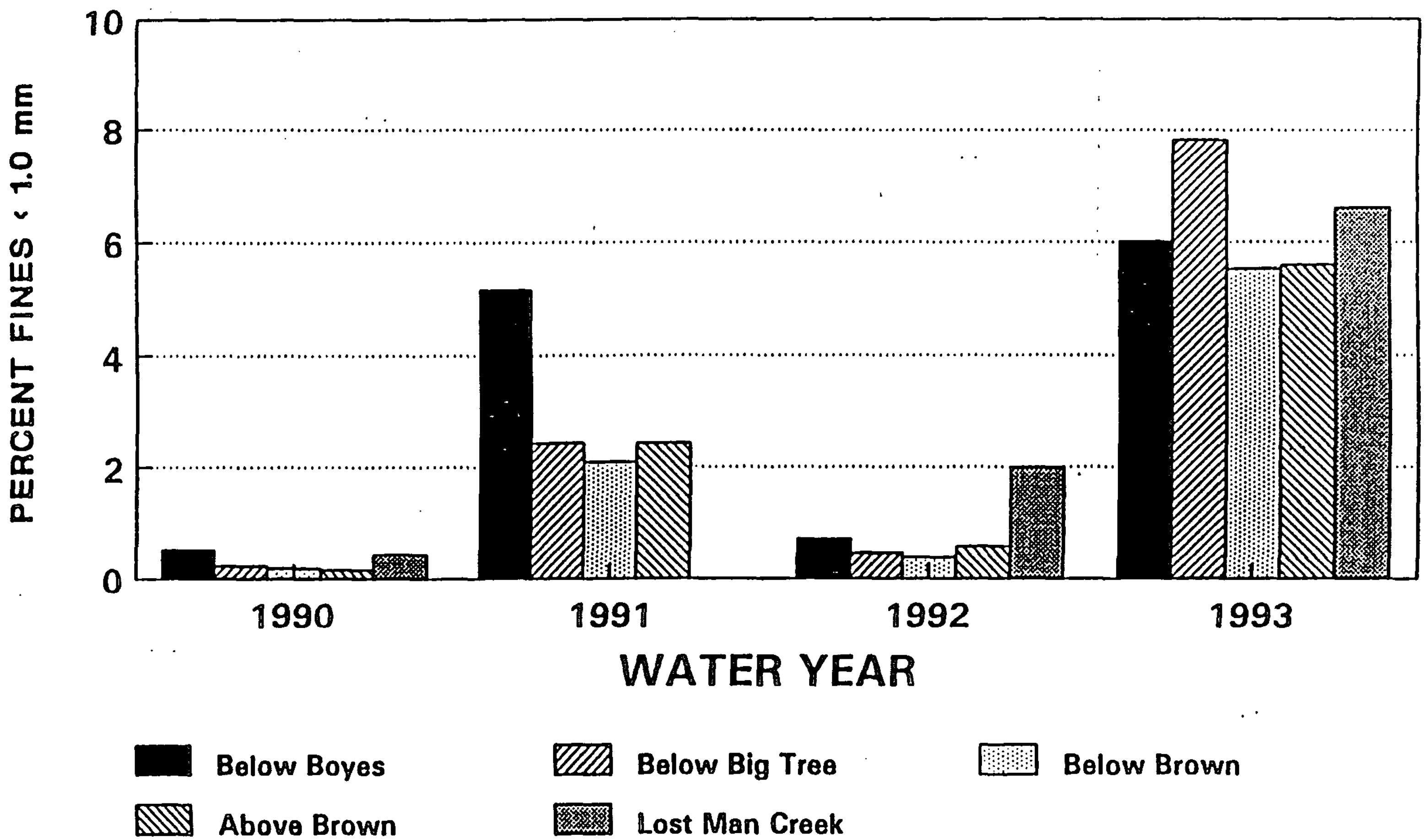


Figure 12. Mean weight of fine sediment < 1.0 mm in reaches of Prairie and Lost Man Creek, WY90-WY93. Prairie above Brown Creek and Lost Man Creek were the control reaches. No Lost Man data was collected for WY91.

Similar to WY92, geometric mean of the gravel framework before fines infiltration was lower in the control reaches than the affected reaches (means of 34.4 vs. 46.4 mm in WY93), so removal of this difference was necessary.

Table 2. Comparison between WY92 and WY93 of percent fines in hatching baskets of artificial redds on the affected and control reaches of Prairie and Lost Man Creeks.

HATCHING BASKET FINES (%)					
		AFFECTED REACH		CONTROL REACHES	
YEAR	PARTICLE SIZE (mm)	MEAN	S.E.	MEAN	S.E.
BY WEIGHT					
WY92	< 4.7	2.01	0.28	3.86	0.70
WY93	< 4.7	11.45	1.73	13.09	4.17
WY92	< 1.0	0.92	0.12	1.75	0.02
WY93	< 1.0	6.45	0.70	6.10	0.50
WY92	< 0.5	0.73	0.08	1.27	0.20
WY93	< 0.5	3.40	0.43	3.19	0.30
BY VOLUME					
WY92	< 4.7	10.03	0.81	11.73	1.44
WY93	< 4.7	28.32	1.60	27.05	0.60
WY92	< 1.0	8.95	0.70	12.15	1.49
WY93	< 1.0	24.13	2.46	20.87	2.98
WY92	< 0.5	8.76	0.67	14.08	1.74
WY93	< 0.5	21.65	2.14	18.30	1.41

Permeability. In the affected and Lost Man Creek control reaches, the inflow rate decreased over the WY93 incubation period. It did not change in the control reach on Prairie Creek. Decreases were greater below Big Tree and Boyes Creeks than WY92 (Figure 13). Although the change was greater in the affected reach in WY93, the final inflow rate in this reach was higher than the control reaches (Table 3). The larger gravel framework in the affected reach may account for this. Final inflow rates were relatively high in the affected reach in WY92 and WY93. All the reaches showed decreases in permeability over the incubation period every year except the control reach on Prairie Creek (Table 4).

Table 3. Comparison of mean inflow rates (cm^3/sec) in artificial redds in the affected and control reaches between WY92 and WY93.

	WY92				WY93			
	Affected Reach		Control Reaches		Affected Reach		Control Reaches	
Permeability	Mean	S.E	Mean	S.E	Mean	S.E	Mean	S.E
Initial January Rate	106.6	3.6	111.0	3.9	109.7	1.7	86.5	1.4
Final March Rate	96.4	2.9	90.9	2.9	87.0	3.4	74.4	2.4
Mean Change in Rate	-10.2	4.1	-20.2	3.7	-21.7	1.7	-12.1	1.4

Table 4. Mean change in inflow rates at the end of the incubation period in artificial redds on individual stream sections, WY92 to WY93. Prairie Creek reaches are listed in order from upstream to downstream.

	Change in Inflow Rate (cm^3/sec)			
Stream Reach	WY93	WY92	WY91[a]	WY90[b]
Lost Man Creek	-24.2	-25.4	--	--
Prairie Creek above Brown Creek	0.0	-15.0	+5.3	-15.7
Prairie Creek below Brown Creek	-5.8	-23.2	-16.2	-26.4
Prairie Creek below Big Tree Creek	-31.1	-6.2	-17.6	--
Prairie Creek below Boyes Creek	-28.3	-17.1	-43.0	-15.2

[a] For at least half the baskets in all reaches, initial permeability was measured after a large storm had changed conditions in the redd.
[b] Initial permeability was measured after a storm had already affected the redds. Rates in the affected reach were measured in 4 artificial redds (Mill Creek pipes) below tributaries, but not in WY90 egg baskets, which were exposed to fewer and smaller storms.

With the exception of the reach below Big Tree Creek, final rates were lower in WY91 than other years (Figure 14). They were also lower in WY90 in the affected reach (Appendix A, Table A-3); however, WY90 pipes were not in the egg baskets (and thus are not included in Figure 14). They were placed in clusters of 4 directly below tributaries, and were exposed to

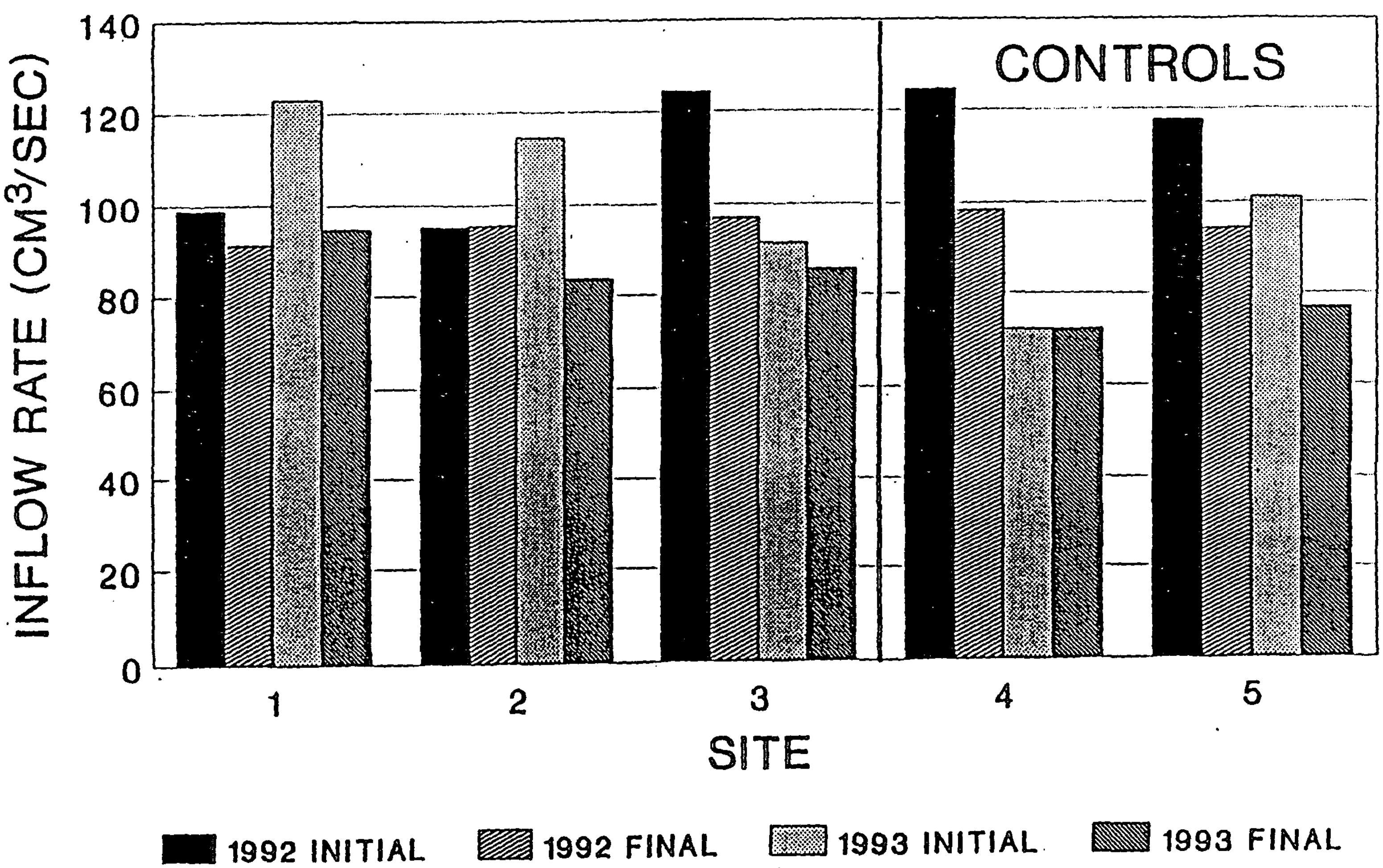


Figure 13. Comparison between WY92 and WY93 of permeability at the beginning and ending of the incubation period in 5 artificial redds created at the same location both years. *Sites 1-4 were on Prairie Creek;* 1 was below Boyes, 2 was below Big Tree, 3 was below Brown, and 4 was above Brown. Site 5 was on Lost Man Creek.

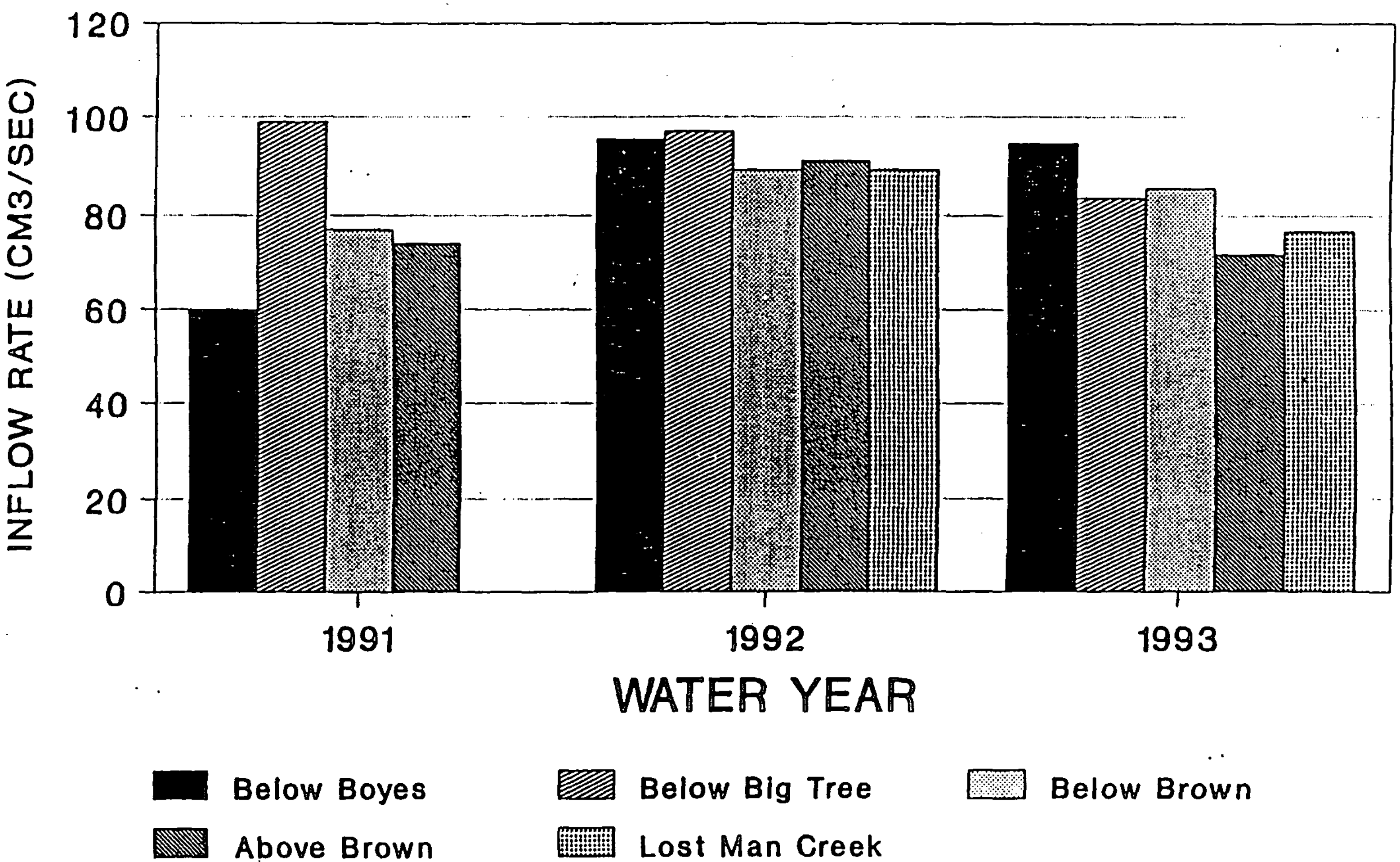


Figure 14. Mean permeability post-incubation period in artificial redds in Prairie and Lost Man Creeks, WY91-93. Prairie Creek above Brown and Lost Man were the control reaches.

larger storms than the egg baskets. Final inflow rates varied more among years than between control and affected reaches (ANOVA, $P = 0.003$ by year, $P=0.843$ by effect, insignificant interaction $P = 0.278$)

Dissolved Oxygen. Mean DO in the buried redd baskets only slightly decreased and never dropped below 11 ppm in any of the redds in WY93 (Figure 15). Final intragravel DO was slightly higher than the previous year and similar to surface water DO (Table 5). DO was also similar between individual stream sections and was relatively high all years (Figure 16 and Appendix A).

Table 5. Comparison of affected and control reaches mean dissolved oxygen concentrations (ppm), initially and after the incubation period in artificial redds and adjacent surface water in WY92 and WY93.

