Appendix C. Studies, Surveys, and Assessments of Covered Species and their Habitats Conducted in the Current Plan Area

CONTENTS

INTRODUCTION

The following are summaries of methods, results and conclusions of numerous investigations Simpson has undertaken on Plan Area properties since at least 1994. These are organized into physical habitat assessments, fish population studies, amphibian surveys, and an analysis and projection of future habitat conditions. Many of these projects have evolved from narrowly focused studies initially employed to answer a single question or monitor relatively few parameters into a comprehensive program across a wide geographic and temporal landscape. The results of these investigations, along with continuing scientific progress in assessing habitat and populations of species inhabiting Simpson's properties have driven the evolution of the methodologies described herein. As they have evolved, many of the monitoring investigations described in this appendix have become the basis for many of the protocols presented and described in the Appendix D of this Plan.

Appendix C1. Channel and Habitat Typing Assessment

CONTENTS

Figures

Tables

C1.1 METHODS

Initial channel and habitat typing assessments were conducted by Simpson Fisheries personnel in 1994 and 1995 following the CDFG methods described by Flosi and Reynolds (1994). Prior to the onset of assessments, Simpson's fisheries field technicians participated in a four-day training seminar sponsored by CDFG in order to become familiar with the methodology. In the 1995 season, Simpson field personnel followed the 10% sampling scheme modification proposed by CDFG to reduce the time required for this assessment (Hopelain 1995). All field data was entered into the Habitat Program (Flosi and Reynolds 1994) and resulting data tabulated, summarized, and discussed below.

During those two years Simpson fisheries personnel assessed sixteen streams on Simpson's ownership in the HPAs, identifying 75 reaches by channel type for a total of over 94 miles of stream channel examined (Table C1-1). The sixteen streams assessed were selected based on their biological significance as producers of salmonids, and the size of Simpson's ownership in the watershed's anadromous reaches.

Additionally, channel and habitat typing assessments of streams on Simpson's ownership in the HPAs also were conducted by the Yurok Tribal Fisheries Program (YTFP) (31 streams during VN1996-1998), the California Conservation Corp (CCC) (3 streams in 1995), the Louisiana Pacific Corp. (4 streams in 1994), and the California Department of Fish and Game (CDFG) (4 streams in 1991 and 1998). Assessments by those entities were conducted on 42 streams covering more than 149 reaches for a total of over 135 miles of channel (Table C1-1).

For the purposes of summarizing and comparing stream channel and habitat parameters several of the channel and habitat typing variables (canopy closure, % conifer canopy, % LWD as structural shelter, and % of stream length in pool) were plotted against stream watershed area. These variables were mean values for the entire length of stream that was surveyed. For comparison purposes to other surveyed streams within each HPA the watershed area was determined at the midpoint of the surveyed reach of stream. The dry sections of channel in the lower portion of the watershed were not included in the overall stream length. The mid point of the wetted channel length normalizes the stream size based on the relative position in the watershed where the survey occurred and the mean values of interest. The least squares regression displayed on these figures was added for comparison purposes only and not intended for statistical analysis. These data were not transformed to find the best fit but just to get a general sense of how conditions in certain HPAs compare with those other HPAs. The R^2 and p-values are also shown on the figures.

To allow the comparison of pool tail-out embeddedness between assessed streams, a stream gradient was determined from the channel types. Each channel type has a delineation criteria based on a range of channel gradients. To derive an average stream gradient, the mean gradient of each channel type criteria was weighted according to the length of each channel type.

Table C1-1. Summary of the channel and habitat typing assessments conducted during 1991-1998 on Simpson's ownership in the HPAs.

HPA	Surveyed By:										Totals	
	Simpson		Yurok Tribal Fisheries Program		Louisiana-Pacific		CCC[1]		CDFG[2]			
	No. streams	Miles	No. streams	Miles	No. streams	Miles	No. streams	Miles	No. streams	Miles	No. streams	Miles
Smith River	4	22.99	x	x	x	x	X	x	x	x	4	22.99
Coastal Klamath	6	35.35	16	52.46	x	x	X	x	x	x	22	87.81
Blue Creek	x	X	4	21.63	x	x	X	x	x	x	4	21.63
Interior Klamath	x	X	11	30.23	x	x	X	x	x	x	11	30.23
Redwood Creek	x	X	x	x	x	x	X	x	x	x	0	0
Coastal Lagoons	x	X	x	x	x	x	X	x	x	x	0	0
Little River	x	X	x	x	4	18.02	X	x	x	x	4	18.02
Mad River	3	11.29	x	x	x	x	X	x	x	x	3	11.29
NF Mad River	2	18.03	x	x	x	x	X	x	x	x	2	18.03
Humboldt Bay	1	7.04	x	X	x	x	3	7.04	x	x	4	14.08
Eel River	x	X	x	X	x	x	X	x	4	5.84	4	5.84
TOTALS	16	94.70	31	104.32	4	18.02	3	7.04	4	5.84	58	229.92

[1]California Conservation Corps
[2]California Department of Fish and Game

C1.2 RESULTS

Results of the channel and habitat typing assessments for the 58 streams are summarized in Tables C1-2 through C1-8. These results are discussed in more detail in the following discussion and conclusions section below.

C1.3 DISCUSSION

The following discussion is based on the results of the channel and habitat typing assessments presented in Tables C1-2 through C1-8.

Table C1-2. Stream assessment summaries for four Plan Area streams in the Smith River HPA.

Parameters	Streams			
	SF Winchuck River	Dominie	Wilson	Rowdy
Year Assessed	1995	1995	1994	1995
Assessed by	Simpson	Simpson	Simpson	Simpson
Total Length of Channel Assessed (feet)	31,961	17,118	35,640	36,668
Mean % Canopy Density	92	94	79	63
% deciduous	98	93	94	97
% conifer	2	7	6	3
% LWD as Structural Shelter in All Pools	6.4	18.2	21.8	5.6
Habitat Types as % of Total Length				
Riffles	41	51	25	24
Flat-water	32	29	41	42
Pools	27	20	28	33
Dry Channel	0	0	7	1
Pool Tailout Embeddedness as % Occurrence				
0-25%	27.3	0.5	37.0	32.5
26-50%	37.2	31.3	35.5	41.0
51-75%	19.1	21.5	28.0	17.5
76-100%	16.4	46.8	0.0	6.3
Maximum Pool Depths as % Occurrence				
<1' deep	0.6	0.9	0.0	20.4
1'-2' deep	4.3	53.7	5.9	2.0
2'-3' deep	40.2	41.7	39.1	7.1
3'-4' deep	39.6	3.7	27.2	33.7
>4' deep	15.2	0.0	27.8	36.7
Index of Embeddedness	3.5	3.1	3.3	2.6
Mid-point Gradient (%)	2.1	4.2	1.1	2.4
Mid-point Watershed Area (acres)	4,336	1,356	5,092	10,990

Table C1-3. Stream assessment summaries for 22 Plan Area streams the Coastal Klamath HPA.

Parameters	Streams							
	Hunter	**EF Hunter**	**High Prairie**	**Mynot**	**HPW**	**NF HPW**	**Terwer**	**EF Terwer**
Year Assessed	1994	1996	1996	1996	1996	1996	1994	1996
Assessed by:	Simpson	YTFP	YTFP	YTFP	YTFP	YTFP	Simpson	YTFP
Total Length of Channel Assessed (ft)	54,399	11,846	18,336	10,880	23,404	4,413	62,416	16,131
Mean % Canopy Density	80	88	80	76	90	95	36	71
% deciduous	93	93	77	85	91	73	75	95
% conifer	7	7	23	15	9	27	25	5
% LWD as Structural Shelter in all Pools	35	55.1	36.4	15.8	46.1	33.1	16.5	6.8
Habitat Types as % of Total Length								
Riffles	8.0	1	8	0	15	22	19.0	7
Flat-water	32.0	41	35	6	28	9	43.0	59
Pools	17.0	15	37	6	19	52	31.0	34
Dry Channel	43.0	44	19	86	38	14	7.0	0
Culvert	0	0	1	0	0	0	0	0
Pool Tailout Embeddedness as % Occurrence								
0-25%	24.7	0	2.3	0	1	0	31.3	9.0
26-50%	57.0	19	46.0	11	19.4	35	45.0	76.0
51-75%	18.2	47	49.4	79	69	63	21.3	15.0
76-100%	0	33	2.8	11	10.6	2	0	0
Maximum Pool Depths as % Occurrence								
<1' deep	0.0	1.8	9.7	21.1	5.0	10.4	0.5	1.6
1'-2' deep	8.0	56.1	55.7	57.9	70.5	60.4	1.5	48.4
2'-3' deep	38.3	31.6	27.8	15.8	22.7	29.2	19.8	36.3
3'-4' deep	32.5	8.8	6.1	0	1.8	2.1	28.9	9.3
>4' deep	21.4	1.8	1.0	5.3	0	0	49.2	4.4
Index of Embeddedness	2.7	2.9	3.0	3.0	1.6	2.6	2.5	1.6
Mid-point Gradient (%)	1.6	NA	3.6	NA	1.7	3.0	1.5	NA
Mid-point Watershed Area (acres)	4,898	1,031	2,134	526	1,012	522	8,602	3,523

Codes

HPW	Hoppaw Creek	NF HPW	North Fork Hoppaw
EF	East Fork	NA	Not applicable, or not available

Table C1-3 Continued. Stream assessment summaries for 22 Plan Area streams in the Coastal Klamath HPA.

Parameters	Streams							
	McG	**WF McG**	**Tarup**	**Omagar**	**APCM**	**APCS**	**APCN**	**A-P Trib**
Year Assessed:	1996	1996	1996	1996	1995	1995	1995	1997
Assessed by	YTFP	YTFP	YTFP	YTFP	Smpsn	Smpsn	Smpsn	YTFP
Total Length of Channel Assessed (feet)	29,085	13,033	26,343	13,276	17,299	8,284	26,669	3,132
Mean % Canopy Density	89	94	97	95	91	95	93	84
% deciduous	92	89	93	90	97	94	89	90
% conifer	8	11	7	10	3	6	11	10
% LWD as Structural Shelter in all Pools	37.8	41.2	25.4	43.4	15.1	35.8	9.6	27.1
Habitat Types as % of Total Length								
Riffles	4	6	10	10	28.0	46.0	37.0	6
Flat-water	25	20	19	39	31.0	29.0	29.0	54
Pools	69	73	71	26	17.0	24.0	25.0	39
Dry Channel	1	1	0	0	24.0	1.0	9.0	1
Culvert	0	0	0	23	0	0	0	0
Pool Tailout Embeddedness as % Occurrence								
0-25%	0.4	0	1.6	7.0	9.0	15.0	9.8	44.1
26-50%	15.5	2.7	26.5	51.0	33.3	23.0	19.3	55.9
51-75%	66.7	62	71.1	38.3	27.9	21.0	27.0	0
76-100%	17.7	35.5	0.9	3.7	24.9	41.0	43.7	0
Maximum Pool Depths as % Occurrence								
<1' deep	6.5	13.9		15.1	2.2	1.5	0.6	19.2
1'-2' deep	42.8	47.5	30.3	56.0	30.1	67.6	29.3	56.2
2'-3' deep	32.1	27	43.9	16.4	45.2	29.4	48.1	20.5
3'-4' deep	10.7	25	16.8	5.0	17.2	1.5	17.1	4.1
>4' deep	7.8	1.6	9.0	0.0	5.4	0.0	5.0	0.0
Index of Embeddedness	1.9	2.1	2.7	2.9	2.7	3.1	3.0	2.2
Mid-point Gradient (%)	1.8	2.7	5.6	3.9	1.7	4.5	2.1	5.6
Mid-point Watershed (acres)	1,672	1,296	1,971	773	2,573	1,290	2,437	1,076

Codes

McG	McGarvey Creek	WF McG	West Fork McGarvey Creek
APCN	North Fork Ah Pah Creek	APCM	Main stem Ah Pah Creek
A-P Trib	Tributary to Main stem Ah Pah	APCS	South Fork Ah Pah Creek

Table C1-3 Continued. Stream assessment summaries for 22 Plan Area streams in the Coastal Klamath HPA.

Parameters	Streams					
	Bear	**Bear (Trib 1)**	**Bear (Trib 2)**	**Surpur**	**Little Surpur**	**Tectah**
Year Assessed	1995	1996	1996	1996	1996	1996
Assessed by	Smpsn	YTFP	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed (feet)	17,581	7,102	4,242	18,046	11,072	66,632
Mean % Canopy Density	88	77	78	89	93	86
% deciduous	93	93	91	94	91	89
% conifer	7	7	9	6	9	11
% LWD as Structural Shelter in all Pools	19.8	9.8	22.7	13.2	18.2	14.6
Habitat Types as % of Total Length						
Riffles	58	14	3	4	0	6
Flat-water	24	53	64	23	33	44
Pools	16	33	31	73	61	48
Dry Channel	2	0	2	0	6	2
Culvert	0	0	0	0	0	0
Pool Tailout Embeddedness as % Occurrence						
0-25%	4.5	1.9	0.0	1.0	0.0	0.0
26-50%	22.3	79.4	73.0	36.0	31.3	68.0
51-75%	54.3	18.4	27.0	61.0	66.7	32.0
76-100%	19.0	0.0	0.0	3.0	2.1	0.0
Maximum Pool Depths as % Occurrence						
<1' deep	60.0	8.2	24.2	0.6	1.6	5.7
1'-2' deep	6.0	71.4	56.1	42.3	42.6	35.9
2'-3' deep	19.0	15.3	15.2	37.2	36	30.6
3'-4' deep	6.0	4.1	4.5	17.3	18.2	14.3
>4' deep	9.0	2.0	0.0	2.6	1.6	13.5
Index of Embeddedness	2.7	2.3	2.9	2.4	2.5	2.3
Mid-point Gradient (%)	3.4	4.2	NA	NA	4.0	NA
Mid-point Watershed (acres)	5,112	1.186	1.442	2.712	1.363	7,434

Table C1-4. Stream assessment summaries for four Plan Area streams in the Blue Creek HPA.

Parameters	Streams			
	Blue	WF Blue	Potato Patch	Slide
Year Assessed	1998	1995	1997	1997
Assessed by	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed	77,144	22,842	2,162	12,050
Mean % Canopy Density	42	87	95	38
% deciduous	66	94	90	23
% conifer	34	6	10	77
% LWD as Structural Shelter in all Pools	4.0	6.0	1.5	3.3
Habitat Types as % of Total Length				
Riffles	16	49	13	16
Flat-water	61	23	56	65
Pools	23	27	30	19
Dry Channel	0	1	0	0
Pool Tailout Embeddedness as % Occurrence				
0-25%	6.1	10.2	0.0	0.9
26-50%	75.1	31.3	28.7	65.3
51-75%	17.5	53.1	68.7	31.0
76-100%	1.3	4.7	2.7	2.8
Maximum Pool Depths as % Occurrence				
<1' deep	0.6	78.4	0	0
1'-2' deep	6.3	1.1	45.5	12.9
2'-3' deep	5.0	8.7	39.4	44.7
3'-4' deep	21.4	8.3	12.1	32.9
>4' deep	66.4	3.5	3.0	9.4
Index of Embeddedness	2.9	2,2	2,1	2,7
Mid-point Gradient (%)	2.0	6.1	5.7	6.6
Mid-point Watershed Area (acres)	38,563	4,372	2,820	3,414

Table C1-5. Stream assessment summaries for 11 Plan Area streams in the Interior Klamath HPA.

Parameters	Streams				
	Johnson	Pecwan	EF Pecan	Mettah	SF Mettah
Year Assessed	1996	1997	1997	1997	1997
Assessed by	YTFP	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed	11,906	4,239	1,836	36,801	8,482
Mean % Canopy Density	94	74	86	86	89
% deciduous	97	69	76	83	78
% conifer	3	31	24	17	22
% LWD as Structural Shelter in all Pools	9.3	1.7	4.3	10.3	19.9
Habitat Types as % of Total Length					
Riffles	3	14	16	10	12
Flat-water	24	62	30	51	64
Pools	60	24	54	40	24
Dry Channel	13	0	0	0	0
Pool Tailout Embeddedness As % Occurrence					
0-25%	0	0	0	0.0	0
26-50%	6.0	7.1	0	23	5.0
51-75%	93.0	92.9	100	76.6	92.0
76-100%	1.0	0	0	0.8	3.0
Maximum Pool Depths as % Occurrence					
<1' deep	4.2	0	0	4.7	0
1'-2' deep	46.9	19.0	10.0	56.5	54.1
2'-3' deep	33.3	33.3	35.0	27.7	38.8
3'-4' deep	11.5	33.3	30.0	8.4	7.1
>4' deep	4.2	14.3	25.0	2.9	0
Index of Embeddedness	3.0	3.0	3.0	2.8	3.0
Mid-point Gradient (%)	NA	3.5	4.1	2,8	3.0
Mid-point Watershed Area (acres)	1,307	17,574	8,401	2,959	1,558

Table C1-5 Continued. **Stream assessment summaries for 11 Plan Area streams in the Interior Klamath HPA.**

Parameters	Streams					
	Roach	**Roach (Trib)**	**Morek**	**Cappel**	**Tully**	**Robbers Ck**
Year Assessed	1997	1997	1997	1997	1997	1997
Assessed by	YTFP	YTFP	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed	38,876	6,235	2,060	3,529	41,995	3,643
Mean % Canopy Density	78	80	85	79	79	84
% deciduous	70	73	66	59	92	92
% conifer	30	27	34	41	8	8
% LWD as Structural Shelter in all Pools	3.5	16.6	6.4	5.7	12.7	10.5
Habitat Types as % of Total Length						
Riffles	4	2	22	27	5	8
Flat-water	48	41	45	31	70	52
Pools	45	53	21	42	24	31
Dry Channel	3	3	13	0	2	1
Pool Tailout Embeddedness As % Occurrence						
0-25%	0	0	0	0	27.6	4.8
26-50%	0	0	16.6	2.0	54.6	32.1
51-75%	100	100	83.4	98.0	0	63.2
76-100%	0	0	0	0	0	0
Maximum Pool Depths as % Occurrence						
<1' deep	1.1	0	9.0	2.3	0.8	6.2
1'-2' deep	30.6	52.4	40.1	14.0	28	43.7
2'-3' deep	30.6	30.2	45.4	65.1	41.4	37.4
3'-4' deep	21.0	12.7	4.5	14.0	19.2	10.4
>4' deep	16.7	4.8	0	4.7	10.7	2.1
Index of Embeddedness	2.4	3.0	2.8	3.0	1.9	3.0
Mid-point Gradient (%)	2.2	2.6	4.7	7.0	4.1	5.0
Mid-point Watershed Area (acres)	10,808	3,548	2,562	5,312	7,264	2,106

Table C1-6. Stream assessment summaries for four Plan Area streams in the Little River HPA.

Parameter	Streams			
	USFLR	**LSFLR**	**RR**	**LR**
Year Assessed	1994	1994	1994	1994
Assessed by	L-P	L-P	L-P	L-P
Total Length of Channel Assessed (feet)	10539	14998	7,262	62,373
Mean % Canopy Density	99	98	98	91
% deciduous	76	67	69	84
% conifer	24	33	31	16
% LWD as Structural Shelter in All Pools	25.9	38.5	26.6	17.3
Habitat Types as % of Total Length				
Riffles	32	30	37	19
Flat-water	20	11	7	25
Pools	45	56	46	53
Dry Channel	3	3	10	3
Pool Tailout Embeddedness as % Occurrence				
0-25%	21.7	14.2	10.5	8.1
26-50%	44.0	46.3	49.2	41.1
51-75%	17.2	31.4	31.9	38.7
76-100%	16.6	8.3	8.1	12.1
Maximum Pool Depths as % Occurrence				
<1' deep	6.8	5.0	26	2.7
1'-2' deep	49.5	43.4	50.0	20.4
2'-3' deep	31.8	31.4	18.7	26.8
3'-4' deep	6.8	7.5	4.4	26
>4' deep	4.5	12.6	1.1	23.6
Index of Embeddedness	2.3	2.3	1.9	3.2
Mid-point Gradient (%)	3.1	1.6	2.9	3.0
Mid-point Watershed Area (acres)	3,095	2,611	1,205	9,475

Codes

USFLR	Upper South Fork Little River
LSFLR	Lower South Fork Little River
RR	Railroad Creek
LR	Mainstem Little River
NA	Not applicable or not available

Table C1-7. Stream assessment summaries for five Plan Area streams in the Mad River HPA and North Fork Mad River HPA.

Parameter	Mad River HPA			North Fork Mad River HPA	
	Streams			Streams	
	CC	DC	LC	NFMR	LPC
Year Assessed	1994	1994	1995	1994	1994
Assessed by	Smpsn	Smpsn	Smpsn	Smpsn	Smpsn
Total Length of Channel Assessed (feet)	24,862	4,512	30,227	80,278	14,928
Mean % Canopy Density	81	92	79	73	95
% deciduous	85	75	79	95	87
% conifer	15	25	21	5	13
% LWD as Structural Shelter in All Pools	16.7	14	26.9	12.1	10.4
Habitat Types as % of Total Length					
Riffles	26	67	9	11	47
Flat-water	27	14	41	38	23
Pools	47	16	50	42	30
Dry Channel	0	3	0	10	0
Pool Tailout Embeddedness as % Occurrence					
0-25%	16.7	30.5	3.0	18.1	6.0
26-50%	41	40.8	16.0	19.3	21.3
51-75%	32.1	18.3	22.0	28.6	20.9
76-100%	11.2	11.1	60.0	33.6	51.9
Maximum Pool Depths as % Occurrence					
<1' deep	1.0	6.1	0.4	07.4	3.5
1'-2' deep	19.6	78.8	12.7	10.7	41.6
2'-3' deep	39.0	9.1	38.3	33.6	39.8
3'-4' deep	22.7	3.03	32.8	26.6	12.6
>4' deep	17.6	3.03	15.6	28.2	2.3
Index of Embeddedness	2.4	2.1	3.4	2.8	2.5
Mid-point Gradient (%)	3.0	3.7	1.0	1.4	2.6
Mid-point Watershed Area (acres)	8,595	1,492	2,985	11,273	4,592

Codes

DC	Dry Creek	NFMR	North Fork Mad River
CC	Cañon Creek	LPC	Long Prairie Creek
LC	Lindsay Creek	NA	Not applicable or not available

Table C1-8. Stream assessment summaries eight Plan Area streams in the Humboldt Bay HPA and Eel River HPA.

Parameter	Humboldt Bay HPA				Eel River HPA			
	Streams				Streams			
	RC	RC(a)	RC(b)	SC	WC	ST	HW	WFH
Year Assessed	1995	1995	1995	1994	1991	1991	1998	1998
Assessed by	CCC	CCC	CCC	Smpsn	CDFG	CDFG	CDFG	CDFG
Total Length of Channel Assessed (feet)	27,682	1,139	8,342	37,153	2,481	5,063	20,975	2,342
Mean % Canopy Density	94	90	88	88	80	67	57	86
% deciduous	68	NA	NA	83	83	71	81	95
% conifer	32	NA	NA	17	17	29	19	5
% LWD as Structural Shelter in all Pools	49.1	17.1	39.8	27.5	10.0	48.2	4.0	0.0
Habitat Types as % of Total Length								
Riffles	5	3	1	27	86	33	65	74
Flat-water	29	16	37	29	10	37	29	18
Pools	65	81	61	44	4	26	6	7
Dry Channel	1	0	0	0	0	5	0	0
Pool Tailout Embeddedness as % Occurrence								
0-25%	7.5	NS*	NS*	9.8	0	63.8	0.9	0.0
26-50%	22.4			24.5	17.8	17.7	22.3	18.0
51-75%	33.5			34.5	17.8	17.3	62.3	73.0
76-100%	36.6			30.6	64.4	1.1	13.8	9.0
Maximum Pool Depths as % Occurrence								
<1' deep	6	19	2.9	0.6	0.0	0.0	0.0	0.0
1'-2' deep	44.8	54.8	43.8	12.6	83.3	43.1	42.0	81.8
2'-3' deep	30.7	19	35.1	42.5	16.7	39.4	52.0	18.2
3'-4' deep	12.2	7.1	13.9	26.5	0.0	10.6	3.8	0.0
>4' deep	6.2	0.0	4.3	17.9	0.0	7.3	2.3	0.0
Index of Embeddedness	3.0	3.0	4.0	2.8	2.9	2.3	2.4	1.9
Mid-point Gradient (%)	1.0	1.0	1.0	1.0	2.6	3.3	2.1	7.0
Mid-point Watershed Area (acres)	3,669	662	1,293	5,399	1,250	3,308	2,594	3,372

Codes

RC Ryan Creek
RC(a) 1st unnamed trib to RC
RC(b) 2nd unnamed trib to RC
SC Salmon Creek
NS* The CCC judged these pools as 'Not suitable for spawning', and did not record pool tailout embeddedness values.
WC Wilson Creek
ST Stevens Creek
HW Howe Creek
WFH West Fork Howe Creek
NA The value was either not recorded or not applicable

C1.3.1 Mean Percent Canopy Closure and Percent Canopy Cover

The mean percent canopy closure along each assessed stream as a function of watershed area is shown as Figure C1-1. The percentage of canopy closure along stream channels is important for the regulation of water temperatures and as a source of nutrients for the aquatic organisms. This assessment also provides information about the species (conifer, deciduous) composition of the riparian zone.

The mean canopy closure in the 58 assessed streams ranged from 36% in Terwer Creek ([Coastal Klamath HPA] Table C1-3), to 99% in Upper South Fork of Little River ([Little River HPA] Table C1-6) and are shown in Figure C1-1. CDFG's Salmonid Restoration Manual recommends that a mean canopy closure of approximately 80% is required/desirable to maintain suitable summer water temperatures for juvenile coho salmon (Flosi and Reynolds 1994). From the assessments conducted 69% of the streams assessed (40 of 58) had mean canopy closures greater than or equal to 80% (Figure C1-1). As shown in this figure the mean canopy closure percentage diminishes with increased stream watershed size.

The percent canopy cover by type (deciduous and conifer) for the assessed streams are shown in Tables C1-2 through C1-8. The mean percent conifer closure plotted against watershed area is shown as Figure C1-2. The percent of conifer cover ranged from a low of 2% in the South Fork Winchuck River ([Smith River HPA] Table C1-2) to 77% on Slide Creek ([Blue Creek HPA] Table C1-4) and are shown in Figure C1-2. As shown in Figure C1-2, deciduous trees dominated the riparian canopy of the assessed streams, with most of the streams (67%) containing less than 20% conifers along the riparian margin. As shown in the figure, there is a trend with a slightly larger percentage of conifer canopy in larger watersheds as compared to smaller watersheds.

C1.3.2 Percent LWD as Structural Shelter in Pool Habitats

To assess habitat complexity, the dominant structural shelter element and the contribution of other shelter components was determined on a percent basis for each habitat type. LWD is an important shelter component that facilitates numerous functions within certain channel types. LWD is a pool-forming component that adds complexity and cover to stream channels. The percentage of in-channel LWD as shelter should reflect the quantity and quality of potential salmonid habitat and possibly the effects of past management practices.

The results of assessment of LWD as structural shelter in all pools surveyed as part of the habitat assessments are summarized in Tables C1-2 through C1-8. LWD as structure in pools in the assessed streams are shown by watershed area in Figure C1-3. As shown in Figure C1-3, the percentage of LWD as shelter was greatest in stream pools. The percentage of LWD as shelter in pools ranged from a low of 0% in West Fork Howe Creek ([Eel River HPA] Table C1-8) to a high of 55% in East Fork Hunter Creek ([Coastal Klamath HPA] Table C1-3).