INITIAL DISSOLVED OXYGEN (ppm)								
	WY92				WY93			
REACH	REDD	S.E.	SURFACE	S.E.	REDD	S.E.	SURFACE	S.E.
AFFECTED	12.0	0.1	11.9	0.1	11.7	0.07	11.5	0.3
CONTROL	12.3	0.2	12.4	0.3	12.1	0.10	12.1	0.2

FINAL DISSOLVED OXYGEN (ppm)								
	WY92				WY93			
REACH	REDD	S.E.	SURFACE	S.E.	REDD	S.E.	SURFACE	S.E.
AFFECTED	10.7	0.1	11.2	0.1	11.4	0.01	11.6	0.07
CONTROL	10.6	0.1	11.5	0.1	11.5	0.07	11.7	0.05

Correlations with Egg Survival and between Variables. Using WY93 data alone, egg survival was highly negatively correlated to weight of fine sediment < 0.5 mm ($r = -0.98$, $P = 0.028$). Other correlations were not significant with the small sample size ($n = 5$). When means for all parameters measured for each redd from WY91, WY92, and WY93 were combined in a dataset ($n = 46$), the strongest correlation was between egg survival and worm abundance (rated high, medium, low, or none) ($r = -0.62$, $P < 0.001$). The correlation between survival and fines was not very strong and was highest with weight of fines < 0.5 mm ($r = -0.49$, $P < 0.001$) (Appendix B); fines of coarser size classes had much weaker correlations with egg survival ($-r < 0.37$). When WY93 data were excluded, the correlation with fines < 0.5 mm strengthened ($r = -0.72$, $P < 0.001$). When Lost Man Creek data were excluded, it increased further ($r = -0.80$, $P < 0.0001$, $n = 36$). Ending permeability over all three years was weakly correlated to egg survival ($r = 0.42$, $P = 0.002$). Correlations were not significant with change in permeability. Cumulative sediment flux over the incubation period explained very little of the variance in egg survival (15%, $r = -0.4$, $P = 0.007$).

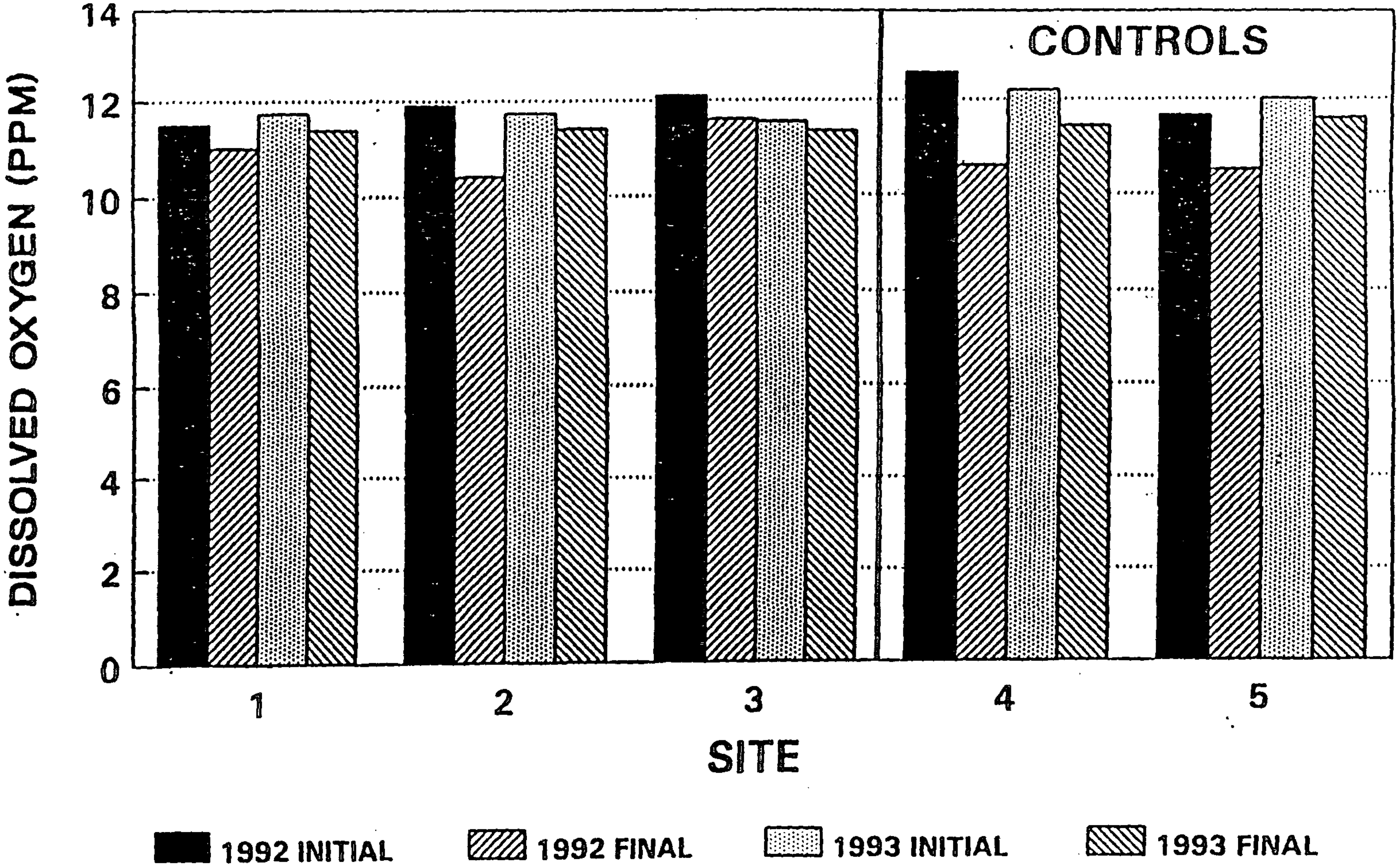


Figure 15. Comparison between WY92 and WY93 of dissolved oxygen at the beginning and ending of the incubation period in 5 artificial redds created at the same location both years. Sites 1-4 were on Prairie Creek; 1 was below Boyes, 2 was below Big Tree, 3 was below Brown, and 4 was above Brown. Site 5 was on Lost Man

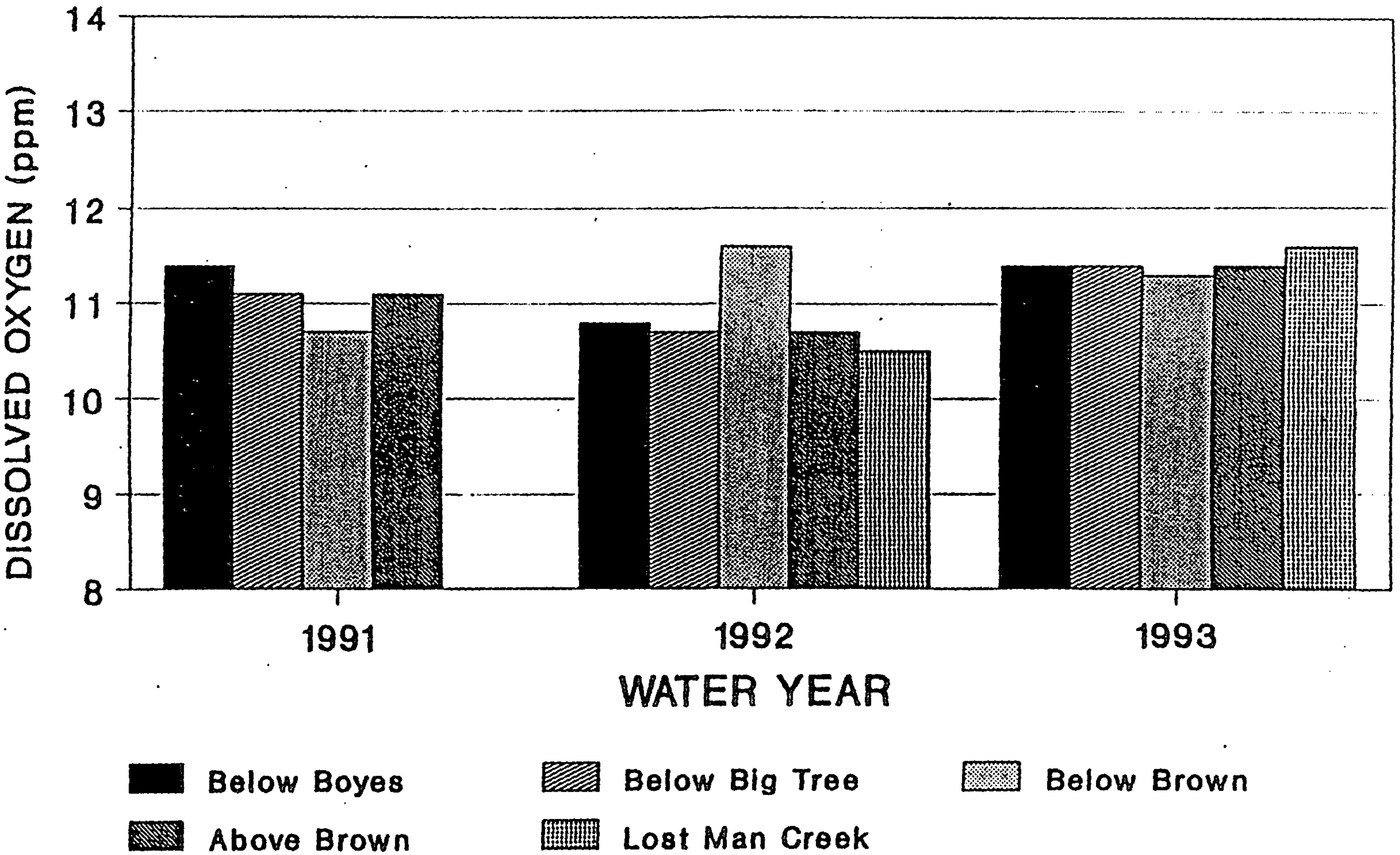


Figure 16. Mean dissolved oxygen post-incubation period in artificial redds in Prairie and Lost Man Creeks, WY91-93. Prairie Creek above Brown and Lost Man were the control reaches.

In a stepwise multiple regression, worms and weight of fines < 0.5 mm were selected and explained 42% of the variance in egg survival (multiple $r = 0.67$, $P < 0.0001$) over all three years. When all redds with worms were removed, correlations between fines and survival were not significant.

After a period of time, worms appear to leave an egg basket, moving to another basket or redd nearby in the same reach. To account for this worm mobility, the entire stream reach (stratum) between tributaries was coded for degree of worm infestation from 0 to 3, 0 being no worms, and 3 being worms found in most redds in that reach. This coded variable was significantly negatively correlated to egg survival ($r = -0.80$, $P < 0.0001$) and explained 63% of the variance. For just the affected reach, this variable accounted for 76% of the variance in egg survival ($r = -0.88$, $P < 0.0001$, $n = 25$). Other variables added little to the variance explained when entered into a multiple regression.

Weight of fine sediment < 0.5 mm was significantly but weakly correlated with abundance of worms over the three years ($r = 0.47$, $P = 0.0017$). Worm infestation coded by stream reach was more correlated to fine sediment ($r = 0.63$, $P < 0.0001$, 31% of variance accounted for). Other variables were not selected in a multiple regression on worm abundance.

Peak stream discharge measured at the nearest gaging station during the incubation period was highest in WY93 and lowest in WY90 (Figure 17). Percent fines (by weight and arcsine square root transformed) was regressed on peak stream discharge during the incubation period for WY91 to WY93. Peak discharge accounted for 77% of the variance in fines < 0.5 mm ($r = 0.87$), 65% for < 1.0 mm, and 53% for < 4.7 mm. When redd fines < 1.0 mm were averaged for each individual stream section for WY90 to WY93 (values from Appendix A: Table A-2), and regressed on peak discharge, 71% of the variance was accounted for (Appendix B: Figure B-2, 1st plot). The slope of the regressions increased from 0.01 for < 0.5 mm to 0.02 for < 1.0 mm to 0.03 for < 4.7 mm (Appendix B: Shows data not transformed).
Cumulative sediment flux in Prairie Creek during just the incubation period was less correlated to percent weight of fine sediment < 0.5 mm, < 1.0, and < 4.7 mm in artificial redds ($r = 0.71$, 0.60, and 0.51, respectively, $P < 0.0005$, $n = 41$) than peak stream discharge vs. fine sediment. In a multiple regression with sediment flux, geometric mean of the gravel framework did not explain any additional variance in redd fines. Ending permeability and fine sediment were weakly negatively correlated ($r = -0.43$, $P = 0.0045$). Change in permeability was weakly negatively correlated with sediment flux ($r = -0.47$, $P = 0.0019$).

Natural Redd Fry Emergence. Only the most upstream natural redd produced fry (399); no fry emerged from the other three redds (Figure 18). The fry were chinook salmon. The trap on the productive redd was partially blown out on April 26 and June 3. The next downstream trap was partially blown out on May 13. All traps experienced clogging of sand and debris. Pacific giant salamanders were found in two of the trap tubes (2,4) in late May and there were no fry in the tubes. Fry were emerging by April 19 (70 days) and stopped by May 19, 1993.

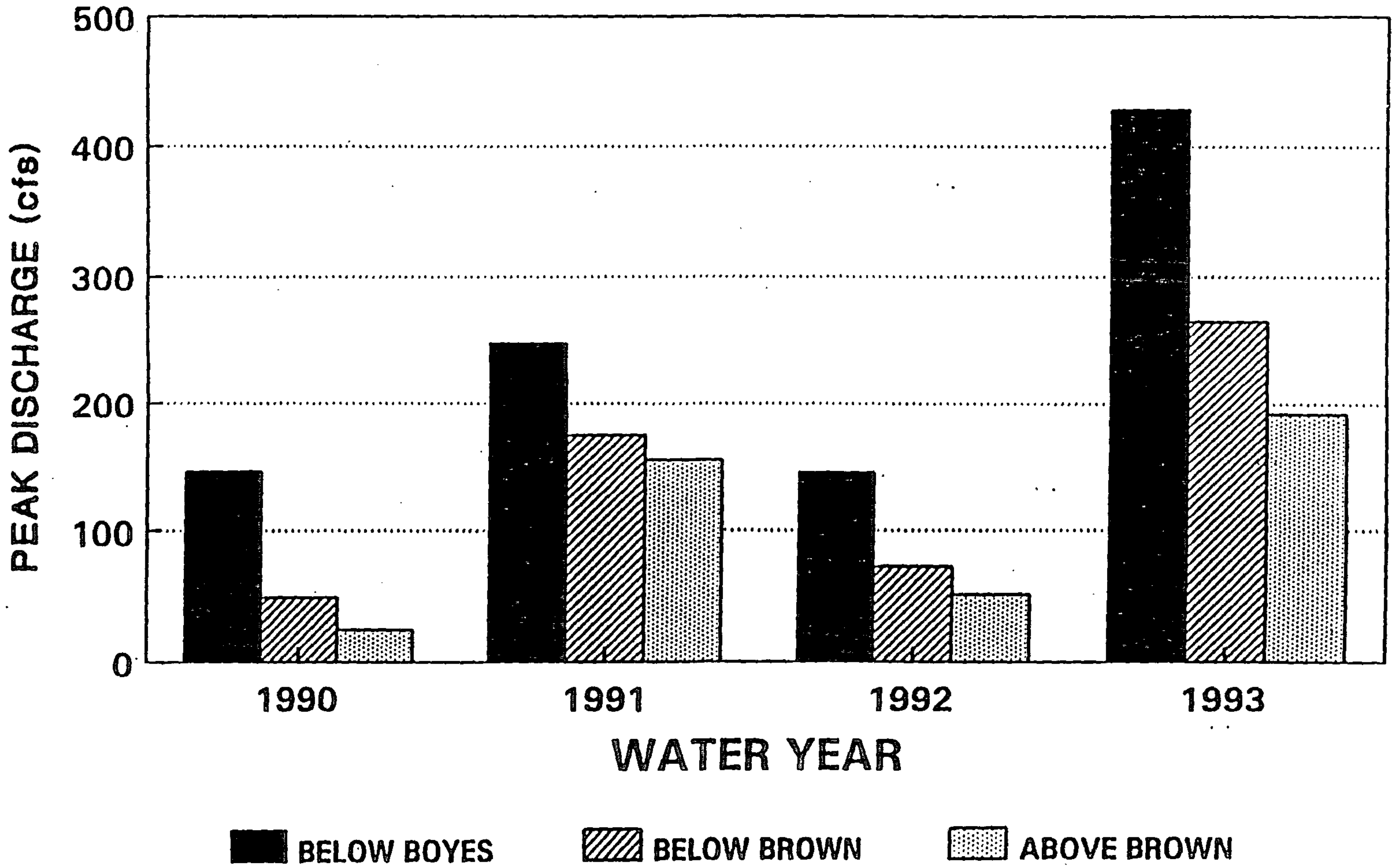


Figure 17. Peak stream discharge during the incubation period for artificial redds, WY90 to WY93. This graph shows the 1991 flows to which the hatching baskets and 5 emergence baskets were exposed.

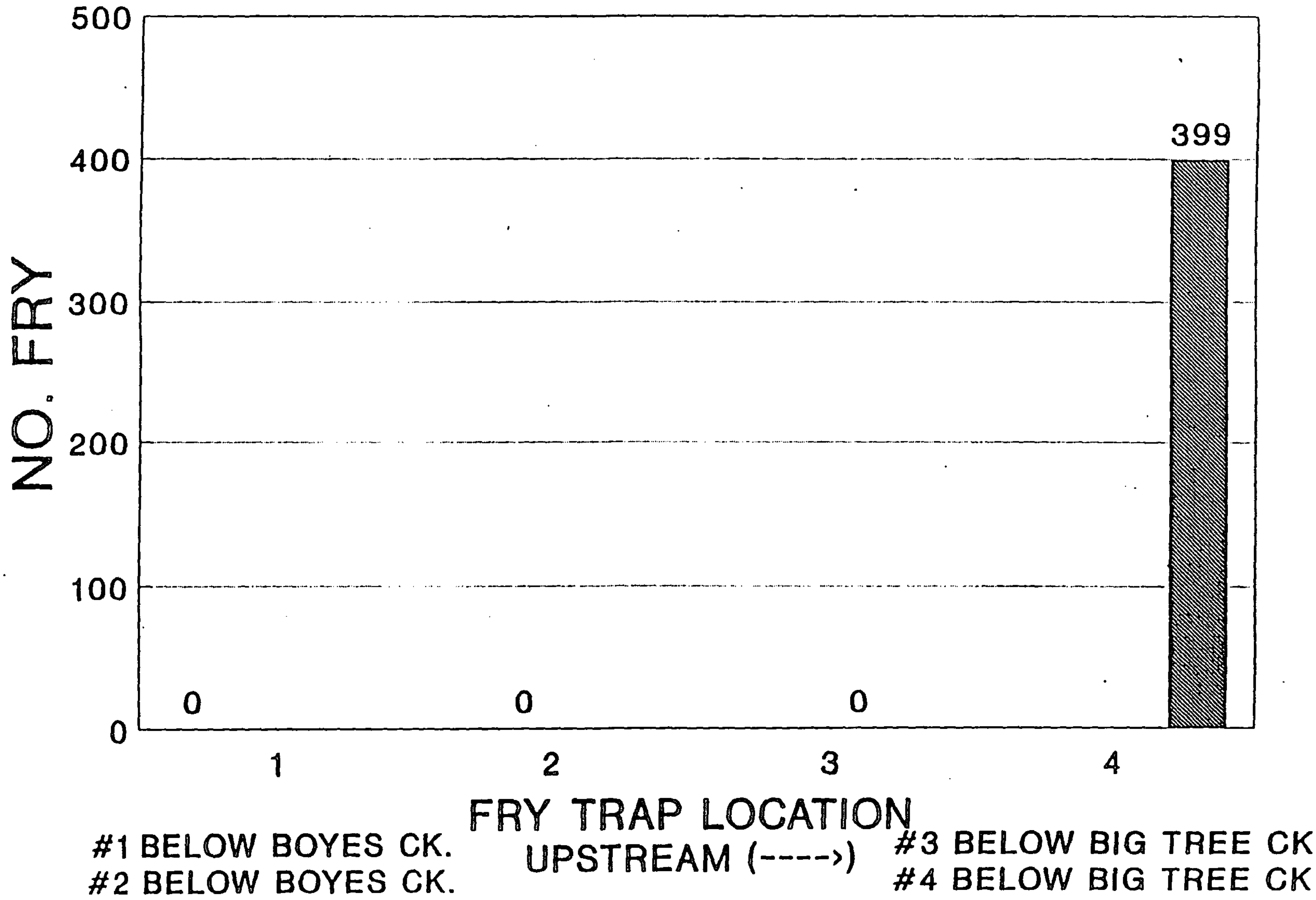


Figure 18. Number of fry that emerged from natural redds that were trapped on Prairie Creek, WY93.

Natural redd results from WY91 to WY93 show that redds trapped below Boyes Creek never produced any fry (Table 6). Results from the other reaches are more variable. When redds on the Lost Man Creek control reach produced fry, they usually produced more than a couple individuals (only one WY92 redd had fewer than 3 fry). In contrast, most of the time, Prairie Creek produced fewer than 3 fry per redd (including the control reach, which only had one redd produce more than 3 fry). Only one redd on Prairie Creek produced more than 1000 fry (redd from WY 91)

Table 6. Number of natural redds producing fry (numerator) of total number of redds trapped (denominator) in individual stream sections over the study period. Prairie Creek reaches are listed in order from upstream to downstream.

Ratio of Trapped Natural Redds Having Emerging Fry to Total Trapped Redds			
Stream Reach	WY93	WY92	WY91
Lost Man Creek	--	5/9*	3/3*
Prairie Creek above Brown Creek	--	3/3*	0/3
Prairie Creek below Brown Creek	--	0/1	2/2*
Prairie Creek below Big Tree Creek	1/2*	2/3	1/2
Prairie Creek below Boyes Creek	0/2	0/3	0/3

*Some or all of the redds produced more than 3 fry.

D. Discussion

D.1 Egg Survival Study Results

Caution is advised when comparing data among years. Sample sizes varied from year to year; results for years with small sample sizes are less reliable. Only one sample was taken for each of the individual stream strata for WY93 and for dissolved oxygen and permeability results in WY90 (see RNP (1991) and Meyer et al. (1994) for other sample sizes). Change in permeability is not comparable among all years because initial rates were measured directly after pipe installation only in WY92 and WY93. Initial rates were not measured until after the largest storm of the year had already affected the gravels in WY90. The same was true in WY91 for all the hatching baskets and five of the emergence baskets in the control stream. Additionally, permeability was not measured within the egg baskets in WY90, although it was measured in cleaned gravels. Also, WY90 beginning and ending permeability rates bracketed a period greater than the egg incubation period of that year or later years. This means the pipes were exposed to more storms, possibly resulting in lower ending rates. Finally, baseflows at which permeability is measured are difficult to keep constant from year to year to allow comparisons.

Flows and sediment flux were much higher in WY93 than previous years, and sediment infiltration into the redds was correspondingly higher. However, peak flows for all study years were low. The peak WY93 flow was only half of the flow of a 2-year recurrence interval stormflow, about 1 1/2 times larger than the next highest peak flow during the study, which was in WY91. Consequently, these study results are only applicable to relatively low flow stream conditions.

Of the three size classes tested, the finest class of sediment (< 0.5 mm) was the most correlated to egg survival. However, despite a 9-factor increase in this class of fine sediment from WY92 to WY93 in redds, egg survival did not sharply decline, except below Boyes Creek where worms were found. In fact, egg survival was high below Brown Creek. Change in permeability was small in this redd, contradicting the concept that permeability will be greatly reduced by high amounts of fines < 4.7 mm (almost 15% of the gravel weight was composed of fines that infiltrated during the incubation period). Corroborating this, correlations between fines and permeability over the study years were poor. Neither fine sediment nor permeability was a major controlling factor in egg survival; worm infestation was a much more important component.

Egg survival was more strongly correlated to fine sediment when just data from WY91 and WY92 were evaluated.. Those years, the combination of worms and higher fine sediment levels was detrimental. In years when very few worms were found in redds, fine sediment was not responsible for most of the egg survival variation. The combination of worms and sediment may be important. We noticed that when worms were abundant, a greater proportion of the fine sediment was within the basket compared to the infiltration bag. The worm mucus might "glue" more of the sediment to the eggs and gravel, preventing fines from washing out of the basket into the infiltration bag during the process of winching up the bag and egg basket. The mucus-laden fine sediment may have deprived eggs of oxygen.

Fine sediment did not appear to be related to timing of hatching. Timing was similar between affected and control streams and did not differ much in years having higher sediment flux. It is unknown why hatchbox eggs hatched sooner than stream eggs. Temperatures were similar or colder for the hatchbox eggs.

Dissolved oxygen measured with probes in pipes buried in the egg baskets did not explain any of the variance in egg survival. Microprobes measuring the DO in the environment immediately adjacent to the eggs would provide more accurate information. Permeability was not a good predictor of fines or survival. Future egg survival studies should focus on fine sediment infiltration, gravel framework characteristics, worm abundance, and egg survival in redds. If possible, fine sediment, biomass of worms, and egg survival should be measured in the same or adjacent artificial redds following each storm to determine fluctuations in these variables over the incubation period.

The dead sac-fry found in the egg baskets when they were opened in WY93 were probably killed during one of the season's larger storms that occurred several days earlier on March 19, 1993.

The high load of sediment in the redds at the time of hatching combined with another sediment dump into the redd from the March storm may have killed the newly hatched fry. Survival from hatching to emergence was not affected by sediment during lower flow years such as WY91, but these results may indicate an adverse effect on sac-fry in higher flow years.

Lost Man Creek had better fry production from natural redds than Prairie Creek, which was contrary to artificial redd results for most years. The assumption of an equal false redd rate between Prairie and Lost Man Creeks may have been wrong. Lost Man Creek receives larger steelhead trout runs than Prairie Creek (Haux and Anderson 1992), and Briggs (1953) found steelhead did not create false redds (trial redds without eggs). We may have trapped a greater proportion of steelhead redds on Lost Man Creek, and thus trapped fewer false redds on this creek (species of fish were not identified on many of the redds).

The finding that 50 to 75% of the redds on Prairie Creek over the 3-year period produced no fry matches Briggs' false redd rates for salmon. Assuming those redds were false redds, only one-third of the "true" redds had numbers greater than a few fry. Only 1/9 of these produced thousands of fry closer to the numbers probably laid (although the one in WY93 was trapped late and may have produced many before the trap was in place). Because the natural redds accumulated large amounts of sediment under the traps in WY92 and WY93, these fry emergence results may be much lower than levels produced from untrapped natural redds.

In all years, the number of fry produced from trapped redds below Boyes Creek was zero. This and the low survival rates in the artificial redds in this reach suggest that egg survival was unnaturally low below Boyes Creek.

WY91 egg survival results throughout Prairie Creek also seemed unnaturally low.

D.2 Impacts of the Bypass Sediment from the October 1989 Storm

To determine whether the bypass sediment from the October 1989 storm was responsible for the very low egg survival in WY91 and below Boyes Creek in the other years, three components need to be examined. First, comparable data collected before the storm should be compared to data collected after the storm. Second, affected stream reaches should be compared to unaffected stream reaches. Third, cause and effect relationships between sediment and egg survival need to be determined.

<u>Comparison of Pre-1989 Storm Data to Post-Storm Data and to Unaffected Streams</u>. Fine sediment in spawning gravels was documented in long-term monitoring sites on Prairie Creek and its tributaries and on Klamath tributaries in years before and after the October 1989 storm (Meyer and Haux--RNP bypass report in preparation). From 1985 to 1992, weight of fine sediment $<$ 1 mm in summer riffle crests was estimated for each reach using freeze-cores (Table 7).

Comparison of mean fines before the October storm shows that sediment levels were very similar on Prairie Creek among reaches used in the WY91 to WY93 egg survival studies (from above May to above Brown Creek). After the storm, only the reach above Brown Creek showed a significant increase in fine sediment. Although statistically insignificant, an increasing trend in fines was apparent on most (all but the most downstream) of the Prairie Creek sites (sample sizes are low for detecting differences).

Table 7. Mean percent of fines < 1.0 mm by weight found in potential spawning areas within summer riffle crests in bypass long-term monitoring sites, WY85-WY92. Except for below Boyes, only one site was located on each stream reach; six samples were taken from each site per year (Meyer and Haux (in preparation)).

STREAM REACH	BEFORE STORM Summer 1985 to 1989	AFTER STORM Summer 1990 to 1992	P
Prairie Above Brown	7.86	10.24	0.045
Prairie Below Big Tree[a]	7.11	8.13	0.123
Prairie Below Boyes[b]	7.84	9.16	0.259
Prairie Below May	10.89	12.79	0.335
Prairie Below Channelization	11.14	11.18	0.913
Lost Man	8.78	8.27	0.608
Brown	7.55	8.29	0.372
Boyes	9.76	9.99	0.812
May	13.39	19.45	0.290
Tarup	7.32	9.99	0.149
Ah Pah	10.30	10.84	0.650
McGarvey	16.03	20.12	0.108

[a] Also called Prairie above Boyes long-term monitoring site.
[b] Average of two sites--Prairie below Boyes and Prairie above May long-term sites.

In two-way analysis of variance among all years (1985 - 1992) and treatment (affected vs. control reach above Brown), fines significantly increased in a monitoring site just below Boyes Creek in conjunction with the increase above Brown (P = 0.025, no significant interaction). The increase was first most notable in 1990 (P = 0.052 with data from 1985 to 1990 only). Increases at other sites were not significant, including the other monitoring site in the reach bounded by Boyes and May Creeks.

An increase in subsurface fines after the storm was not evident in the Lost Man Creek control site, but neither was it evident in the sites in the impacted tributaries of Brown and Boyes Creeks. May Creek had the highest suspended sediment concentration during the October storm (12,100 mg/l, RNP 1991), which likely explains the apparent (not significant) increase in fines on that tributary. May Creek, however, is not considered in this report because impacts were expected and mitigated prior to the October storm. Unfortunately, only one site was monitored on each tributary, and trends on one site are likely not indicative of the entire tributary.

These data suggest (1) subsurface fines are increasing slightly on Prairie Creek mainstem, (2) subsurface fines on some sites on the lower portions of the tributaries of interest have not changed much, and (3) the Prairie Creek reach above Brown Creek was a poor control site. The latter point is important because the reach above Brown was used as a control site for the egg survival studies. We believed that impacts from Ten Tapo Creek sediment inputs would be small compared to inputs from tributaries farther downstream. However, the October storm delivered about 100 tons of suspended sediment into this reach, which was six times greater than sediment delivered during the following January storm (RNP 1991). More sediment would be expected to be transported naturally during the January storm than the October storm because the precipitation from the January storm was about three times greater than the October storm. Thus, the October storm clearly delivered a significant amount of unnatural bypass-derived sediment into the control reach. Although the suspended sediment discharge downstream of this reach was much greater (over 3 times), the lower reaches also have higher flows and greater ability to move sediment. Therefore, impacts could be similar throughout Prairie Creek. Changes over time on creeks completely unaffected by the bypass compared to Prairie Creek may provide a better comparison than the Prairie Creek "control reach" to determine impacts.

Egg survival, redd fines, permeability, and dissolved oxygen were monitored only on one stream unaffected by the bypass: Lost Man Creek. In general, trends in Lost Man Creek values for these parameters over the years were the same as in Prairie Creek. One could argue that the October storm caused Prairie Creek conditions to deteriorate to Lost Man Creek's level, since Lost Man Creek had been previously impacted by logging. Subsurface fines values on Lost Man Creek did not change after the storm, yet Prairie Creek, particularly in the reach below Boyes and above Brown Creek, tended to change. Although this supports that subsurface impacts occurred in Prairie Creek, it is of note that Tarup Creek, a creek unaffected by the bypass, also showed increases in subsurface fines following the October storm (Table 7). Tarup Creek may have been affected by logging activities, however. More pre-October storm data need to be evaluated to determine the extent of the subsurface impacts.

When freeze-core probes are pounded into the gravel, silt or clays may go in suspension and be lost from the sample. Also, most of these sized particles do not settle out in 45 minutes in the Imhoff cones. The long-term freeze-core data may underrepresent silt, the main size class observed during the October 1989 storm (RNP 1991). The freeze-core data may be documenting trends in sand more than silt. Trends in percent of area covered by surface sand vs. silt in the summer were documented on benthic habitat maps at the same long-term monitoring sites before and after the storm (Meyer and Haux, in preparation). We found that between 1988 and 1992, mean coverage by surface silt (dominant or second-most dominant substrate type) and thick vegetation (which grew on silty substrates and replaced benthic habitat in our sites over time) increased 23% (from 20 to 43%) on Prairie Creek below Brown Creek and 34% on affected Prairie Creek tributaries, but only 2% on Prairie above Brown Creek, 19% on Lost Man Creek, and 18% on Klamath tributaries. When sand was included, however, percent surface fines and vegetation did not change much on Prairie because sand decreased (silt was deposited on sands, but sands shifted, too). The exception was below May Creek where surface fines increased. Except for Lost Man Creek, surface fines (including sand) increased on the tributaries, although sand was still lost from these sites after the October 1989 storm. The finding that subsurface fines did not increase on the tributaries using freeze-cores after the October storm may be more influenced by the declining sand trend rather than the increasing silt trend.

Boyes Creek and sites on Prairie Creek below Boyes Creek showed a large increase in surface silt and vegetation in comparison to other upstream sites (Figures 19 and 20), but the larger increases began in 1988 before the October 1989 storm. For the other bypass-affected sites, increases (including above Brown Creek) were most noticeable after the storm. Silt in the unaffected site on Lost Man Creek did not increase above 1985 levels when first sampled after the storm.

Boyes Creek may have gained silt in 1988 from earlier bypass construction activities, watershed restoration activities, or delayed sediment releases from logging impacts. In 1988, a short stretch of Prairie Creek below May Creek was channelized, which explains the increased silt in these areas. These events were more detrimental than the October storm. However, the October storm appears to still have contributed substantial amounts of silt to all bypass-affected sites. The succession of drought years (1985 and 1987 to 1992 were below mean annual rainfall) likely also increased silt, exacerbating the habitat degradation.