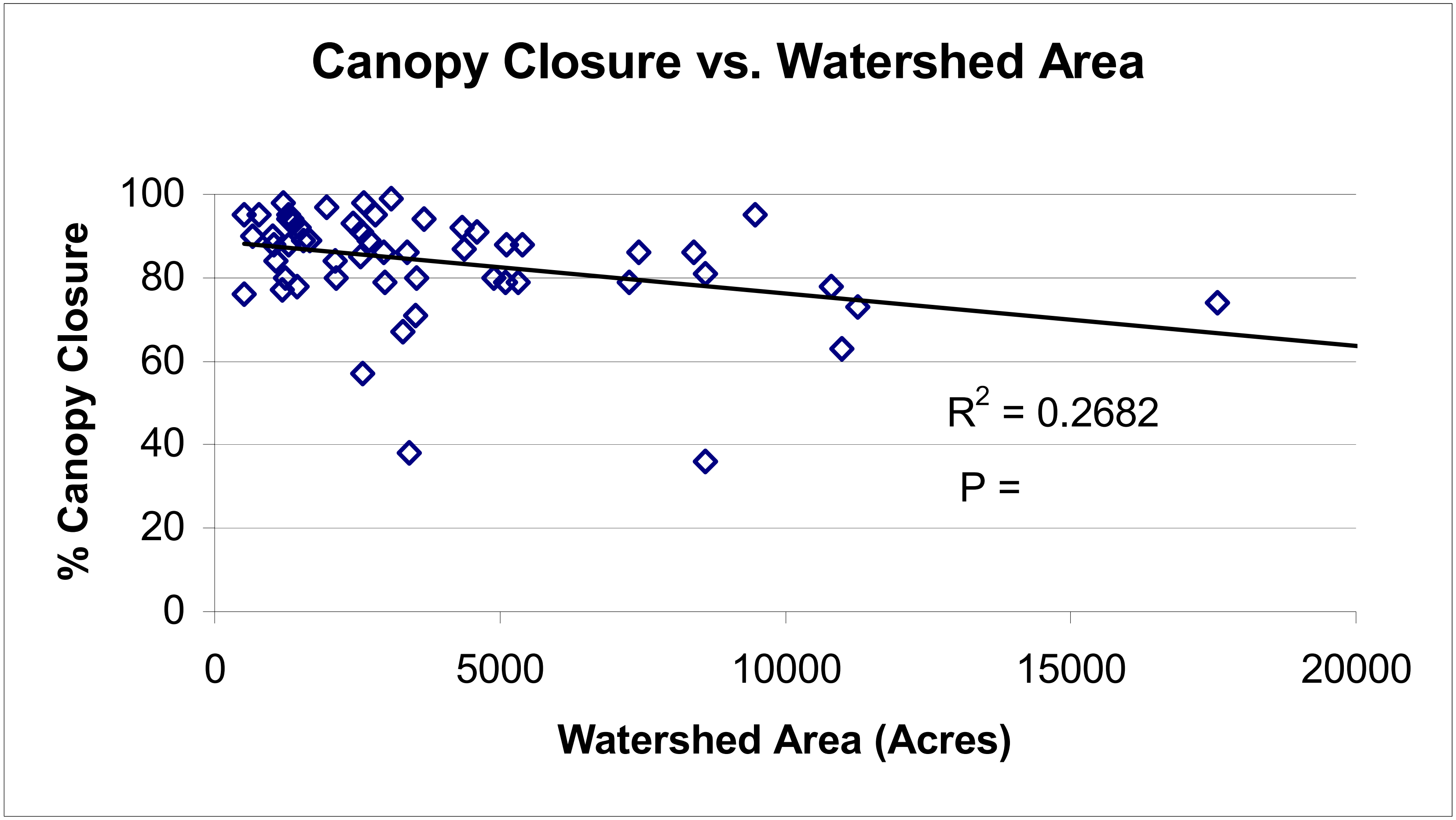

Figure C1-1. **Canopy closure versus watershed area for all assessed streams in which habitat typing surveys were conducted.**

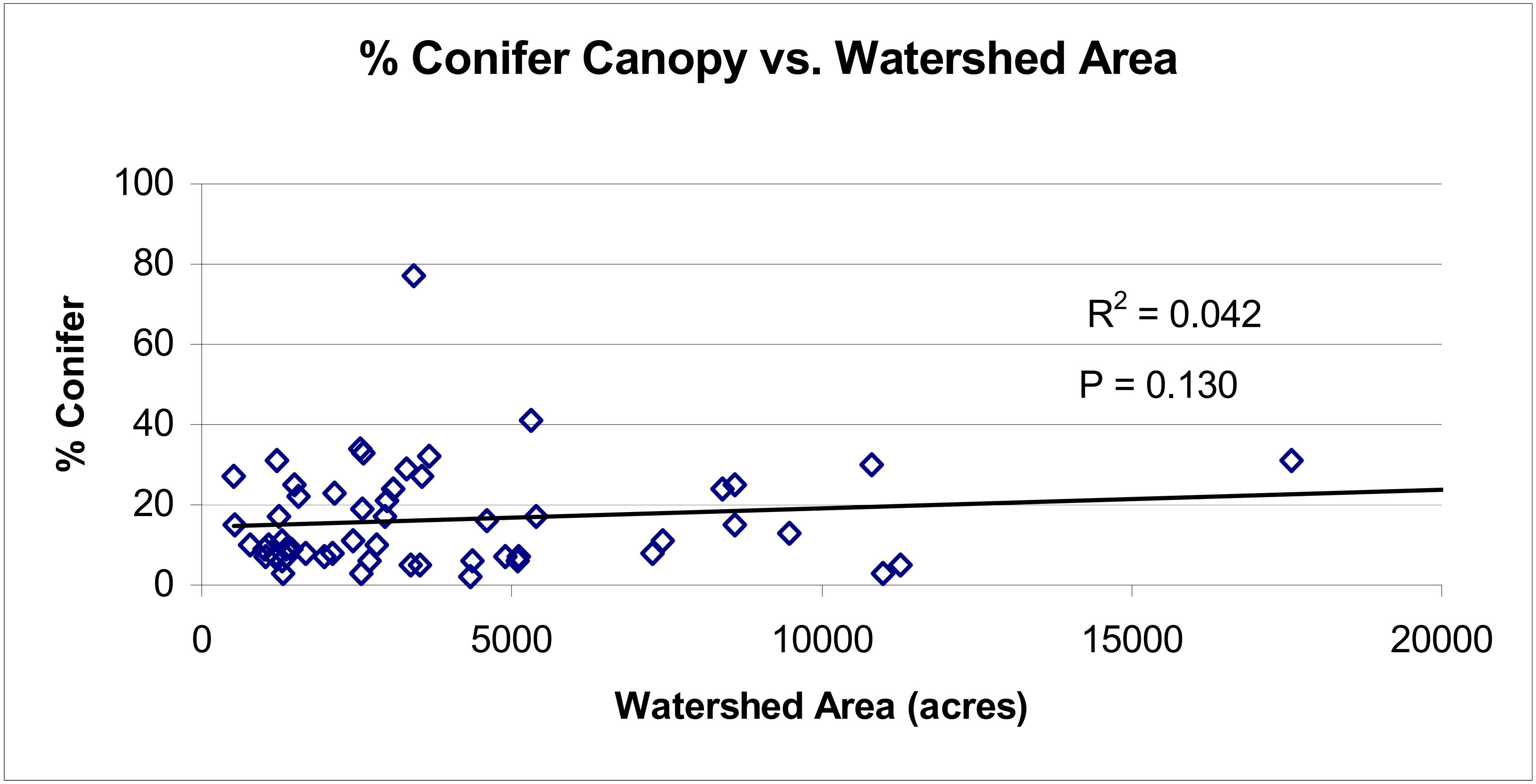

Figure C1-2. **Percent conifer canopy versus watershed area for all assessed streams in which habitat typing surveys were conducted.**

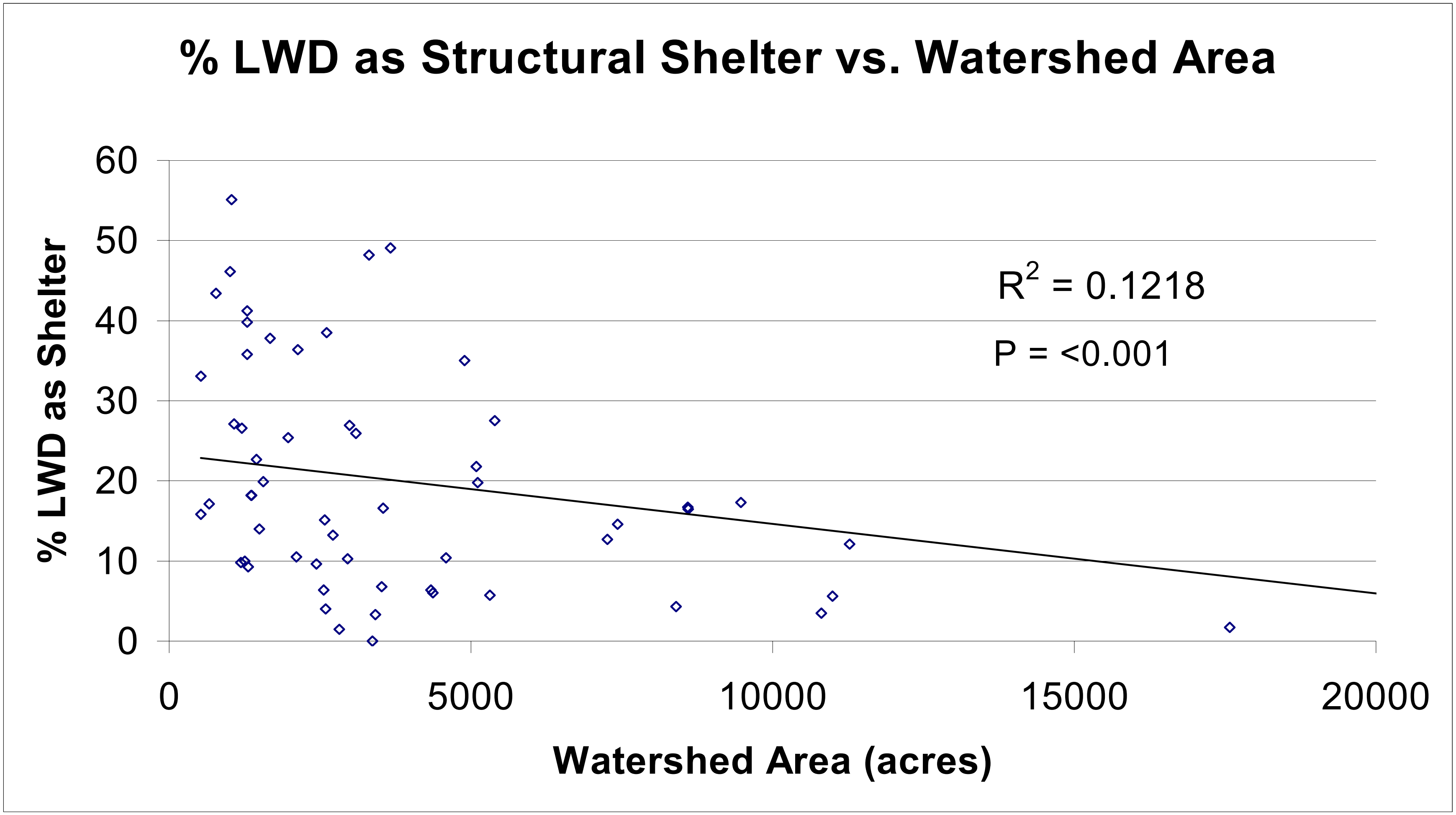

Figure C1-3. **Percentage of LWD as structural shelter versus watershed area for the assessed streams.**

East Fork Hunter Creek was the only stream assessed in which LWD was the dominant (>50%) structural cover. Two additional streams, Ryan Creek ([Humboldt HPA] Table C2-7) with 49%, and Stevens Creek ([Eel River HPA] Table C1-8) with 48% had nearly 50% LWD as structural cover. Of all 58 of the streams assessed, approximately 36% (21 of 58 streams) had LWD as a structural shelter component greater than 20% of all in-stream cover present (Figure C1-3). As shown in that figure there is generally a trend of lower percentages of LWD as structural shelter in pools within streams with larger watershed areas.

The relatively higher amounts of LWD as structural shelter in Hunter Creek, Ryan and Stevens Creeks are probably due to past management practices which retained some riparian cover and also did not aggressively clear the channel of LWD. These watersheds may additionally have some inherent geologic instability that still provides episodic inputs of LWD and sediments to their channels. The lower percentages of LWD in the North Fork Mad River can be attributed to extensive clearing of LWD from the channel. Historic photographs from the mid-1950's show sections of channel clogged with immense jams of logging slash and giant pieces of redwood LWD. Presently, these same sections of channel are nearly devoid of LWD as a result of aggressive stream cleaning efforts during the late 1960's and 1970's. At the time, clearing stream channels of debris jams was deemed by the best available information as a means of fisheries restoration (stream cleaning was also a response to the damage incurred to bridges and roads by debris during the 1955 and 1964 floods). Unfortunately many of these efforts went far beyond improving fish passage and removed what are now regarded as vital habitat components.

C1.3.3 Habitat Types as a Percent of Total Length

Level II (Flosi and Reynolds 1994) partitioning of habitat units separates the stream channel into riffles, flat-water, pools and dry channel. Generally, forming conclusions about the relative health of a stream with respect to salmonids from a level II partitioning of habitat units is difficult. Local geology, channel type, water level, and channel gradient will all influence the relative proportions of each habitat type. However, an extremely high proportion of a certain habitat unit may indicate a channel response to major (either natural or management influenced) watershed disturbances.

Excessive aggradation of stream reaches may lead to a high proportion of riffle habitat as well as an increase in seasonal stretches of dry channel as pools and runs get filled in with sediment. Intermittence is common in steep mountainous watersheds where a majority of the channel is confined and sediments are transported through these areas and are deposited on the wide, low gradient reaches near the mouths. Depending on the watershed this aggradation of sediment can be quite extensive. During low flow conditions the stream will go sub-surface, percolating through the sediment deposits. Many stream channel segments assessed were dry during the assessment surveys.

The summary of the habitat types as a percent of total length of each assessed stream and plotted by watershed area are shown in Tables C1-2 through C1-8. Of the 58 streams evaluated, there were 59% (34 out of 58) which had at least 1% of their total length of stream channel classified as dry channel. Three streams had greater than 40% of their total channel classified as dry: Hunter Creek (43%), East Fork Hunter Creek (44%) and Mynot Creek (86%) all within the Coastal Klamath HPA (Table C1-3).

Many watersheds within the Plan Area exhibit this naturally occurring phenomenon. However, the increased sediment loads from hillslope failures often associated with logging activities and road construction can amplify the spatial and temporal extent of intermittency (Hicks et al. 1991). The impact of intermittency on salmonid populations has not been quantified, but probably affects the out-migration of juveniles or may result in the stranding of juveniles in isolated pools where they would be susceptible to threshold temperatures and increased predation.

For the streams assessed, the percent of stream length of pools ranged from 4% in Wilson Creek ([Eel River HPA] Table C2-8) to 81% in Ryan Creek ([Humboldt Bay HPA] Table C2-7). The percent of stream length of pools by watershed area are shown in Figure C1-4. As shown in Figure C1-4 the percentage of stream length of pools were widely variable in smaller watersheds (less than 5000 acres). For the 58 streams assessed, the percent of total stream length of riffles ranged from 0% in Mynot Creek and Little Surper Creek ([Coastal Klamath HPA] Table C1-3) to 86% in Wilson Creek ([Eel River HPA] Table C1-8). The percentage of stream length of flat-water habitats ranged from 6% in Mynot Creek ([Coastal Klamath River HPA] Table C2-3) to 70% in Tully Creek in the Interior Klamath River HPA (Table C1-4). The trend is that as watershed size increases beyond 5,000 acres, the variability in pool lengths as a total of stream length decreases.

C1.3.4 Pool Tail-out Embeddedness as Percent Occurrence

Summary of pool-tail out embeddedness estimates are shown in Tables C1-2 through C1-8. The embeddedness of channel substrate in pool tail-outs is a gross indication of the amount of fines present in spawning gravels which, in turn, may reduce the survival to emergence of salmonid alevins. However, the measurement is subjective and probably not accurately repeatable. If embeddedness was considered high (>50%), a more rigorous monitoring of substrate composition may be warranted to document amount of fines within pool tail-outs. Of the 58 assessed streams, 60% (35 out of 58) had embeddedness occurrences greater than 50%. From these assessments, 3 streams: East Fork Pecwan, Roach Creek, and a tributary to Roach Creek (all in the Interior Klamath HPA) had pool tail-out embeddedness occurrences of 100%.

An index of Pool tail-out embeddedness as a function of stream gradient for the assessed streams is shown in (Figure C1-5). Using embeddedness index categories of 1 through 4 which correspond to estimates of percent embeddedness of: 0-25% =1; 26-50% = 2; 51-75% = 3; and 76-100% = 4 the streams were categorized as shown in Figure C1-5 (Flosi et al. 1998). As shown in Figure C1-5 the estimated embeddedness for all Plan Area streams assessed generally were found to fall within the range of Index values of 2 to 3 regardless of stream gradient and the average index rating only diminished slightly for streams with larger watersheds.

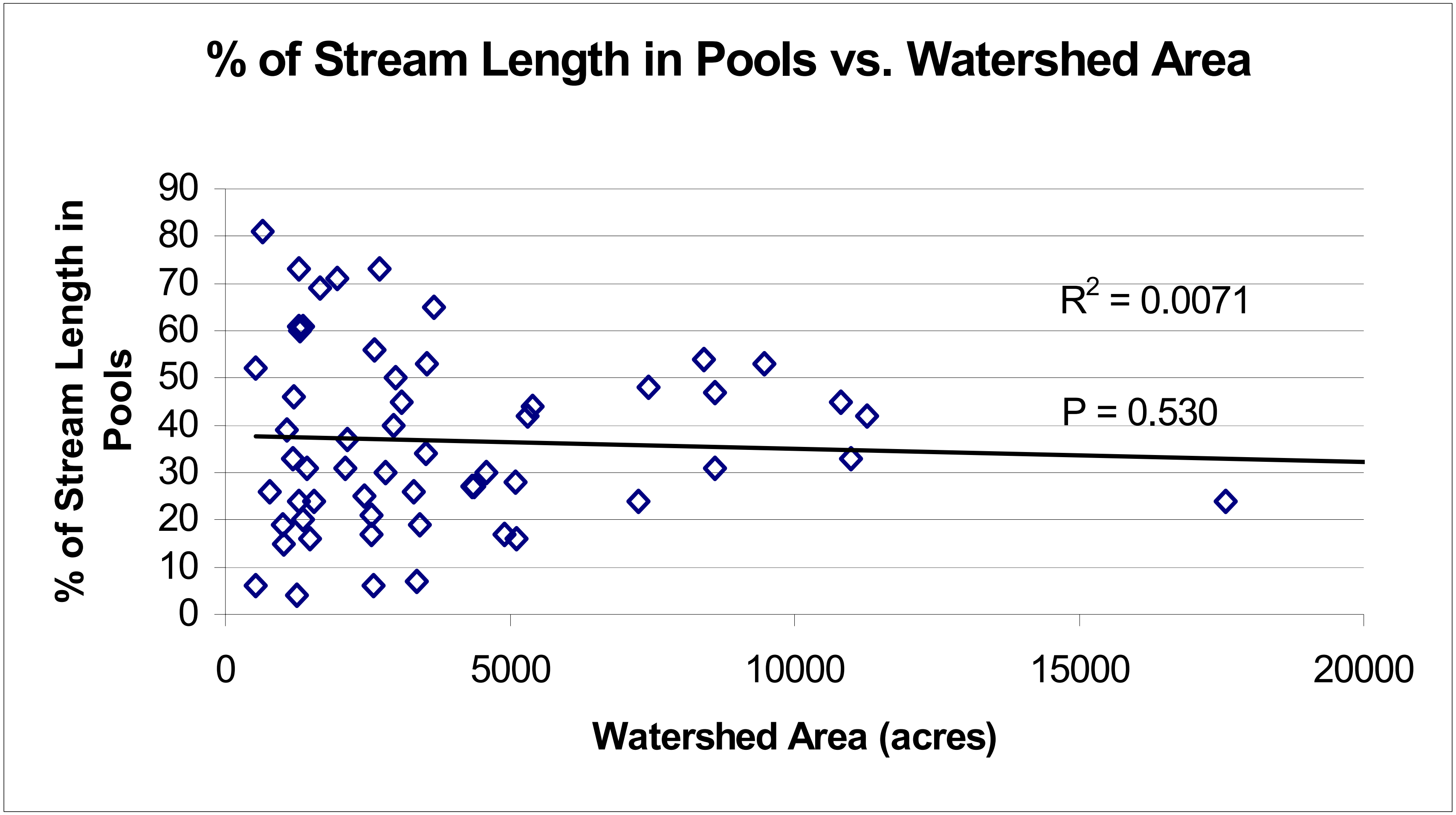

Figure C1-4. **Percent of stream length in pools plotted by watershed area for all streams assessed during the habitat assessments.**

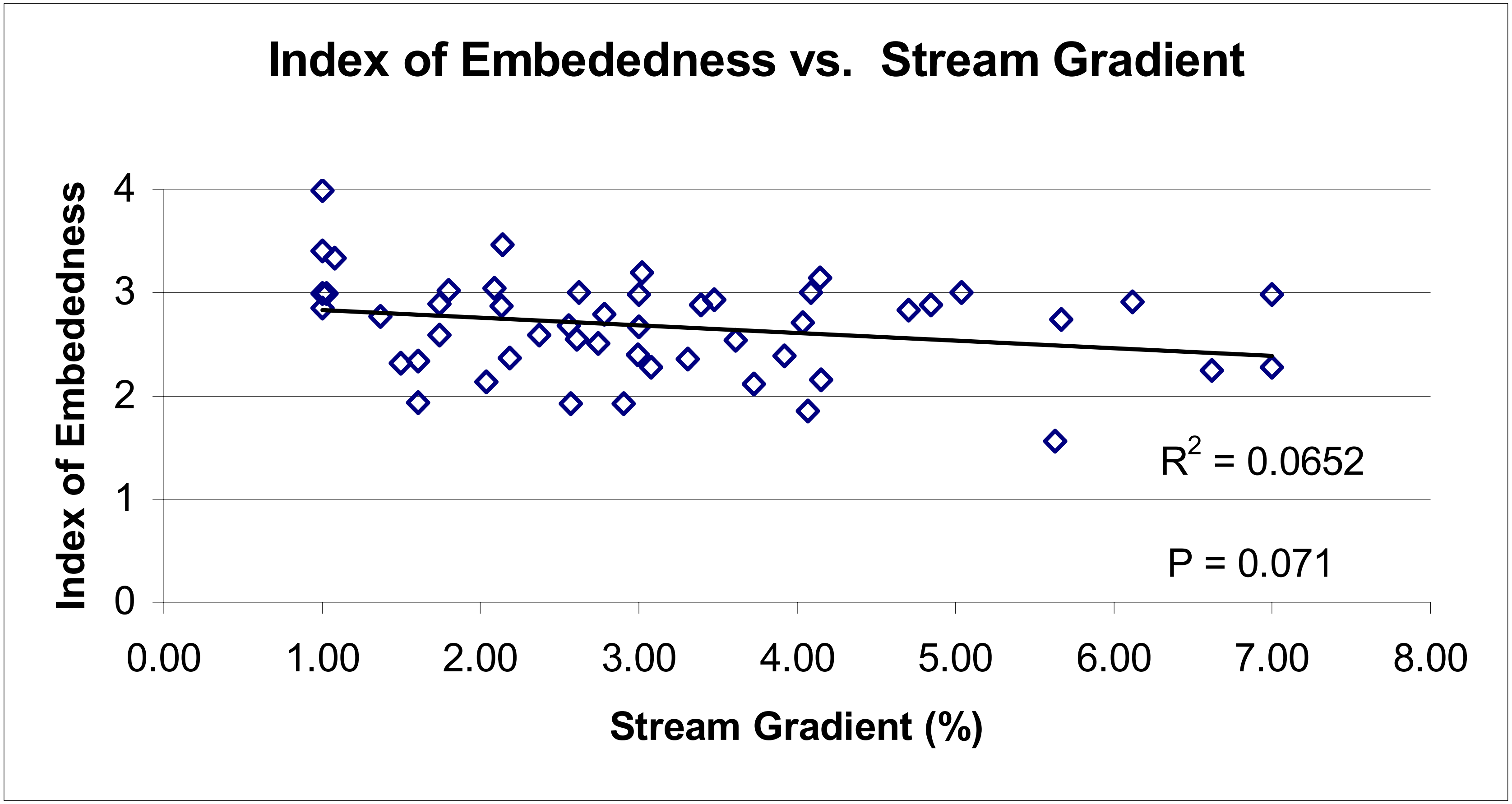

Figure C1-5. **Index of streambed embeddedness as a function of stream gradient for all assessed streams.**

C1.3.5 Maximum Residual Pool Depth as Percent Occurrence

Maximum pool depths are used by CDFG to calculate the percentage of primary pools, which are known to provide critical summer habitat for juvenile coho and steelhead under low flow conditions (Flosi et al., 1998). From CDFG's habitat typing assessments, there are indications that the better coastal coho streams may have as much as 40% of their total habitat length in primary pools (Flosi et al., 1998). A primary pool in a third order or larger stream would be expected to have a depth of three feet or greater. A primary pool in a first and second order stream is considered to be a depth of 2 feet or greater (Flosi and Reynolds 1994). Watershed area may be a confounding factor in comparing this variable, as smaller drainages with lower discharges tend to have shallower pools.

A summary of the residual pool depths for all assessed streams is shown in Tables C1-2 through C1-8. Of the 58 streams assessed, 14 (24%) had greater than 40% of their total pool habitat in primary pools (residual depths greater than 3') (Figure C1-6). These included three creeks that had in excess of 70% of their pools greater than 3' in depth: Rowdy Creek ([Smith River HPA] 70.4%), Terwer Creek ([Interior Klamath River HPA] 78.1%), and Blue Creek ([Blue Creek HPA] 87.8%) (Figure C1-6). On the average, the mean maximum residual pool depth was 2 feet for the assessed streams. In general, the streams with larger watershed areas contain deeper pools, on the average, than those with smaller watershed areas. Most of the assessed streams are in small drainages and are smaller than third order streams. Pools with residual depths greater than 2 feet or greater in many of these small streams may act as primary pools and provide temperature refugia. If these pools were considered as primary pools, functioning as summer habitat for juvenile salmonids during low flow conditions, then 71% of the assessed streams (41 out of 58) have greater than 40% of their pools classified as primary pools. Twenty-one percent of total streams assessed (12 out of 58 streams), have over 80% of their total pools greater than 2' in depth (Figure C1-6).

C1.4 CONCLUSIONS

The stream channel and habitat typing assessments indicated that habitat conditions for salmonids varied significantly among and within the 58 assessed streams. Taken together, the assessments suggested that there were:

1. A lack of complex pool habitat with low levels of LWD as shelter;

2. Dense, alder dominant riparian zones that provided excellent canopy closure, yet lacked the LWD recruitment potential of larger, more persistent, conifers;

3. Embedded gravels in many pool tails; and

4. Aggraded conditions in the lower reaches of some streams.

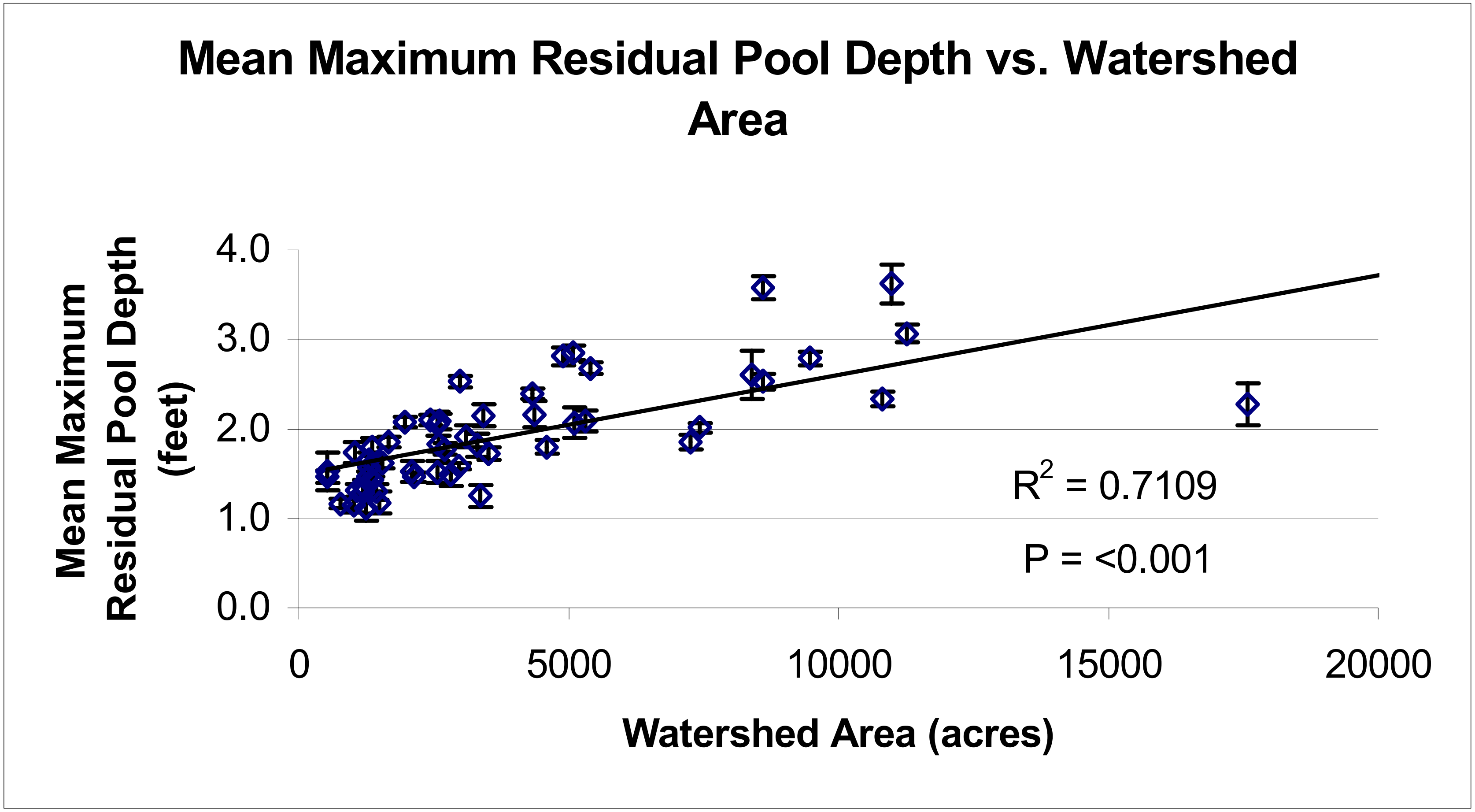

Figure C1-6. **Mean maximum pool depths plotted against watershed acres for the assessed streams. Error bars represent plus or minus one standard error.**

C1.5 REFERENCES

Flosi, G. and F.L. Reynolds. 1994. California salmonid stream habitat restoration manual. Second Edition. IFD, CDFG, Sacramento, CA.

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 1998. California salmonid stream habitat restoration manual. Third Edition. IFD, CDFG, Sacramento, CA.

Hicks, B.J., J.D. Hall, P.A. Bisson and J.R. Sedell. 1991. Responses of salmonids to habitat changes. American Fisheries Society Special Publication 19:483-518.

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Appendix C2. Large Woody Debris Surveys

CONTENTS

Figures

Tables

C2.1 OBJECTIVES AND METHODS

In the following description, there is a difference between an inventory and a sample. A sample is a type of survey where the crewmember only counts and measures LWD pieces within a certain percentage (i.e. 20% sample) of the stream length. An inventory is a survey in which all pieces are counted and measured for the entire anadromous stream length.

C2.1.1 Number of Streams Sampled and/or Inventoried

An in-channel and recruitment zone large woody debris (LWD) survey was conducted on 16 streams on Simpson's ownership in the HPAs: eight in in 1994 and eight additional streams in 1995. Information regarding the distribution of LWD was also obtained in the channel and habitat typing assessment process, but the importance of LWD to biological and physical processes in the stream channel justified the need for a more thorough assessment of this critical habitat component. The LWD surveys covered two distinct zones:

- LWD within the bankfull discharge area of the stream channel; and
- LWD and live trees within the "recruitment zone," defined as the area encompassing the floodplain and 50 feet of the hillslope beyond the bankfull channel margin.

The objectives of the LWD survey include:

- Accurately documenting the current abundance, distribution, and characteristics of instream LWD.
- Providing a repeatable methodology for monitoring long-term changes in the abundance, distribution, and characteristics of instream LWD.
- Accurately identifying the source of instream LWD (naturally recruited or restoration structure) and the species composition of instream LWD (hardwood or conifer).

The LWD survey was conducted using the CDFG methods (Flosi and Reynolds, 1994). This methodology is a 20% sample that was designed with the objective of quickly identifying stream reaches lacking in LWD for prioritizing restoration projects. Each stream reach is delineated by Rosgen Channel Type during the CDFG Habitat Typing process. During these LWD surveys 200' out of every 1000' of each channel type would be inventoried for both inchannel LWD and recruitment zone LWD.

Little River and three of its primary tributaries were inventoried for LWD in 1994 by Louisiana Pacific (LP) Fisheries Biologists. In 1998 Simpson Timber acquired the LP timberlands as well as their historical fisheries data for Little River. LP's LWD survey was a 100% inventory that tallied all inchannel pieces of LWD within the Bankfull margins. In LP's survey no riparian or recruitment zone inventory was conducted and the inchannel inventory grouped the 3' – 4' category with the >4' category. This lack of information is noted in the following tables that summarize the Little River LWD data.