Surface silt subdominant with gravel or larger rocks increased on Brown, Boyes, and May Creeks, but not on Lost Man Creek or on most of Prairie Creek mainstem after the storm (Meyer and Haux (in preparation)). In general, sand as a subdominant decreased on most sites. This suggests the spawning gravels of the affected tributaries may have received the greatest infiltration of fines and impacts from the bypass. The finding that Brown Creek and especially Boyes Creek had much lower permeability in streambed gravel than the mainstem after the October storm supports this (Appendix D in RNP 1991). Also, Brown Creek redds had twice the fine sediment infiltration rate of Prairie Creek below Brown Creek (Meyer et al. 1994). Although the impacts were less on the mainstem, the substantial increase in available silt (and vegetation on silt) in the habitat surrounding the gravels probably had some adverse effect on the spawning gravels because storms can move such sediment into the gravels. However, the very low survival below Boyes Creek probably can not be fully attributed to the October 1989 sediment input because the largest increases occurred before the storm. (This is not to say that bypass grading activities had no effect on Boyes Creek in 1988).

<u>Causes of Low Egg Survival</u>. The sudden additional October sediment input could have led to an increase in subsurface silt, worms, and subsequent low egg survival found in Prairie Creek

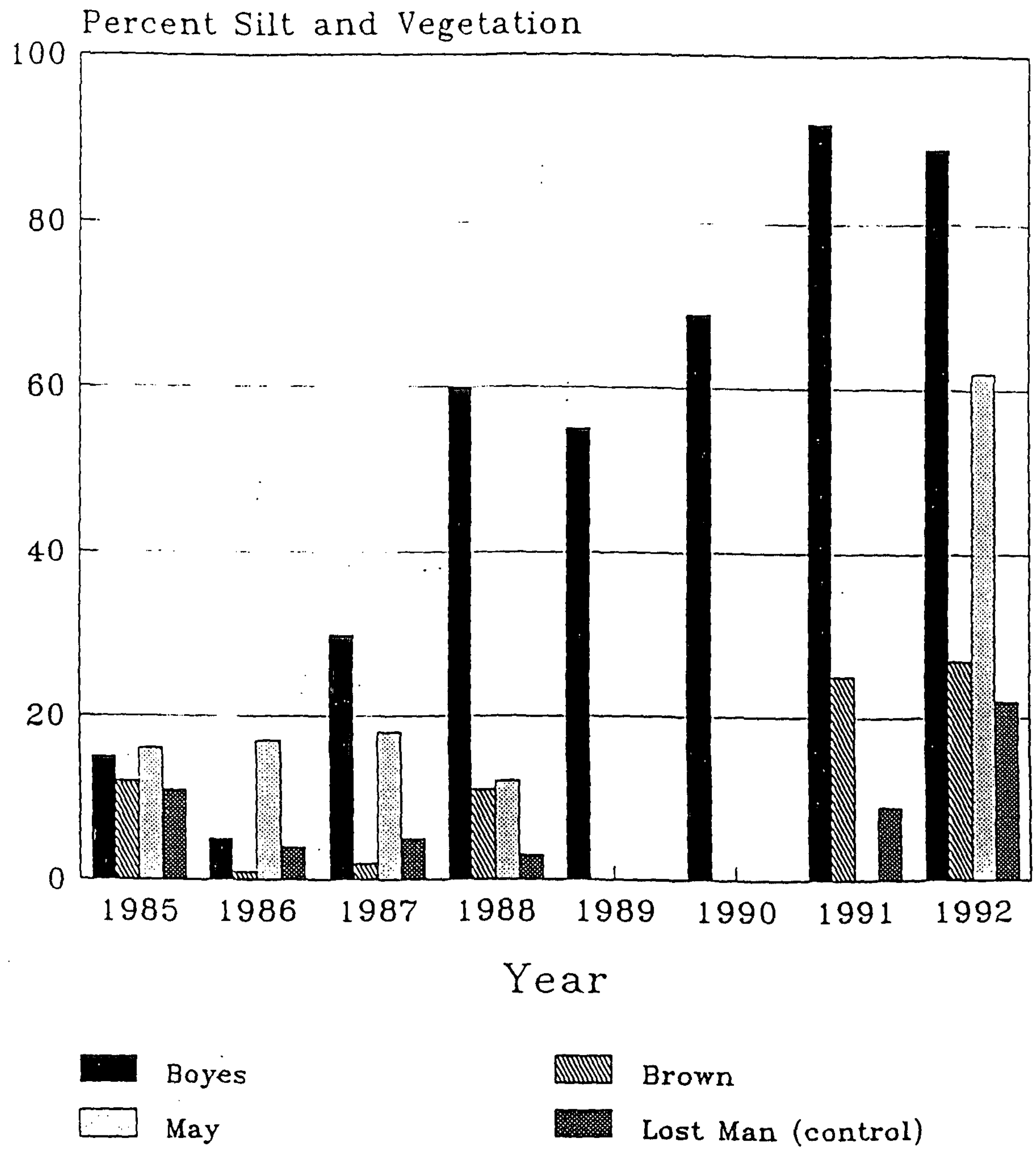


Figure 19. Percent of stream substrate in which silt or vegetation growing on fine sediment is the dominant or second-most dominant substrate type on long-term monitoring sites on Prairie Creek tributaries in summer or early fall. All but the Lost Man Creek site were affected by the October 1989 storm. Data were not collected or analyzed for all sites from 1989 - 1991 (from RNP bypass report in preparation).

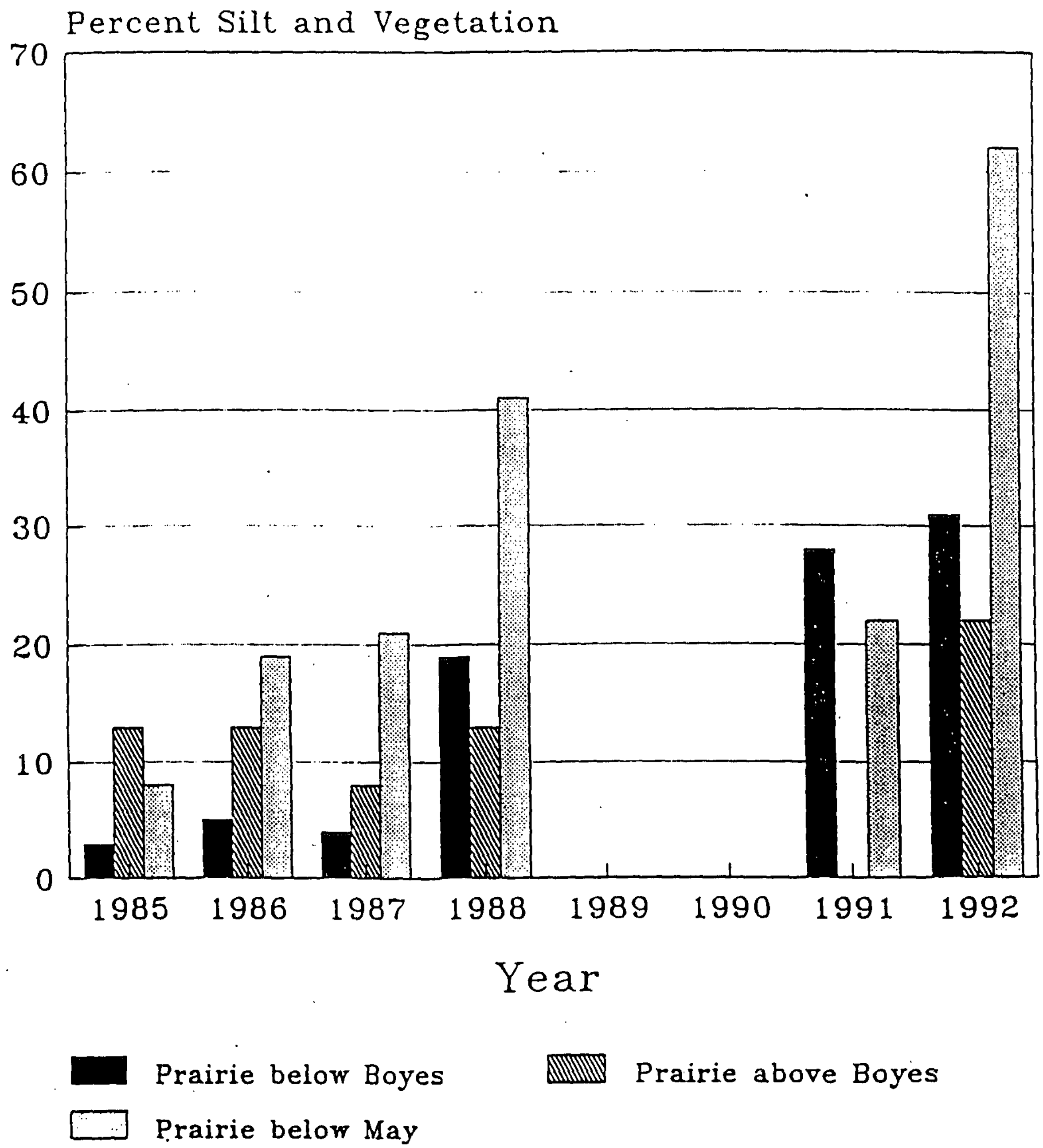


Figure 20. Percent of stream substrate in which silt or vegetation growing on fine sediment is the dominant or second-most dominant substrate type on long-term monitoring sites averaged for each reach on Prairie Creek ~~tributaries~~ in summer or early fall. All sites were affected by the October 1989 storm. Data were not collected or analyzed for all sites from 1989 - 1991 (from RNP bypass report in preparation).

in WY91. However, the direct cause-effect relationship between sediment and egg survival was not very strong over all the study years. It is possible the October 1989 sediment disturbance also created an imbalance in the aquatic macroinvertebrate communities or predators, giving a competitive edge to aquatic worms. More of the WY90 worm eggs may have been laid and/or survived to WY91, creating the proliferation of worms observed throughout Prairie Creek that year. If this was the case, even the level of disturbance that occurred in the control reach above Brown Creek would have to have been enough to increase worm abundance. These worms are a natural part of Prairie Creek, but infestations recorded in WY49 and WY51 were lower (27% vs. 62% infestation rate for WY91) and less damaging to egg survival (average coho salmon egg survival in redds with worms was 45% vs. 74% for all coho redds in Briggs 1953). The imbalance could have been corrected as soon as the aquatic invertebrate community or predators recovered, which would explain to WY92 redds' relative lack of worms. Benthic invertebrates can rebound or recolonize quickly following a major sediment-laden storm event (Iwatsubo 1981, RNP 1991). The imbalance may still exist below Boyes Creek. This might be because it is the most downstream reach and repeatedly receives large amounts of sediment from upstream sources.
An imbalance in the aquatic macroinvertebrate community was not obvious when its status in 1990 was evaluated to the order level and compared to variation in unaffected stream reaches and previous years' data including 1980-81, 1985, 1986, 1987, 1990 (RNP 1986, 1991 and Meyer and Haux, in preparation). Changes could have occurred at lower taxonomic levels, but this was not evaluated. The predator community may have changed, however. Welsh and Ollivier (1992) demonstrated that amphibians declined in affected Prairie Creek tributaries following the October 1989 storm. Some of these amphibians may have been important predators on the worms; their reduction may have temporarily increased worms if their reduction was also temporary.

Another possibility is that worms undergo population cycles, periodically breaking out in large numbers. Unfortunately, other than Briggs' data, pre-1989 storm worm data is minimal and insufficient to determine population trends of this species. Summer and fall invertebrate sampling of the surface of Prairie Creek riffles in WY85, WY87, and WY90 has not captured this particular species of worm. Winter surface sampling in WY81 also did not reveal much of this species. These results, Briggs' findings, and Barnhardt's recent WY92 studies (personal communication) suggest the worms are mainly found deeper in the gravel, particularly in areas where eggs or decaying animal matter is found. A laboratory study (n = 1 for treatment and n =1 for control) of the effect of worms on survival in clean gravels showed survival to hatching was not affected by worms, but hatched fry were affected (Kimsey 1955). This differs from our results, which may be due to the addition of sediment. Future monitoring of worm abundance in Prairie Creek and research on the worm's biology is recommended to effectively evaluate the impacts of the bypass sediment.

Mean coho salmon egg survival on Prairie Creek and a few tributaries was much higher in the late 1940's and early 1950's than our results for the early 1990's (74% vs. 21%). Briggs' methods differed from this study, however. Briggs estimated survival-to-hatching by periodically digging up natural redds and counting the number of dead eggs, live eggs, and hatched fry. This method can overestimate egg survival because many dead eggs disintegrate or perhaps are eaten by the worms or other creek fauna. In fact, Briggs' total eggs and/or fry in each redd was less than 500 in most cases. Fecundity per coho salmon female was over 2,000 eggs in 1991 (Farro 1991). Briggs conducted a small experiment showing rate of dead egg disintegration in cans was small in the creek (9% loss after 60 days); however, our study shows that the majority of dead eggs totally disappeared by the time the eggs hatched. Also, it is easy to miss eggs and larvae using this method. Thus, it makes it difficult to compare survival rates from our studies directly to Briggs'.

D.3 Comparison to Other Studies on Other Streams

In a Washington study, Cederholem et al. (1981) found a 2% reduction in coho salmon egg survival-to-emergence for each 1% increase in volume of fine sediment < 0.85 mm over natural levels. Both Cederholm et al. and Tagart (1976) reported a 14% drop (down to 18 and 3%, respectively) in coho salmon egg survival-to-emergence when fine sediment within natural redds exceeded 20%. Using net-capped redds and estimating egg deposition with carcass lengths, Koski (1966) found mean egg survival to emergence in an undisturbed Oregon watershed was 27%. This is much lower than Briggs' estimate (which, as stated earlier, is probably an overestimate). Koski's method may have trapped trial redds or redds with only part of the female's full egg load, biasing his estimate downwards (Everest et al. 1987). Also, as we found in this study, redd caps can accumulate fines and adversely affect survival.

To even decrease egg survival to 30% or lower (which occurred below Boyes Creek most years and throughout Prairie Creek in WY91), fine sediment levels < 0.85 mm had to reach 15% based on the regression line from Cederholm's field study. In Oregon, Hall and Lantz (1969) also found a negative correlation between volume of fines < 0.83 mm and survival-to-emergence in natural redds using fry traps ($r = -0.62$). To decrease egg survival to below 30%, fine sediment levels in natural redds had to reach 25% in Hall and Lantz's study. In WY90, fine sediment levels in the affected reach in Prairie Creek generally did not reach 15% in artificial redds (even when more comparable wet volumes were calculated instead of dry volumes---see Meyer et al. 1994 for methods) in WY90 and WY92. However, fines < 1 mm in natural redds (which were exposed to more storms than the artificial redds) located in the affected reach ranged from at least 15 to 31% in WY90. In WY91, mean wet volumes < 0.85 mm were between 16 and 17% in artificial redds and 11 and 13% in natural redds. In WY93, artificial redd volumes of fines were between 31 and 34%. Based on Cederholm's regression, survival would be predicted to be at least 20% in WY91 and no more than 15% in WY93. Actual survival means were 7% in WY91 and 41% in WY93. Clearly, fines levels alone during this study were not the main cause of egg mortality in the redds.

Peak discharge during the incubation period was a good predictor of fine sediment infiltration. Extrapolating the regression line to higher flows, a 5-year recurrence interval flood would increase weight of fines in redds below Boyes Creek up to 18%, 32%, and 50% for particle sizes < 0.5 mm, < 1.0 mm, and < 4.7, respectively (these estimates are for weight; it would be much higher by volume). Fines would increase to 22%, 39%, and 61% for the same sizes for a 10-year recurrence interval flood. In reality, the regression line is probably not linear at higher flows, but based on other studies this suggests a flow as low as a 5-year recurrence interval flood could increase fines to detrimental levels.

D.4 Conclusion

If the worm increase was a natural phenomenon, then the October 1989 sediment could not be blamed for the low egg survival in the mainstem unless the properties of the small increase in subsurface silt and sand from the storm became very detrimental when combined with worm mucous. This could be tested in a laboratory. If the worm increase was a response to the sediment disturbance, then the responsibility probably lies with the bypass. Egg survival studies were not conducted in the tributaries, but silt inputs were much greater there, suggesting that impacts to redds of coho salmon, steelhead trout, and cutthroat trout may have been greater in the affected tributaries than on the mainstem. Higher magnitude floods in the future may be detrimental to egg survival even without worms in the bypass sediment persists.

IV. EFFECT ON JUVENILE PRODUCTION AND SPAWNERS

The California Department of Fish and Game requested that all fishery data on or relevant to Prairie Creek be evaluated to supplement the egg survival results and evaluate the impacts of the bypass. The following evaluates downstream migrant trapping data and spawning surveys before and after the October 1989 storm.

A. Effect of Bypass and Hatchbox Program on Chinook Salmon Production.

Egg survival data can be weighted by proportion of spawners in each reach of Prairie Creek. Doing this for chinook salmon (assuming survival rates for coho salmon eggs do not greatly differ from chinook salmon eggs), egg survival in all of Prairie Creek for this species was about 4% in WY91, 30% in WY92, and 44% in WY93. To determine whether juvenile production was correspondingly low and whether low production in Prairie Creek occurred before the bypass, pre-bypass juvenile production data must be compared to post-bypass data. Unfortunately, data were not collected with this comparison in mind; data are missing, and many assumptions have to be made to make comparisons. These assumptions are certainly subject to criticisms, but managers need to make decisions on whether to continue the hatchbox rearing program and must use what data is available to make management recommendations. The following compares RNP's pre-bypass downstream salmonid migrant data to Farro's (1990,1991,1992,1993,1994) post-bypass downstream salmonid migrant data. It also assesses other data collected in the Redwood Creek estuary and basin to evaluate the hatchbox program's importance.