C2.1.2 Index of LWD Volume

An index of volume was developed for the purposes of depicting and comparing the amount of LWD in each stream to the watershed area. At the time of the survey/inventory, LWD pieces were categorized as follows based on their length: 6-20 feet, and >20 feet. In addition the LWD pieces were categorized as follows based on their maximum diameter: 1-2 feet, 2-3 feet, 3-4 feet, and >4 feet. The volume index was calculated by multiplying the mean diameter class times the "mean" length class. The mean diameter classes used for calculating the volume index were: 1.5 feet for the 1-2' class, 2.5 feet for the 2-3', 3.5 feet for the 3-4' class, and 4 feet for the >4' class. The "mean" lengths used for calculating the volume index were: 13 feet for the 6- 20' class and 20 feet for the >20' class. The index of volume was based on the instream average pieces per 100 feet. Since the actual diameters and lengths were not measured for each piece, the calculated volume in not a "true" volume but rather an index of volume. The index allows comparison between streams on Simpson property within the different HPAs.

C2.1.3 100% In-Channel Inventory

During Simpson's 1994 surveys field crews noted that a 20% sample could significantly underestimate or overestimate the actual pieces per 100 feet of channel. For example within a short channel type, where only 400 or 600 feet of channel were sampled, it is possible that one large log jam could skew the survey results to indicate that there are more pieces per 100 feet than actually exist in the reach. Conversely, if in that same short reach of channel the survey locations randomly missed most of the LWD, the results would be artificially low. To test these possibilities, an additional 100% inventory was conducted on all of the streams surveyed in 1995. The 100% inventory and the CDFG 20% sample were conducted simultaneously. This data allows a direct comparison of the CDFG methodology to a known inventory and thus is an indicator of the accuracy of a 20% sample.

C2.1.4 1999 Prairie Creek Inventory by Redwood National Park

In-channel and recruitment zone LWD data from undisturbed watersheds in coastal California are needed to compare with data from managed forests in the same area. This need led to the cooperative effort with Redwood National Park (RNP) and National Marine Fisheries Service (NMFS) to inventory inchannel LWD in Prairie Creek. In 1999 RNP and NMFS conducted a 100% inventory of 4.3 miles of Prairie Creek in Prairie Creek National Park. Prairie Creek is considered to be the best remaining example of a watershed dominated by old growth redwood forest. While this survey focused on quantifying LWD volume rather than a piece count per unit length, the data has been summarized by size categories of inchannel pieces (Kramer, pers. Comm.). This data should be considered as a known or true piece count of a relatively undisturbed watershed that may be directly compared to both the CDFG 20% samples and the 100% inventories conducted in Plan Area streams. However, when comparing Prairie Creek and many of the assessed Plan Area streams, the differences in their channel morphology must be considered. Prairie Creek is a low–gradient alluvial channel in a relatively wide valley bottom, while many of the Plan Area streams are higher gradient in more incised channels.

C2.2 RESULTS

C2.2.1 LWD Sampling Survey Results

Results of Simpson's 1994 and 1995 LWD surveys and the 1994 Louisiana Pacific LWD inventories are summarized in Tables C2-1 through C2-14. Tables C2-1 through C2-7 contains the estimated overall LWD piece count, displayed as average pieces per 100 feet of channel, delineated by Rosgen Channel Type, condition (dead vs. live), and live species. Figure C2-1 depicts each stream's mean count of instream LWD per 100 feet of stream channel plotted against the stream's watershed area. Figure C2-2 graphically depicts, for each stream surveyed, the mean number of LWD pieces in the riparian recruitment zone per 100 feet of stream channel. Tables C2-8 through C2-14) provides summaries of the LWD data delineated by size categories both in the channel and in the riparian recruitment zone. In Figure C2-3, the index of LWD volume for each stream surveyed is plotted against that stream's watershed area.

In the 20 streams surveyed, the average amount of inchannel LWD ranged from zero pieces per 100 linear feet of an A2 channel type in North Fork Mad River (North Fork Mad River HPA) to 16.3 pieces per 100 linear feet of an F3 channel in Salmon Creek (Humboldt Bay HPA). The average amount of live conifers in the recruitment zone (50 feet beyond the bankfull channel) that could potentially become instream LWD ranged from 0 pieces per 100 linear feet in three sections of Long Prairie Creek (Mad River HPA) to 9.5 pieces per 100 linear feet of channel in the upper reaches of Salmon Creek (Humboldt Bay HPA). The survey also divided LWD pieces into eight size classes by length (greater or less than 20') and by diameter (1'-2', 2'-3', 3'-4', and over 4') to identify dominant size classes of LWD. Of the twenty streams surveyed in 1994 and 1995, the dominant, or co-dominant size class of inchannel LWD for all streams was 1'-2' diameter and less than 20' in length. The dominant size class in the riparian zone for all sixteen streams with Recruitment Zone surveys was consistently 1'-2' diameter and greater than 20' in length. The summarized results of the LWD surveys are presented in the tables below.

As shown in Figure C2-1, the mean number of instream LWD pieces per 100 feet of stream channel decreased significantly with increased watershed area. While there is some variability the trend for streams with less than approximately 4,000 acres in the watershed, the number of instream pieces of LWD is generally greater than 3 per 100 feet of channel (Figure C2-1). For streams with watershed areas greater than approximately 4,000 acres, the mean number of instream pieces of LWD is generally less than 3 pieces per 100 feet of stream channel (Figure C2-1).

The number of pieces of LWD within the stream recruitment zone for each of the Streams surveyed is shown in Figure C2-2. As shown in Figure C2-2, the mean number of pieces of LWD per 100 feet of channel in the riparian recruitment zone ranged from approximately 3.5 in Wilson Creek (Smith River HPA) to 12.5 for the South Fork Ah Pah Creek (Coastal Klamath River HPA). Streams within in the Coastal Klamath and Blue Creek HPAs had 5 of the 7 greatest mean number of LWD pieces (7.7 to 12.6 pieces) in the recruitment zone per 100 feet of stream channel of all streams surveyed.

Table C2-1. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Smith River HPA.

South Fork Winchuck River		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	C4	0.5	0.1	0.1	0.2	7.1	1.2	16
2	F4	0.2	0.3	0.0	0.5	7.8	0.3	3
3	C4	1.3	0.1	0.0	0.9	5.9	2.4	7
4	D3	0.5	0.0	0.0	0.5	3.5	0.0	1
5	A2	1.5	1.0	0.5	0.5	6.4	3.0	4
Rowdy Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	D4	0.3	0.0	0.0	0.3	1.1	0.7	12
2	B3	0.4	0.4	0.2	0.5	3.6	1.4	16
3	B2	0.2	0.5	0.3	0.7	5.5	0.5	6
4	F3	0.8	0.3	0.0	1.5	8.5	0.2	3
Dominie Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	F3	1.0	0.4	0.1	0.6	3.2	1.8	8
2	A3	0.8	0.2	0.0	1.7	6.2	3.3	3
3	F3	3.0	1.0	0.0	3.5	2.0	1.0	1
4	A2	0.9	0.5	1.0	2.1	2.9	6.9	4
Wilson Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	F4	1.7	0.2	0.1	1.2	4.1	2.0	35
2	B3	2.5	2.0	0.2	1.8	2.2	2.7	3

Table C2-2. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Coastal Klamath HPA.

Hunter Creek		Recruitment Zone					In-Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F4	0.5	0.0	0.0	0.3	8.2	0.4	8
2	D4	0.6	0.1	0.1	0.5	2.9	1.8	25
3	B4	1.2	0.2	0.0	1.5	4.7	3.4	11
4	F3	2.2	0.5	0.0	1.2	4.7	3.7	3
5	F4	3.8	0.7	0.4	1.4	2.9	5.2	9
Terwer Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F4	1.6	0.2	0.1	1.5	2.0	3.6	18
2	F3	2.1	1.5	0.2	2.7	5.3	3.5	13
3	F2	4.1	1.9	0.1	3.8	6.4	1.5	15
4	F4	3.3	3.9	0.2	2.6	0.8	3.3	16
North Fork Ah Pah Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F4	0.2	0.3	0.0	3.2	2.1	1.7	5
2	A2	5.0	1.5	0.0	2.0	7.5	6.5	1
3	B3	3.6	1.1	0.0	3.4	7.1	5.8	4
4	B2	4.8	1.8	0.0	5.8	8.5	4.5	2
5	A2	5.2	0.8	0.2	4.7	7.0	4.7	3
6	F4	2.4	1.8	0.2	4.8	6.4	5.8	13
South Fork Ah Pah Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	B4	4.8	0.1	0.1	1.1	2.6	2.1	5
2	A3	5.8	0.2	0.4	3.0	2.8	7.9	5
Ah Pah Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	C4	0.8	0.2	0.7	2.7	2.5	2.1	6
2	D4	3.5	1.2	0.0	2.3	2.7	3.3	3
3	F3	3.5	1.3	0.0	5.3	1.3	2.3	2
4	A2	8.0	0.0	0.0	1.5	0.5	6.0	1
5	F4	6.6	0.3	0.0	3.3	1.4	7.0	4
6	A2	7.0	0.5	0.0	2.5	5.5	7.0	1
7	F3	4.4	1.0	0.4	2.6	4.6	5.8	4

Table C2-3. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Blue Creek HPA.

West Fork Blue Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	B2	0.8	0.2	0.0	1.5	3.5	1.8	5
2	A2	3.7	0.7	0.1	2.6	2.8	3.2	18

Table C2-4. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Little River HPA.

Little River		Recruitment Zone (N/A)					In Channel	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	B3	N/A	N/A	N/A	N/A	N/A	2.2	1614
2	B2	N/A	N/A	N/A	N/A	N/A	1.5	5506
3	B3	N/A	N/A	N/A	N/A	N/A	2.8	3526
4	F2	N/A	N/A	N/A	N/A	N/A	3.2	3214
5	F3	N/A	N/A	N/A	N/A	N/A	1.4	1366
6	B2	N/A	N/A	N/A	N/A	N/A	2.0	10902
7	B4	N/A	N/A	N/A	N/A	N/A	2.5	9876
8	B2	N/A	N/A	N/A	N/A	N/A	2.4	6347
9	A2	N/A	N/A	N/A	N/A	N/A	3.2	1062
10	B2	N/A	N/A	N/A	N/A	N/A	4.2	9415
11	B3	N/A	N/A	N/A	N/A	N/A	5.1	2412
12	B2	N/A	N/A	N/A	N/A	N/A	8.8	2644
13	B4	N/A	N/A	N/A	N/A	N/A	10.2	3339
14	A2	N/A	N/A	N/A	N/A	N/A	9.8	1546
Railroad Cr.		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	F4	N/A	N/A	N/A	N/A	N/A	4.1	748
2	B2	N/A	N/A	N/A	N/A	N/A	6.7	3901
3	B3	N/A	N/A	N/A	N/A	N/A	7.8	1998
4	B4	N/A	N/A	N/A	N/A	N/A	13.1	1244
Lower South Fork Little River		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	F4	N/A	N/A	N/A	N/A	N/A	5.9	7594
2	F3	N/A	N/A	N/A	N/A	N/A	8.4	2042
3	B2	N/A	N/A	N/A	N/A	N/A	9.3	961
4	C4	N/A	N/A	N/A	N/A	N/A	9.4	1679
5	F3	N/A	N/A	N/A	N/A	N/A	10.9	1628
Upper South Fork Little River		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	B3	N/A	N/A	N/A	N/A	N/A	4.4	2437
2	B2	N/A	N/A	N/A	N/A	N/A	3.4	1250
3	A2	N/A	N/A	N/A	N/A	N/A	6.3	2190
4	F3	N/A	N/A	N/A	N/A	N/A	6.0	3942
5	B4	N/A	N/A	N/A	N/A	N/A	14.8	583

Table C2-5. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Mad River HPA.

Lindsay Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F5	0.9	0.5	0.1	4.9	2.9	3.6	28
Cañon Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B4	0.5	1.0	0.5	5.8	2.3	1.3	2
2	D4	0.5	0.3	0.8	4.1	2.6	4.9	4
3	B3	2.6	0.5	0.4	5.0	3.5	1.5	4
4	F3	1.1	0.3	0.0	6.4	2.1	0.3	8
5	A2	1.3	0.1	0.4	6.6	3.4	1.8	6
Dry Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B4	0.9	1.1	0.3	2.8	1.8	1.8	4
2	A3	2.0	0.5	0.0	1.5	3.5	0.5	1
3	B3	0.0	1.0	1.0	2.5	10.0	6.5	1

Table C2-6. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), North Fork Mad River HPA.

North Fork Mad River		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F4	0.2	0.0	0.1	1.1	0.8	0.6	12
2	B3	1.3	0.1	0.1	4.0	1.1	0.4	4
3	F2	0.3	0.1	0.3	3.2	0.8	0.2	6
4	A2	1.8	0.0	0.1	1.0	2.5	0.0	4
5	F2	1.4	0.4	0.3	6.2	4.7	1.1	36
6	F4	1.7	1.2	0.1	7.7	3.1	1.7	6
7	F3	1.4	1.0	0.1	6.6	2.6	1.4	7
8	F4	1.3	0.4	0.2	5.7	2.9	2.2	9
Long Prairie Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B3	1.9	2.5	0.4	2.6	9.7	2.4	7
2	B2	3.0	0.0	1.5	0.5	5.5	1.5	1
3	B3	2.0	1.2	0.3	5.8	6.3	5.3	3
4	F3	1.3	0.0	0.0	0.0	11.8	0.0	2
5	B2	3.5	0.0	1.5	0.0	6.0	3.5	1
6	F3	2.0	0.0	1.0	4.3	3.5	0.5	2
7	B2	6.5	0.5	0.0	0.0	4.0	0.0	1

Table C2-7. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Humboldt Bay HPA.

Salmon Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F3	1.3	0.3	0.4	1.9	1.8	1.8	19
2	F1	0.8	0.5	0.5	3.8	1.8	3.0	2
3	F3	4.5	0.3	0.3	5.5	0.8	16.3	2
4	F1	0.0	0.0	0.0	1.0	0.0	5.0	1
5	F3	1.9	0.3	0.3	5.7	2.3	4.5	8
6	B2	3.3	0.7	1.2	9.5	6.4	6.1	7

Table C2-8. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Smith River HPA.

Stream	Size Classes of In-channel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
SF WINCHUCK									
Instream LWD	0.8	0.4	0.2	0.1	0.1	0.0	0.1	0.0	1.7
Riparian	0.2	4.2	0.1	0.7	0.1	0.2	0.0	0.1	5.6
Total	1.0	4.6	0.3	0.8	0.2	0.2	0.1	0.1	7.3
ROWDY CREEK									
Instream LWD	0.2	0.2	0.1	0.0	0.2	0.1	0.1	0.0	0.9
Riparian	0.3	2.1	0.0	0.8	0.0	0.2	0.0	0.1	3.5
Total	0.5	2.3	0.1	0.8	0.2	0.3	0.1	0.1	4.4
DOMINIE CREEK									
Instream LWD	1.7	0.3	0.5	0.4	0.2	0.2	0.1	0.0	3.4
Riparian	0.5	3.8	0.2	1.3	0.1	0.4	0.0	0.1	6.4
Total	2.2	4.1	0.7	1.7	0.3	0.6	0.1	0.1	9.8
WILSON CREEK									
Instream LWD	0.4	0.4	0.4	0.3	0.1	0.1	0.2	0.2	2.1
Riparian	0.4	2.8	0.4	0.9	0.1	0.2	0.3	0.2	5.3
Total	0.8	3.2	0.8	1.2	0.2	0.3	0.5	0.4	7.4

[a] = maximum diameter of LWD piece

Table C2-9. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Coastal Klamath HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
HUNTER CREEK									
Instream LWD	0.8	0.4	0.3	0.3	0.2	0.2	0.3	0.2	2.7
Riparian	0.3	3.1	0.3	0.6	0.1	0.3	0.2	0.2	5.1
Total	1.1	3.5	0.6	0.9	0.3	0.5	0.5	0.4	7.8
TERWER									
Instream LWD	0.7	0.6	0.3	0.4	0.2	0.3	0.2	0.4	3.1
Riparian	0.6	4.5	0.3	1.1	0.1	0.5	0.2	0.4	7.7
Total	1.3	5.1	0.6	1.5	0.3	0.8	0.4	0.8	10.8
AH PAH									
Instream LWD	2.0	0.7	0.8	0.3	0.3	0.1	0.2	0.2	4.6
Riparian	1.3	4.1	0.5	1.2	0.5	0.4	0.6	0.4	9.0
Total	3.3	4.8	1.3	1.5	0.8	0.5	0.8	0.6	13.6
NORTH FORK AH PAH									
Instream LWD	2.1	0.7	1.0	0.2	0.2	0.1	0.5	0.2	5.0
Riparian	0.7	6.9	0.6	1.0	0.3	0.4	0.8	0.6	11.3
Total	2.8	7.6	1.6	1.2	0.5	0.5	1.3	0.8	16.3
SOUTH FORK AH PAH									
Instream LWD	2.6	0.3	1.0	0.5	0.3	0.3	0.3	0.3	5.6
Riparian	1.2	6.1	1.1	1.6	0.6	0.6	0.7	0.8	12.7
Total	3.8	6.4	2.1	2.1	0.9	0.9	1.0	1.1	18.3

Table C2-10. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Blue Creek HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
WEST FORK BLUE CREEK									
Instream LWD	1.4	0.9	0.4	0.1	0.2	0.1	0.1	0.0	3.2
Riparian	1.7	4.6	0.5	0.8	0.1	0.1	0.0	0.0	7.8
Total	3.1	5.5	0.9	0.9	0.3	0.2	0.1	0.0	11.0

Table C2-11. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Little River HPA.

Stream	Size Classes of In-channel LWD and Wood within Riparian Recruitment Zone						
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	>3' max dia.[a]; <20'	>3' max dia.[a]; >20'	All Size Classes
LITTLE RIVER							
Instream LWD	1.2	0.9	0.5	0.4	0.3	0.2	3.5
RAILROAD							
Instream LWD	3.0	1.4	1.9	1.0	0.4	0.3	8.0
LOWER SOUTH FORK LITTLE RIVER							
Instream LWD	3.6	1.2	1.6	0.7	0.5	0.4	8.0
UPPER SOUTH FORK LITTLE RIVER							
Instream LWD	2.8	0.8	1.2	0.4	0.5	0.2	5.9
[a] = maximum diameter of LWD piece							

Table C2-12. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), North Fork Mad River HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
NF MAD RIVER									
Instream LWD	0.2	0.3	0.1	0.1	0.1	0.0	0.2	0.0	1.0
Riparian	0.2	4.1	0.1	1.2	0.1	0.3	0.2	0.1	6.3
Total	0.4	4.4	0.2	1.3	0.2	0.3	0.4	0.1	7.3
LONG PRAIRIE CREEK									
Instream LWD	1.0	0.5	0.1	0.4	0.0	0.2	0.0	0.0	2.2
Riparian	1.5	6.2	0.1	1.5	0.0	0.5	0.0	0.1	9.9
Total	2.5	6.7	0.2	1.9	0.0	0.7	0.0	0.1	12.1
[a] = maximum diameter of LWD piece									

Table C2-13. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Mad River HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
LINDSAY									
Instream LWD	1.9	0.3	0.6	0.2	0.3	0.1	0.2	0.1	3.7
Riparian	0.4	4.1	0.1	1.6	0.1	0.6	0.2	0.6	7.7
Total	2.3	4.4	0.7	1.8	0.4	0.7	0.4	0.7	11.4
DRY CREEK									
Instream LWD	0.9	0.1	0.3	0.1	0.0	0.0	0.0	0.0	1.4
Riparian	0.6	3.2	0.1	1.1	0.2	0.2	0.7	0.1	6.2
Total	1.5	3.3	0.4	1.2	0.2	0.2	0.7	0.1	7.6
CAÑON CR.									
Instream LWD	0.6	0.6	0.2	0.1	0.1	0.0	0.2	0.0	1.8
Riparian	0.9	3.8	0.1	1.7	0.2	0.3	0.1	0.1	7.2
Total	1.5	4.4	0.3	1.8	0.3	0.3	0.3	0.1	9.0
[a] = maximum diameter of LWD piece									

Table C2-14. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Humboldt Bay HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
SALMON CREEK									
Instream LWD	0.8	0.8	0.5	0.3	0.4	0.4	0.4	0.4	4.0
Riparian	0.5	4.1	0.3	1.0	0.2	0.4	0.4	0.2	7.1
Total	1.3	4.9	0.8	1.3	0.6	0.8	0.8	0.6	11.1
[a] = maximum diameter of LWD piece									

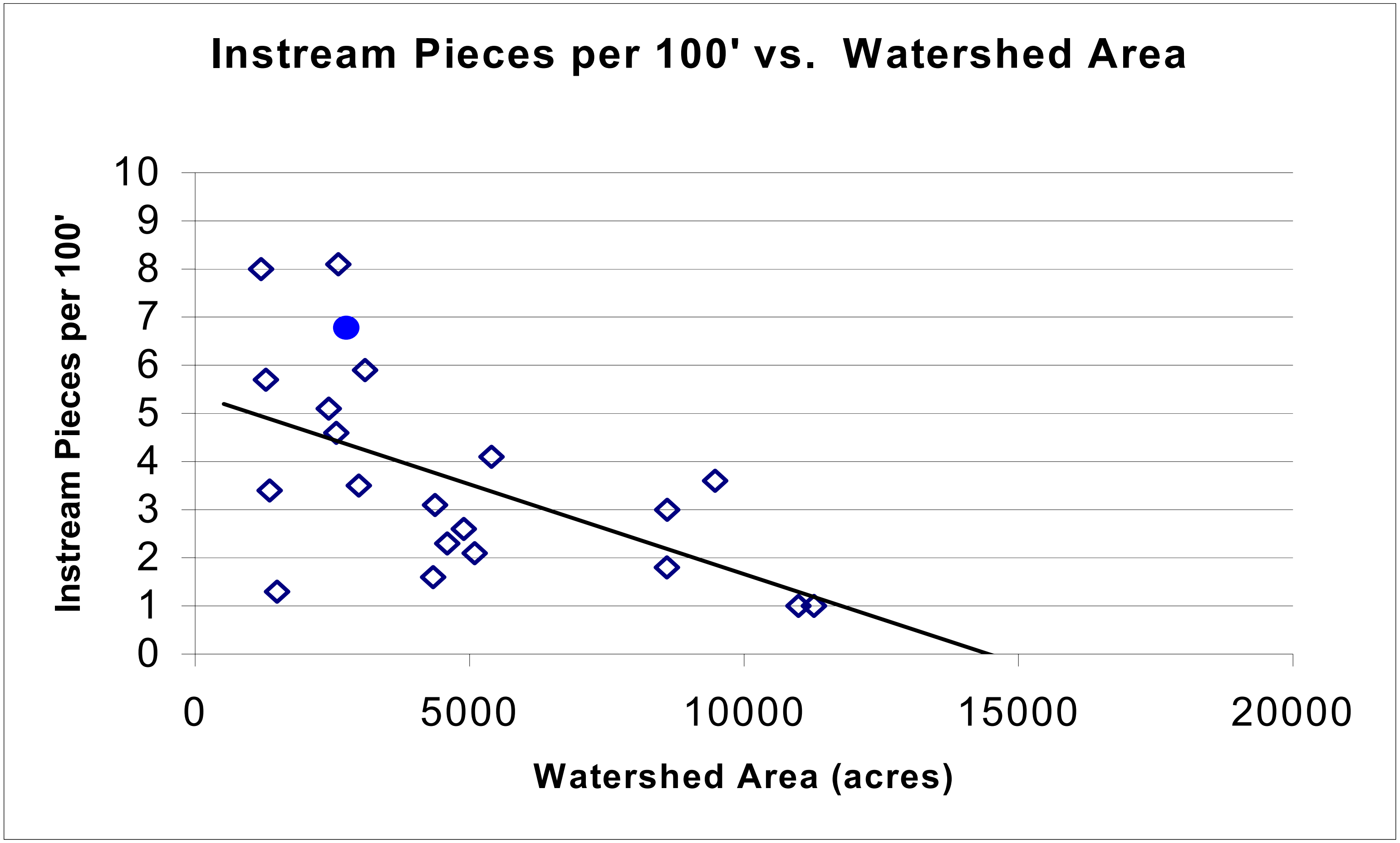

Figure C2-1. **Summary of mean number of instream LWD pieces per 100 feet of stream channel versus stream watershed area for 20 Plan Area streams. (Note: solid circle depicts Prairie Creek for reference.)**

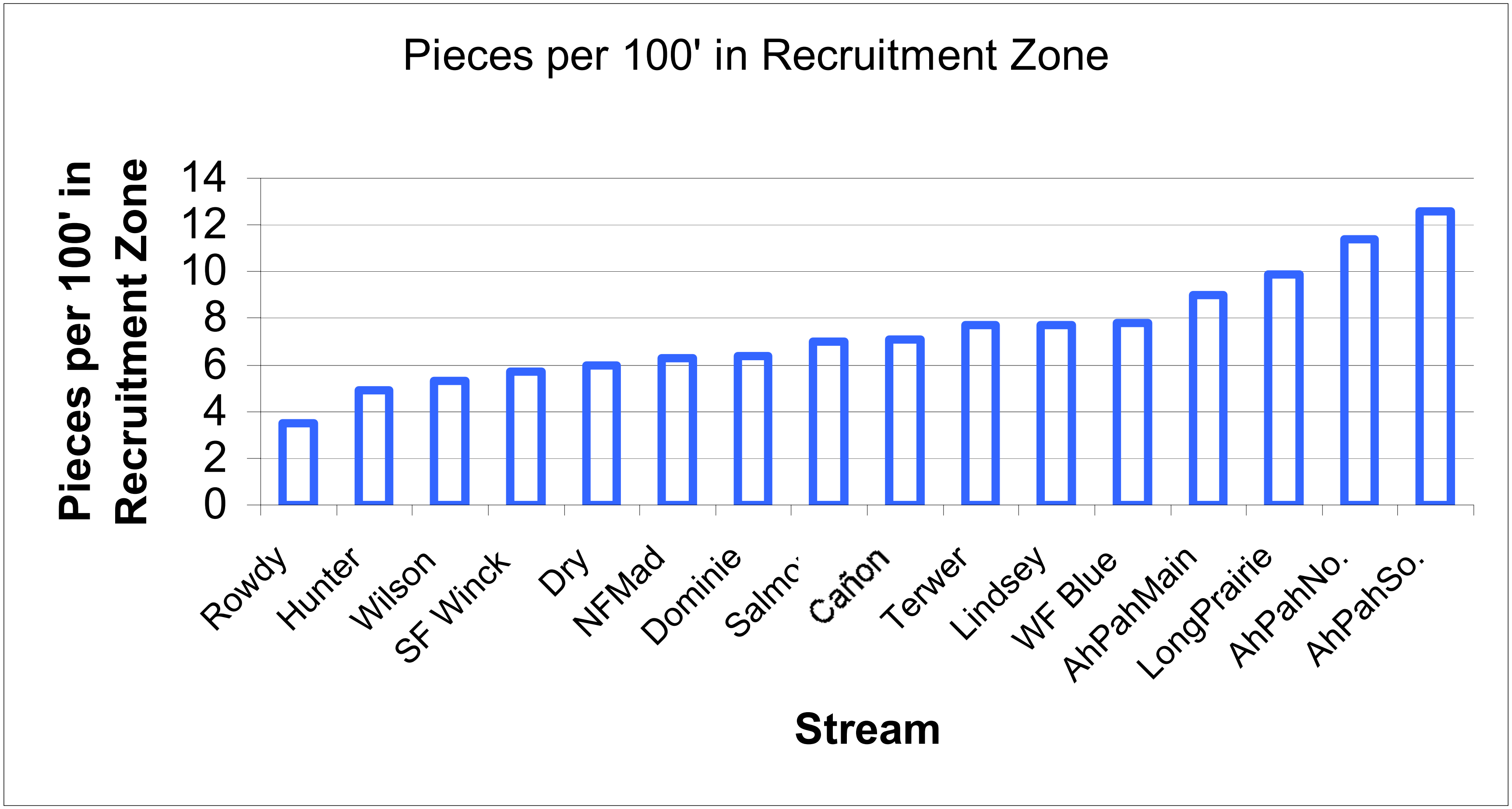

Figure C2-2. **Summary of the mean number of LWD pieces in the recruitment zone per 100 feet of stream channel for 16 Plan Area streams.**

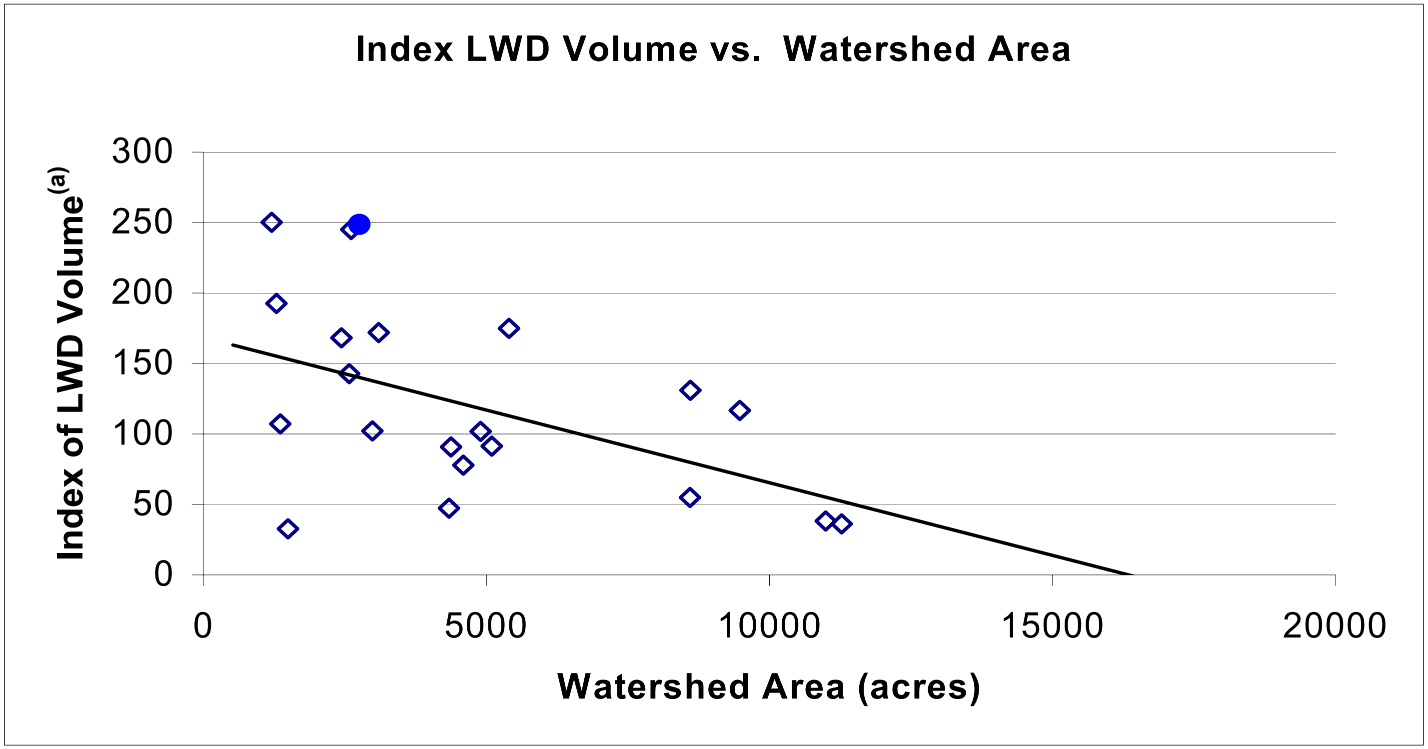

Figure C2-3. LWD volume index versus watershed area for 20 Plan Area streams (Note: solid circle represents Prairie Creek for reference). (Index equals the maximum diameter times the mid-point of the LWD length class.)