From early May to early August 1983 and 1984, RNP trapped downstream migrating chinook salmon (1 night/week, most fish migrate at night) on Prairie Creek 0.5 km upstream of its mouth (McKeon 1985 and Meyer 1994). RNP used a weir and inclined plane traps, which fully spanned Prairie Creek. The chinook salmon catch was 594 in 1983 and 127 in 1984. Multiplying these numbers by 7 to simulate continuous trapping (7 nights/week) gives 4,158 fish in 1983 and 889 fish in 1984. High flows blew out the trap for no more than a few hours in 1983. One sampling day was missed in 1984, which was possibly during the peak migration period.

During post-October 1989 storm studies of outmigrating chinook salmon for the bypass project, PCFWWRA attempted to trap Prairie Creek below May Creek (just below the channelized section) continuously from no later than May to August (1990 to 1994). They used various combinations of pipe traps and fyke net traps that did not fully span the creek. Stormflows blew out traps periodically (especially in WY90 and WY93), and estimates were best in WY91 and WY94 when the traps were in the creek the entire period. Comparing Prairie Creek WY93 results to WY93 Little River downstream migrant results during the same period (unpublished data, U.S. Fish and Wildlife Service, Arcata, CA) suggests that a couple thousand fish may have been missed that year during the 2 1/2 week period when the traps were not in place.

Annual outmigrant wild smolt estimates peaked at around 12,000 in WY94 (Table 8). WY90 and WY92 had very low production of wild chinook smolts from Prairie Creek. In WY91, it is unknown what percentage of the 7,500 fish migrating past the trap were wild.

Table 8. Number of chinook salmon migrating downstream below May Creek on Prairie Creek from WY90 to WY94. PCFWWRA's trap was located just downstream of channelized section of Prairie Creek (from Farro 1990, 1991, 1992, 1993, 1994 and Anderson 1990, 1991, 1992, 1993, 1994)(1994 reports in preparation)

Outmigrant Chinook Salmon	WY90	WY91	WY92	WY93	WY94
Total hatchbox smolts released in Prairie Creek	33,900	32,226	8,190	62,410	83,000
# hatchbox smolts released above trap	33,900	19,135	1,850	13,380	700
# hatchbox smolts past trap[a]	--	<7,574	< 50[b]	5,420	--
# wild smolts past trap	887	<7,574	< 50	2,835	12,000
# in estuary in late June[c]	6,390	<1,000	48,830[d]	17,990[e]	63,390
# hatchbox smolts in estuary	--	--	--	950	-

[a] Hatchbox fish were marked only in WY93. They could be differentiated from wild smolts in WY90 in traps based on size. Traps were blown out for several weeks in WY91 and WY93 (so numbers underestimate those years).
[b] In WY92, smolts were missed because 24% of redds and carcasses were below the trap due to low winter flows.
[c] Estuary was open during estimate all years shown. 95% confidence intervals for estimates did not overlap except in WY94, which overlapped with WY93 and WY92.
[d] 1,850 hatchbox fish were released into Prairie Creek after this estimate.
[e] No hatchbox fish were included in this estimate (they showed up in the estuary mostly in July and August).

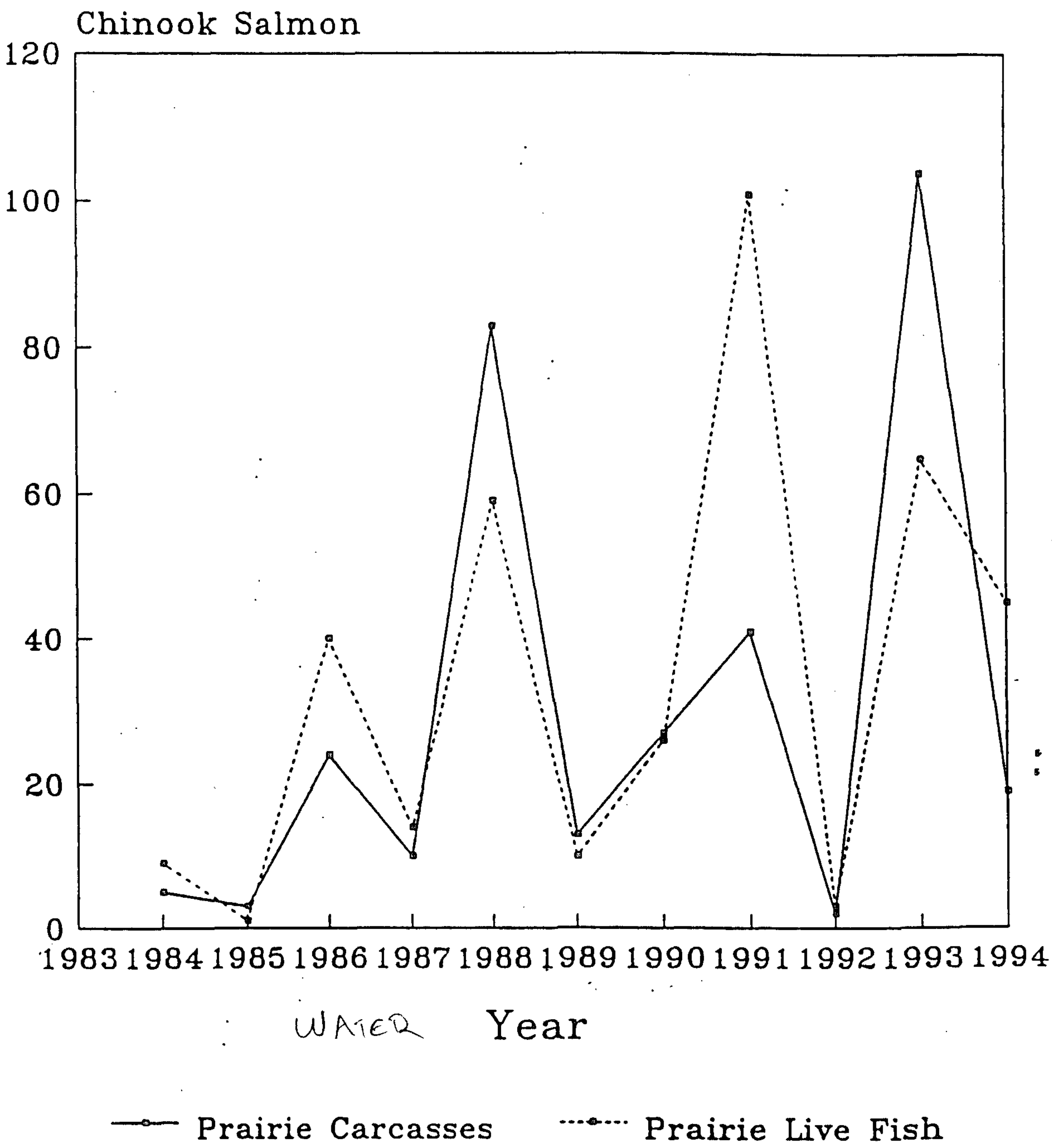
To compare PCFWWRA's data to RNP's, PCFWWRA 1994 data were re-analyzed using a 1 night/week sampling scheme. If PCFWWRA had sampled only 1 night/week in WY94, they would have captured 439 fish that year. Expanding their trap numbers to account for trapping efficiency (based on mark-recapture tests), about 2,000 fish would have been captured in WY94. Multiplying this estimate by 7 to simulate continuous trapping gives 14,000 fish, which is close to the 12,000 estimate based on continuous trapping. This estimate is much higher than the 1983 or 1984 estimate.

It is possible that PCFWWRA's mark-recapture tests caused some fish mortality (McKeon 1985), which would inflate the population estimates, or that the full-spanning weir missed fish. The pipe trap efficiency (PCFWWRA used the pipe traps most years) was compared to a full-spanning weir without using a marking procedure (Farro 1990). In 1990, a full-spanning weir was placed behind two pipe traps and seined to capture fish missed by the pipe traps. This took place during higher May flows in 1990. This test showed the pipe trap caught no more than 49% of the fish passing by. The pipe traps usually become more efficient as flows decrease (Farro 1993). If the pipe traps were 3/4 to half as efficient as the weir over the season, 590 to 880 fish might have been caught if a full-spanning weir had been used in 1994 on a 1 night/week trapping schedule (these numbers increase if the efficiency is even lower). This suggests numbers in 1994 were at least as high as 1983 (594) and higher than the number caught in 1984 (127).

McKeon conducted mark-recapture tests of the full-spanning weir efficiency in 1983 and found it averaged only 27%. Data from the estuary support that perhaps his efficiency was that low (see Appendix D). If so, his numbers could be expanded to 15,400 fish in 1983 and 3,300 fish in 1984. This and the previous analysis support that 1994 produced about the same order of magnitude of wild fish as 1983 and many more fish than were produced in 1984. WY93 estimates were of the same order of magnitude as WY84, but WY90 and WY92 produced fewer wild fish than Wy84.

These chinook salmon numbers must be evaluated against the spawning run strength of the year in which outmigrants were trapped. The number of salmon carcasses (including coho because they were not separated by species in 1983 before surveys were standardized) counted in Prairie Creek in WY83 and WY84 were ranked approximately in the lower third of all years spawning surveys have been conducted (1983 to 1994)(WY84 was lower than WY83 -- unpublished data). The number of chinook salmon carcasses (and all salmon carcasses) counted on the creek in WY94 was ranked at the midpoint level of all survey years since WY84 (Figure 21, Meyer 1994). Thus, production might be expected to be somewhat higher in WY94 than WY83 or WY84, all other factors being equal. For other years, WY92 had very low numbers of chinook salmon carcasses in Prairie Creek similar or lower than WY84; WY90 had average numbers, and WY91 and WY93 had high numbers. Based on the year's run, outmigrant production in WY92 might be expected to be much lower than WY83 and about the same as WY84. WY91 and WY93 would be expected to have much higher production, whereas WY90 would have slightly higher numbers. WY90's very low wild outmigrant estimate compared to WY84 suggests the October 1989 storm had a negative effect on chinook production that year. Unfortunately, no data exists for wild chinook in WY91, another year suspected of having very poor production based on the egg survival studies. Due to the very low spawning run, WY92 would probably have had low chinook salmon production even without the October storm. WY93 should have been a more productive year than what was observed, supporting that the poor creek conditions below Boyes Creek were still problematic. WY94, on the other hand, had numbers close to expected based on spawning run strength.

The mainstem of Prairie Creek is very important to chinook salmon, which usually do not spawn in the Prairie Creek tributaries (Briggs 1953). From 1984 to 1989, on average 2/3 of carcasses were chinook salmon and about half of the spawners (including steelhead) were chinook salmon. Numbers of redds (all salmonid species) on Prairie Creek averaged 60 (55 with standardized



Standardized to 3 surveys (no March)

Figure 21. Number of chinook salmon carcasses and live fish counted on Prairie Creek during spawning surveys from November to February, 1984 to 1994. Numbers were standardized to 3 surveys to facilitate comparison among years (from Meyer 1994).

effort). From 1990 to 1994, redds per survey effort averaged 155 (116 with standardized effort) and on average 70% of carcasses were chinook salmon and about half (45%) of the spawners were chinook salmon (Meyer 1994). The increase in redds is partially due to extension of the survey period into March in 1990 and subsequent years. Even with this change, it is apparent that the spawning run has not decreased on Prairie Creek since the October storm (see Figure 21). In contrast, the chinook spawning run has been decreasing on the two other creeks surveyed in the Redwood Creek basin: Bridge and Tom McDonald Creeks (Meyer 1994). Still, without much of a contribution from Prairie Creek, the basin can produce at least 50,000 juvenile chinook salmon, as was estimated for fish in the estuary in WY92 (Table 8), a drought year when Prairie Creek chinook salmon runs were extremely low.

B. Effect of Bypass and Hatchbox Program on Coho Salmon Production.

Weighted by distribution of coho salmon carcasses, coho salmon egg survival in Prairie Creek mainstem for WY91 would have been 11% in WY91, 30% in WY92, and 21% in WY93. Many coho salmon spawn in Prairie Creek's tributaries where egg survival was not evaluated (Briggs 1953). The exception was Lost Man Creek, where egg survival was 20% and 28% in WY92 and WY93, respectively. The WY92 coho spawning run in Lost Man Creek was over 3 times greater than Prairie Creek (Anderson and McGuire 1994), so egg survival for coho that year was closer to Lost Man's lower estimate (although the drought and not the bypass was responsible for Lost Man's greater importance and lowering of egg survival). In WY93, coho salmon were much more abundant in Prairie Creek than Lost Man Creek, typical of other years. Although abundance of spawners was not estimated in the other tributaries, one tributary--Brown Creek--had about twice the rate of sediment infiltration of redds as Prairie Creek from WY90 to WY92 (Meyer et al. 1994). Other tributaries were not affected by the bypass. Thus, it is unknown whether average coho salmon egg survival throughout the Prairie Creek system would have been much lower or the same as the mainstem. The following compares coho salmon juvenile production data during pre- and post-October 1989 storm periods to substantiate whether lowered egg survival did occur during post-October storm years.

Coho salmon usually spend a year in the creek system, and then most migrate in April and May. Downstream migrant traps were usually not in place until May, so total outmigrant population estimates were not possible. In WY92, expanded estimates of numbers passing the traps from May to July were 2724 coho salmon smolts (age 1+): 1836 hatchbox (67%), 815 wild (30%), and 73 Prairie Creek hatchery smolts (3%) (Farro 1992). In WY94, another dry year, coho salmon smolts were more abundant during this same time period (about 3700 1+ fish and 600 0+ fish from May 3 to August 1)(unpublished data from Farro). It is unknown what percentage were wild fish in WY94. Estimates for other post-storm years have not been made although the data is available.

Expansion of numbers caught in RNP's traps (McKeon 1985, Meyer 1994)(using 27% efficiency) gives 1400 and 169 coho salmon during the sampling period of early May to early August, 1983 and 1984, respectively. Unfortunately, April data were not collected. WY83 and WY84 had much higher rainfall in March and April than WY92 and WY94 (from South

Operations Center rain gauge in Orick, CA), and therefore possibly a greater proportion of smolts migrated in April, the missed sampling period. In WY94, about 30% of the trapped juvenile salmon in Prairie Creek from May to July were coho salmon (0+ and 1+) compared to 9% and 5% in WY83 and WY84 (unpublished data from Farro), which may also be due to earlier migration in WY83 and WY84. WY92 and WY94's estimates could be inflated relative to pre-storm estimates, or else, since the early 1980's, coho salmon have become more abundant in Prairie Creek, similar to other creeks in the Redwood Creek basin (Meyer 1994).

WY84 had few spawners (Figure 22) and thus low numbers of coho smolts may have been produced before the October storm. WY91, however was a good spawning year for coho salmon and would be expected to have produced many more fish than observed. Although the data are incomplete, they suggest WY91 produced low numbers of young-of-the-year wild coho salmon, which is corroborated by the egg survival data. A high mortality of hatchbox fish probably occurred over the year these fish resided in Prairie Creek because only 7% were estimated to pass the trap in WY92. Even if this estimate was quintupled to account for earlier migration, it would still only be 35% juvenile survival. Still, the hatchbox program may have more than tripled the production of yearling coho salmon that year.

Smolt numbers (wild and hatchbox) in May and June 1994 were higher than the same time period in 1992. The spawning run during the brood years for these fish (WY91 and WY93) was about the same; and fewer hatchbox fish were released in WY93 than WY91. Hence, WY93 probably had better production of wild coho salmon than WY91. This agrees with the egg survival data.

In the summer WY90, coho salmon 0+ densities in bypass-affected sites on Prairie Creek and its tributaries were much lower than sites unaffected (Farro 1990). For instance, Little Lost Man Creek and Godwood Creek sites had 2 to 5 times greater densities of coho salmon than Brown, Boyes, and Prairie Creeks. Upper Prairie Creek (sampled on the long-term monitoring site on the control reach in Prairie Creek) had numbers similar to lower Prairie Creek, but as previously mentioned, this reach was affected by the bypass and is a poor control reach. In September 1994, the site on upper Prairie Creek was electroshocked again and coho densities were an order of magnitude larger than June-July or August-September estimates in WY90, even though the spawning run strength was about the same (Table 9)(unpublished RNP data). Little Lost Man Creek densities were only about 4 times greater in WY94 than WY90. In August-September 1981 before the October storm (Anderson 1988), the density was similar to 1994 on Prairie Creek (within 95% confidence interval), but an order of magnitude smaller on Streelow Creek (which was unaffected by the bypass but affected by logging) and Little Lost Man Creek. This suggests that production had improved by WY94 to pre-bypass levels at least on upper Prairie Creek.

Electroshocking data was also collected on Godwood Creek (unaffected creek) and May Creek (affected creek) periodically from 1985 to 1991 (Meyer and Haux, in preparation), Table 10). Using the average width from the 1990 electroshocking site on lower Godwood Creek (from Farro 1990), densities were similar between WY91 and WY90 for this creek. In WY90,

Table 9. Densities (fish/m^2) of coho salmon and steelhead trout in Prairie Creek and unaffected tributaries.[a]

	COHO SALMON			STEELHEAD TROUT		
CREEK	1981	1990	1994	1981	1990	1994
Prairie above Brown Creek	0.18	0.01[b]	0.23[b]	0.02	0.02[b]	0.07[b]
Little Lost Man Creek	0.00[b]	0.10[b]	0.43	0.40[b]	0.36[b]	0.30
Streelow Creek[c]	0.04	--	0.40	0.05	--	0.19

[a] 1994 data are preliminary and unpublished. 1981 data are from (Anderson 1988). 1990 data are from Farro (1990).
[b] The same site was sampled on the creek.
[c] Located just south of Godwood Creek.

Godwood Creek had coho salmon densities between 0.04 m^2 and 0.08 m^2, which was much higher than upper or lower Prairie Creek (0.01 fish/m^2)(Farro 1990). In WY91, a site on Godwood Creek had more or the same numbers of coho than years before the October storm, and a site on May Creek had fewer coho than years before the storm. This shows that another old-growth unaffected stream besides Lost Man Creek did not decrease in numbers following the October storm, but an affected stream did. Interestingly, park staff observed numerous 1+ and 2+ coho salmon when electroshocking in May Creek just this November, 1994. Even lower May Creek may be recovering.

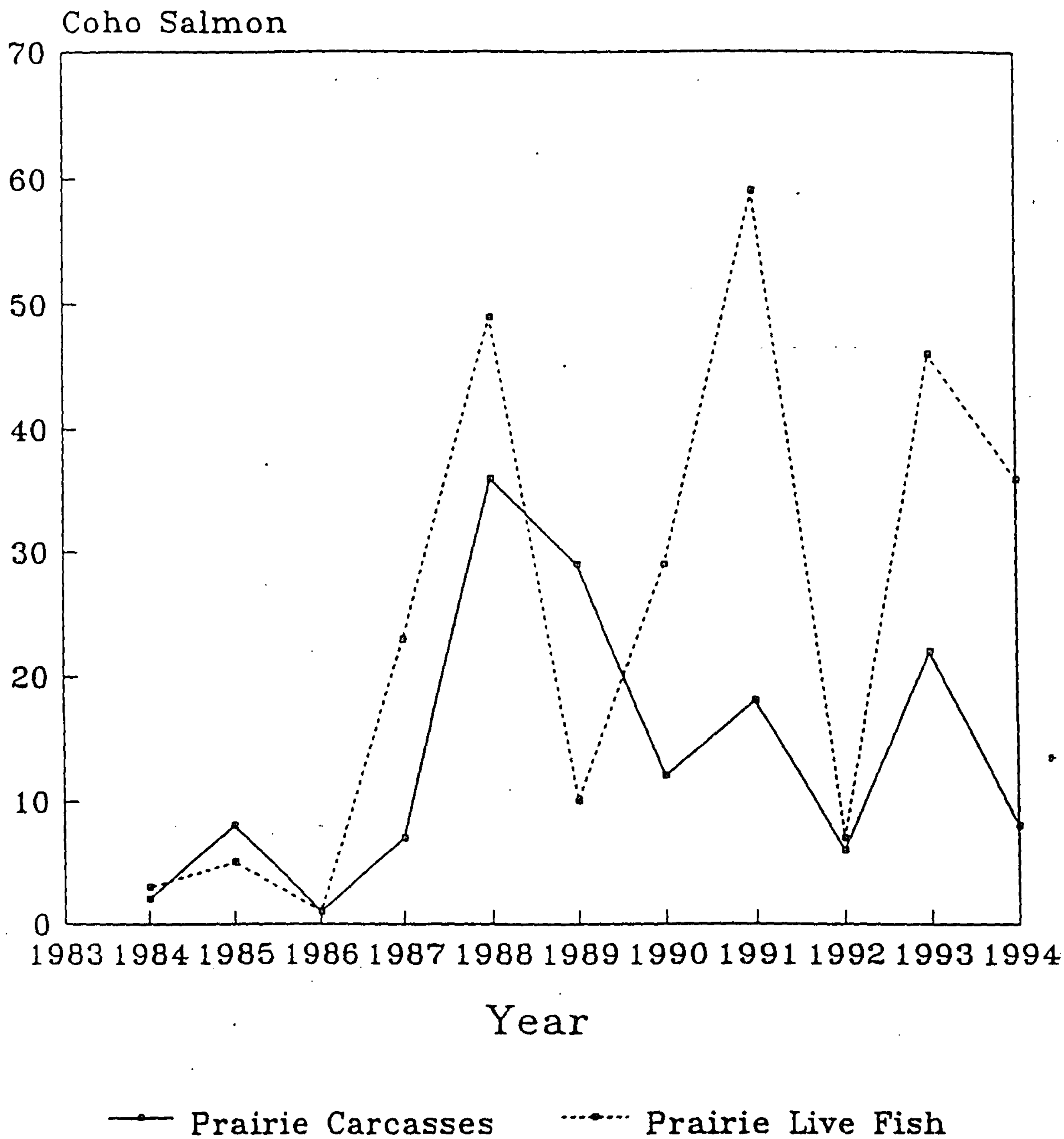
Coho salmon hatchbox releases were 35,645 fish in WY90, 24,880 in WY91, 26,710 in WY92 and 17,430 in WY93 (Farro 1990, 1991, 1992, 1993). Coho salmon were not released in WY94, which probably was a good decision because the year had low flows, good numbers of juveniles in upper Prairie Creek, and likely had improved production over WY90 and WY91 as was found in WY93.

In WY81, WY82, and WY83 of McKeon's (1985) study, less than 2% of the salmon in his Redwood Creek traps were coho salmon. Outside of Prairie Creek and its tributaries, very few tributaries in the Redwood Creek basin contained juvenile coho salmon in 1981 (only Tom McDonald, Coyote, and Karen Creeks)(Anderson 1988). By WY94, Emerald, Copper, and Bridge Creeks also contained coho salmon, but creeks surveyed outside the park still did not (unpublished RNP data). Live or dead (carcasses) coho salmon spawners were not observed in Bridge and Tom McDonald Creek until the early 1990's (Meyer 1990). This indicates coho salmon have been making a recovery in the park, but not in the upper basin. Spawning surveys in Prairie Creek, however, show trends in numbers of live and dead adult coho salmon have not changed from 1987 to 1994. Only the latter two years would have contained many offspring produced after the October storm, and numbers during those years were in the range of variability before the storm (Figure 22). These numbers, particularly in WY93, may have been

Table 10. Abundance and composition of juvenile fish estimated for 30 meter (100 ft) sections of May and Godwood Creeks, U.S. Highway 101 bypass, September 10, 1985, August 22, 1986, September 3, 1987, and October 17, 1991.[a]

	MAY CREEK				GODWOOD CREEK			
	1985	1986	1987	1991	1985	1986	1987	1991
Total Estimate	51	69	28	7	40	44	46	14
95% C. I.	2	4	2	1	3	8	4	1
Coho Salmon								
Subtotal Estimate	44	10	13	4	3	8	4	9
Average Fork Length (cm)	5.8	6.5	5.3	7.6	5.5	6.4	6.1	7.1
Fork Length Range (cm)	5.0-7.0	5.6-7.6	3.9-7.3	5.8-9.0	4.5-8.5	4.8-8.4	5.4-7.1	6.1-8.3
Steelhead Trout								
Subtotal Estimate	4	58	15	3	12	23	31	5
Average Fork Length (cm)	7.8	4.6	3.9	7.9	6.4	4.7	5.1	7.1
Fork Length Range (cm)	7.0-8.5	3.2-6.4	3.0-6.0	5.2-10.8	5.5-9.0	3.5-5.6	3.5-8.7	6.1-9.0
Cutthroat Trout								
Subtotal Estimate	3	1	0	0	2	1	3	0
Average Fork Length (cm)	7.8	10.7	N/A	N/A	7.8	7.9	7.5	N/A
Fork Length Range (cm)	6.0-9.5	10.7	N/A	N/A	5.5-10.0	7.9	5.4-11.0	N/A

[a] From (Meyer and Haux (in preparation)).



Standardized ~~to 3 surveys~~

Figure 22. Number of coho salmon carcasses and live fish counted on Prairie Creek during spawning surveys from November to April, 1984 to 1994. Numbers were standardized to 3 surveys to facilitate comparison among years (from Meyer 1994).

In summary, the various data suggest coho salmon production was adversely impacted by the bypass in WY90 (even without worms). WY91 had poor production of young-of-the year coho salmon, and this could be attributed to the bypass. Migration data for yearling coho salmon have not been analyzed for WY93, but the greater dependence of the run on Lost Man Creek suggests egg survival was lowered in WY92, but higher than the previous 2 years or WY93. WY92's lowered survival, however, was more attributable to the low flows than the bypass. WY93 seemed to have better young-of-the-year production than WY91. The only data for WY94 are for young-of-the-year coho salmon in upper Prairie Creek, which were as abundant as before the October storm.

C. <u>Effect of Bypass and Hatchbox Program on Steelhead Trout Production</u>

Many steelhead spawn in the tributaries to Prairie Creek (Anderson 1988), especially Lost Man Creek, which has over 3 times more steelhead spawners than Prairie Creek (Haux and Anderson 1992, Anderson and McGuire 1994). In Lost Man Creek, steelhead egg survival was high and similar to Prairie Creek under low flow conditions in WY90. The majority of steelhead usually do not enter the creeks until February (Farro 1990), whereas the salmon arrive in late November (Farro 1990, 1991, 1992, 1993). Because of this, redds are often exposed to less severe storms. Also, the incubation period is shorter because of the higher stream temperatures. These conditions might allow steelhead eggs to have better success than salmon eggs.

Expansion of RNP data show 1,740 and 770 steelhead trout passing through or by the Prairie Creek trap in WY83 and WY84, respectively, from May to early August (using 27% trap efficiency, Meyer 1994). PCFWWRA's data for this same time period in WY94 show 2,030 wild trout. Estimates for other years have not been made. Estimates both years include trout too small to identify as steelhead or cutthroat trout. Shapovalov and Taft (1954) reported that peak emigration of steelhead usually occurs from April to June with minor emigration the entire year. On Redwood Creek, peak steelhead migration occurred in June in 1981 and in July in 1982, 1983, and 1984 (Meyer 1994). In WY94, a smaller peak of mainly 1+ fish occurred in June and the larger peak in July included 0+ fish. The peak migration occurred in July for Prairie Creek in 1983, 1984, and 1994. Although the trapping period (set up for chinook salmon) was too narrow for steelhead trout, we believe it captured a good proportion of the fish (although data show peak population estimates in the estuary often occur in mid- to late August, and steelhead often reside in the creek just above the estuary in late July (Anderson 1990, Larson 1987). The data suggest that steelhead production in WY94 may not have been strongly affected by the bypass.

During electroshocking surveys in summer WY90, young-of-the year steelhead, cutthroat, and resident trout were in high densities in Little Lost Man Creek, a creek unaffected by the bypass. Next highest was an affected creek, upper Brown Creek, although the data suggests the high numbers were due to cutthroat trout, not steelhead (Farro 1990). Godwood Creek, another control creek, did not have substantially more trout than affected tributaries. Upper Prairie Creek (control reach in the egg survival study) had very few steelhead trout in WY90, but numbers were equivalent to WY81 (Table 9). Little Lost Man Creek also had similar densities between WY81 and WY90. Trends in trout numbers on Godwood Creek and May Creek before and after the October storm are similar (Table 10). Godwood Creek densities were similar between WY90 and WY91. These data do not support that the bypass had an adverse effect on steelhead in WY90 and WY91. Steelhead runs were later in the year than for salmon on Prairie Creek and steelhead missed the largest storm events in WY90 and WY91 (Meyer 1994). The egg survival study with steelhead eggs in WY90 indicated survival was very good. For these reasons, steelhead trout were not raised in hatchboxes after the first year in WY90, when 24,800 were released without a tag or mark (Farro 1990).

D. Evaluation of Hatchbox Rearing Program

The results suggest that wild chinook populations needed supplementation to offset bypass losses in WY90, WY91, and WY93. However, the effectiveness of the hatchbox rearing program in providing surviving chinook salmon appeared low during the one year chinook salmon were tagged. Less than 1,000 of the 5,420 hatchbox chinook outmigrating were estimated to reside in the estuary in summer 1993 (Table 8). Estuary numbers increased from practically no hatchbox fish in June to about 1,000 in August, no more than 5% of the estuarine chinook juvenile population. Also, when fish were released above the trap, survival of these fish to migrating past the trap was low (Table 8), possibly due to high predation.

Returns for tagged adult chinook will not be known until WY97, making it difficult to evaluate the program contribution to chinook salmon at this time. However, WY94 had a higher spawning run than most years despite the mild winter (Figure 21). This may be attributable to the hatchbox program in WY90. Also, despite the increase in the number of chinook salmon removed from spawning in the creek in the 1990's (in the mid to late 1980's Prairie Creek hatchery removed no more than 13 chinook salmon each year (RNP 1991), whereas PCFWWRA removed between 15 and 30 females (Farro 1990, 1991, 1992, 1993)), numbers of chinook salmon carcasses and live fish counted in the early 1990's were not lower than the 1980's.

Tagged coho salmon released in WY90, some of which entered the ocean that same year (Anderson 1991), composed 43% of the adults trapped in WY93 (Farro 1993), and possibly over 20% of the jacks in WY92 (Farro, personal communication). Coho released in WY91 composed about 10% of the adults trapped in WY94 and 13% of the jacks in WY93 (Farro 1994). (Coho juveniles were tagged in WY90 and WY91; chinook were tagged in WY93; none were tagged in WY94.) The lower percentage in WY91 may have been because about twice as many coho salmon spawned on Prairie Creek in WY91 as in WY90 (Figure 22) and fewer hatchbox coho were released in WY91.

Coho salmon populations appeared to need supplementation to offset bypass losses in WY90, WY91, and perhaps to a lesser extent, WY93. Supplementation was probably not needed in WY94. It appears that hatchbox fish might have lower survival in the ocean than wild fish. When outmigrating in WY92, the WY91 hatchbox coho cohort composed 67% of the coho salmon, which decreased to 10% of the returning adults.

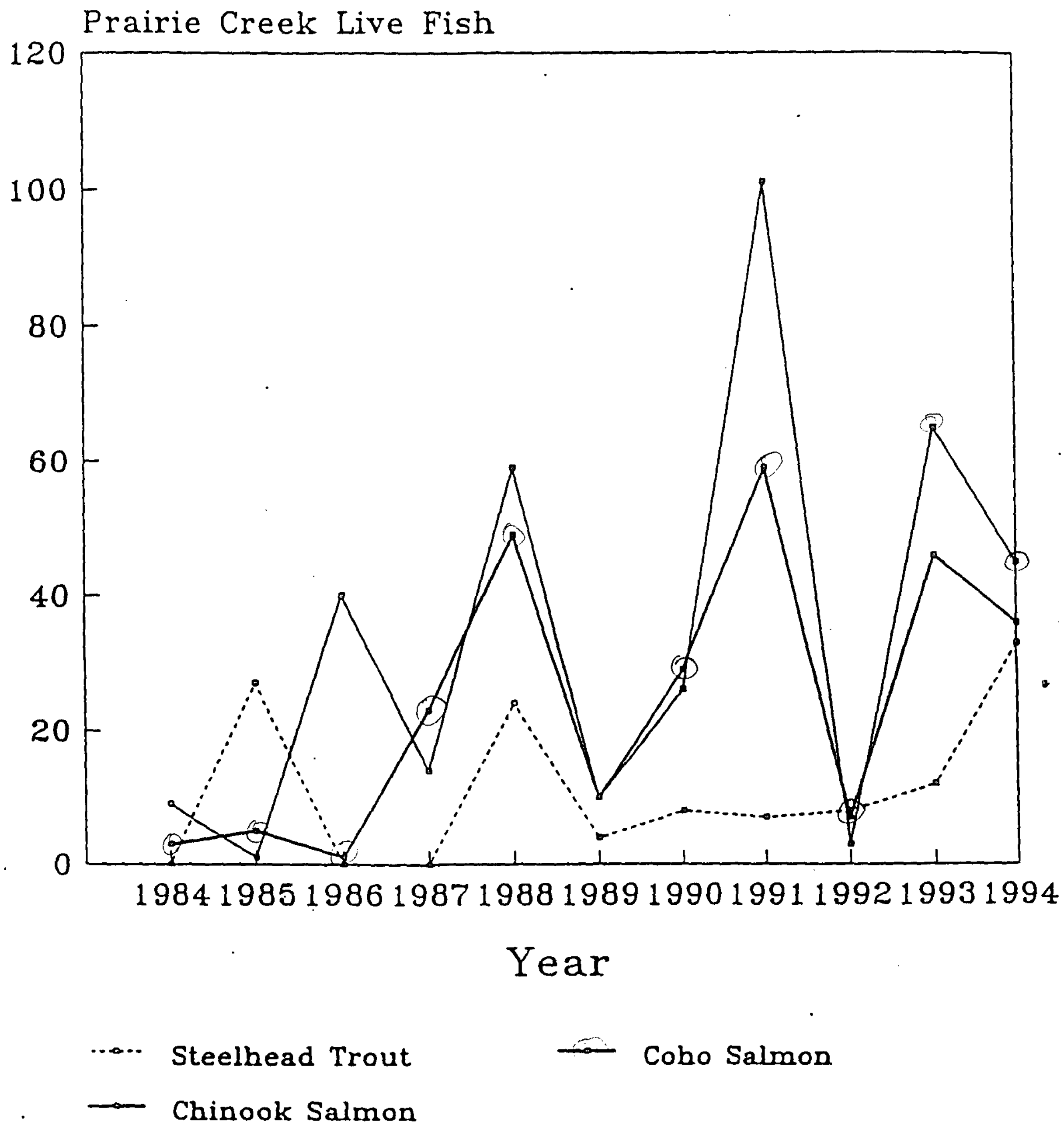
Stocking of hatchbox fish did not significantly decrease densities of wild coho salmon in upper and lower Prairie Creek in WY90 (Farro 1990). The program has likely made a significant contribution to Prairie Creek for WY90 and WY91 when it was needed most (assuming the system was undersaturated with wild fish due to poor egg survival). However, because egg survival and fish production have been improving over time, supplementation was less needed in other years, particularly in WY94.