The results of the LWD surveys indicate that most streams surveyed had low amounts of inchannel LWD that consisted of the smallest size categories. Eleven of the sixteen streams with riparian surveys had low amounts of conifer abundance (relative to hardwoods) within the recruitment zone. These results support the conclusions drawn from the channel and habitat typing assessment: there are generally low levels of inchannel LWD available to function as shelter or to promote formation of pools in the surveyed streams. The dominant size class of inchannel LWD also parallels channel assessment descriptions of smaller diameter, alder dominated riparian zones with low numbers of large conifer (greater than 3' in diameter) as potential LWD.

As shown in Figure C2-3, an index of LWD volume for each stream surveyed was calculated and plotted against each stream's watershed area. Similar to the trend shown in Figure C2-1, (fewer pieces per 100 feet of channel with larger watershed areas) volume of LWD generally decreased with increases in watershed area (Figure C2-3).

C2.2.2 LWD Inventory Results

Results of Simpson's 1995 Inchannel LWD inventory are summarized in Tables C2-15 through C2-17. These tables summarize the 100% inchannel inventory displaying average pieces per 100 feet by Rosgen Channel Type and piece size category. The last two lines for each stream are the weighted average pieces per 100 feet of channel as determined by both the inventory and the 20% sample.

The results of the 1995 100% Inchannel LWD Inventory suggest that the 20% sample is comparable. CDFG's 20% sample is adequate for an estimate of average pieces per linear distance but does not address any volume or function related issues. The overall goal of the survey as designed by CDFG was to identify specific stream reaches that are in need of restoration in the form of additional LWD. To address the issues of total volume or inchannel function more detailed surveys will be needed.

C2.2.3 Prairie Creek LWD Inventory Results

The Prairie Creek inventory data is displayed in Table C2-18 as average pieces per 100 feet of channel in the various size categories. For a graphic comparison of the LWD data for Prairie Creek and the surveyed Plan Area streams, see Figures C2-1 and C2-3 above.

The section of Prairie Creek that was inventoried is a low gradient, small cobble dominated channel (Rosgen Channel Type of C4) that is considered to be a relatively undisturbed reach. Results of the Prairie Creek LWD data revealed that inchannel LWD occurred at an average of 6.8 pieces per 100 linear feet of channel for the 4.3 miles of channel inventoried (Kramer, pers. comm.)(Figure C2-1). This value exceeds all but two of the ranges calculated for any single average for the surveyed Plan Area streams (1.0 - 8.1 pieces/100'). Two tributaries in the Little River HPA, Lower South Fork and Railroad, had average piece counts at 8.1 and 8.0 pieces/100' respectively.

Table C2-15. Summary of 1995 100% in-channel LWD inventory (average pieces per 100 feet by channel type and size category), Smith River HPA.

Stream	Size Classes of In-channel LWD								
	1'-2' max dia.[a]; <20'	**1'-2' max dia.[a]; >20'**	**2'-3' max dia.[a]; <20'**	**2'-3' max dia.[a]; >20'**	**3'-4' max dia.[a]; <20'**	**3'-4' max dia.[a]; >20'**	**>4' max dia.[a]; <20'**	**>4' max dia.[a]; >20'**	**All Size Classes**
SOUTH FORK WINCHUCK RIVER									
C4	0.4	0.3	0.2	0.1	0.1	0.0	0.1	0.0	1.2
F4	0.7	0.8	0.2	0.1	0.0	0.0	0.1	0.0	1.9
C4	0.6	0.4	0.1	0.1	0.1	0.1	0.0	0.0	1.4
D3	2.3	0.4	0.0	0.0	0.2	0.2	0.0	0.0	3.2
A2	2.7	0.7	0.7	0.2	0.2	0.1	0.3	0.1	4.9
Weighted Average	0.8	0.4	0.2	0.1	0.1	0.0	0.1	0.0	1.7
20% sample	0.8	0.4	0.2	0.1	0.1	0.0	0.1	0.0	1.6
ROWDY CREEK									
D4	0.3	0.2	0.1	0.1	0.0	0.1	0.0	0.0	0.8
B3	0.1	0.1	0.1	0.0	0.0	0.0	0.1	0.0	0.4
B2	0.1	0.2	0.1	0.1	0.1	0.0	0.1	0.0	0.6
F3	0.9	0.4	0.2	0.2	0.0	0.1	0.1	0.0	2.0
Weighted Average	0.2	0.2	0.1	0.1	0.0	0.0	0.0	0.0	0.7
20% sample	0.2	0.2	0.1	0.0	0.2	0.1	0.1	0.0	0.9
DOMINIE CREEK									
F3	0.6	0.4	0.2	0.2	0.1	0.2	0.1	0.0	1.7
A3	2.6	1.0	1.3	0.7	0.4	0.5	0.1	0.1	6.6
F3	0.6	0.2	0.5	0.2	0.2	0.2	0.0	0.0	1.8
A2	2.6	0.4	0.9	0.3	0.4	0.2	0.2	0.1	5.0
Weighted Average	1.7	0.5	0.6	0.3	0.3	0.2	0.1	0.0	3.8
20% sample	1.7	0.3	0.5	0.4	0.2	0.2	0.1	0.0	3.4

[a] = maximum diameter of LWD piece

Table C2-16. Summary of 1995 100% in-channel LWD inventory (average pieces per 100 feet by channel type and size category), Coastal Klamath HPA.

Stream	Size Classes of Inchannel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
AH PAH CREEK									
C4	1.7	0.7	0.4	0.2	0.1	0.1	0.2	0.1	3.4
D4	1.9	1.7	0.3	0.3	0.3	0.1	0.4	0.1	5.2
F3	2.4	0.3	0.5	0.3	0.1	0.1	0.1	0.1	4.0
A2	1.4	0.5	0.9	0.0	0.8	0.0	0.8	0.0	4.3
F4	2.5	0.3	1.3	0.0	0.6	0.1	0.3	0.1	5.2
A2	5.6	1.0	1.2	0.6	0.7	0.4	0.7	0.3	10.5
F3	3.1	0.3	1.4	0.3	0.9	0.1	0.7	0.1	6.9
Weighted Average	2.4	0.7	0.8	0.2	0.5	0.1	0.4	0.1	5.1
20% sample	2.0	0.7	0.8	0.3	0.3	0.1	0.2	0.2	4.6
NORTH FORK AH PAH CREEK									
F4	0.8	0.4	0.1	0.0	0.1	0.0	0.2	0.0	1.7
A2	3.7	0.6	0.7	0.2	0.2	0.1	0.3	0.1	5.9
B3	1.7	0.4	0.8	0.2	0.3	0.1	0.6	0.3	4.4
B2	1.5	0.9	1.3	0.1	0.2	0.1	0.4	0.3	4.9
A2	2.5	1.1	1.2	0.1	0.5	0.0	0.7	0.2	6.4
F4	2.0	0.5	0.8	0.1	0.3	0.1	0.5	0.1	4.4
Weighted Average	1.8	0.5	0.7	0.1	0.3	0.1	0.5	0.1	4.2
20% sample	2.1	0.7	1.0	0.2	0.2	0.1	0.5	0.2	5.1
SOUTH FORK AH PAH CREEK									
B4	1.2	0.6	0.5	0.3	0.1	0.2	0.2	0.1	3.1
A3	3.8	0.8	1.7	0.7	0.7	0.5	0.9	0.6	9.6
Weighted Average	2.4	0.7	1.0	0.5	0.3	0.3	0.5	0.3	6.1
20% sample	2.6	0.3	1.0	0.5	0.3	0.3	0.3	0.3	5.6

[a] = maximum diameter of LWD piece

Table C2-17. Summary of 1995 100% in-channel LWD inventory (average pieces per 100 feet by channel type and size category), Mad River HPA.

Stream	Size Classes of Inchannel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
LINDSAY CREEK									
F5	1.8	0.4	0.5	0.2	0.2	0.1	0.1	0.1	3.4
20% sample	1.9	0.3	0.6	0.2	0.3	0.1	0.2	0.1	3.5
[a] = maximum diameter of LWD piece									

Table C2-18. Summary of 1999 100% in-channel LWD inventory (average pieces per 100 feet by size category), Prairie Creek.

Stream	Size Classes of Inchannel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
PRAIRIE CREEK									
	2.8	1.1	0.8	0.7	0.3	0.4	0.2	0.6	6.8
[a] = maximum diameter of LWD piece									

Additionally, in five separate reaches within the Little River HPA and Salmon Creek, LWD tallies exceeded 6.8 pieces per 100 feet. When comparing the Prairie Creek results only to low gradient (<2%) stream reaches (Rosgen Channel Types C, D and F), five reaches in the surveyed Plan Area streams (three F3, one F4 and one C4 channel types) exceed the Prairie Creek values. These are Salmon Creek (16.3 pieces per 100') and Lower South Fork Little River (8.4, 9.4 and 10.9 pieces per 100') and Ah Pah Creek (7.0 pieces per 100'). In general, the surveyed Plan Area streams had, on average, more pieces per 100' in the higher gradient and more confined channel types. This intuitively makes sense; the smaller and steeper the stream the more likely it is for an individual LWD piece to be retained in the system.

In Prairie Creek the dominant category of inchannel LWD was in the 1' - 2' and less than 20' long" category (Table C2-18). This compares to the dominant, or co-dominant category of inchannel LWD for all but one of the surveyed Plan Area streams. The dominant inchannel category for the North Fork of the Mad River was the "1' to 2' and greater than 20' long". This difference can probably be attributed to the relatively larger size of the North Fork Mad River. In this stream an individual LWD piece less than 20 feet long would tend to be delivered through the system rather than be retained. The Prairie Creek results accurately reflect the LWD piece size for a relatively undisturbed coastal drainage. However, comparisons between Prairie Creek and many Plan Area streams may not be valid, because of differences in their morphology. Prairie Creek is a low–gradient alluvial channel in a relatively wide valley bottom, while many Plan Area streams are higher gradient in more incised channels.

Numerous factors influence the frequency, size, distribution and function of LWD including: geographic location, dominant tree species, channel width, channel gradient and drainage area. As a result, comparing LWD inventories from Simpson's California timberlands with data from undisturbed watersheds in other states could be inappropriate or misleading. LWD inventories from additional undisturbed watersheds including an inland, Douglas fir dominated forest, and a coastal redwood forest with steeper channel gradients than those found in Lower Prairie Creek would aid in the analysis of the existing LWD results, as these conditions are common on Simpson timberlands. Inventories on undisturbed watersheds of varying drainage area and channel gradient would also aid in differentiating between the many factors that influence LWD distributions

C2.3 DISCUSSION

The LWD survey results reflect the effects of past timber management practices and early habitat improvement efforts. Throughout the surveyed Plan Area streams, there were generally low amounts of LWD; and the predominate size of the existing LWD was small (primarily 1'-2' diameter pieces). The lack of large pieces of LWD (> 4' diameter and > 20' long) suggests that surveyed stream channels have been subjected to extensive channel clearing as part of past timber harvesting practices and/or early habitat improvement efforts. The relative lack of large live trees (conifers with > 4' diameters) within the recruitment zone reflects the effects of pre-FPRs management practices that removed most merchantable conifers from riparian zones adjacent to stream channels and failed to re-establish conifers in these areas. As a result, most riparian zones in sampled watersheds tend to be dominated by alder, willow, and younger conifers.

Comparisons of logged and unlogged streams or reaches provide insights into management impacts on LWD loading, recruitment rate and downstream transport. Numerous studies have compared LWD in old growth, mature second growth and recently clear-cut watersheds in Alaska, British Columbia, Washington and Oregon (Sullivan et al. 1987; Bibly and Ward 1989, 1991; Murphy and Koski 1989; Ralph et al. 1994; McHenry et al. 1998). Some studies indicated that LWD frequency was reduced in managed watersheds (Bilby and Ward 1991, McHenry et. al. 1998) and others failed to prove or detect a difference in piece counts (Ralph et al. 1994). However, every study confirmed a statistically significant reduction in sizes of LWD pieces in managed watersheds, suggesting that size and volume of LWD pieces are more important than frequency of pieces in forming and maintaining complex habitat features.

The LWD structures placed by restoration groups are often undersized (mainly in length as opposed to maximum width) for several reasons, including: 1) monetary limits per structure as required by CDFG-administered restoration funds, 2) size constraints by the cull logs available at or near a work site or donated by timber companies, and/or 3) size constraints of cull logs that restoration groups can maneuver with their equipment. Most restoration projects have also failed to mimic natural conditions, tending to locate LWD structures along channel margins with minimal amounts of wood lying within the main channel, and rarely, if ever, fully spanning the channel with large conifer.

Comparing the results of the Prairie Creek inventory with the inventories for the surveyed Plan Area streams suggests that the occurrence of larger in-channel pieces is lower in managed streams within the Plan Area than in unmanaged streams nearby. Several of the surveyed Plan Area streams had average overall piece counts per 100' within specific size categories that approached or exceeded the values seen in Prairie Creek. However, the piece lengths in these managed streams were shorter than the piece lengths in Prairie Creek, especially in similar channel types. In the 20 surveyed Plan Area streams, most of the larger diameter LWD was either: 1) old-growth root wads with little or no bole attached to them, or 2) instream restoration projects consisting of short, stubby pieces of cull logs anchored to bedrock, boulders, or riparian trees. Both of these types of LWD often provide marginal habitat compared to intact trees recruited from the riparian zone. Old-growth redwood rootwads contain fairly large volumes of wood, yet their short length provides minimal surface area for capturing and retaining additional LWD to form complex salmonid habitat. The short length of these rootwads also increases their likelihood of mobilizing during moderate storm events (as occurred during the winters of 1995-96 and 1996-97).

C2.4 CONCLUSION

LWD within Plan Area streams will be reassessed periodically during the 50-year life of the Plan with the objective of documenting increases in conifer piece frequency, size, and functionality. Improvements in the current LWD inventories and sampling designs are needed to more accurately assess the changes in volume and function of LWD debris over longer periods of time. Conditions can be expected to gradually improve as a result of current FPRs and the increased riparian standards implemented under the Plan. The hardwood dominated riparian zones now prevalent on various Plan Area streams will eventually be succeeded by redwoods and other conifers, resulting in increasing recruitment of large diameter LWD for Plan Area streams. It has been suggested (McHenry et al. 1998, Emminghamm and Hibb 1996) that without active management of riparian zones; protection of existing conifers, conifer release and/or planting that conifer succession will be extremely slow or even effectively precluded.

C2.5 REFERENCES

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Appendix C3. Long-Term Channel Monitoring

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Figures

Tables

C3.1 BACKGROUND

Simpson implemented the initial long-term monitoring program of its California watersheds in 1993. The first two years of the monitoring program was based on two U.S. Forest Service publications which address monitoring strategies of both instream and riparian conditions (Platts et al. 1983; Platts et al. 1987). At the conception of this early monitoring study, the selection of watersheds was primarily influenced by the concerns of the Regional Water Quality Control Board and the CDFG regarding possible cumulative effects of Simpson's activities in several basins. The primary watersheds of concern were Salmon Creek and Jacoby Creek, both tributaries to Humboldt Bay. The Salmon Creek watershed was of concern due to its highly unstable and erosive geology (Wildcat Formation) and past management practices. The Jacoby Creek watershed has sections of erosive Franciscan Formations, a diverse mix of ownership and a complex history of watershed disturbances (logging, grazing and residential development). Additional watersheds were selected to distribute the monitoring across the ownership.

The next step in designing the early monitoring program was the selection of sample stream sections within watersheds. Two approaches were utilized in selecting sampling sections:

- Paired reference (control) and test (treatment) sections; and
- A general watershed approach.

When employing the paired reference and test sections, the sections were selected on the basis of their location relative to a potential impact from a management activity (e.g., sedimentation from a timber harvest). Sections established upstream from the activity site were the reference sections and those downstream were the test sections. The data collected from the reference and test sections were compared to evaluate potential impacts. However, to make data comparable, sections above and below the management activity must be selected from stream reaches that matched according to valley bottom and riverine habitat types. Once similar stream reaches were selected, each reach was divided into 300-foot sections from which two 300-foot sections were randomly selected. A minimum of two reference and two test sections were identified for each of Simpson's anticipated management activities within a watershed.

Because the location of potential impacts within a watershed cannot always be identified in advance, a general watershed approach must occasionally be utilized. With this approach, the 300-foot stream sections were randomly selected throughout a watershed without identifying them as either reference and test sections. Statistically, a minimum of five to eight sections were sampled, depending on the complexity of the watershed, to insure that suitable reference and test sections would be available following future timber harvest activities. Sampling was conducted following the protocol established by Platts et al. (1983 and 1987).

These pilot projects provided valuable information regarding effective methods and response variables, and the difficulties of analyzing the resulting data. Using the information gathered in these pilot studies, a revised methodology was developed and first implemented in Cañon Creek beginning in 1995.

To fine tune the long-term monitoring methodology, Simpson consulted with William Trush, a watershed scientist from Humboldt State University. Trush reviewed the channel monitoring program and suggested modifying the program to reduce data collection time and improve the ability to detect changes in channel response. His review indicated that:

- Most variables measured were flow dependant and generated significant differences in channel conditions with slight changes in base summer flow;
- The systematic selection of monitoring cross sections at ten foot intervals ignored geomorphic characteristics of certain channel features and processes; and
- Flow dependant variables resulted in significant differences regardless of management activities, while systematically selected monitoring cross sections created high variance estimates.

These comments assisted Simpson in revising its selection of stream reaches to capture specific channel responses to significant hydrologic events (and possibly management activities) and measuring only variables that were independent of flow. This protocol was implemented on Cañon Creek (a Mad River tributary) in 1995. During 1996, Simpson field personnel again monitored the Cañon Creek site and established additional channel monitoring reaches on the South Fork Winchuck River (a tributary in Smith hydrographic unit), Hunter Creek (a lower Klamath River tributary), and Salmon Creek (a Humboldt Bay tributary). These surveys have continued with scheduled re-surveys every two years or after a five year flood event. Data collected on all of the monitoring sites since 1998 are scheduled for analysis in 2003. Each monitoring reach should have at least 3 years of data prior to the first analysis and updated biennially to coincide with the biennial report to the Services (see Section 6 regarding report). The purpose of that monitoring protocol was to document the recovery of Plan Area watersheds from past timber harvesting practices and to evaluate the effects of current and future harvesting practices on watershed condition and recovery. The long-term channel monitoring protocol also has potential to evaluate the effectiveness of “storm-proofing” techniques, currently in vogue, in reducing road-related erosion sources.

C3.2 METHODOLOGY

In early 1998, Simpson hired a statistical consultant (Trent McDonald) to assist in refining and developing methods to analyze the long-term channel monitoring data. The consultant confirmed that the data being collected was valid and rendered itself to analysis. Using the previous developed monitoring data collection methods the results were analyzed as described below.

The monitoring objective of the Class I channel monitoring project was to track long term trends in the sediment budget of Class I watercourses as evidenced by changes in channel dimensions. Initially 3 and later 9 monitoring reaches were established in 8 streams across the Plan Area. Two additional reaches were also established with a reduced protocol (thalwag profile only), because the sites did not meet the criteria necessary for doing the full protocol. The initial three streams: Cañon, Hunter, and Canyon creeks were chosen for monitoring and analysis. A section of each creek was selected for monitoring activities and field sampling was carried out on those reaches using Simpson’s monitoring protocols as described above. Monitored sections were chosen to be the highest (closest to headwaters) depositional reach in each creek. Depositional reaches were characterized by relatively low gradient where sediment was expected to be deposited. The reasoning behind establishment of these monitoring reaches was that if changes in sediment load or other stream morphology parameters occurred anywhere in the watershed, such changes were likely to be reflected in the first depositional reach downstream. The three stream systems under study were small enough that there was only one depositional reach contained in each stream.

Three creeks in the Plan Area (Cañon Creek, Hunter Creek, and Canyon Creek) were chosen for monitoring and analysis. A section of each creek was chosen for monitoring activities and field sampling was carried out on those reaches under Simpson protocol. Monitored sections were chosen to be the highest (closest to headwaters) depositional reach in each creek. Depositional reaches were characterized by relatively low gradient where sediment was expected to be deposited. The reasoning behind establishment of these monitoring reaches was that if changes in sediment load or other stream morphology parameters occurred anywhere in the watershed, such changes were likely to be reflected in the first depositional reach downstream. The three stream systems under study were small enough that there was only one depositional reach contained in each stream.

Sampling occurred at Cañon Creek in 1995, 1996, and 1997. Sampling occurred in 1996 and 1997 at the other two creeks (Hunter and Canyon). Each year, thalweg elevation (defined as the height of the deepest part of the channel), bank full width, active channel width, and substrate (pebble) sizes were recorded on the monitoring reaches. Thalweg elevation residuals (see below) were analyzed for changes in variance. A change in thalweg residual variance indicates an improvement (or degradation) of pools via changes in pool depth. Bank full and active channel widths were analyzed for changes in average width. Substrate sizes were analyzed for changes in distribution.

C3.2.1 Analysis of the Thalweg

Thalweg elevation was analyzed for change in mean elevation and thalweg residuals (from a spatial polynomial regression of elevation on distance from the upper end of the reach) were analyzed for change in variance. Both sets of analyses used statistical models appropriate for correlated data. The basic data were pairs of points, (d_i, y_i), where y_i was thalweg elevation and d_i was the distance from the upper terminus of the reach to the point where y_i was measured. Because thalweg elevations were measured relatively close together (approximately every 10 feet) the measurements (i.e., the y_i) were potentially spatially correlated and did not represent independent observations. Therefore, the analyses accounted for this lack of independence by adjusting model coefficients and significance levels using a one dimensional spatial regression model (Cressie 1991; Venables and Ripley 1994). The spatial regression model estimated a one dimensional correlation function among residuals then adjusted estimates and p-values via generalized least squares regression techniques. The spatial regression techniques and the adjustment for auto-correlation is described in more detail in Attachment C3-A.

For the analysis of thalweg elevation, a regression model relating elevation of the thalweg to a cubic polynomial in distance was estimated. Included in this model was a year factor so that the interaction between year and the cubic polynomial in distance could also be estimated. In equation form and provided the reach will be monitored for three years, the regression relationship was:

$$
\begin{aligned}
E[y_i] = {} & \beta_0 + \beta_1 x_{1,i} + \beta_2 x_{2,i} \\
& + \beta_3 d_i + \beta_4 d_i^2 + \beta_5 d_i^3 \\
& + \beta_6 d_i x_{1,i} + \beta_7 d_i^2 x_{1,i} + \beta_8 d_i^3 x_{1,i} \\
& + \beta_9 d_i x_{2,i} + \beta_{10} d_i^2 x_{2,i} + \beta_{11} d_i^3 x_{2,i}
\end{aligned}
$$

where y_i was thalwag elevation measured at a distance of d_i meters from the top of the reach, $x_{1,i}$ was an indicator variable for year 1 (i.e., 1 if observation $_i$ was taken in year 1, 0 otherwise), and $x_{2,i}$ was an indicator variable for year 2 (i.e., 1 if observation $_i$ was taken in year 2, 0 otherwise). For reaches which were monitored only two years, $x_{2,i}$ and all interactions involving it were eliminated from the model (i.e., β_2, β_9, β_{10}, and β_{11} were not present in the model). These models effectively fit separate cubic polynomials in d_i each year.

The analysis for change in thalweg residual variance was a statistical test designed to detect increased (or decreased) variance in residuals which is indicative of increased (or decreased) pool depths and complexity of the reach habitat. Thalweg residuals were defined as the residuals of thalweg elevation in the above regression model; $r_{yi} = y_{yi} - \hat{y}_{yi}$, where y_{yi} was observed elevation at distance d_i in year y and $\hat{y}_{yi}$ was the predicted elevation at distance d_i in year y. The test for change in thalweg residual variance was carried out using a modified version of Levene's test (Neter et al. 1991). Absolute deviations of the residuals from their median were calculated as $d_{yi} = |r_{yi} - m_y|$, where d_{yi} was the absolute deviation associated with the i-th observation in the y-th year and m_y was the median of residuals in the y-th year. Levene's test entailed carrying out a one-way analysis of variance on the d_{yi}, with year defining the groups. Because the *ryi* were potentially (spatially) correlated, the d_{yi} were also potentially correlated and the one-way analysis of variance was adjusted using the spatial regression techniques outlined in Attachment C3-A. Variance of the original residuals was deemed significantly different across years if the (spatially adjusted) one-way analysis of variance rejected the hypothesis of equal average deviations. The distribution of thalweg residuals was also plotted as a visual interpretation aid.

C3.2.2 Analysis of Width

Both bank full and active channel widths were analyzed for changes across years. To conduct this analysis, a systematic sample of widths was computed from available data after field sampling was complete. Such a systematic sample of widths was necessary because field-sampling protocol dictated that each bank of the creek is measured separately. Consequently, width measurements were not taken completely across the creek, but rather from each bank to a center tape. Furthermore, measurements from one bank to the center tape were not necessarily in the same place as measurements to the opposite bank. Therefore width could not be computed directly from the raw data and consequently a systematic sample of widths was computed and analyzed by the following methods. The systematic sample of widths was computed by first connecting left and right bank width measurements with straight lines to form an approximate stream channel. A random starting point along the center tape was then chosen and widths (across the whole channel) were computed at regular intervals along the center tape. The number of systematic points in the sample was equal to the smaller of the two sample sizes taken on each bank. For example, if 50 measurements were taken on the left bank and 75 measurements were taken on the right bank, 50 systematic measurements of width were taken to analyze. A picture of the systematic sample of widths computed at Cañon Creek in 1996 is presented in Figure C3-1 below.

The systematic sample of widths was computed each year for each creek. Average width was analyzed using one-way analysis of variance (anova) techniques analogous to the modified Levene's test described for analysis of thalweg residual. A one-way analysis of variance (two sample t-test if only two years) was computed, with year as the grouping factor, to test for changes in mean stream width. Because measurements in the field were taken relatively close together and because spacing of the systematic sample of widths was relatively tight, computed widths were potentially correlated and consequently the analysis of variance was modified to adjust for spatial correlations using the techniques outlined in Attachment C3-A. This analysis of variance was parallel to the modified Levene's test described for analysis of thalweg residual variance.

C3.2.3 Analysis of Substrate Size

Substrate size, or pebble size, was measured at between 5 and 10 sites within each monitored reach. Each site was approximately 50 feet by 50 feet in size and consisted of sand bars, lee banks, and other rocky areas in the stream. At each site, field personnel measured the secondary axis of rocks (pebbles) which were collected by selecting one near the toe of their right foot as transects were walked around the site. Collection and measurement continued until 150 rocks were measured. All measurements were reported in millimeters and the smallest measurement was one millimeter.

The distribution of pebble size was plotted and analyzed for changes across years assuming independence of the measurements. Due to the large distances (relative to average pebble size) at which rocks were measured and the fact that several independent systematic samples were taken at each site, spatial correlations among observations were highly unlikely and consequently no adjustments for such correlation were made. The hypothesis of no change in distribution was tested using two sample Wilcoxon rank sum tests (Wilcoxon 1945, Hollander and Wolf 1973) or three sample Kruskal-Wallis tests (Lehmann 1975; Hollander and Wolf 1979), depending on the number of years data were collected from a stream. Substrate size measurements from all sites within a year were combined for testing because site to site differences in substrate size were not of interest and, if such differences existed, would tend to inflate the distribution's variance and provide a conservative analysis. Treating the systematic measurements as if they were purely random (i.e., by assuming independence) also inflates the distribution's variance and further contributes to a conservative analysis.

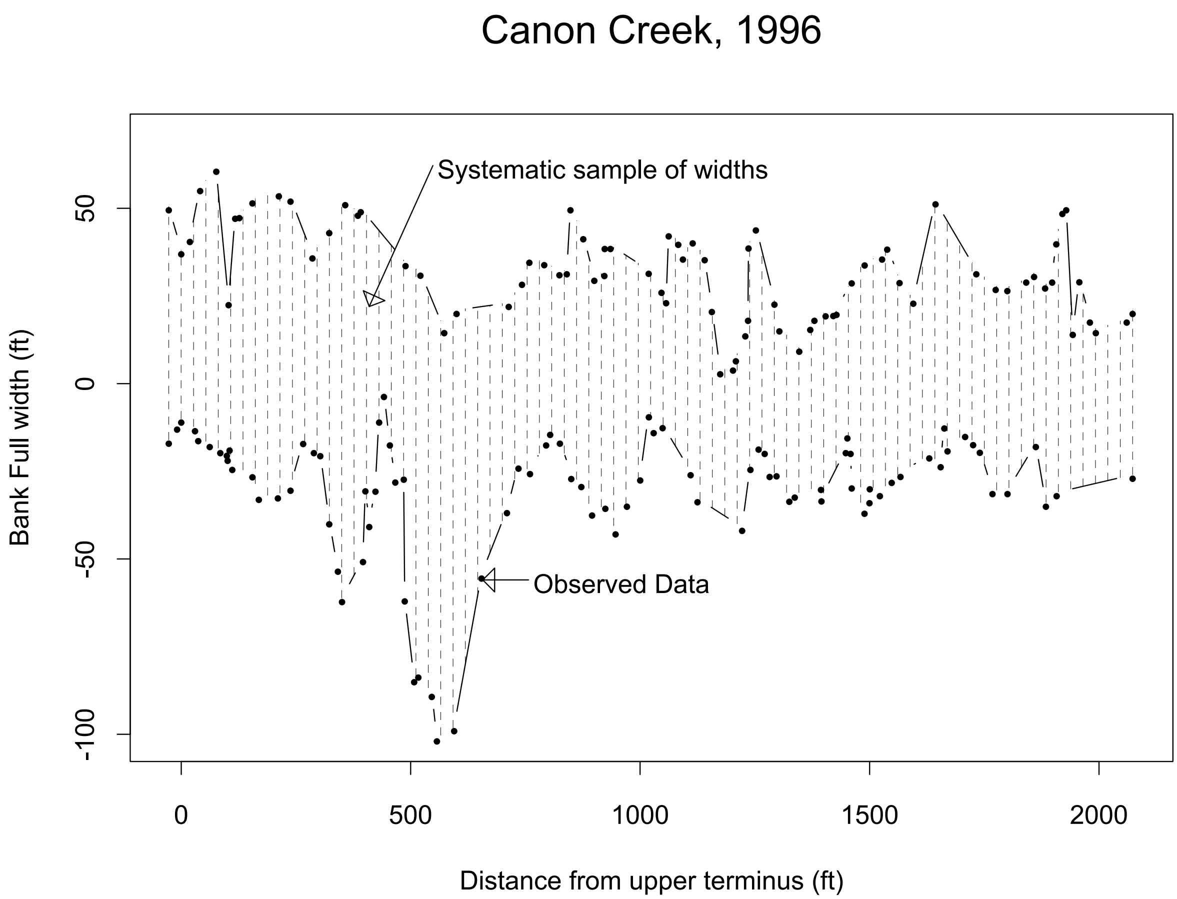

Figure C3-1. Diagram of the systematic sample of widths taken for the investigation of width (Cañon Creek 1996). This example shows bank full width at Cañon Creek in 1996. Zero in vertical dimension represents the center tape while negative numbers represent the left bank and positive numbers represent the right. Dots are observed bank full measurements with linear interpolation between each. Dashed lines show the systematic sample of widths.