Although data are limited, steelhead trout appear to be more resilient to sedimentation due to timing of their redds and shortness of their incubation period. Supplementation of this species was probably not necessary. However, despite the low flows, return of spawning adults was much higher in WY94 than previous years, which may have been the result of hatchbox supplementation in WY90 (Figure 23).

E. Recommendations

Wild fish may be forced to compete with the hatchbox fish in the streams or estuary during years of low flows and few worms. More data are needed on when worms are present and the effect of high flow years on egg survival. However, without this information, we recommend that supplementation be discontinued unless conditions outlined below are met. The current trend of having low rainfall years appears to be continuing, which, barring worm outbreaks, probably supports good wild fish production.

Starting in WY95, trends in worm abundance and sediment on Prairie Creek should be monitored using at least 8 baskets filled with bait (eggs or meat) and placing infiltration bags underneath. This allows one to look for multi-year cyclic worm patterns and test for correlations with annual sediment infiltration rates. Alternatively sediment flux could be monitored in place of infiltration rates. Spawning surveys on Prairie Creek should be continued, at least 3 surveys each season, until at least WY97 to monitor if bypass-affected adult returns are abnormally low. After a high flow year ($\geq$ 2-year recurrence interval), downstream migrating juvenile chinook salmon should be counted, and stream sites that were electroshocked in WY90 and WY94 should be resampled in the summer to determine if production is substantially lower than WY94 estimates. If it is lower, data should be examined to see if abundance of worms, infiltrating sediment, or scarcity of spawners were important factors that year. If the low production appears unrelated to sediment, fish stocks should not be supplemented and Caltrans obligations should be suspended. If data are unclear, a simpler egg survival study (no permeability or DO monitoring) with infiltration bags under egg baskets should be conducted on Prairie Creek during a year that has high flows. If sediment is strongly implicated, hatchbox supplementation of coho and chinook salmon may be advisable during high flow years until streamflows mobilize the streambed to at least 1 foot depth, removing the bypass sediment (which should continue to be monitored with scour chains). Following such an event, egg survival and sediment flux relative to stream discharge should be tested in the section below and just above Boyes Creek to verify success (realizing lower survival below Boyes Creek may be acceptable because it may be caused by other factors than the October 1989 storm). If egg survival is good in a high water year, Caltrans obligations could then be suspended. Finally, in WY97, returning adult chinook



Standardized to 3 surveys

Figure 23. Number of live steelhead, coho and chinook salmon counted on Prairie Creek during spawning surveys from 1984 to 1994. Numbers were standardized to 3 surveys to facilitate comparisons among years (from Meyer 1994).

salmon should be trapped, counted, and passed over the trap to determine percentages of returning adults that were tagged hatchbox fish. The percentage might be expected to be similar to hatchbox coho salmon returns from the WY91 cohort, which was 10%. If the percentage is less than 5%, which is low for a year when supplementation seemed to be needed, the hatchbox program is probably not a profitable alternative for chinook salmon, even if it is needed.

V. LITERATURE CITED

Anderson, D.G. 1990. Redwood Creek estuary monitoring and management: 1990 progress report. Redwood National Park, Orick, CA.

------------. 1991. Redwood Creek estuary monitoring : 1991 progress report. Redwood National Park, Orick, CA.

------------. 1992. Redwood Creek estuary monitoring : 1992 progress report. Redwood National Park, Orick, CA. 11pp.

------------. 1993. 1993 Redwood Creek estuary annual monitoring report. Redwood National Park, Orick, CA. 17pp.

------------. 1994. 1993 Redwood Creek estuary annual monitoring report (in preparation). Redwood National Park, Orick, CA.

Anderson, D., and H. McGuire. 1994. Redwood Creek basin 1991 - 1992 spawning and carcass survey, Annual progress report. Fish and Wildlife Branch, Research and Resource Management Division. Redwood National Park, Orick, CA.

Barnhardt, R. 1994. Personal communication. Professor and Unit Leader, Humboldt State University Fisheries Cooperative Unit, Arcata, CA.

Briggs, J.C. 1953. The behavior and reproduction of salmonid fishes in a small coastal stream. California Department of Fish and Game Marine Fisheries Branch. Fisheries Bulletin No. 94.

Brown, R.A. 1988. Physical rearing habitat for anadromous salmonids in Redwood Creek basin, Humboldt County, CA. M.S. Thesis. Humboldt State University, Arcata, CA. 132pp.

Cederholm, C.J., Reid, L.M., and E.O. Salo. 1981. Cumulative effects of logging road sediment on salmonid populations on the Clearwater River, Jefferson County, WA. In: Proceedings from the conference on salmonid spawning gravel: A renewable resource in the Pacific Northwest? Report 39. State of Washington Water Research Center, Pullman, WA. pp38-74.

Everest, F.H., R.L. Beschta, J.C. Scirvener, K.V. Koski, J.R. Sedell and C.J. Cederholm. 1987. Fine sediment and salmonid production: a paradox. pp. 98-141 in E.O. Salo and T.W. Cundy (editors). Streamside Management: Forestry and Fishery Interactions. College of Forest Resources. University of Washington, Seattle. Contribution No. 57. Proceedings of Symposium held at the University of Washington, Feb. 12-14, 1986.

Farro, M. Trinidad Fisherman's Salmon Enhancement, Prairie Creek Project 1989 - 1990, Final Report. Pacific Coast Federation of Fisherman's Associations Inc., Trinidad, CA.

--------. Trinidad Fisherman's Salmon Enhancement, Prairie Creek Project 1990 - 1991, Final Report. Pacific Coast Federation of Fisherman's Associations Inc., Trinidad, CA.

--------. Prairie Creek Salmon Restoration, Preliminary Report 1991 - 1992 season. Pacific Coast Fish, Wildlife & Wetlands Restoration Association, Arcata, CA.

--------. Prairie Creek Salmon Rèstoration, Preliminary Report 1992 - 1993 season. Pacific Coast Fish, Wildlife & Wetlands Restoration Association, Arcata, CA.

--------. 1994. Personal communication.

Hall, J.D., and R.L. Lantz. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. Oregon Agricultural Experiment Station, Technical Paper No. 2570, Corvallis, OR. 375pp.

Haux, S., and D.A. Anderson. 1993. Redwood Creek Basin 1990-1991 spawning and carcass Survey. Annual Progress Report, Fish and Wildlife, Research and Resource Management Division, Redwood National Park, Orick, CA.

Iwatsubo, R.T. and R.C. Averett. 1981. Aquatic biology of Redwood Creek and Mill Creek drainage basins. U.S. Department of Interior, Geographical Survey Open File Report -B 143. 115pp.

Kimsey, J.B. 1955. Experimental hatching of steelhead eggs in the presence of oligochaete worms with a suggested program of further investigation. Inland Fisheries Administrative Report 55-7. California Department of Fish and Game. 6 pp.

Koski, K.V. 1966. The survival of coho salmon (Oncorhynchus kisutch) from egg depositon to emergence in three Oregon coastal streams. M.S. Thesis, Oregon State University, Corvalis, OR. 84pp.

Larson, J.P. 1987. Utilization of the Redwood Creek Estuary, Humboldt County, CA by juvenile salmonids. M.S. Thesis. Humboldt State University, Arcata, CA. 79pp.

McKeon, J.F. 1985. Downstream migration, growth and condition of juvenile fall chinook salmon in Redwood Creek, Humboldt County, CA. M.S. Thesis. Humboldt State University, Arcata, CA. 90pp.

Meyer, C.M. 1994. Redwood Creek basin fisheries summary: 1980 - 1994. Redwood National Park, Orick, CA.

Meyer, C.M. and S. Haux (in preparation). Monitoring of aquatic impacts associated with construction of the U.S. 101 bypass: Ten year report. Redwood National Park, Arcata, CA.

Redwood National Park. 1985. Estuarine Management and Research Activities, mouth of Redwood Creek, 1984. Crescent City, CA. 15pp.

Redwood National Park. 1986. Monitoring of aquatic impacts associated with construction of the U.S. 101 bypass. U.S. Department of the Interior, Arcata, CA. 98pp.

Redwood National Park (RNP). 1991. Monitoring the Impacts and Persistance of Fines in the Prairie Creek Watershed. Progress Report prepared for the California Department of Transportation. National Park Service, Arcata, CA. 123 pp.

Shapavalov, L. and A.C. Taft. 1954. The life histories of the steelhead rainbow trout and silver salmon. California Department of Fish and Game. Fish Bulletin No. 98.

Tagart, J.V. 1976. The survival from egg deposition to emergence of coho salmon in the Clearwater River, Jefferson County, WA. M.S. Thesis. University of Washington, Seattle, WA. 101pp. (Cited in Cederholm et al. 1981).

Welsh, H.H. and L.M. Ollivier. 1992. Effects of sediments from the Redwood National Park bypass project (Caltrans) on the amphibian communities in streams in Prairie Creek State Park. USDA Forest Service, Pacific Southwest Research Station, 1700 Bayview Drive, Arcata, CA.

VI. APPENDICES

APPENDIX A

Mean Values for Data Collected from Artificial Redds from WY90 to WY93

Table A-1. Mean egg survival in individual stream sections, winter 1991-1992. Prairie Creek reaches are listed in order from upstream to downstream.

Percent Egg Survival				
Stream Reach	WY93	WY92	WY91	WY90[a]
Lost Man Creek	20.6	28.3	--	85.0
Prairie Creek above Brown Creek	59.5	36.0	15.1	77.0
Prairie Creek below Brown Creek	87.3	54.4	12.0	95.0
Prairie Creek below Big Tree Creek	33.3	52.3	9.5	92.5
Prairie Creek below Boyes Creek	1.6	30.3	0.1	89.0

[a] WY90 eggs were eyed steelhead eggs rather than the fertilized "green" coho salmon eggs used in other years. Green steelhead eggs placed in two redds below Big Tree Creek in WY90 averaged 61% survival compared to 93% for eyed.

Table A-2. Mean percent fine sediment < 1.0 mm by weight in artificial redds in individual stream sections over the study period. Prairie Creek reaches are listed in order from upstream to downstream.

Percent Fines				
Stream Reach	WY93	WY92	WY91	WY90[a]
Lost Man Creek	6.60	1.99	--	0.44
Prairie Creek above Brown Creek	5.60	0.57	2.43	0.16
Prairie Creek below Brown Creek	5.53	0.38	2.09	0.19
Prairie Creek below Big Tree Creek	7.82	0.45	2.43	0.23
Prairie Creek below Boyes Creek	6.01	0.71	5.18	0.52

[a] Weight of fines for WY90 were calculated by dividing volume of fines < 1.0 mm by 8.5, the approximate mean ratio of volume to weight in WY92 (another low flow study period). Other years, weight was measured directly.

Table A-3. Mean final inflow rates over the study period at the end of the incubation period in artificial redds (basket pipes) in individual stream sections. Prairie Creek reaches are listed in order from upstream to downstream.

	Mean Final Inflow Rate (cm^3/sec)			
Stream Reach	WY93	WY92	WY91	WY90[a]
Lost Man Creek	76.8	89.4	--	--
Prairie Creek above Brown Creek	72.0	91.2	74.1	96.8
Prairie Creek below Brown Creek	85.6	89.3	77.0	38.7
Prairie Creek below Big Tree Creek	83.6	97.1	98.9	--
Prairie Creek below Boyes Creek	94.7	95.6	60.0	38.9

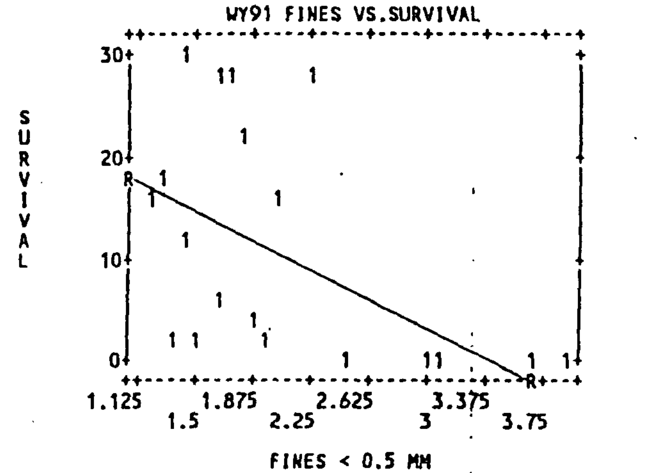
[a] Rates in the affected reach were measured below tributaries in artificial redds (Mill Creek pipes), but not in WY90 egg baskets, which were exposed to fewer and smaller storms.

Table A-4. Mean dissolved oxygen at the end of the incubation period in artificial redds individual stream sections. Prairie Creek reaches are listed in order from upstream to downstream.

	Dissolved Oxygen (ppm)			
Stream Reach	WY93	WY92	WY91	WY90
Lost Man Creek	11.6	10.5	--	--
Prairie Creek above Brown Creek	11.4	10.7	11.1	10.2
Prairie Creek below Brown Creek	11.3	11.6	10.7	10.0
Prairie Creek below Big Tree Creek	11.4	10.7	11.1	--
Prairie Creek below Boyes Creek	11.4	10.8	11.4	8.2

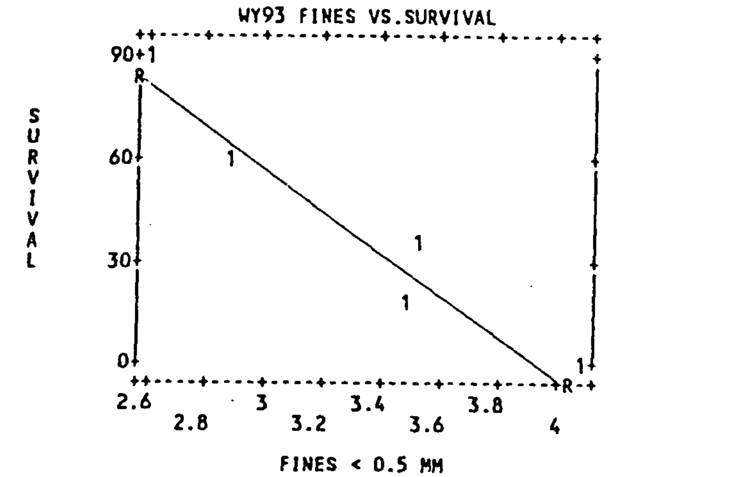
APPENDIX B

Regression Plots



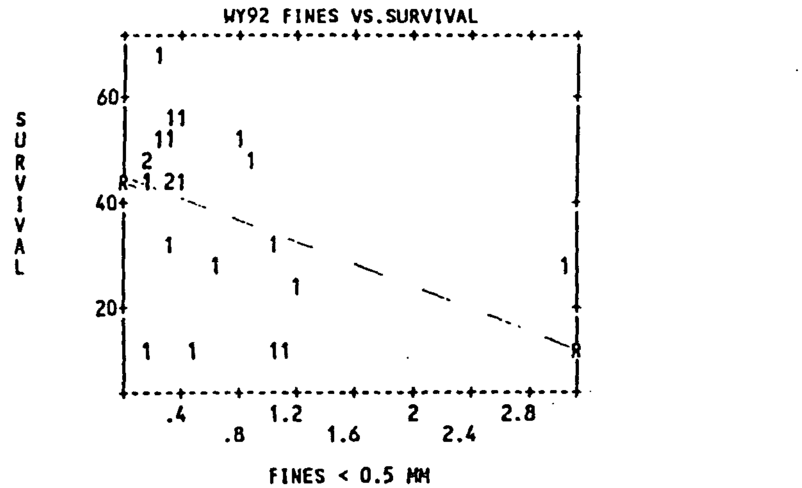
SPSS/PC+

19 cases plotted. Regression statistics of SURV on W0.5:
Correlation -.50105 R Squared .25106 S.E. of Est 10.20975 Sig. .0289
Intercept(S.E.) 26.04769(6.58181) Slope(S.E.) -7.10605(2.97676)



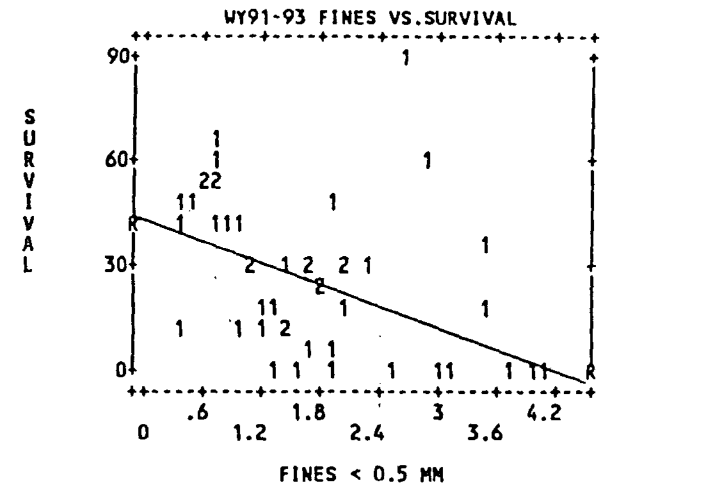
SPSS/PC+

5 cases plotted. Regression statistics of AVGSRV on AW0.5:
Correlation -.97629 R Squared .95315 S.E. of Est 8.39486 Sig. .0044
Intercept(S.E.) 229.55417(24.49234) Slope(S.E.) -56.98553(7.29442)



SPSS/PC+

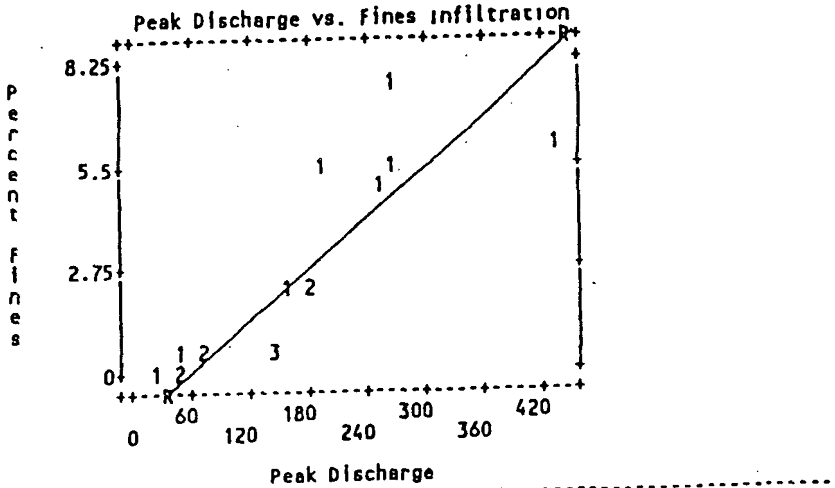
22 cases plotted. Regression statistics of AVGSRV on AVGW0.5:
Correlation -.39155 R Squared .15332 S.E. of Est 15.83269 Sig. .0715
Intercept(S.E.) 44.63692(4.72912) Slope(S.E.) -10.20372(5.36182)



SPSS/PC+

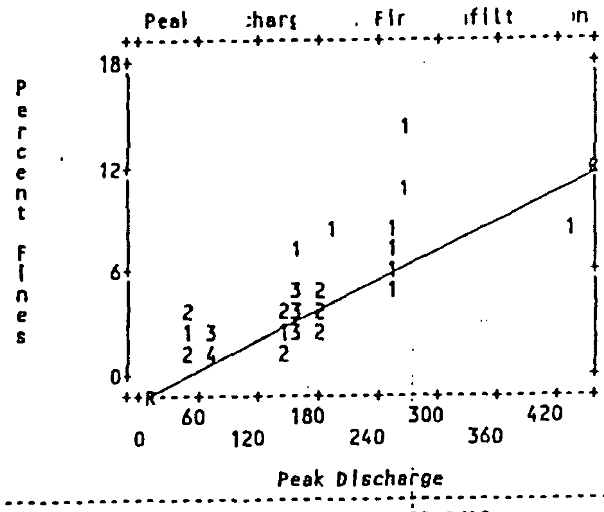
46 cases plotted. Regression statistics of AVGSRV on IW0.5:
Correlation -.41540 R Squared .17256 S.E. of Est 19.95288 Sig. .0041
Intercept(S.E.) 42.35135(5.73836) Slope(S.E.) -8.87446(2.92963)

Figure B-1. Regression plots of weight of percent fine sediment < 0.5 mm vs. salmon egg survival for each water year and WY91-93 combined. Statistics may vary slightly from text because values in text were based on arcsin square root transformed values.



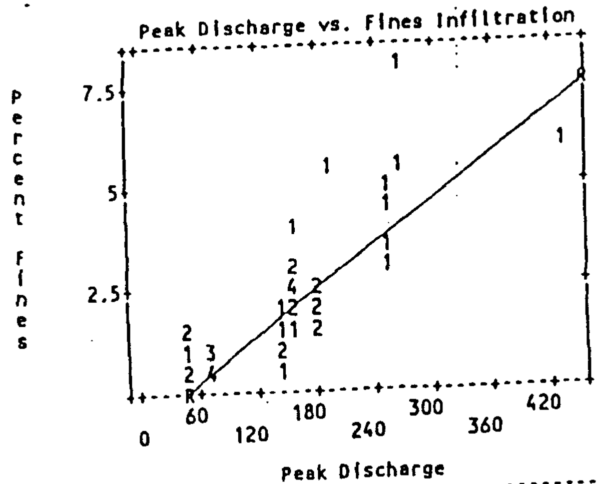
SPSS/PC+

17 cases plotted. Regression statistics of MEANFINE on PEAKQ:
Correlation .84221 R Squared .70932 S.E. of Est 1.43610 Sig. .0000
Intercept(S.E.) -.84262(.63855) Slope(S.E.) .02074(.00343)



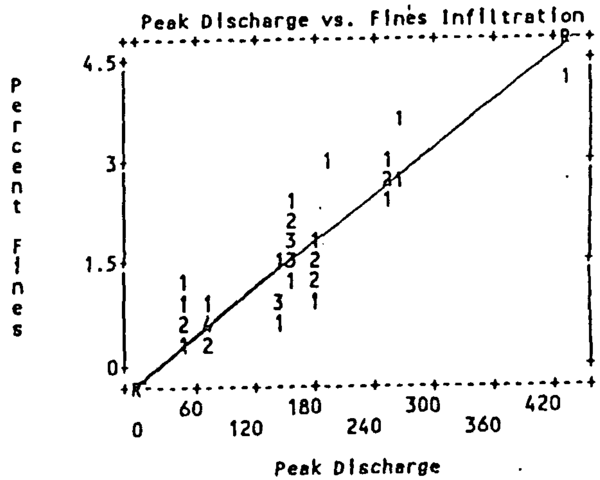
SPSS/PC+

41 cases plotted. Regression statistics of IW4.75 on PEAKQ:
Correlation .72783 R Squared .52974 S.E. of Est 2.04981 Sig. .0000
Intercept(S.E.) -.17688(.71513) Slope(S.E.) .02785(.00420)



SPSS/PC+

41 cases plotted. Regression statistics of IW1.0 on PEAKQ:
Correlation .80499 R Squared .64801 S.E. of Est 1.04541 Sig. .0000
Intercept(S.E.) -.47399(.36472) Slope(S.E.) .01816(.00214)



SPSS/PC+

41 cases plotted. Regression statistics of IW0.5 on PEAKQ:
Correlation .87469 R Squared .76508 S.E. of Est .45316 Sig. .0000
Intercept(S.E.) -.06079(.15810) Slope(S.E.) .01047(.00093)

Figure B-2. Regression plots of peak discharge during the incubation period vs. weight of fine sediment < 1 mm (left plots) and <4.7mm (upper right) and < 0.5mm (lower right).

APPENDIX C
Raw data collected from artificial redds in WY93

Table C. Summary of data collected from infiltration bags within artificial redds on Prairie Creek, CA 1993.

SITE*	SURVIVAL (%)	WORM WEIGHT (GRAMS)	FINES <0.5 mm VOLUME (%)	FINES <0.5 mm WEIGHT (%)	FINES <1.0 mm VOLUME (%)	FINES <1.0 mm WEIGHT (%)	FINES <4.7 mm VOLUME (%)	FINES <4.7 mm WEIGHT (%)	% OF <4.7 mm FRACTION <1.0 mm, VOLUME	CHANGE IN INFLOW (cm^3/sec)	BEGINNING INFLOW (JANUARY) (cm^3/sec)	ENDING INFLOW (MARCH) (cm^3/sec)
1	1.59	131.70	21.41	4.07	23.02	6.01	25.46	8.91	90.42	-28.28	122.97	94.69
2	33.33	0.00	25.46	3.53	28.84	7.82	31.00	10.56	96.13	-31.06	114.64	83.58
3	87.30	0.00	18.07	2.61	20.54	5.53	28.51	14.87	72.04	-5.81	91.41	85.60
4	59.52	0.00	21.17	2.89	23.55	5.60	26.45	8.91	89.04	0.00	71.96	71.96
5	20.63	0.00	15.42	3.49	18.19	6.60	27.65	17.26	65.79	-24.24	101.00	76.76

*1 = below Boyes, 2 = below Big Tree, 3 = below Brown, 4 = above Brown, 5 = Lost Man Creek

APPENDIX D
COMPARISON TO ESTUARY DATA

Comparison of Chinook Salmon Production Data to RNP Estuary Data. A second way to estimate production from Prairie Creek in 1983 and 1984 is to examine chinook juvenile population estimates in the estuary for those years and determine the proportion that came from Prairie Creek. The population estimate of juvenile chinook salmon in the estuary the summer of 1983 was 26,927 fish in late June; it dropped to 8,300 fish in mid-July when estuarine water levels dropped to very low levels, and increased to 15,161 by early August (1984 RNP memorandum, file code N1423).

McKeon (McKeon 1985) trapped downstream migrants on Redwood Creek below Prairie Creek and on Prairie Creek in 1983. These data suggest over 100% of the fish he counted going into the estuary were from Prairie Creek, which is questionable, particularly when his seines on Redwood Creek just above Prairie Creek produced substantial numbers of fish (certainly in the range of numbers caught on Prairie Creek). More likely, his Redwood Creek trap, which did not span the entire creek, was even less efficient than he estimated. Best guess would be that no more than half of the chinook came from Prairie Creek in 1983. Assuming 18,627 fish left the estuary in early July, and that 50% of the fish came from Prairie Creek, no more than 17,000 estuarine fish could have come from Prairie Creek. Based on lower bounds of 95% confidence intervals for the estuary estimates, the minimum produced would have been about 8,000. Of course, if Prairie Creek's contribution to basin production was less than 50%, these numbers would be much lower.

In 1984, the downstream migrant trap was moved to above Prairie Creek on Redwood Creek and 358 chinook salmon were trapped during similar time periods that 127 were trapped in Prairie Creek (Meyer 1994). Unlike Prairie Creek, the weir on Redwood Creek did not fully span the creek (only 1/3 to 1/2 of the stream width was sampled on June 4), so Redwood Creek clearly produced many more fish than Prairie Creek in 1984. Best guess based on width of Redwood Creek sampled, Prairie Creek contributed about 15% of the chinook salmon entering the estuary in 1984 (no mark-recapture efficiency tests were conducted in 1984). The estuary contained approximately 15,000 chinook on June 21 (Anderson 1990). At that time, only about 1/2 to 2/3 of the fish had migrated downstream, so the peak number of chinook entering the estuary that year might be around 20,000 to 30,000. (The estuary mouth was breached before the July estuary population estimate of 7,500; fish probably left the estuary when the mouth was open (RNP 1985). Prairie Creek's contribution (at 15%) would have been between 3,000 to 4,500 chinook salmon.

Other data do not contradict the estimates of Prairie Creek's percent contribution of chinook salmon. More salmon spawners were in Bridge Creek in 1984 than 1983, whereas the reverse trend appeared in Prairie Creek (Meyer 1994). If Bridge Creek represented conditions in the rest of the Redwood Creek basin, Prairie Creek would be expected to have contributed a greater percentage of the fish in 1983 than 1984. Eleven percent of the total length of streams available for chinook spawning (5th and 6th order streams) in the basin are in Prairie Creek (Brown

1988). Because much of Prairie Creek was in fairly pristine condition before the bypass compared to the rest of the Redwood Creek basin, one would expect Prairie Creek to contribute a greater percentage of chinook salmon than its proportion of available habitat. In a normal year, the proportion would be expected to be at least 15%.

The population estimates from Prairie Creek based on the estuary data are about 4 times greater than numbers obtained from the RNP downstream migrant traps that fully spanned the creek. Interestingly, McKeon (1985) conducted several mark-recapture efficiency tests on Prairie Creek in 1983 and found that on average only 27% of the marked fish were recaptured. His tests were not very effective because the recapture period was very short (often no more than 7 night hours), but the estuary data are suggestive that many fish were missed even with that full-spanning weir. One time, fish were recorded escaping from the holding box.

If the pipe traps placed in Prairie Creek in the 1990's were about 75% as efficient as the full-spanning weir trap, PCFWWRA should have caught about 20% (75% of 27%) of chinook actually migrating down Prairie Creek into the estuary. PCFWWRA's efficiency estimates were usually higher than 20%, averaging between 25 and 50% (Farro 1990, 1991, 1992, 1993, 1994). Data are nonexistent for proportions of salmon coming from Prairie Creek vs. Redwood Creek for the 1990's. However, estuary data from years in which fish probably did not immediately enter the ocean from the estuary can be compared to PCFWWRA's downstream migrant estimates. Of the 1990's, 1992 and 1994 had higher water and an estuary configuration that might have held fish (Anderson 1990, 1991, 1992, 1993, 1994 (latter in preparation)). Using PCFWWRA's trap efficiencies, Prairie Creek would have contributed 0.1% in 1992 and 17% in 1994. These percentages seem reasonable given the strength of the spawning run in Prairie Creek for those years.

These results point out that RNP's downstream migrant data in the 1980's are difficult to compare to PCFWWRA's data. When compared to the estuary, RNP's weekly data do not hold up as well as Farro's continuous data. It is probably best to use the 27% efficiency estimate for the full-spanning weir on Prairie Creek in 1983 and 1984 and multiply by 7 to compare these pre-October storm migrant data to PCFWWRA's post-October storm migrant data.

Estimated steelhead production using estuary data. To test how informative downstream migrant data are for steelhead trout, estuary estimates for steelhead can be compared to downstream migrant estimates in WY83 and WY84. RNP's (Meyer 1994) total downstream migrant steelhead catches adjusted for trap efficiency in 1983 were very similar between Prairie Creek and Redwood Creek below Prairie Creek (Prairie was 96% of Redwood Creek below Prairie Creek). Yet, seining showed that numbers of steelhead caught on Redwood above Prairie Creek were higher than numbers estimated to pass the Redwood Creek trap (even when adjusted for trap efficiency).

In 1984, Prairie Creek contributed about 7% of the downstream migrating steelhead trout from May to July (adjusting Redwood Creek trap numbers for estimated proportion of stream width sampled (Meyer 1994)). Estimates of steelhead in the estuary in 1984 reached a maximum of

31,000 in mid-August 1984 (27,000 was lower bound of 95% confidence interval)(RNP 1985). Although this number likely does not include many fish that left when the water elevation was lowered (the estuary was breached 19 times in 1984 and 22 times in 1983), the estimate is about in the middle of the range for peak estuary estimates for steelhead from WY82 to WY94 (Meyer 1994). Using the 7% figure, a minimum of 1,900 to 2,200 steelhead juveniles would have come from Prairie Creek in WY84. Assuming fish migrated from May to early August, this would be possible if the Prairie Creek trap efficiency in WY84 was 10% or less, which is unlikely. More likely, many Prairie Creek steelhead probably migrated into the estuary before May and after August when the trap was not in. Using 7% and 28,000 estuary steelhead (8,000 was lower bound of 95% confidence interval) estimated for WY83, 2,000 fish would have been produced in Prairie Creek in WY83 (600 using the lower bound) , which would require a 25% trap efficiency that year, close to the estimated efficiency figure. The 1983 estuary estimate was from an early July sampling period, so the effect of August migration was removed. Still, substantial loss to the ocean probably occurred by July as seen with the chinook salmon. More likely, Prairie Creek contributed a greater number of steelhead than 2,000 in 1983. Downstream migrant data for steelhead should be interpreted keeping in mind that the trapping period was inadequate to fully represent steelhead.

Eighteen percent of spawning habitat (3rd to 6th order streams) accessible to steelhead trout in the Redwood Creek basin is in Prairie Creek; however, steelhead are much less numerous in Prairie and its tributaries than other creeks because of apparent competition from coho salmon and cutthroat trout, (Brown 1988, Anderson 1988) suggesting the proportions may more likely be between 7 and 18%.