Three quantiles from each substrate distribution were estimated. The 16-th, 50-th, and 84-th quantiles were estimated from each distribution to facilitate comparison with sediment movement models developed elsewhere (USEPA 2000). The 16-th quantile was defined as that point in the distribution that was greater than 16% of the observations and less than 84% of the observations. By symmetry, the 84-th quantile was defined as that point in the distribution that was greater than 84% of the observations and less than 16% of the observations. The 50-th quantile was defined similarly and corresponded to the median. The standard error of each quantile was estimated using standard bootstrap methods (Manly 1997).

C3.3 RESULTS

C3.3.1 Analysis of the Thalweg

At Cañon Creek, thalweg elevation measurements were significantly correlated with other thalweg elevations measured nearby. Correlation of thalweg residuals (i.e., residuals computed from the initial regression) within 8 feet of one another was 0.52 in 1995 (95% confidence interval 0.21 - 0.83), 0.81 in 1996 (95% confidence interval = 0.46 - 1.0), and 0.73 in 1997 (95% confidence interval = 0.52 - 0.95).

A graph of the final spatial regression model for Cañon Creek appears in Figure C3-2. There was a significant difference in overall curvature of the thalweg profile at Cañon Creek between 1995 and later years ($p<0.0001$ for 1995 vs. 1996; $p<0.0001$ for 1995 vs. 1997). The overall curvature of the thalweg profile was negative in 1995 while in 1996 and 1997 curvature was positive. Inspection of Figure C3-2 shows that the middle half (approximately) of the Cañon Creek monitoring reach remained at roughly the same elevation in all three years, but that the upper and lower quarters (approximately) were lower in 1995 and than in 1996 and 1997. No significant differences existed in the linear or cubic trends between 1995, 1996, and 1997. No significant differences existed in overall thalweg trend between 1996 and 1997 ($p=0.29$ for linear trend, $p=0.37$ for quadratic trend, $p=0.77$ for cubic trend).

Thalweg elevation measurements in Hunter Creek were significantly correlated with similar measurements taken nearby. Correlation of thalweg residuals within 8 feet of one another was 0.44 in 1996 (95% confidence interval 0.11 - 0.78), and 0.98 in 1997 (95% confidence interval 0.64 - 1.0).

A graph of the final spatial regression model for Hunter Creek appears in Figure C3-3. A marginally significant difference existed in the coefficient of the cubic trend term between 1996 and 1997 at Hunter Creek ($p=0.072$). This difference in third order trend, if deemed significant, was caused by a drop in thalweg elevation from 1996 to 1997 near the bottom third of the monitoring reach, between 1500 and 2200 feet from the upper terminus of the reach.

Thalweg elevation measurements in Canyon Creek were significantly correlated with similar measurements taken nearby. Correlation of thalweg residuals in Canyon Creek within 8 feet of one another was 0.69 in 1996 (95% confidence interval = 0.42 - 0.97), and 0.65 in 1997 (95% confidence interval = 0.43 - 0.87).

A graph of the final spatial regression model for Canyon Creek appears in Figure C3-4. No significant differences occurred in overall thalweg elevation in Canyon Creek between 1996 and 1997 (p= 0.36 for year*linear term, p=0.78 for year*quadratic term, p=0.10 for year*cubic term). Because yearly interaction was not significant, interaction was dropped from the final regression at Canyon Creek and consequently the lines in Figure C3-4 were forced to be exactly parallel. There was no difference in the parallel lines of Figure C3-4 (p=0.67).

The distributions of thalweg residual for Cañon, Hunter, and Canyon creeks appear in Figure C3-5, Figure C3-6 and Figure C3-7. In addition to standard histograms, these figures display a (Gaussian) kernel smooth density estimate for each distribution. Absolute deviations from the median, used in Levene's test, measured near one another were significantly correlated in every creek every year.

Table C3-1 contains estimates and confidence intervals for correlation between absolute deviations within 8 feet of one another. After adjustment for spatial correlation using the method outlined in Attachment C3-A, there remained a significant decrease in thalweg residual variance at Cañon creek between 1995 and latter years (p=0.0019 for 1995 vs. 1996; p=0.0013 for 1995 vs 1997).

Inspection of the histograms in Figure C3-5 confirm that there were more large negative thalweg residuals in 1995 than there were in 1996 and 1997. There was no significant difference in thalweg residual variance between 1996 and 1997 at Cañon Creek (p=0.5379). Thalweg residuals at Hunter and Canyon creeks displayed changes similar to those at Cañon Creek. Variance of thalweg residuals was higher in 1996 than 1997 at both Hunter and Canyon creeks (p=0.0465 for Hunter, p=0.0365 for Canyon). Inspection of Figure C3-6 and Figure C3-7 confirm that there were more large negative residuals in 1996 than in 1997 at both creeks.

Table C3-1. Estimated correlations among absolute thalweg residual deviations from the median measured less than 8 feet apart.

Creek	Year	Estimated Correlation	Approximate 95% confidence interval	
			Low	High
Cañon	1995	0.50	0.19	0.81
	1996	0.83	0.49	1.00
	1997	0.70	0.49	0.91
Hunter	1996	0.38	0.05	0.72
	1997	0.89	0.55	1.0
Canyon	1996	0.70	0.42	0.97
	1997	0.60	0.38	0.82

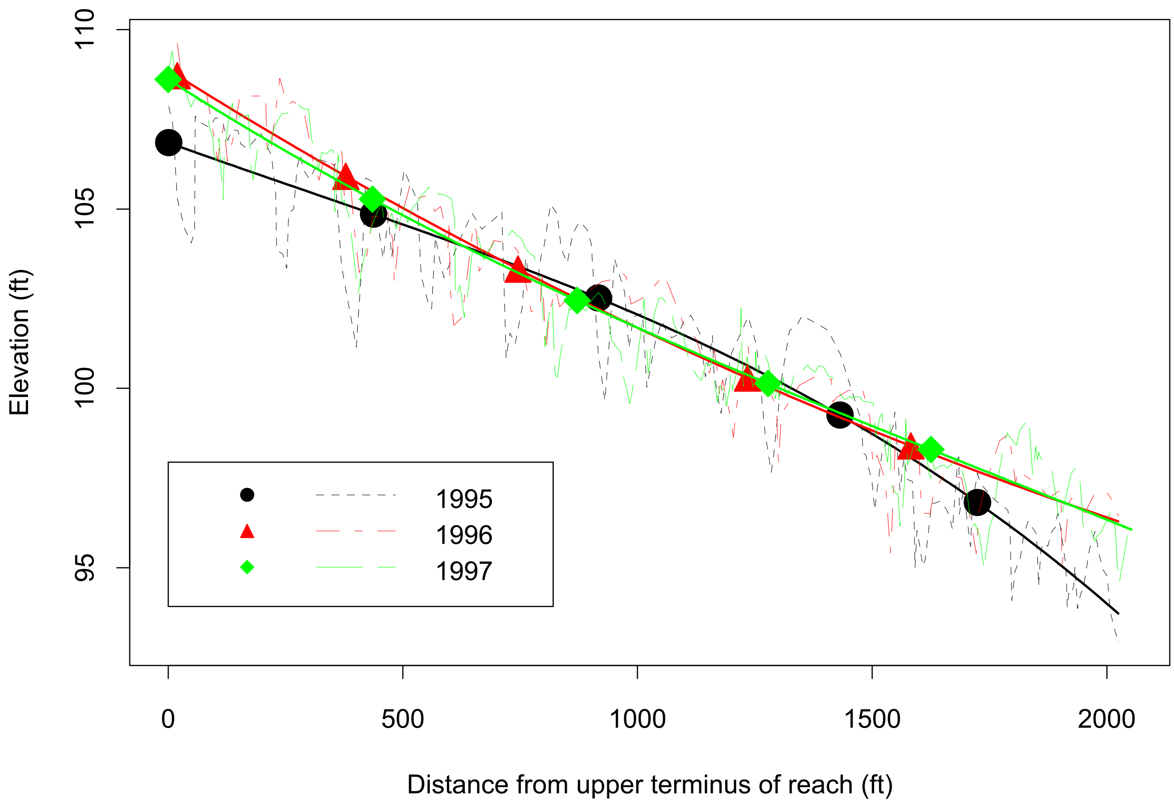

Figure C3-2. Thalweg elevation profile for the Cañon Creek monitoring reach, 1995, 1996, and 1997. Dashed lines show measured elevations. Solid lines show trend estimated by spatial regression that adjusted for auto-correlation in residuals. Curvature (2nd derivative) was negative in 1995, positive in 1996 and 1997.

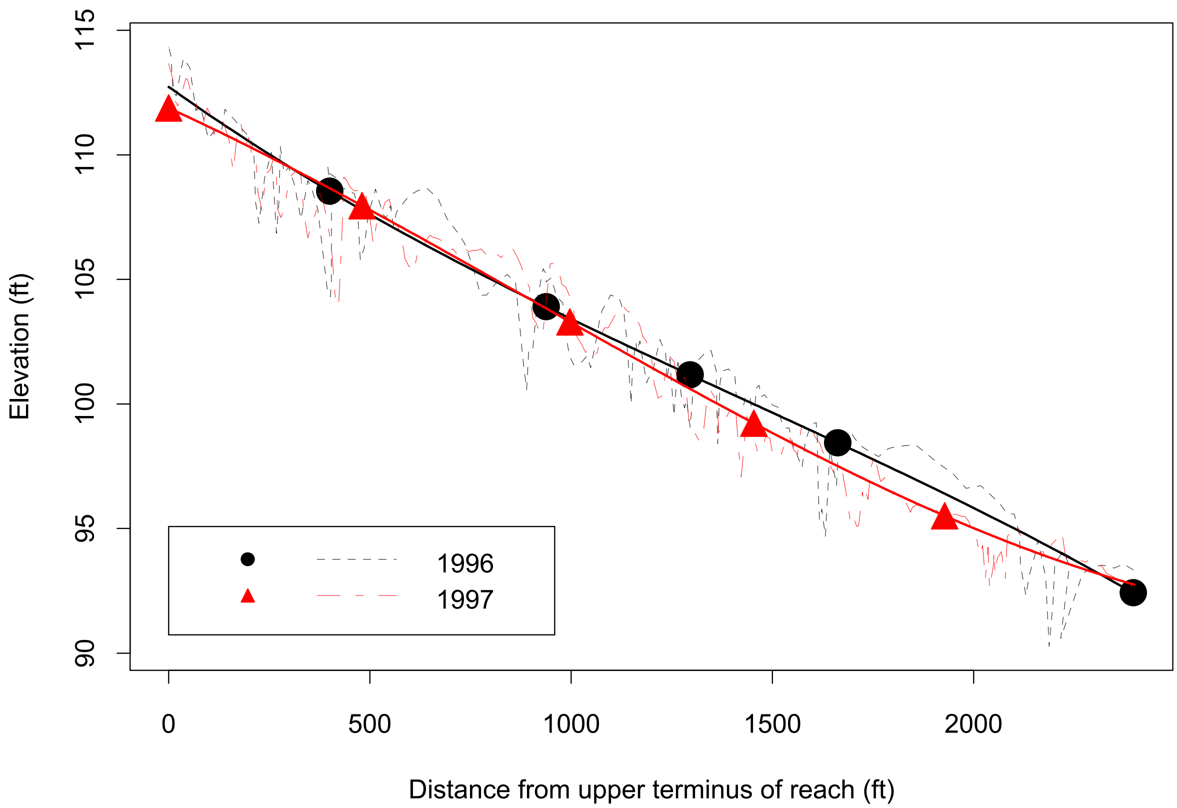

Figure C3-3. Thalweg elevation profile for the Hunter Creek monitoring reach in 1996 and 1997. Dashed lines show measured elevations. Solid lines show trend estimated by spatial regression that adjusted for auto-correlation in residuals.

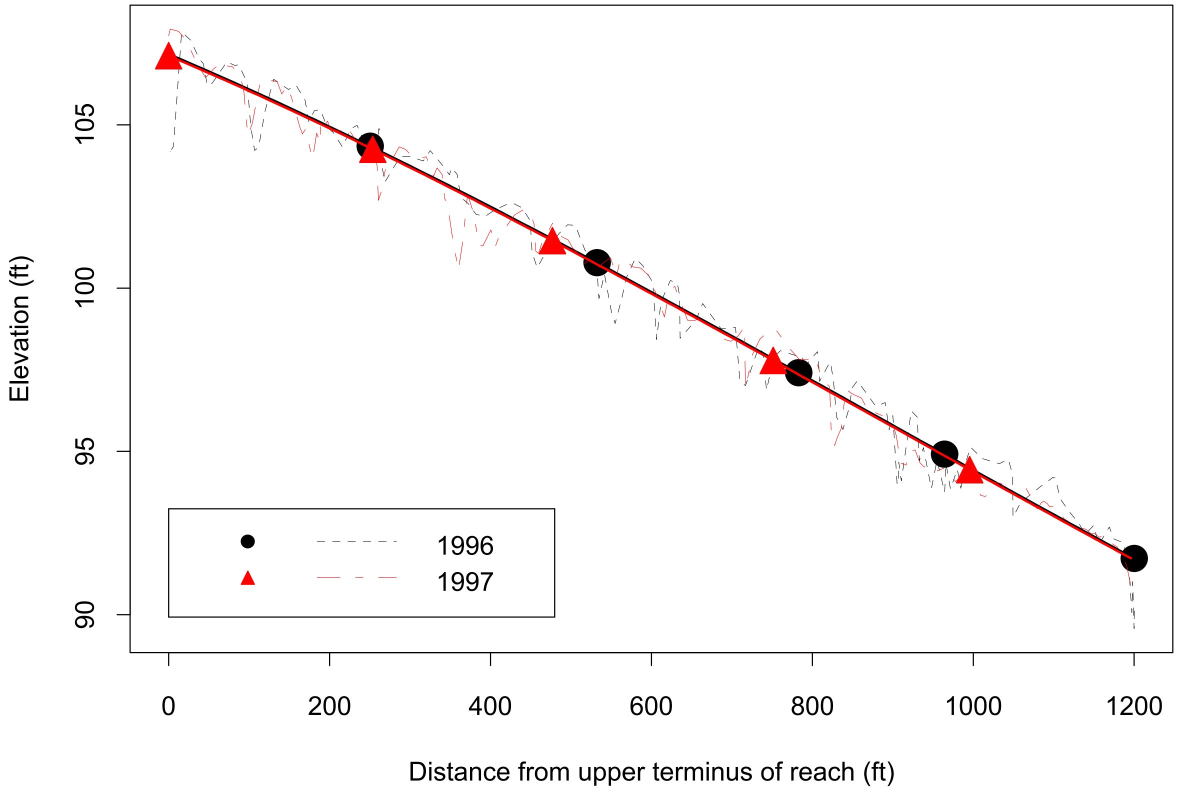

Figure C3-4. **Thalweg elevation profile for the Canyon Creek monitoring reach in 1996 and 1997. Dashed lines show measured elevations. Solid lines show trend estimated by spatial regression that adjusted for auto-correlation in residuals.**

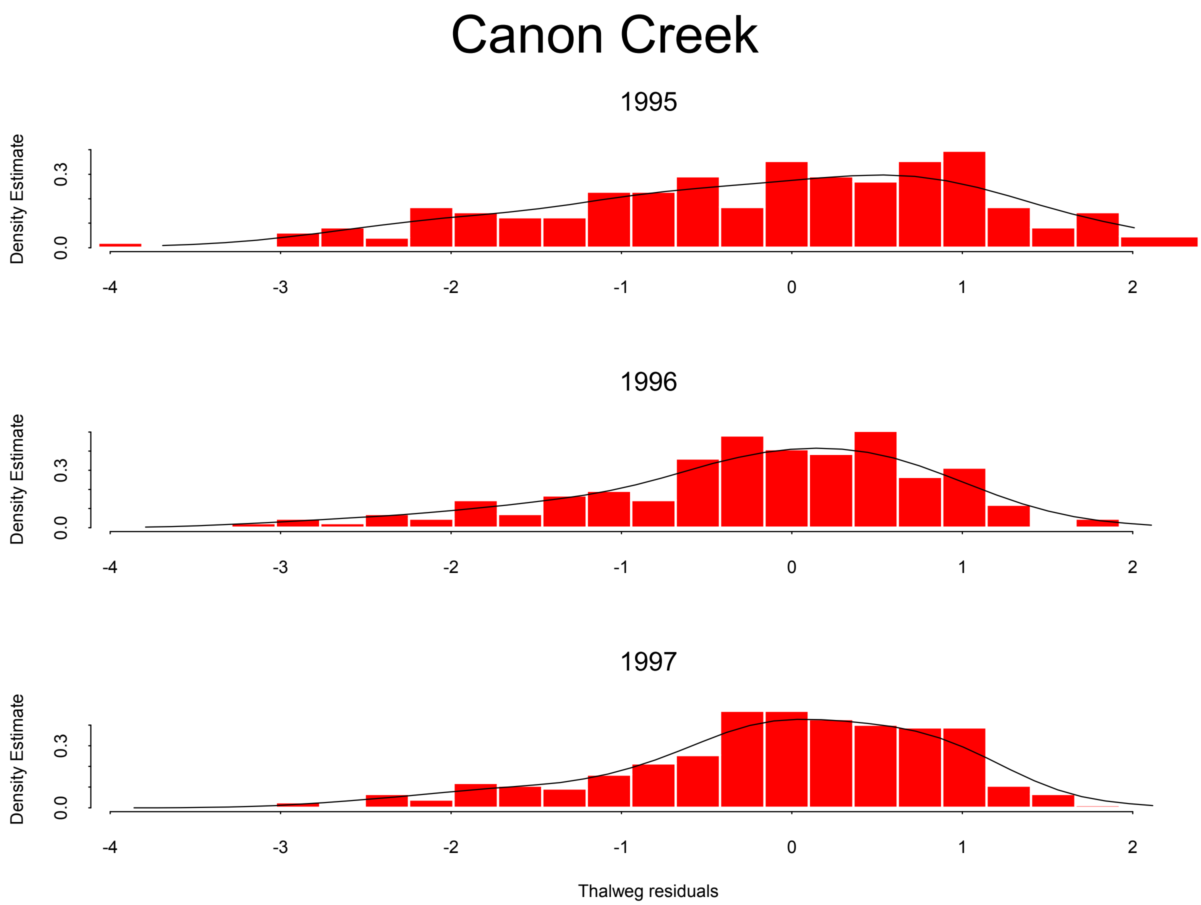

Figure C3-5. **Histograms of thalweg residuals at Cañon Creek, 1995 through 1997, used to compare variance of residuals among years. Residuals computed using models fit in Figure C3-1. Solid line is Gaussian kernel smoothed density estimate.**

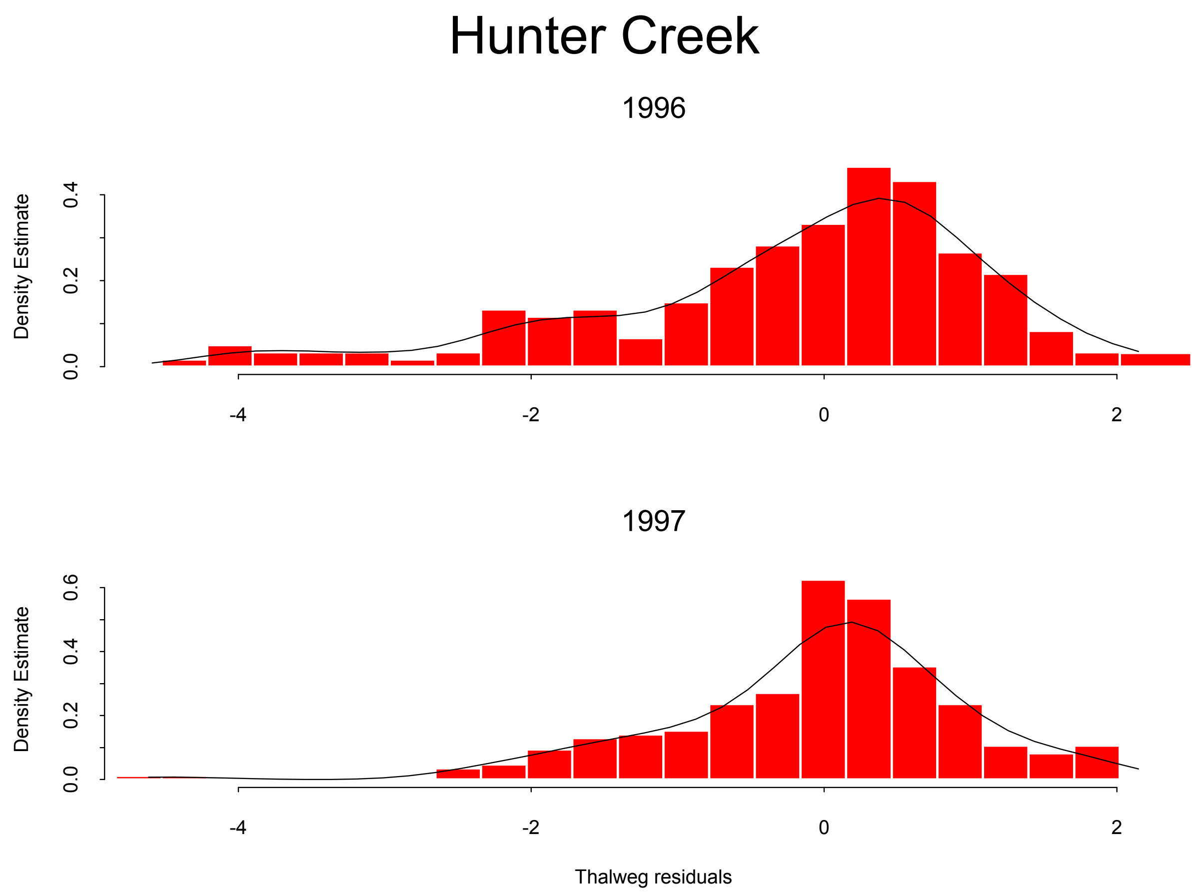

Figure C3-6. Histograms of thalweg residuals at Hunter Creek, 1996 and 1997, used to compare variance of residuals among years. Residuals computed using models fit in Figure C3-2. Solid line is Gaussian kernel smoothed density estimate.

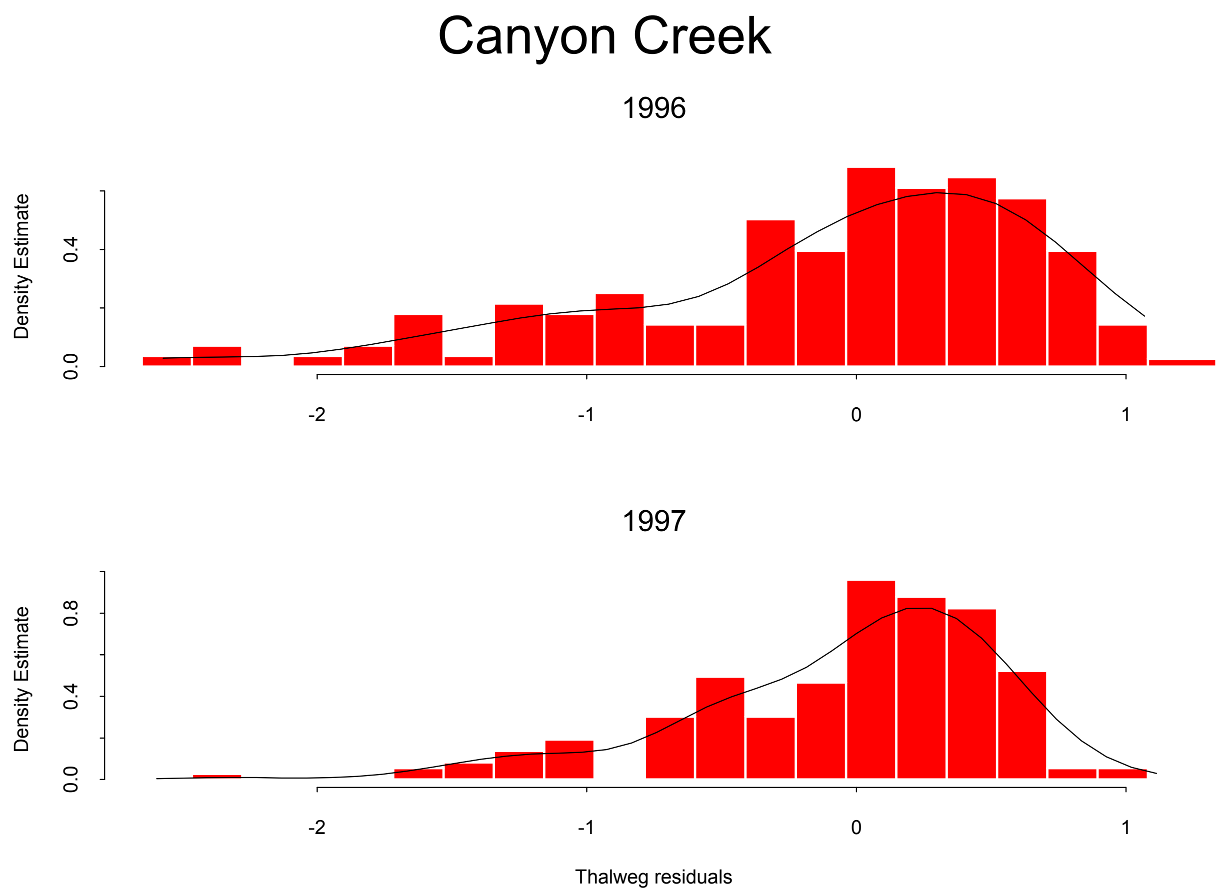

Figure C3-7. Histograms of thalweg residuals at Canyon Creek, 1996 and 1997, used to compare variance of residuals among years. Residuals computed using models fit in Figure C3-3. Solid line is Gaussian kernel smoothed density estimate.

C3.3.2 Analysis of Width

Both bankfull and active channel width measurements were significantly correlated when measured close together. For bank full width at Cañon Creek, the estimated correlation among measurements within 100 feet of one another was generally greater than 0.5 in all years and never lower than 0.32. The estimated correlation among active channel width measurements at Cañon Creek which were within 100 feet of one another was greater than 0.47 in all years and as high as 0.82 for measurements within 25 feet of one another. Similar high spatial correlations were observed in Hunter and Canyon creeks. Correlation of both bankfull and active channel widths measured within 50 to 75 feet of one another was generally greater than 0.5. Consequently, substantial adjustments were made to the estimates and p-values when correlations were accounted for.

Table C3-2 contains estimated mean bankfull and active channel widths for all years of the study. Values reported in Table C3-2 were obtained from the coefficients of the spatial regression (anova) model and standard errors are adjusted for estimated correlations. At Cañon Creek, the observed increase in mean bank full width from 1995 to 1996 was almost statistically significant at the $\alpha=0.05$ level ($p=0.054$). Mean bank full width at Cañon Creek was significantly bigger in 1997 when compared to 1995 ($p=0.015$), but there was no difference in bankfull width between 1996 and 1997 ($p=0.57$). Active channel widths followed a pattern similar to bankfull. Active channel width at Cañon Creek increased significantly between 1995 and subsequent years ($p<0.0001$ for 1995 vs. 1996; $p<0.0001$ for 1995 vs. 1997), but remained constant between 1996 and 1997 ($p=0.45$ for 1996 vs. 1997). At Hunter Creek, neither bank full and active channel width changed significantly between 1996 and 1997 ($p=0.90$ for bankfull, $p=0.88$ for active channel). At Canyon Creek, the change in bankfull width between 1996 and 1997 was almost statistically significant at the $\alpha=0.05$ level ($p=0.057$). Active channel width at Canyon Creek was not significantly different between 1996 and 1997 ($p=0.25$).

Table C3-2. Estimated bankfull and active channel width for all years of the study.[1]

Creek	Year	Estimated Mean Bankfull Width (ft)	Standard Error, Bankfull	Estimated Mean Active Channel Width (ft)	Standard Error, Active Channel
Cañon	1995	47.39	4.68	29.51	2.64
	1996	62.06	5.97	47.16	2.36
	1997	67.15	6.61	50.78	4.11
Hunter	1996	56.2	3.42	38.5	3.15
	1997	57.0	5.13	37.8	3.40
Canyon	1996	33.4	1.39	20.8	1.04
	1997	27.0	3.00	18.6	1.58

Note

1 Estimates and standard errors were computed from the spatial regression model that accounted for spatial correlation. All measurements in feet. Significance levels can be found in the text.

C3.3.3 Analysis of Substrate Size

Figure C3-8, Figure C3-9, and Figure C3-10 display estimates of substrate size distribution for the three monitored creeks for all years of the study. Table C3-3 contains the estimated 16-th, 50-th, and 84-th quantiles from each distribution depicted in the figures, as well as each quantile's bootstrap standard error.

Table C3- 3. Estimated quantiles of substrate distributions found in three monitored creeks.[1]

Creek	Year	16th Quantile (Standard Err.)	50th Quantile (Standard Err.)	84th Quantile (Standard Err.)
Cañon	1995	14 (0.59)	36 (0.94)	68 (1.62)
	1996	11 (0.60)	29 (0.91)	63 (1.77)
	1997	16 (1.59)	44.5 (1.91)	80 (2.29)
Hunter	1996	17 (0.85)	41 (1.69)	85 (2.60)
	1997	15 (0.76)	44 (1.55)	98 (3.36)
Canyon	1996	9 (0.73)	35 (1.22)	67 (1.58)
	1997	15 (1.25)	43.5 (1.53)	84 (2.45)

Note
1 Standard errors of each quantile computed using 1000 bootstrap iterations. All measurements in millimeters (mm). 50-th quantile is the median.

The three distributions of pebble size at Cañon Creek, depicted in Figure C3-8, were all significantly different from one another ($p<0.0001$, Kruskal-Wallis; $p<0.0001$ Wilcoxon 1995 vs. 1996; $p<0.0001$, Wilcoxon, 1995 vs. 1997; and $p<0.0001$, Wilcoxon, 1996 vs. 1997). Although marginally difficult to visualize in Figure C3-8, the tests and values in Table C3-3 indicated that, in general, the distribution of pebble size shifted to the left (smaller) from 1995 to 1996 and then shifted back to the right (larger) from 1996 to 1997. Most of the distributional differences among years at Cañon Creek can be attributed to differences in the right hand tail of the distribution, with relatively more small substrate observed in 1996.

The distribution of pebble size at Hunter Creek was marginally significantly different between 1996 and 1997 ($p=0.061$, Wilcoxon). Quantiles reported in Table C3-3 indicated that the change in distribution, although not significant at the $\alpha=0.05$ level, involved a slight increase in the relative frequency of larger pebbles in 1997, relative to 1996.

The distribution of pebble size at Canyon Creek increased from 1996 to 1997 ($p<0.0001$, Wilcoxon). Inspection of Table C3-3 and Figure C3-10 reveals that almost all of the distribution of pebble size shifted to the right (larger) in 1997 at Canyon Creek, relative to 1996.

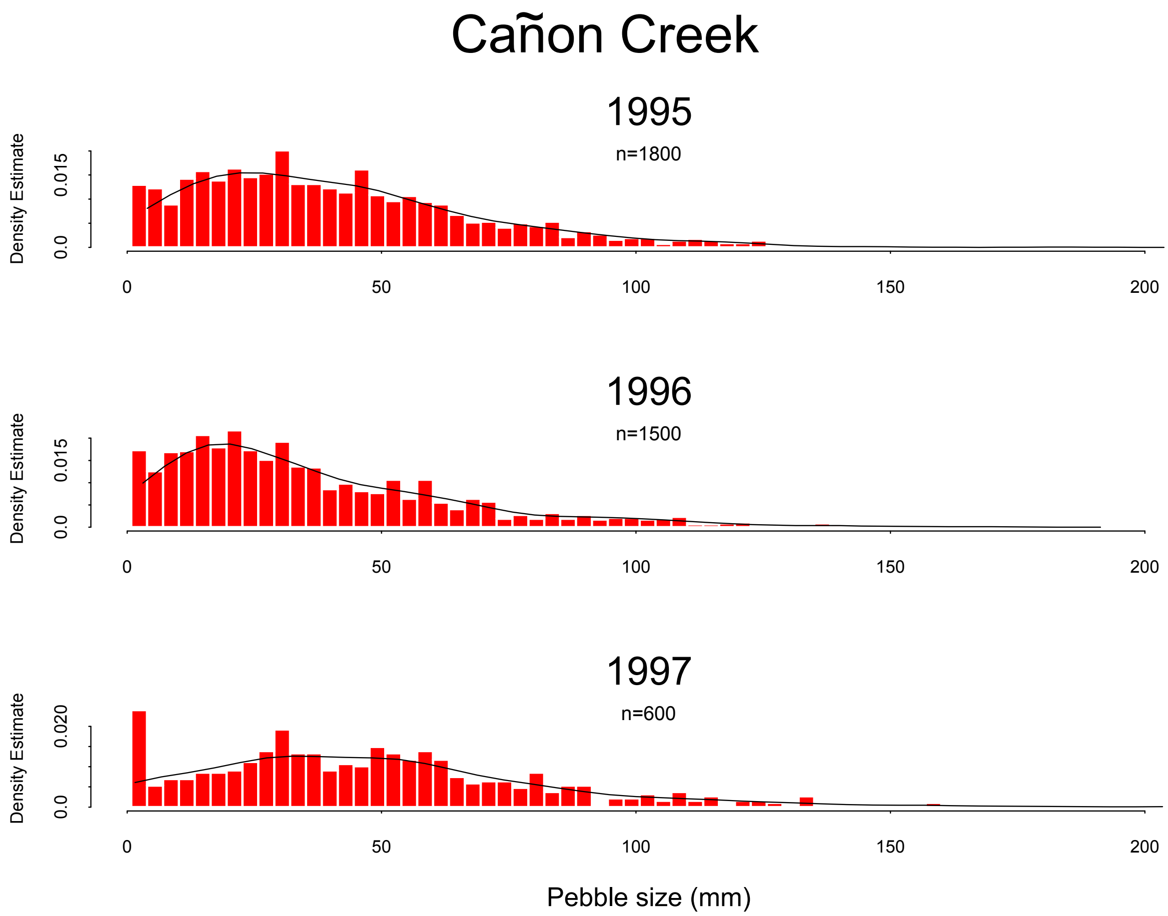

Figure C3-8. Estimated distributions of pebble size in Cañon Creek during the study. Solid lines are Gaussian kernel smooth density estimates.

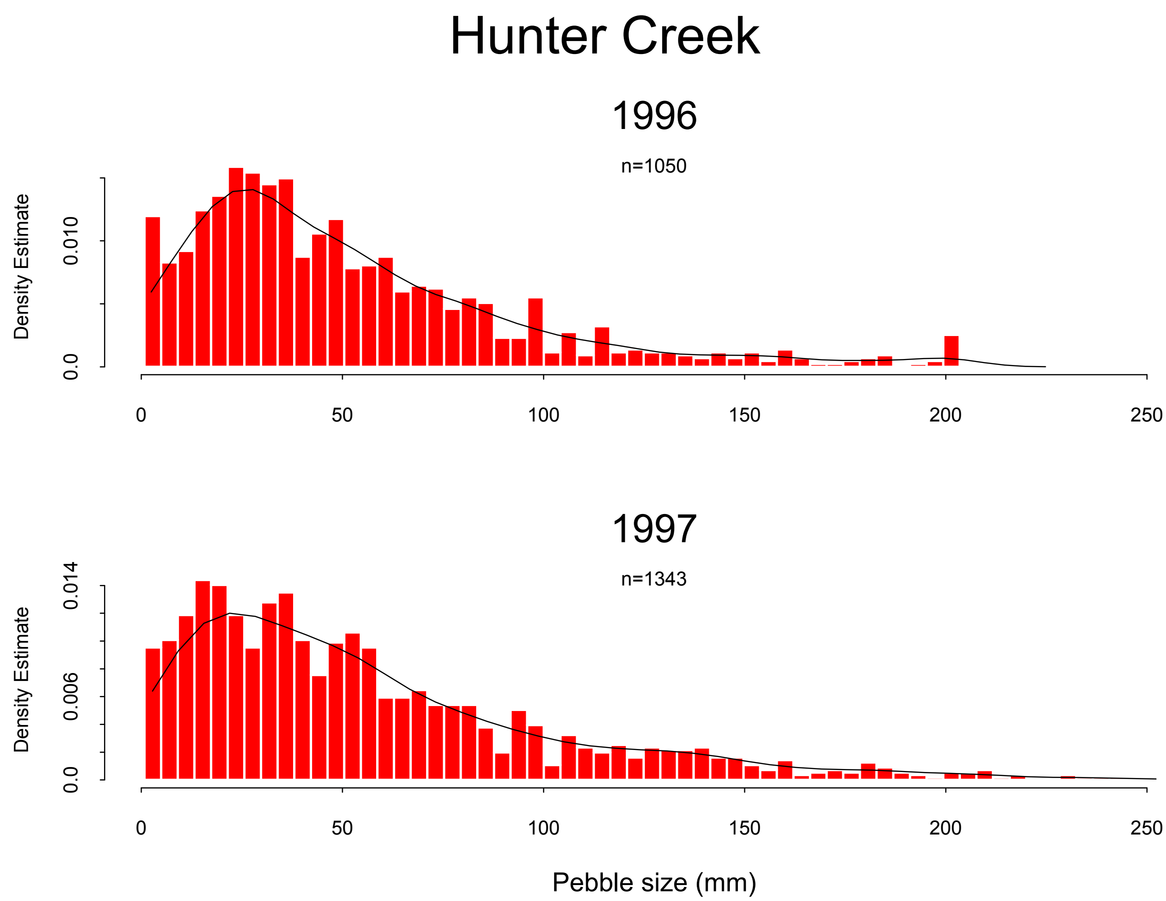

Figure C3-9. Estimated distributions of pebble size in Hunter Creek during the study. Solid lines are Gaussian kernel smooth density estimates.

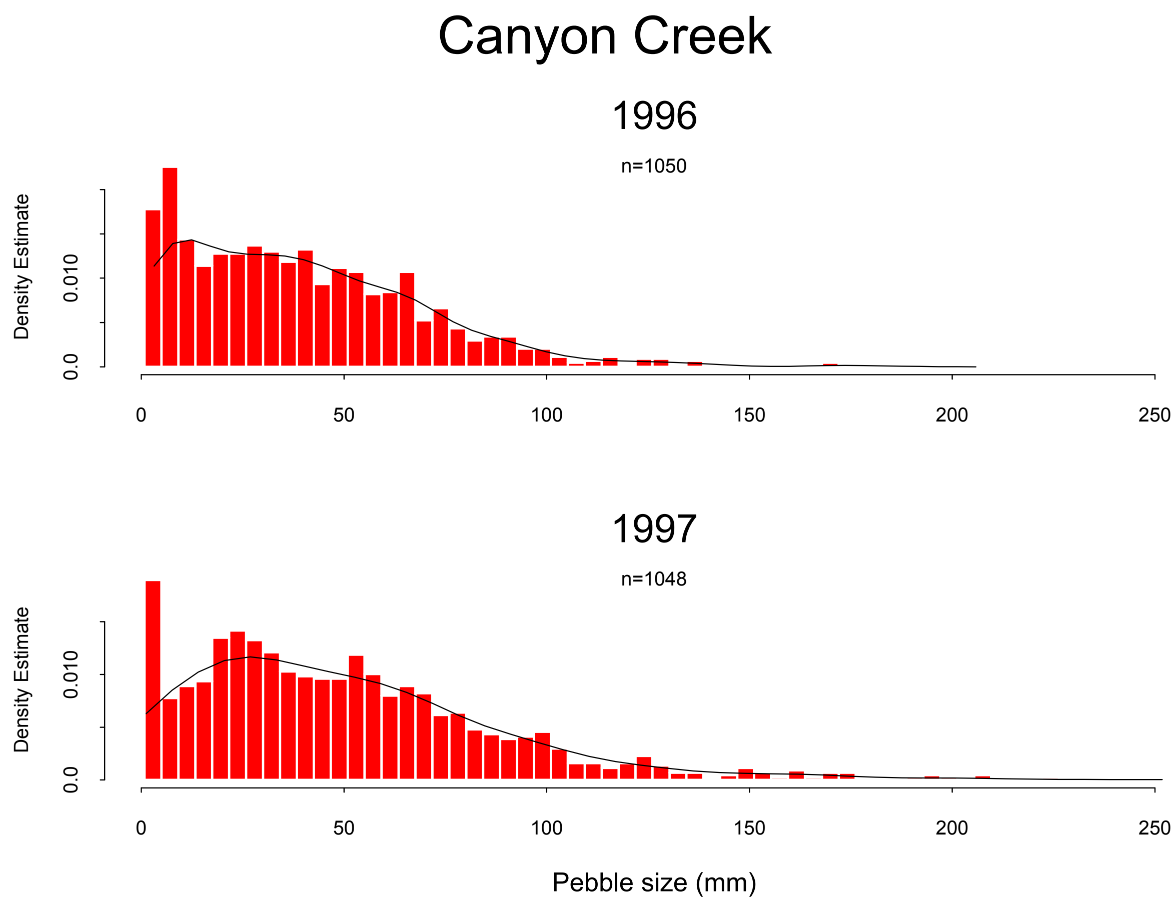

Figure C3-10. Estimated distributions of pebble size in Canyon Creek during the study. Solid lines are Gaussian kernel smooth density estimates.

As a caution when interpreting the results of this section, note that the number of pebbles measured in each creek each year was quite high (number of pebbles measured is given as >n= in Figure C3-8 through Figure C3-10). Such large sample sizes caused high statistical power to detect even relatively small differences in distributions. Small differences, although statistically significant, should be judged as to whether or not they are of any practical importance before any management decisions are made.

C3.4 DISCUSSION

The fundamental assumption associated with the long term channel monitoring is that the morphology of a depositional stream reach acts as a response surface for upslope sediment inputs. When sediment delivery increase beyond the capacity of the stream to transport it, depositional reaches will become aggraded, reduced sediment inputs will result in the opposite response. Although the morphological changes of stream reaches due to upslope sediment inputs have been well documented (Swanston 1991; Benda 1990; Benda and Dunne 1987; Hagans et al. 1986; Heede 1980), there are limitations associated with using this phenomenon for monitoring hillslope sediment production.

Quantification of some of the complex changes in channel morphology that result from changes in sediment supply can be problematic. Some changes such as the degree of sinuousity of a given stream reach generally follow predictable patterns depending on changes in the sediment load, but quantification in a statistically rigorous manner may not be possible. To deal with this potential problem, the channel monitoring protocol has been refined over time to focus on variables that respond in predictable ways and lend themselves to statistical analysis. The primary response variables that were determined to be suitable for measurement with minimum subjectivity and rigorous statistical analysis include changes in thalweg elevation and residuals, bankfull and active channel width, and substrate particle size distribution.

One of the most commonly raised concerns related to using channel morphology for monitoring is the lag times that can be associated with upslope sediment inputs and the corresponding response in the depositional reach. There is also a potential problem associated with separating natural sediment inputs from management related inputs. Both of these limitations are exacerbated with increasing distances between the upslope sediment sources and the depositional reach. As a result, the use of this monitoring approach was limited to depositional stream reaches that are closely coupled to transport reaches and potential hillslope sediment sources. Ideally, each monitoring reach is located in the watershed such that it is the first depositional reach immediately below continuously confined high gradient reaches that deliver sediment from upslope delivery sites with no capacity to store sediments in route. In reality, it is usually not possible to find the ideal monitoring reach and the selected reaches vary in how closely they are located to transport reaches and the extent to which sediments can be stored upstream of the monitoring site.

However, the response variables were found to be sensitive to mass wasting and major storm events, which have been shown to significantly change the channel dimensions. For example in Canon Creek, there was a significant decrease in the thalweg residual variance between 1995 and 1996. Between these two sampling years, there was a 10-15 year flood event (January 1996) that altered the channel morphology. The resurvey during the summer following the January 1996 flood indicated that the frequency of large deep pools decreased and the upstream and downstream ends of the monitoring reach aggraded. In this particular case, the response time was rapid in terms of showing changes in the morphology of the reach following a storm. However, Canon Creek has several miles of upstream transitional reaches that have the capacity of storing sediment, so that the aggrading of the channel did not necessarily indicate increased hillslope sediment inputs during the 1996 flood. This short coming of some of the first monitoring reaches has been recognized, and subsequent monitoring reaches have been placed so that this problem will be minimized. Although the data have not yet been analyzed, there is strong evidence that a second Hunter Creek monitoring reach located further upstream responded dramatically to a mass wasting event triggered higher up in the watershed during a November 1998 storm. The changes in the monitoring reach appeared to occur within days of the storm event. Given the differences in their placement, Simpson believes that the current monitoring sites have a range of response times that can vary from days to 1-2 years following a >5-year storm event. The individual response time of each monitoring site will be confirmed over time through additional monitoring.

An additional challenge associated with using channel dynamics for monitoring purposes is understanding the range of natural variability that is associated with any given stream. As a result, it likely will be necessary to continue monitoring for extended periods of time to develop a full understanding of the natural relationship between storm recurrence intervals and stream morphology. Even though it may be difficult to delineate natural variability from anthropogenic changes in the near term, Simpson believes that many useful insights will be gained in understanding the link between hillslope processes and channel morphology.

C3.5 CONCLUSION

This is a long term monitoring study, and therefore Simpson does not expect to be able to determine trends in the sediment budget of Class I watercourses for possibly 10-15 years. Threshold values for monitoring can not be established until lag times and the range of natural variability for individual watersheds or sub-basins are understood. In the interim period, Simpson expects to gain useful insights concerning the relationship between channel dynamics and hillslope processes within the Plan Area. By integrating data from different monitoring approaches, Simpson believes that channel monitoring will ultimately be a powerful tool for better understanding of the relationship between management activities and stream habitat condition for the Covered Species in the Plan Area.

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ATTACHMENT C3-A

This attachment describes the spatial regression technique used in the analysis of mean thalweg elevation, thalweg residuals, and mean channel width. This spatial regression analysis attempted to account for spatial correlations in the responses, which arise because measurements were taken close together. The technique can be described in three steps; 1) ordinary least squares parameter estimation, 2) auto-correlation modeling, and 3) weighted linear regression. Each step is described below.

Step one of the spatial regression analysis estimated a regular (Normal theory) regression of responses (i.e., thalweg elevation, thalweg residual, or channel width) onto a set of indicator variables and/or other explanatory study covariates. For example, the analysis for change in average thalweg elevation related elevation of the thalweg to a cubic polynomial of distance. The models for thalweg residual and channel width were analysis of variance (anova) models and contained indicator functions delineating the years of the study. More details about the models used for each response can be found in the main body of this report.

Step two of the spatial regression analysis estimated and modeled the auto-correlation among observed regression residuals. Estimated auto-correlations among residuals were deemed significant at various distances if an approximate 95% confidence interval surrounding Moran's *I* statistic (Moran 1950) did not contain zero. Moran's *I* was computed for relatively short lag distances, longer lag distances were ignored. If significant auto-correlation were found in the residuals, a non-linear correlation model which predicted correlation as a function of the distance between measurements was fit to the estimated correlations (see below for the form of the variance model). Auto-correlations (if significant) were modeled (spatially) within year and no (temporal) correlation was allowed across years.

If significant auto-correlations existed, a *spherical* variance model (Cressie 1991) was fit to model correlations as a function of distance. The spherical variance model had the form $v(d_{ij}) = c_1(1-1.5(d_{ij}/h_0)-0.5(d_{ij}/h_0)^3)$ if d_{ij} #h_0 and 0 if $d_{ij} > h_0$ where d_{ij} was the distance between measurements $_i$ and $_j$, and c_1 and h_0 were parameters to be estimated (c_1 is commonly called the intercept and h_0 is commonly called the range). The parameters c_1 and h_0 were estimated by forming all possible statistics $z_{ij} = (r_i - \mu\, r)(r_j - \mu r)/s_r^2$, where r_i was the regression residual from the $_i$-th observation and s_r^2 was the sample variance of the residuals, and plotting the z_{ij} against d_{ij}. This graph was then smoothed using a Gaussian kernel smoother (Venables and Ripley 1994; Statistical Sciences 1995) and the spherical model was fit to the smoothed estimates using non-linear least squares estimation techniques (Statistical Sciences 1994, documentation for nlminb function). Kernel smoothing was carried out by the S-Plus function ksmooth (Statistical Sciences, 1995).

Step three of the spatial regression analysis used the estimated variance-covariance matrix derived from the variance model computed in step two as a weight matrix to re-compute coefficients, standard errors, and p-values obtained at step one. This weighted regression step is described next. Assume X was the original design matrix used in the regression model at step one which contained indicator variables and/or polynomials in distance. Assume Y was the vector of responses, and V was the estimated variance-covariance matrix obtained at step two. The re-computed vector of coefficients, $\hat{\beta}$, and variance was,

$$\hat{\beta} = (X'V^{-1}X)^{-1}X'V^{-1}Y$$
$$\mathrm{var}(\hat{\beta}) = (X'V^{-1}X)^{-1}.$$

Significance of an element in $\hat{\beta}$ was assessed by comparing the ratio of the element to its standard error to a (Student's) T distribution having *n*-*p* degrees of freedom (*n* was total number of observations, *p* was the number of columns in X). This test is commonly referred to as a Wald t-test (Venables and Ripley 1994).

Appendix C4. Assessment of Erosion and Sedimentation in Class III Watercourses: A Retrospective Study

CONTENTS

Figures

Tables

C4.1 INTRODUCTION

California forest practice rules during the period of this study (1992-1998) required that Class III watercourses (typically first order streams that do not support aquatic life) be delineated as equipment exclusion zones and that ground disturbance be minimized, but they did not require retention of existing forest canopy. Concerns have been raised that complete removal of trees from Class IIIs will result in destabilizing these headwater areas resulting in an upslope extension of the channel and increased risk of shallow rapid landslides. The mechanisms that could trigger these potential effects may not be fully mitigated by the existing forest practice regulations: loss of root strength in the soil column that could increase mass wasting, decrease bank stability and increased incident precipitation and storm runoff that could increase mass wasting and fluvial erosion processes in Class III watercourses. There is some evidence suggesting the latter from Caspar Creek (Lewis 1998). The net effect is that there could be significant increases in sediment production from watercourses even though Class I and II watercourses may have ample buffer retention. Because the majority of a channel network is made up of the first order channels, the overall impact of destabilized Class IIIs may be quite large even though increased sediment delivery in any given Class III is small. There is also the concern that if a debris torrent is triggered from one of these Class III areas, there will be no opportunity for delivering LWD into the channel below if no trees are retained in the uppermost reaches of these watercourses. The role of LWD in erosion and sedimentation processes in Class III channels is also potentially significant. LWD provides sediment storage sites, controls channel grade by preventing channel bed erosion, and deflects and concentrates stream flow thereby both protecting banks from erosion and magnifying fluvial bank erosion processes.

However, there are few empirical data available to assess the magnitude of these potential problems in northern California forestlands. To begin with, the proportion of first order streams that are designated as Class IIIs in current timber harvest plans (THPs) has not been quantified. Since any headwater channel that is judged to support “aquatic life” must be classified as a Class II, an unknown but increasingly higher proportion of first order channels are receiving protection as Class II watercourses. Although the forest practice rules have not changed, this trend has occurred primarily due to the southern torrent salamander. The transition began at Simpson in 1992 when its biological staff began demonstrating to the foresters that many first order channels supported torrent salamanders. The rest of the California north coast region followed suit when the torrent salamander was petitioned to be state listed in 1995. The species was not listed, but a mandatory training program to learn to identify the habitat of the salamander was instituted for all registered professional foresters that wished to submit THPs within the range of the species. Region wide, this had a dramatic effect on watercourse classification and in some areas there are few Class IIIs at the head of a Class II watercourse. The channel begins as a Class II, because it has intermittent habitat for torrent salamanders.

In addition to not knowing the extent of Class IIIs in THPs, there are no data on the changes that result in these watercourses following timber harvest. In particular, it is important to know the degree to which channel extension or head-cutting is occurring along with some quantification of the amount of sediment that is being generated from the existing channel banks due to bank erosion or channel scour. It is also important to know if destabilized Class IIIs are contributing to increases in shallow rapid landslides.

Past protection of Class III watercourses during timber harvest was a combination of both compliance and effectiveness of the forest practice rules as they were implemented through the THP process. Therefore, completed THPs were used as the basis for the selection and assessment of the condition of Class IIIs. A retrospective approach was used to randomly select completed THPs from across the ownership, and quantify the number and extent of both Class II and III watercourses that were identified by the RPF prior to harvesting. The selected watercourses were visited, and data were gathered on the physical condition of the Class III watercourse. Since this was a retrospective study and it was not possible to utilize controls, subtle changes in Class IIIs following timber harvest could not be quantified. Rather the objective was to assess the extent to which major changes occurred in Class IIIs that were responsible for substantial increases in management related sediment production. Specifically, the objectives were to: 1) collect data to characterize and describe Class III channels following clearcut harvest under the past Forest Practice Rules and Simpson's spotted owl HCP; and 2) explore potential relationships between key response variables that correlate strongly with sediment production (e.g. bank erosion and number of landslides) and other important stream variables. There also was the opportunity to compare pre-harvest characteristics of Class III watercourses that were assessed as part of the Little River monitoring study to a sub-set of the streams from the retrospective study that were located within or adjacent to the Little River HPA. Unfortunately, this was not a pre and post-treatment assessment of the same streams, but it did allow for general comparisons of characteristics before and after harvest.

It is important to reiterate that this was a retrospective study and comparisons to untreated control streams (i.e., unharvested Class III watercourses in advanced second growth or virgin old growth) were not possible. Therefore, conclusions from the study were limited in scope. The primary objectives were to provide a description of key variables of Class III watercourses sampled and quantify gross changes that might have occurred following clearcut timber harvesting. A stratified random sampling design was followed, so it was appropriate to draw inferences to the total sampling universe. However, since the sampling was tied to recent harvesting (1992-1998), the inferences need to be restricted to that portion of the total ownership that has experienced significant harvesting in recent years. Despite these limitations, the study has significant value simply because there is so little known about the characteristics of Class III watercourses or the impact of timber harvest on them.

C4.2 METHODS

C4.2.1 Site Selection

The Class III retrospective survey was conducted across all of Simpson's property with the exception of some of outlying areas (e.g. South Fork Mountain, Supply and Goose Creeks) where logistical constraints would have drastically reduced the efficiency of the project. All of Simpson's ownership within the Mad River was included in the study, including lands outside the HPAs. A stratified random sampling of Class III watercourses was employed throughout the remaining tracts (management units) of the ownership. All Class IIIs in completed THPs from 1992-1998 were classified as either a "run-through" or "within" (Figure C4-1). A "run-through" refers to a Class III watercourse where the beginning of the channel is outside the harvest unit, but if the channel was initiated within the boundaries of the harvest unit, it was designated "within." The number of Class IIIs was then randomly sampled at frequency of 2:9 within streams and 1:9 run-throughs. The sampling was weighted toward within streams in order to focus on channel extension of Class IIIs. The original THP map for each selected unit was reviewed as well as aerial photos to ensure that selected units were true clearcuts. Units that had non-clearcut prescriptions (i.e. seed tree removal, selection harvest or commercial thinning) were not included in the sample. In addition, a minimum apparent channel length of 200 feet on the THP map was required to be included in the sample. However, in the field, the actual channels varied from minimums of 113 and 58 feet, and maximums of 1146 and 1295 feet for run-through and within channels, respectively.

Figure 1. “Within” versus “Run-through” Channels

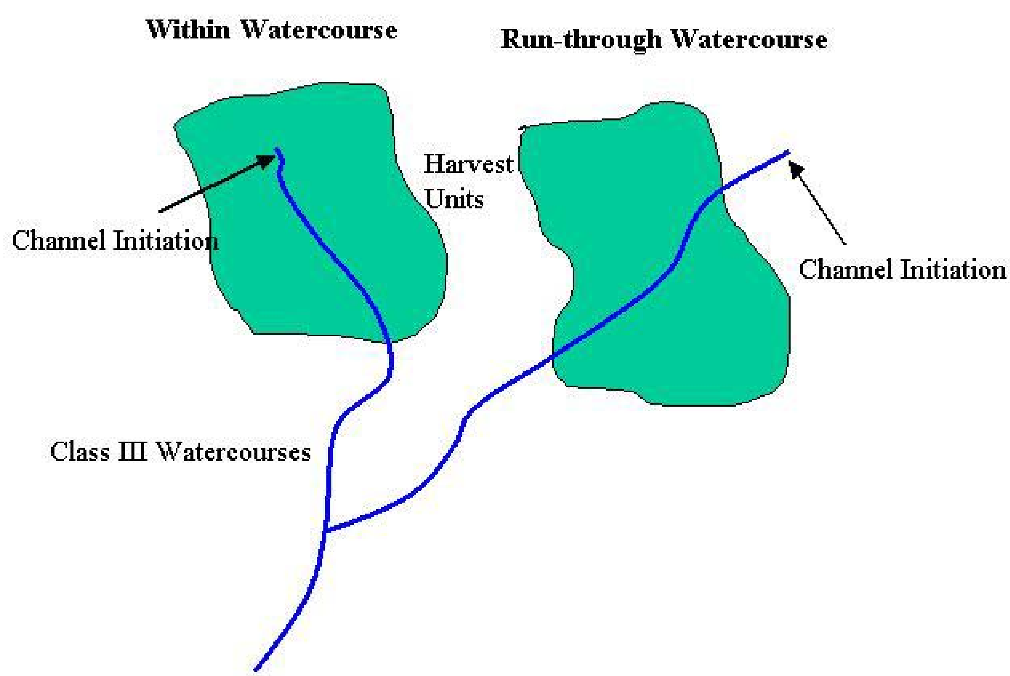


Figure C4-1. "Within" versus "run-through" channels.

Bedrock geology underlying each study site was determined based on USGS geologic maps and characterized as “consolidated” or “unconsolidated” by Oscar Huber (retired geologist, CDF). Consolidated bedrock geology included the Franciscan series (undifferentiated, melange, sandstone with siltstone, rocks and schist), Galice and ultramafic rocks. The undifferentiated Wildcat Group, Hookton and Falor Formations, Alluvium, Quaternary marine terraces and coastal plain sediment were considered unconsolidated bedrock geology.

THPs were not selected before 1992, because of a property-wide shift in the designation of Class II versus III watercourses. Prior to that year, many small intermittent channels were classified as Class IIIs that would have been designated a Class II after 1992. (This shift resulted from the recognition of southern torrent salamander habitat as noted above.) THPs were not selected after 1998 to insure that Class IIIs had experienced at least one winter of storms.

C4.2.2 Field Protocol

Before going into the field, Simpson delineated the Class III drainage as mapped on the original THP map. Assessment of the watercourse began at the lowest point on the channel within the THP unit. If the lowest end was within a riparian protection zone or habitat retention area (HRA), then Simpson began the channel measurements at the uppermost edge of the standing timber. Measurements were taken systematically up the channel at 10-foot intervals based on a random start within the first 10-foot interval. At each 10-foot sampling interval, the active channel width, maximum depth, was measured, and it was determined if there was evidence of an exposed active channel (channel bed exposed by fluvial processes). The linear length of exposed bank within 15 feet of the channel on both banks also was measured. If the exposed bank was part of an earth flow or slide, the entire limit of the exposed ground was measured. Game trails and animal burrows were not included in measurements of exposed banks, but their occurrences were noted. Watershed drainage area at the downstream end of study sites was also determined.

At every 50-foot interval, the bank angle perpendicular to the channel on the left and right banks was measured. At every 100-foot interval, the mean understory vegetation height was measured, and percent overstory canopy closure was determined using a densiometer. The channel gradient was measured with a clinometer at the beginning of the layout and at all major gradient breaks in slope throughout the remaining channel layout. Large woody debris (LWD) greater than 6 inch diameter with no minimum length requirement was measured (length and average diameter) wherever it occurred throughout the channel. It was recorded if the LWD was hardwood or conifer (if not clear which, “hardwood” was recorded, which provides a more conservative estimate of the longevity of the LWD), and it was noted if the LWD was acting as a control point. (A control point was any in-channel feature retaining sediment and/or preventing head-cutting.) The location and type of all other control points (roots, boulders, bedrock, etc.) were recorded in addition to LWD, and the size (with the exception of bedrock) and the vertical drop below the control point were measured. The area and location of any significant (generally greater than 3 feet in length) bank erosion were measured, and the predominant channel substrate, presence and flow of water, changes in predominant vegetation, and the occurrence of any aquatic vertebrates were noted.

Simpson photo documented the site, looking upstream at the beginning of the layout, both directions in the middle, and downstream at the end. In addition, any major gradient breaks in the channel that precluded visibility, any significant mass wasting, large scours, or other major features that affected the channel were photo documented.

The in-channel survey was continued until the Class III channel ended at a headwall, or at the harvest unit boundary, if the channel was a run-through. Simpson assessed the channel for evidence of head cutting by looking for evidence of recent scour or bank erosion at the initiation of the channel. In addition, Simpson compared the mapped initiation of the channel from the THP map relative to the current initiation of the channel. Simpson surveyed the associated road system within the sub-basin and sketched the drainage area onto a topographic map. Simpson recorded any stream piracy or diversions associated with the road system and include it in the drainage area. On the topographic map, Simpson recorded road failures, inner gorge slides or other larger scale sediment delivery features within the sub-basin. Data collected are summarized in Table C4-1.

An ongoing monitoring program in the Little River watershed utilizing a BACI (before-after-control-impact) experimental design allowed for a partial comparison of pre-treatment (advanced second growth with no recent timber harvesting activities) Class III watercourses to some of the post-treatment streams from this retrospective study. The same protocols described above were applied to the pre-treatment assessment of 26 Class III watercourses in the Little River, which were compared to 29 post-treatment (retrospective) watercourses located within or adjacent to the Little River watershed.

Table C4-1. Summary of continuous and categorical variables measured on surveyed Class III watercourses.[1]

Continuous	Categorical
Width and depth of active channel	Exposed active channel
Length of surveyed channel	Exposed banks
Channel gradient	Channel initiation (run-through vs. within)
Bank slope	Bedrock geology
Number of years (winters) since harvest	Type of harvest (tractor vs. cable)
Drainage area above the channel	Burn history
Height of ground vegetation	
Total canopy closure	
LWD: #, length, diameter and volume	
Bank erosion: number and area	
Slides: number and area	

Note
1 Exposed active channel and exposed banks were assessed as a categorical variable at each 10-foot sample interval, but summarized as a percentage of the total samples intervals measured. Response variables are highlighted.

C4.2.3 Data Analysis

Simpson selected four variables that best reflected potential sediment delivery to the lower portions of a watershed as the primary response variables for analysis. These variables were cross-sectional area (product of the active channel depth and width measurement), percent exposed active channel, frequency of sites with bank erosion and number of slides relative to channel length. Forward stepwise regression was performed using function step.glm (generalized linear model) in the computer program S-Plus. Step.glm added variables from the pool of potential explanatory (independent) variables, one at a time, until the model AIC (Akaike's Information Criterion) would not decrease if another variable was entered. The variable chosen for inclusion at each step was the variable that provided the greatest improvement of the modeled likelihood among variables that were not yet in the model. This addition amounted to adding the variable at each step with the most significant likelihood score statistic. Significance of terms in the final model was assessed using an approximate F-test based on the drop-in-deviance likelihood ratio. GLM R^2 values were calculated, which are equivalent in interpretation (amount of the variation in the dependent variable explained by the independent variable) to R^2 values from regression based on a normal distribution.

Response variables 'bank erosion' and 'number of slides' were modeled using a Poisson regression that included an "offset" to relate the count to the length of sampled stream segment. 'Percent exposed active channel' was modeled using binomial regression. 'Cross-sectional area' of the channel was modeled using Normal regression theory, but was first transformed by computing the natural log of the variable. To meet assumptions of normality, cross-sectional area and percent exposed active channel were also transformed (natural log for area and square root for percent scour) before performing t-tests or analysis of covariance (ANCOVA). For stepwise regression, geology was treated as a categorical variable with two levels: 'unconsolidated' and 'consolidated'.

C4.3 RESULTS

There were 899 THP units operated within the study area from 1992-1998. To find units that meet the criteria of having a Class III watercourse located within a clearcut block, 553 harvest units were initially selected using a stratified random sampling design. From these units, 110 Class III watercourses were identified that appeared to have met the criteria for inclusion in the survey. On field inspection, some of these Class III watercourses had to be to be eliminated (e.g. trees were retained in the Class III to meet habitat retention guidelines under Simpson's spotted owl HCP), which resulted in 100 channels ultimately being assessed across Simpson's ownership (Figure C4-2). Forty-seven of the channels were run-throughs (channel initiated outside the harvest unit) and 53 were within channels (initiated within the harvest unit). Because the selection of Class IIIs was dependent on recent (1992-1998) harvesting activities, the number of channels assessed per HPA was not necessarily proportional to the area of the HPA. In addition, the number of Class III watercourses associated with each unit varied across the study area. The majority of harvest units within most of the study area had no or only one Class III watercourse within or adjacent to the unit, while the majority of units had multiple Class III watercourses in the two most southerly HPAs (Table C4-2). The greatest number of channels (25) was assessed in the Mad River HPA, followed by Smith River (20), North Fork Mad River (14), Little River (13), Humboldt Bay (11), Eel River (6), the area in the Mad River that is outside the Plan Area (3), and two each for Redwood Creek, Coastal Lagoons, Coastal and Interior Klamath HPAs. Of the 100 watercourses selected to be assessed as Class IIIs based on the original THP, 16 were judged to have at least a small portion that was a Class II watercourse based on Simpson's current more thorough and conservative approach to evaluating streams for the presence of headwater amphibians or their habitat.

The mean length and cross-section area of run-through channels were greater than within channels (Table C4-3), as might be expected because they were generally lower in the watershed and had greater drainage area. However, the mean cross-sectional areas were not significantly different ($t = 1.81$, d.f. = 96, $P = 0.073$) between run-through and within channels.

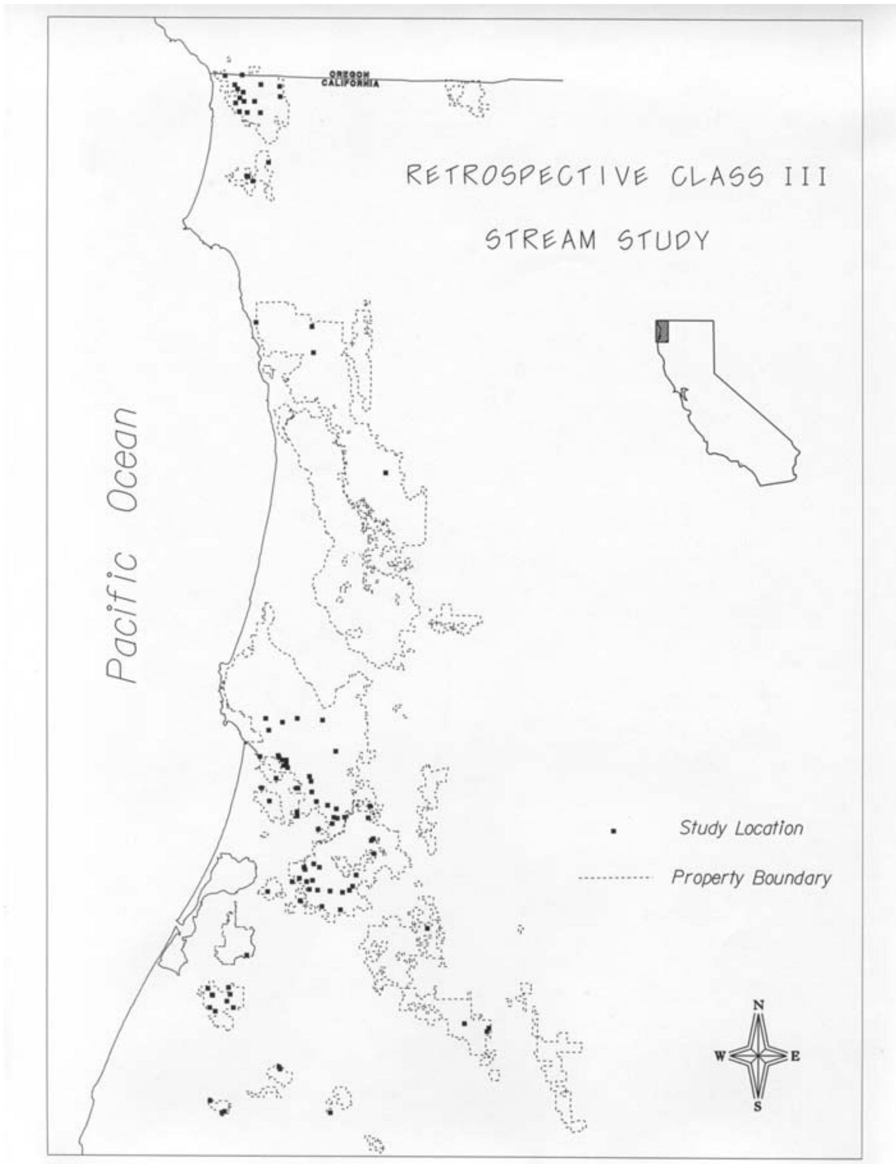


Figure C4-2. Location of Class III channels assessed on Simpson's ownership.

Table C4-2. Summary of harvest units operated from 1992-1998 within each Hydrographic Planning Area and the number of units with no or only one Class III watercourse within or adjacent to the harvest unit.[1]

Hydrographic Planning Area	Harvest Units	Percentage with no Class III	Percentage with one Class III
Smith River	141	36.2	24.1
Blue Creek	53	34.0	35.8
Coastal Klamath	152	38.1	31.6
Interior Klamath	145	39.3	27.6
Redwood Creek	51	62.7	21.6
Coastal Lagoon	11	27.3	27.3
Little River	38[2]	5.3[2]	15.8[2]
NF Mad River	61	23.0	29.5
Mad River	126	17.5	26.2
Humboldt Bay	42	14.3	16.7
Eel River	42	11.9	16.7
Area outside the Plan Area	64	43.8	28.1
Total	899	32.0	26.2

Notes

1 Summary includes all units whether or not there were any type of watercourses associated with the harvest unit.

2 Harvest units in this HPA were developed and operated by a previous owner.

Table C4-3. Summary of Class III watercourse characteristics.[1]

Variables	Run-through		Within		Total	
	N	mean (SE)	N	mean (SE)	N	mean (SE)
Drainage area (acres)	47	10.5 (2.48)	53	5.6 (0.66)	100	7.9 (1.24)
Channel length (ft)	47	451.5 (31.62)	53	346.1 (34.46)	100	395.6 (24.02)
Channel width (ft)	47	2.55 (0.147)	53	2.69 (0.234)	100	2.62 (0.140)
Channel depth (ft)	47	0.33 (0.029)	53	0.25 (0.002)	100	0.29 (0.019)
X-section area (ft^2)	47	0.96 (0.146)	53	0.67 (0.083)	100	0.81 (0.083)
Channel gradient (%)	47	31.5 (1.79)	53	35.2 (1.81)	100	33.4 (1.28)
Bank slope (%)	47	47.4 (2.481)	53	43.0 (2.61)	100	45.1 (1.81)
Exposed bank (%)	47	0.66 (0.113)	53	1.00 (0.343)	100	0.84 (0.189)

Note

1 Cross-sectional area of the channel represents the product of the active channel depth and width measurement. RT = run-through channels and Within = within channels.

Simpson conducted a forward stepwise regression analysis to determine which of the independent variables explained variation in mean channel cross-sectional area. The first variable to enter the model was drainage *area* (F = 20.80, d.f. = 1,92, P < 0.001, improvement R^2 = 0.237, model coefficient = 0.044), followed by underlying bedrock geology (F = 8.23, d.f. = 1,92, P = 0.005, improvement R^2 = 0.061, model coefficient = -0.455) indicating greater channel width in unconsolidated bedrock geology), stream gradient (F = 9.16, d.f. = 1,92, P = 0.003, improvement R^2 = 0.051, model coefficient = -0.016) and number of rock controls (F = 3.93, d.f. = 1,92, P = 0.051, improvement R^2 = 0.027, model coefficient = 0.937). The full model explained 37.5% of the variation in cross-sectional area of channels among streams. The cross-sectional area of channels with consolidated underlying geologic materials was significantly less when corrected for drainage area than channels in unconsolidated geology (consolidated area: n = 74, $\bar{x}$ =

0.61, SE = 0.048; unconsolidated area: n = 24, $\bar{x}$ = 1.41, SE = 0.273; ANCOVA: F = 13.52, d.f. = 1,95, P < 0.001). This relationship between drainage area and cross-sectional area of the active channel is illustrated in Figure C4-3.

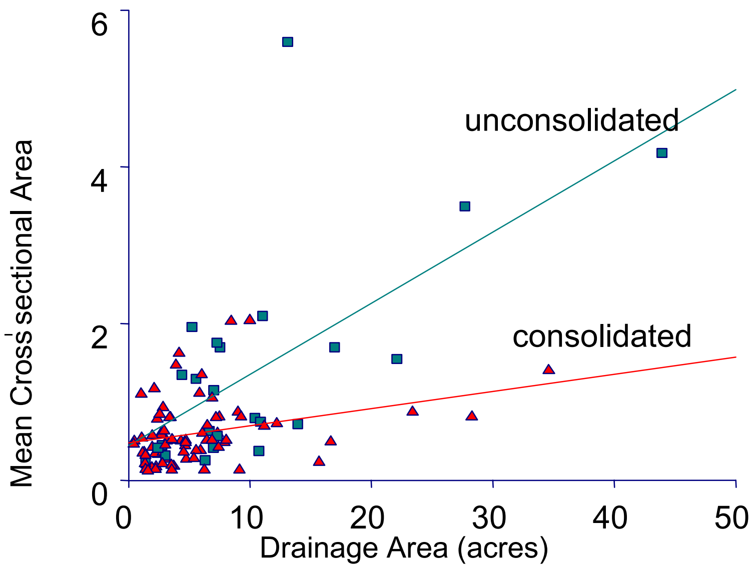


Figure C4-3. Mean cross sectional area (ft^2) of channels versus drainage area in consolidated and unconsolidated bedrock geology. Triangles represent consolidated geology and squares unconsolidated geology. Regression equation for consolidated geology: Y = 0.477 + 0.022*drainage, R^2 = 0.096; unconsolidated geology: Y = 0.447 + 0.091*drainage, R^2 = 0.409.

Consistent with being higher in slope position, within channels had somewhat higher mean stream gradient ($\bar{x}$ = 35.2, SE = 1.82) compared to run-through channels ($\bar{x}$ = 31.5, SE = 1.79), although the differences were not statistically significant (t = 1.44, d.f. = 98, P = 0.153). In addition, the distribution of stream gradients indicated that both types of Class III channels had a similar wide range of stream gradients (Figure C4-4). There was no difference in channel gradient or bank slope between consolidated and unconsolidated bedrock geologies with drainage area as the covariate (ANCOVA: stream gradient – F = 0.51, d.f. = 1,97, P = 0.478; bank slope – F= 1.02, d.f. = 1,97, P = 0.315). The mean number of LWD pieces per 100 feet of Class III channel was 4.80 (SE = 0.318), while mean volume was 226.6 (SE = 25.02) cubic feet per 100 feet of channel. However, the distribution in the number and volume of LWD (Figure C4-5) indicated that most channels had relatively low amounts with a small proportion of channels having high amounts of LWD. Of the LWD associated with these channels, 85.0% (SE = 2.59) was determined to be conifer.

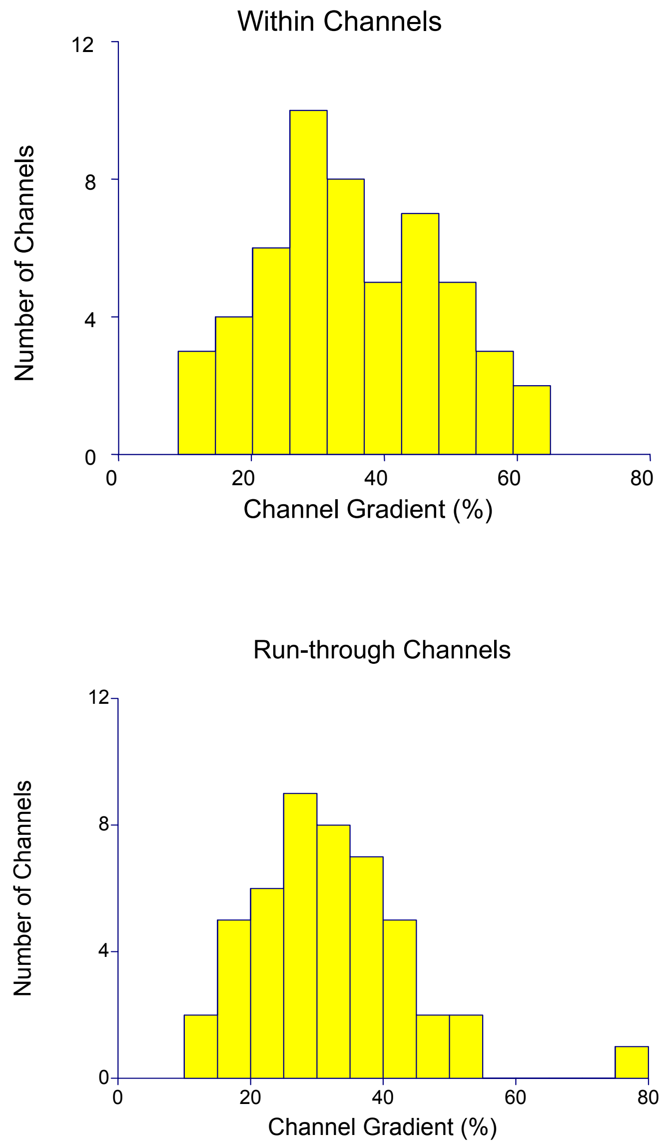
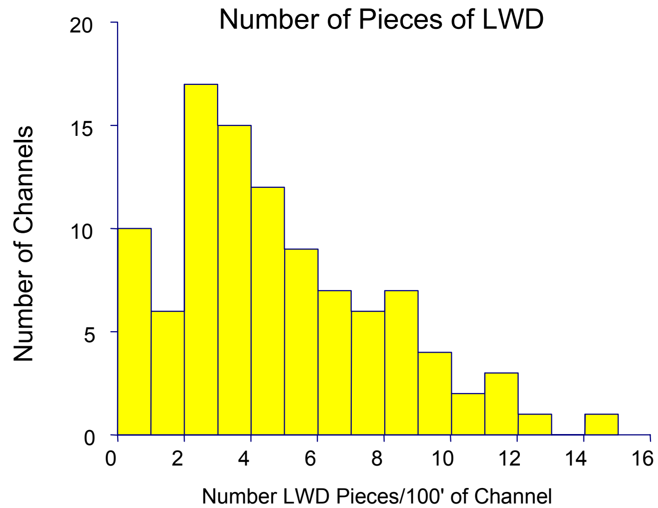


Figure C4-4. Distribution of stream gradients for "within" and "run-through" Class III watercourses.



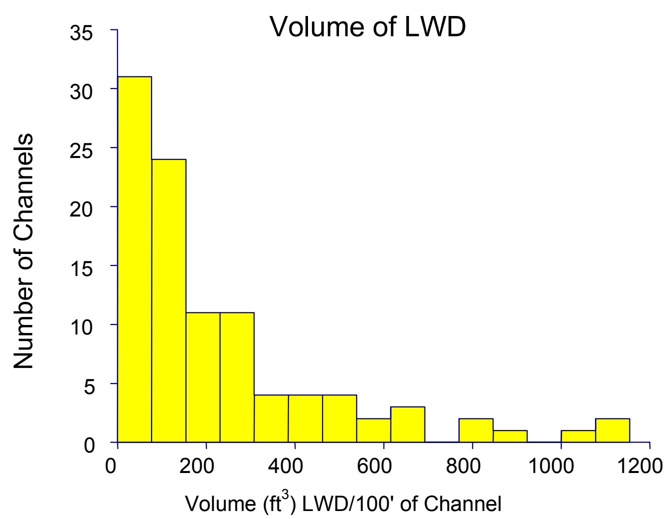


Figure C4-5. Distribution among surveyed Class III watercourses of the number and volume of LWD per 100 feet of channel.

The mean number of total control points per 100 feet of Class III channel was 0.93 (SE = 0.121) with most (>75%) of the controls being formed by LWD (Figure C4-6). Roots and rocks (large rock or bedrock) were particularly rare in forming control points in these Class III channels.

Mean percent exposed active channel (EAC – percent of 10-foot sample intervals with evidence of an exposed active channel) for within and run-through channels was 23.3 (SE = 2.88) and 24.6 (SE = 2.55), respectively. The difference was not statistically different (t = 1.097, d.f. = 97, P = 0.275) so the two channel types were combined for additional analysis. The distribution of mean percent EAC channel (Figure C4-7) was highly skewed to the left with most channels showing little or no EAC. Simpson conducted a forward stepwise regression to further explore the relationship between EAC and other independent variables measured. The first variable to enter the model was the total number of channel control points (F = 41.427, d.f. = 1,93, P < 0.001, improvement R^2 = 0.232, model coefficient = 0.474), followed by mean height of riparian ground vegetation (F = 6.75, d.f. = 1,93, P = 0.011, improvement R^2 = 0.047, model coefficient = 0.220), and underlying bedrock geology (F = 5.33, d.f. = 1,93, P = 0.023, improvement R^2 = 0.036, model coefficient = -0.498). The full model explained 31.5% of the variation in EAC of channels among streams. Simpson expected channel scour to be positively correlated with stream gradient, but it did not enter the stepwise regression model. To graphically explore the relationship, Simpson produced a scatter plot of EAC and gradient (Figure C4-8), which further illustrates the lack of correlation between these two variables.

The preponderance of LWD as channel controls and the apparent positive correlation between channel controls and EAC prompted us to graphically look at the relationship between LWD controls and EAC (Figure C4-9). Although there is considerable variation, it is apparent that there was a positive relationship between the number of LWD controls and percent EAC.

Sites along the banks of the Class III channels with bare mineral soil that were the result of undercutting or sloughing were termed bank erosion. Relative to the axis of the channel, these sites were longer (mean length = 9.6 feet, SE = 0.81) than wide (mean width = 5.3 feet, SE = 0.47). Among the 100 channels surveyed, there were 107 total sites with bank erosion. Most sites (57%) had no bank erosion, while a few streams had relatively frequent bank erosion (Figure C4-10). Simpson conducted a forward stepwise regression to further explore the relationship between bank erosion and other independent variables measured. The only variables to enter the model were underlying bedrock geology (F = 8.05, d.f. = 1,93, P = 0.006 improvement GLM R^2 = 0.258, model coefficient = -0.787) (greater bank erosion in unconsolidated geology), followed by total canopy closure (F = 7.75, d.f. = 1,93, P = 0.007, improvement GLM R^2 = 0.086, model coefficient = -0.030) (less bank erosion with greater canopy closure) and volume of LWD (F = 3.21, d.f. = 1,93, P = 0.077, improvement GLM R^2 = 0.026, model coefficient = 0.001) (greater bank erosion with more LWD). The full model explained 37.1% of the variation in bank erosion among streams.

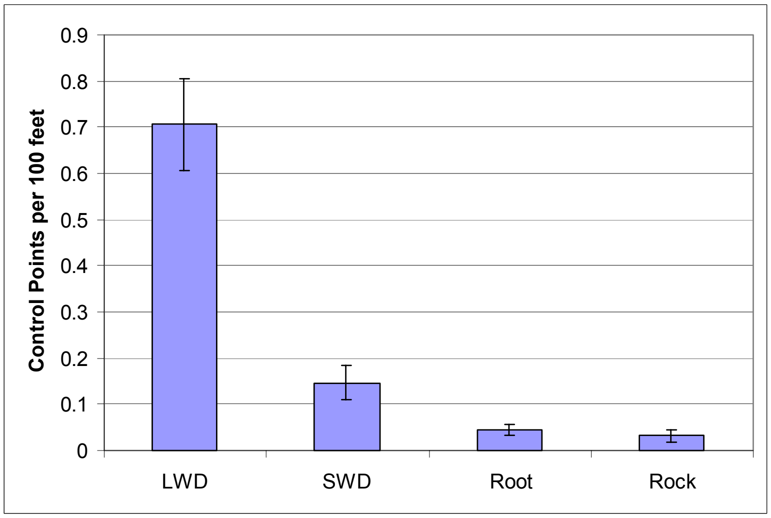


Figure C4-6. **Mean number of control points per 100 feet of channel with standard error bars. LWD = control points formed from large woody debris (>6 inches), SWD = control points formed from collections of small woody debris (<6 inches), root = control points formed by tree roots and rock = control points formed from large rocks or bedrock.**

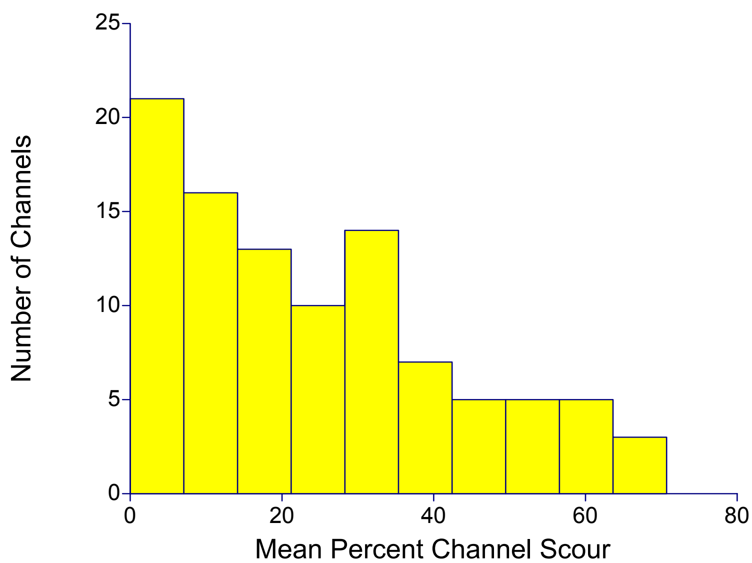


Figure C4-7. **Distribution of mean percent exposed active channel (EAC) among surveyed Class III watercourses.**

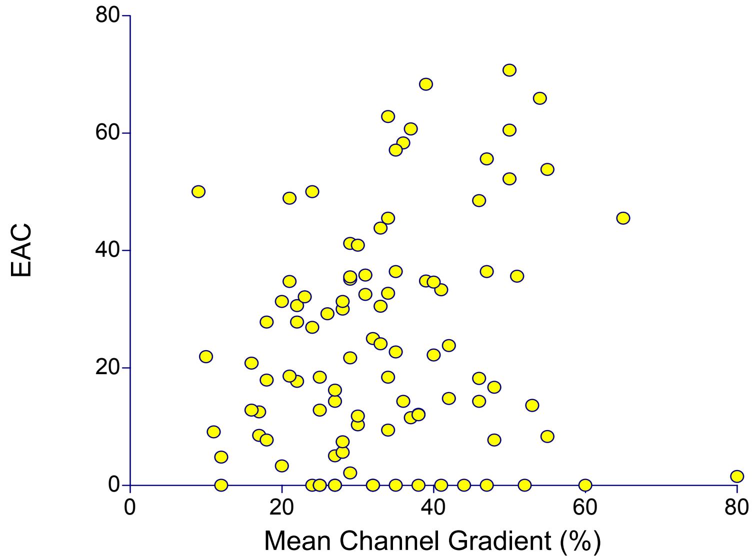


Figure C4-8. Mean channel gradient versus mean percent exposed active channel (EAC) for individual watercourses.

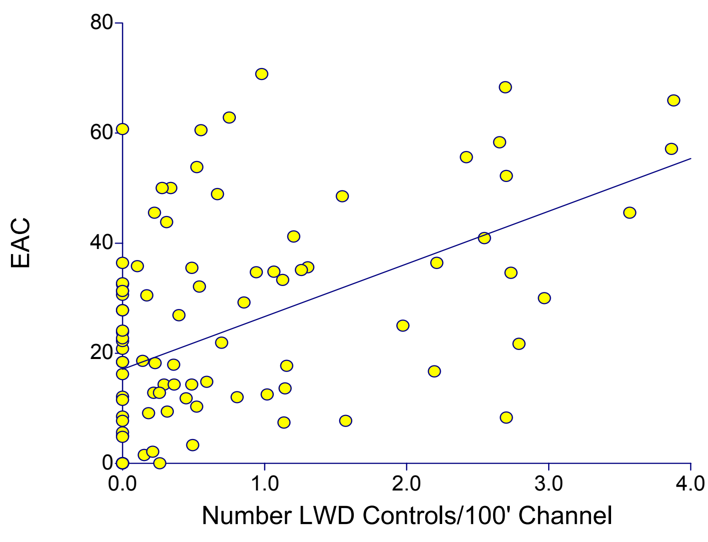


Figure C4-9. Number of LWD control points per 100 feet of channel versus mean percent exposed active channel. Trend line is the least squares regression line. Regression equation: Y = 0.010 + 0.026*EAC, R^2 = 0.245.

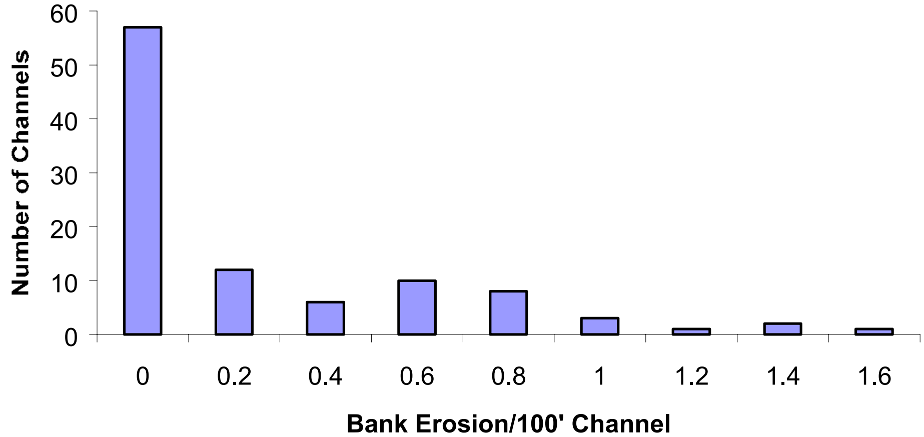


Figure C4-10. Distribution of sites with bank erosion among surveyed Class III watercourses. Each value in the figure represents the mean value from a given stream.

Twenty-four shallow rapid landslides were identified while surveying the 100 Class III watercourses. One slide was associated with a road and not included in further analysis, while all of the rest of the slides were associated with an inner gorge or steep streamside slope. There were no debris torrents associated with any of the channels surveyed. The distribution of landslides among surveyed channels (Figure C4-11) indicated that most (85%) had no slides with a few of the channels accounting for the majority of the slides. The cumulative frequency distribution of the length (maximum head scarp distance) of the landslides indicated that 80% of the slides were located within less than 20 feet of the channel (Figure C4-12). The results of a forward stepwise regression analysis of the relationship between landslides (number/100 feet of channel) and other independent variables measured indicated that the first variable to enter the models was stream gradient (F = 7.17, d.f. = 1,91, P = 0. 009, improvement GLM R^2 = 0.350, model coefficient = 0.027). This was followed by mean height of ground vegetation (F = 30.15, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.093, model coefficient = -1.128), mean bank slope (F = 25.74, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.072, model coefficient = 0.054), number of LWD controls (F = 14.56, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.051, model coefficient = 0.473) and years since harvest (F = 14.57, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.071, model coefficient = 0.322). The full model explained 63.6% of the variation in the number of slides among streams.

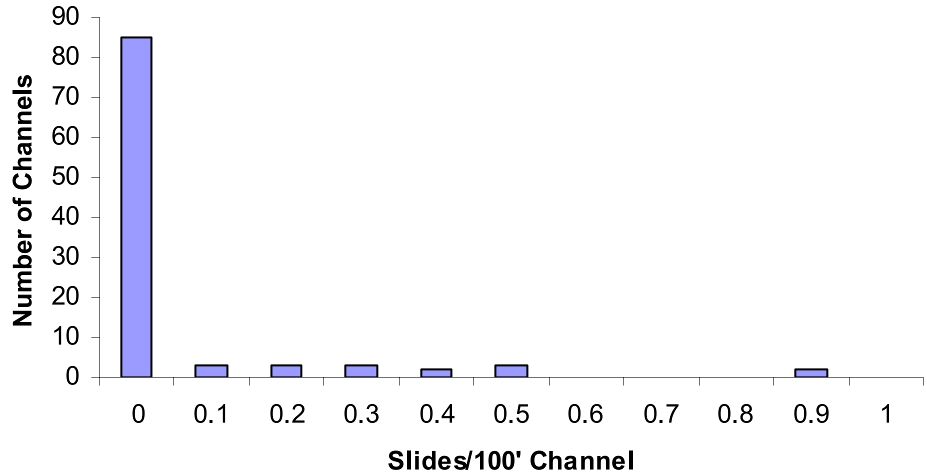


Figure C4-11. Distribution of landslides among surveyed class III watercourses. Each value in the figure represents the mean value from a given stream.

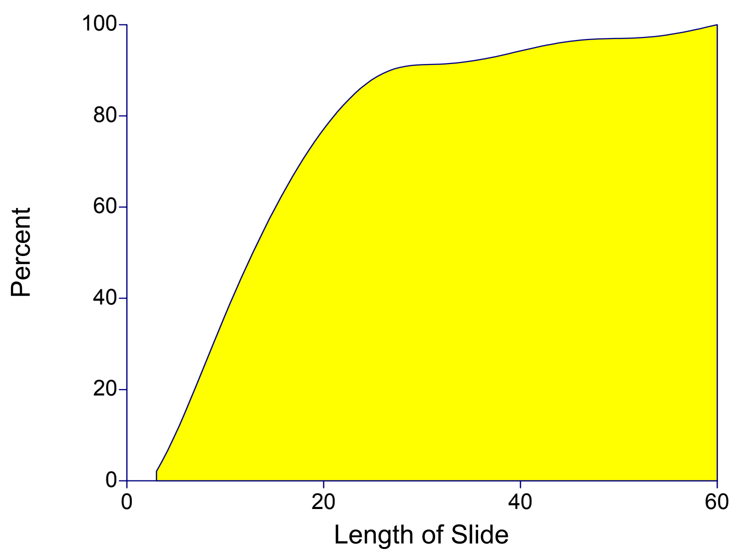


Figure C4-12. Cumulative frequency distribution of the length (maximum head scarp distance) of 23 inner gorge or steep streamside slope landslides associated with surveyed Class III watercourses.

Bank erosion or slides at the initiation of Class III watercourses are evidence of head cutting or channel extension. In the 53 within channels where this could be assessed, the only channel extension or head cutting observed was due to runoff from roads. This occurred in both in within and run-through channels and was typically associated with improper road drainage. There was no direct evidence for head cutting or channel extension due to hillslope processes. There was also no evidence of channel extension based on the mapped initiation of the channel in the THP map, but these maps were not considered very precise.

C4.3.1 Comparisons with Pre-treatment Steams

There were 26 Class III watercourses that were assessed as part of the Little River monitoring program. These were compared to 29 Class III watercourses in or adjacent to the Little River HPA that were assessed as part of this retrospective study. Although these streams were spatially and temporally separated, most characteristics were similar (Table C4-4).

Using ANCOVA with drainage area as a covariate, cross-sectional area and percent EAC (square root transformed) for pre and post-treatment streams were not significantly different (Cross-sectional area: $F = 0.31$, d.f. = 1,49, $P = 0.583$; Percent scour: $F = 2.72$, d.f. = 1,52 $P = 0.105$).

Table C4-4. Summary of pre- and post-treatment Class III watercourse characteristics.

Variables	Pre-treatment		Post-treatment	
	N	mean (SE)	N	mean (SE)
Drainage area (acres)	26	8.0 (1.40)	29	8.7 (3.60)
Active channel length (ft)	26	374.9 (51.81)	29	405.2 (50.54)
Active channel width (ft)	24	2.05 (0.156)	28	2.42 (0.231)
Active channel depth (ft)	24	0.28 (0.024)	28	0.26 (0.106)
Cross-sectional area (ft^2)	24	0.567 (0.063)	28	0.617 (0.063)
Channel gradient (%)	24	28.5 (2.10)	29	30.4 (2.19)
Bank slope	26	16.8 (1.21)	29	21.8 (1.41)
Percent exposed active channel	26	15.0 (2.47)	29	27.7 (4.26)
Bank erosion sites/100 ft	26	0.46 (0.127)	29	0.33 (0.084)
Slides/100 ft	26	0.03 (0.033)	29	0.05 (0.034)

C4.4 DISCUSSION

C4.4.1 Limitations

The preceding data are retrospective in nature and do not provide comparisons to untreated control streams (i.e. unharvested Class III watersheds in advanced second growth or virgin old growth.) Therefore, it is important to identify the type of conclusions that one should expect to be able to draw from the data. Most of the data were descriptive in nature, which allowed us to create an “image” of the characteristics of Class III watercourses sampled. Simpson followed a stratified random sampling design, so it was appropriate to draw inferences to the total sampling universe. However, since the sampling was tied to recent harvesting (1992-1998), the inferences should be restricted to that portion of the total ownership that has experienced significant harvesting in recent years. In addition to descriptive characterizations of these watercourses, the objective was to assess the extent to which major changes occurred in Class IIIs that were responsible for substantial increases in management related sediment production. Caution must always be used when attempting to establish treatment effects or cause and effect relationships using a retrospective study design, but this type of study can be useful in identifying major or gross changes that occurred in Class III watercourses following clearcut timber harvest. It should be noted that most knowledge concerning the impact of timber harvest on geologic or hydrological processes comes from studies that were retrospective in nature. Before-after-control-impact (BACI) experiments (Skalski and Robson 1992; McDonald et al. 2000)) are the only approach to definitively assess the impact of a treatment on a response variable, and there have been few studies that utilize such an experimental approach on landscape level geologic or hydrologic processes.

Despite these limitations, the pre-treatment data set from the Little River HPA indicates that there were not gross differences between treated and untreated control streams for this HPA. This suggests that the results of the retrospective study may be interpreted with greater confidence than might otherwise be possible for a retrospective study. However, it is also recognized that conclusions from this one region may not hold for other HPAs with steeper topography or unconsolidated geology.

C4.4.2 Channel Size

An expected feature of these first order channels associated with Class III watercourses was that they were generally steep with an overall mean channel gradient of 33.4%. However, there was also considerable variation in gradient with a range from 9-80%. The size of the active channel was also quite small with a mean cross-sectional area (product of the channel depth and width measurement) of 0.81 ft^2, which can also be represented by a mean volume (volume of substrate that was transported to produce the existing channel) of 8.07 ft^3/100 feet of channel. In addition, this was a maximum estimate since Simpson only measured the maximum depth of the channel at each 10-foot sampling interval. It was also important to note the influence that geology had on the size of Class III channels. Channels with unconsolidated underlying geology (i.e. most of the channels in the Humboldt Bay and Eel River HPAs), had channels approximately twice the cross-sectional area than channels in consolidated geology. Qualitative field observations further support that Class III watercourses were much larger in areas with unconsolidated geology. The suggestion that underlying geology is an important determinant of the size and hydrologic response of Class III watercourses is generally consistent with findings from the Freshwater Watershed Analysis. In Freshwater, Class III channels draining the extremely weak Wildcat Group enlarged significantly following initial harvest, while Class III watercourses in Franciscan Formation sandstones did not. Recent harvest, however, did not appear to have dramatic effects on Class III channels in either of the major bedrock formations (Freshwater Watershed Analysis, Stream Channel Module).

C4.4.3 Exposed Active Channel and Control Points

Observations of EAC can be interpreted as an indicator of fluvial erosion or deposition. The fact that the percentage of the bed showing EAC was correlated with control points suggests that fluvial erosion and deposition processes as expressed by EAC were associated with control points

Simpson has no information by which to judge the relative merits of the number of control points per unit length of stream channel identified from this study. A high proportion of control points were made up of LWD, but it was of interest to note that even collections of small woody debris (SWD) could serve as control points in these channels. LWD and SWD in the channel created plunge pools that were responsible for streambed scour immediately below the control point. Evidence for this was provided by the fact that the total number of control points was the first variable to enter the regression model (with a positive coefficient) with EAC as the dependent variable. It is generally thought that although control points may cause scour in short waterfalls immediately below the control point, they prevent overall channel down-cutting. Control points may also correlate with the abundance of roughness elements that cause lateral scour. With a retrospective study, Simpson was not able to detect subtle changes in mean channel bed elevation, and apparently, there were sufficient control points in all streams to prevent any major “unraveling” of the channels.

One of the potential effects of harvest is an increase in peak storm runoff in Class III channels. The potential for channel bed erosion (down cutting) is limited by erosion-resistant elements of the channel bed. Roots and rocks (large rock or bedrock) rarely formed control points. LWD was the dominant channel element forming control points in these Class III channels. This is consistent with the conceptualization of Class III channels as ephemeral streams with low sediment transport capacity; these would be expected to be colluvial channels with weak fluvial sorting of hillslope material and relatively fine bed texture. The fact that EAC occurred in only 25% of 10-foot channel measurements also demonstrates that fluvial processes were spatially intermittent in these Class III channels. Consequently, few bedrock or coarse sediment exposures in the channel bed may be expected and proportionately more might be expected in Class II channels or larger Class III channels as suggested by the stepwise regression for channel cross-section area.

The abundance of LWD is significant in relation to the frequency of control points. Simpson has no data on the amount or distribution of LWD in Class II watercourses for comparison, but LWD surveys from the smallest Class I watercourses produced a mean of 5-6 pieces per 100 feet of channel in comparison to 4.8 for the Class III watercourses. However, these comparisons may not be appropriate, because the LWD surveys were conducted following different protocols. Simpson saw no evidence of transport of LWD in Class III watercourses. LWD was primarily composed of conifer in these Class III channels, which was generally not the case for Class I watercourses. However, this was consistent with the general observation of relatively few hardwoods such as red alder in upslope positions, while alder was a predominant component in many Class I watercourses.

Sites with bank erosion (bare mineral soil on the bank of the channel that was the result of undercutting or sloughing) were generally not large (about 50 ft^2) and did not occur in most channels. Relatively few channels were responsible for most of the bank erosion reported (Figure C4-10). Underlying bedrock geology (more bank erosion in unconsolidated geology), total canopy closure (less bank erosion with greater total canopy) and volume of LWD (more bank erosion with greater of amounts of LWD) were the only dependent variables that entered a stepwise regression analysis of bank erosion versus all appropriate independent variables measured. Increases in bank erosion in unconsolidated geology were expected, as was a decrease in bank erosion with increases in total canopy. (Canopy closure was coming from the regrowth of shrubs and trees since the streams were all in clearcuts with no tree retention.) However, the positive relationship between bank erosion and LWD was not as intuitive. Presumably, LWD directs flow into the banks of the channel thus increasing the sites with bank erosion.

C4.4.4 Slides and Debris Flows

There were relatively few total slides associated with these Class III watercourses and most of the slides occurred in just a few of the channels. In addition, the maximum head scarp distance for 80% of the slides was only 20 feet. It was also notable that there were no debris flows associated with any of these channels even though some had mean stream gradients as high as 80%. Number of LWD control points per 100 feet of channel (positive coefficient), stream gradient (positive coefficient), mean height of ground vegetation (negative coefficient), bank slope (positive coefficient), and number of years since harvest (positive coefficient) were the dependent variables that entered a stepwise regression analysis of the number of landslides versus all appropriate independent variables measured. The positive association between landslides and stream gradient as well as bank slope was predictable, given the importance of slope angle in slope stability. These two variables explained over 40% of the variation in landslides among streams and accounted for over two-thirds of the variation explained by the full regression model. A negative association with ground vegetation might be expected due to increased root strength, but this variable only explained 9% of the variation in the model. Positive correlation between years since harvest and landslide frequency may also be explained relative to root strength (initially declining following harvest), but the variable only explained 7% of the variation in the model making further speculation unwarranted. The potential reason for the positive association between inner gorge landslides and LWD control points was not so intuitive. Simpson believes that the apparent association was most likely created by landslides bringing LWD into the channel, and not that LWD in the channel had any direct effect on the rate of landslides. However, once again the variable contributed so little (5%) to explaining variation in the model that conclusions are unwarranted.

C4.5 CONCLUSIONS

This study suggests that there were no gross short-term effects of timber harvest on erosion in and near Class III channels for the period 1992-1998. There were few sites that experienced extensive bank erosion and less than 25% of 10-foot channel intervals contained exposed active channel (EAC). Furthermore, in the 100 sites examined, there were no debris flows. This is significant in that there were several potential triggering storms in 1996 and 1998 and there was above average (generally 120-140% of normal) total rainfall in all years except 1992 and 1994. In addition, 53% of the streams surveyed were harvested from 1996-1998 when the potential effects of increased incident precipitation (caused by reduced forest canopy) on soil erosion should have been greatest immediately following harvest. However, there is an expected lag effect of approximately 5 to 20 years associated with reduced root strength (Zeimer 1981; Sidle 1992), and a concomitant increased rate of landsliding (Sidle et al. 1985, p. 73-76). It may therefore be concluded that under the recent regime of harvest practices, Class III channels were not responding to harvest in the short-term by unraveling and causing the potential for major increases in sedimentation downstream. However, these results do not rule out the possibility that there were increases in sediment production from more subtle and chronic sources, or that a longer period of study might reveal changes not recognized in this investigation. The tendency for most of the sediment production from Class IIIs to be limited to a relatively few streams, particularly in regions with unconsolidated geology, suggest that effective mitigation can be provided by site specific geologic review where conditions warrant.

Since there were no controls, this study was not capable of assessing whether the observed erosion indicators differ significantly from either virgin old growth or advanced second growth forest stand conditions. In particular, it provides no clear evidence regarding whether predicted increases in peak runoff have induced significant increases in rates of fluvial erosion. This study was very similar to the retrospective study of the impact of timber harvest on water temperature in Class II watercourses (see Appendix C5), in that, potential short-term impacts of timber harvest were too subtle to be readily detected with a retrospective study design. That led to a BACI experimental design for Class II water temperature (see Appendix C5), and the BACI design has also recently been initiated for sediment production of Class III watercourses. The initial data set from the Little River HPA suggests that control-treatment comparisons may not show significant effects in that region.

The landslides recorded in this study that delivered sediment to Class III watercourses were associated with steeper stream gradients and bank slopes, shorter vegetation (a combination of silvicultural treatment, site preparation and time since harvest) greater time interval since harvest and more LWD in the channel. These findings were consistent with expectations regarding known triggering mechanisms for landslides (Sidle et al. 1985). The dominant predictor of landslide potential was the slope of the stream and its banks. Collectively it explained over 40% of the variation in landslides among streams and accounted for over two-thirds of the variation explained by the full regression model. However, it was much more difficult to determine potential management effects from this study. To begin with, the two variables that had management implications (height of ground vegetation and time since harvest) collectively only explained a small fraction of the variation of slides among streams. In addition, the height of ground vegetation could represent the influence of multiple management factors. Moreover, height of ground vegetation, had the opposite model coefficient as the direct measurement of time since harvest. It is likely that this retrospective study design is not capable of detecting management effects on landsliding. A more effective study design would include control streams, before-after data or both (BACI experiment).

Without reference or control streams for comparison, it was not possible to assess the quantity of LWD in Class III watercourses in the study area. However, LWD was the predominate element in the formation of channel bed grade control points. In addition, LWD was positively correlated with exposed active channels and bank erosion and, in some cases, with slides. Hence, there was evidence that LWD interacts with fluvial processes in Class III watercourses, but it was not possible to predict the impact of changes in the volume of LWD in Class III watercourses from this study.

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Appendix C5. Water Temperature Monitoring

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Figures

Tables

C5.1 GENERAL WATER TEMPERATURE MONITORING

C5.1.1 1994-1995 Water Temperature Monitoring Program

C5.1.1.1 Objectives

- Document diurnal and seasonal temperature fluctuations;
- Determine maxima and duration of daily peak water temperatures; and
- Identify stream reaches with temperatures that may exceed the thresholds of juvenile salmonids (especially coho salmon).

C5.1.1.2 Methods

Water temperatures were recorded with HOBO® (Onset Computer Corp.) temperature recorders. These devices automatically recorded temperatures at specified time intervals and were left in use for extended periods (up to six months). Two different models where deployed in 1994 and 1995; the HOBO® HTI –05/37°C with an accuracy of +/- 0.2°C and the HOBO®® HTI –37/46°C with an accuracy of +/-0.5°C. No attempt at calibration was made during the first two years of temperature monitoring. The manufacturer's specifications where well within the expected requirements of the temperature monitoring. Each thermograph is capable of recording approximately 1800 data points. The length of deployment depends on the selected recording interval. A recorder launched at a 0.8 hr interval will have to be downloaded and restarted within 45 days and thus runs a risk of missing a peak temperature while the recorder is out of the water. An interval of 1.2 hours records 20 temperatures per day and will last 90 days until the memory is full. The hottest three months of the year (July, August and September) were targeted as the summer monitoring window. To test the assumption that a 1.2 hour interval was enough to catch the entire diurnal range in 1994 three thermographs were launched at an interval of 0.6 hours and placed "piggy-back" on thermographs launched at 1.2 hour intervals. A third data set at 2.4 hours was created by deleting every other record in the 1.2 hr. data set. The 1.2 hour interval accurately represents average temperatures but has the potential to miss the absolute extremes by up to two or three tenths of a degree. Since this is within the accuracy of the thermograph (+/- 0.2°C) it was determined for practical reasons (i.e. deployment length of 90 days) that 1.2 hours was adequate.

The HOBO®s were typically deployed in the upper, middle and lower reaches of the larger streams with fewer HOBO®s in smaller streams. Actual site selection often depended upon property ownership and access issues. In larger streams the lowest monitoring site in the watershed would frequently be just inside Simpson's property boundary. The placement of each HOBO® was in the thalweg of a riffle or the head of a pool where water was mixed (to avoid thermal gradients). The HOBO®s were started between mid- June and early July and recorded temperatures throughout the summer months. They were removed between late September to early November. Time intervals of either 1.2 or 0.8 hours were used to accurately capture both diurnal temperature fluctuations and daily maximum temperatures. During the summer of 1994, 40 HOBO® temperature recorders were placed in fish bearing stream reaches distributed throughout Simpson's California property in areas that reflect a wide variety of stream conditions. In 1995, 28 Class I reaches where monitored (Table C5-1).

Table C5-1. Watersheds and number of reaches in 1994-1995 temperature monitoring program.

Watershed	No. of Reaches Monitored in 1994	No. of Reaches Monitored in 1995
South Fork Winchuck River	2	1
Rowdy Creek	2	1
South Fork Rowdy	2	0
Dominie Creek	2	0
Wilson Creek	3	1
Hunter Creek	2	2
Turwar Creek	3	3
McGarvey Creek	2	0
Blue Creek	1	1
Potato Patch Creek	1	1
West Fork Blue Creek	2	1
Ah Pah Creek	0	2
Bear Creek	1	3
Tectah	0	2
Tully	0	1
Roach	0	1
Pecwan Creek	1	3
Coyote Creek	1	0
Lindsay Creek	1	1
North Fork Mad River	3	1
Long Prairie Creek	1	0
Dry Creek	1	0
Cañon Creek	3	1
Maple Creek	1	0
Boulder Creek	1	1
Jacoby Creek	2	0
Salmon Creek	2	1

C5.1.1.3 Results

The 1994-95 monitoring effectively documented both diurnal (the difference between daily maximum and minimum temperatures) and seasonal temperature variations. Simpson calculated maximum weekly average temperatures (MWAT) for the 1994-1996 data as defined by the 1997 document Aquatic properly functioning condition matrix, a.k.a. the “Inter-Agency Matrix” (NMFS 1997). MWATs were generated by identifying the 7-day interval with the peak temperature and then calculating a mean temperature from all the data points recorded by the HOBO® device. For example, because Simpson has set their HOBO®s to record temperatures at 1.2 hour intervals (20 recordings for a 24-hour period), a MWAT would be the average of 140 data points for the hottest 7-day interval of the monitoring period. The MWAT for that creek was to be compared to established MWAT thresholds for a specific life stage and species. The MWAT threshold of 17.4°C for Coho summer rearing habitat was suggested in the “Inter-Agency Matrix” document. The temperature data indicated that on a Plan Area scale summer water temperatures were probably not limiting summer rearing habitat for salmonids. Of the 68 monitoring sites in 1994 and 95, 94% were below the suggested MWAT threshold of 17.4°C. The four sites that exceeded the MWAT of 17.4°C were all large order streams with watersheds more than 15,000 acres upstream of the recorder.

(See Tables C5-2 through C5-12 for a complete summary of Simpson's Class I and Class II summer temperature monitoring). Simpson believes that the single MWAT threshold value failed to account for natural variations in water temperature due to geographic location and drainage area of the monitored sub-basin. Also, depending on the method used to test the upper incipient lethal temperature of juvenile salmonids, a critical MWAT can range from 16.8°C to 18.4°C (Armour 1991; Thomas et al. 1986; Becker and Genoway 1979).

Following the 1994-5 temperature monitoring seasons improvements to the temperature monitoring protocol included collecting information relating to riparian canopy closure, channel morphology, flow and drainage area above the location of HOBO®s. Temperature monitoring was continued annually in selected stream reaches, either those that exhibit excessive temperatures or those of special biological significance (extremely diverse or abundant salmonid populations). In 1995 Simpson conducted some experimental Class II temperature monitoring which was formalized and expanded in 1996.

C5.1.2 Water Temperature Monitoring Program (1996 to the Present)

C5.1.2.1 Objectives

- Document the highest:

    (a) 7DMAVG (highest 7-day moving average of all recorded temperatures),

    (b) 7DMMX (highest 7-day moving average of the maximum daily temperatures),

    (c) seasonal temperature fluctuations for each site for both Class I and Class II watercourses.

- Identify stream reaches with temperatures that have the potential to exceed the monitoring thresholds relative to the drainage area above the monitoring site for both Class I and Class II watercourses. (To account for the relationship between water temperature and drainage area, water temperature was regressed on the square root of watershed drainage area at locations known to support populations of southern torrent salamanders, tailed frogs or coho salmon throughout Simpson's ownership in the HPAs.

- Directly assess the effects of timber harvest on water temperatures in Class II watercourses (Before, After, Control, Impact [BACI] experiments).

One of the major changes in the monitoring protocols occurred in the analysis of the data. Initially the analysis of the MWAT was a manual search through the data file to find the seasonal peak and then it was assumed that the encompassing seven-day period would provide the highest average temperature. This process was automated in 1996 with an Excel Macro that actually calculated the average for every 7-day period and then selected the highest average as the critical metric.

Table C5-2. Summer water temperature monitoring summary, Smith River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Goose (high)	13020101	1	1999	1:12	14.0	8/26	15.1	8/25	16.2	7/13	12.8	297.6
Goose (high)	13020101	1	2000	1:12	14.8	8/2	16.2	8/2	16.7	8/8	13.9	297.6
Wilson (low)	14010801	1	1994	0:48	12.2	7/4	13.7	7/4	13.9	6/27	11.4	7930.0
Goose (low)	14020201	1	1998	1:12	17.0	7/25	19.0	7/25	19.8	7/26	15.9	14752.0
Goose Low	14020201	1	1999	1:12	16.0	8/26	17.7	8/25	19.0	7/13	14.6	14752.0
Goose Trib	14020202	1	1998	1:12	15.4	7/25	16.8	7/24	17.8	7/26	14.6	4197.0
Goose Trib	14020202	1	1999	1:12	14.8	8/26	16.0	8/25	16.5	7/13	13.3	4197.0
Goose Mid	14022601	1	1999	1:12	14.8	8/26	15.8	8/25	16.1	8/25	14.1	663.6
Goose Mid	14022601	1	2000	1:12	15.5	8/2	16.8	8/2	17.3	8/1	14.7	663.6
Goose, East Fork	14022602	1	1999	1:12	13.2	8/26	13.7	8/25	13.9	8/25	12.6	1114.1
Goose, East Fork	14022602	1	2000	1:12	13.9	8/2	14.5	8/2	14.9	8/3	13.2	1114.1
Wilson Trib.	14510401	2	1996	1:12	12.1	8/30	12.5	8/30	12.7	8/30	11.9	679.1
Wilson (high)	15012901	1	1994	0:48	13.6	8/15	14.2	8/16	14.5	8/13	13.3	2146.0
Wilson (mid)	15013201	1	1994	0:48	13.4	8/16	14.0	8/16	14.2	8/13	13.1	3961.0
Wilson (mid)	15013201	1	1995	0:48	13.8	8/4	14.5	7/30	14.8	7/31	13.3	3961.0
Wilson (mid)	15013201	1	1996	1:12	13.8	8/30	16.1	8/30	16.5	8/30	12.9	3961.0
Wilson (mid)	15013201	1	1997	1:12	14.3	9/3	15.3	9/2	15.4	8/27	13.7	3961.0
Wilson (mid)	15013201	1	1998	0:08	13.8	8/15	14.4	8/13	14.8	7/26	13.4	3961.0
Wilson (mid)	15013201	1	1999	1:12	13.7	8/27	14.1	8/27	14.2	8/26	13.6	3961.0
Wilson (mid)	15013201	1	2000	1:12	13.7	9/22	14.5	7/31	14.6	7/29	13.3	3961.0
Goose (really low)	15023501	1	1997	1:12	17.3	8/9	19.6	8/9	20.6	8/7	16.1	22067.7
Little Mill	17010701	1	1998	1:12	13.5	8/13	14.2	8/13	14.7	7/26	13.0	2274.0
Little Mill	17010701	1	1999	1:12	13.5	8/27	14.0	8/25	14.2	8/26	13.6	2274.0
Sultan	17011901	1	1997	1:12	15.2	8/5	16.9	8/7	17.7	8/7	13.6	1281.0
Sultan	17011901	1	1999	1:12	13.7	8/26	14.2	8/25	14.5	8/26	13.7	1281.0
Peacock	17012901	1	1998	1:12	13.2	9/15	14.0	8/13	14.5	7/26	12.7	846.0
Peacock	17012901	1	2000	1:12	14.0	9/16	14.6	9/16	14.5	6/28	12.6	846.0
Campsix	17512501	1	1998	1:12	13.2	8/13	13.5	8/13	13.8	8/12	13.0	233.0
Campsix	17512501	1	1999	1:12	13.3	8/27	13.5	8/27	13.7	8/26	13.3	233.0
SF Winchuck Trib #1	18010601	2	1995	1:12	13.4	8/2	14.3	8/2	14.6	7/31	13.1	13.7
SF Winchuck Trib #2	18010602	2	1995	1:12	13.2	8/3	13.6	8/2	13.9	7/31	13.3	24.6
Rowdy trib R1700	18010901	2	1999	1:12	13.5	8/25	13.8	8/25	14.2	8/27	13.3	124.2
Rowdy trib R1700	18010901	2	2000	1:12	13.1	6/27	13.7	6/26	14.3	6/27	13.3	124.2

Table C5-2 Continued. **Summer water temperature monitoring summary, Smith River HPA.**

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Rowdy (high)	18011601	1	1994	1:12	14.8	8/15	16.3	8/15	16.5	7/14	13.1	7667.0
Rowdy (high)	18011601	1	1999	1:12	14.9	8/26	15.8	7/12	16.6	7/12	13.3	7667.0
Rowdy (high)	18011601	1	2000	1:12	15.6	9/19	16.9	9/19	17.8	9/19	15.8	7667.0
Ravine	18011701	1	1998	0:08	13.6	9/15	14.3	9/3	15.0	9/13	13.8	803.0
Ravine	18011701	1	2000	1:12	14.3	9/19	14.8	9/19	15.6	9/19	14.8	803.0
Rowdy trib. #3	18011801	2	1995	1:12	12.9	8/2	13.2	8/2	13.6	7/31	13.1	72.5
Rowdy trib. #4	18011901	2	1995	1:12	14.6	9/21	17.6	9/21	19.8	9/20	13.4	260.4
Rowdy Trib	18012001	2	1996	1:12	12.7	8/31	13.3	8/31	13.4	8/30	12.0	296.5
Rowdy (low)	18013001	1	1994	1:12	16.4	8/14	19.0	8/15	19.8	7/14	12.8	12587.0
Rowdy (low)	18013001	1	1995	1:12	16.6	8/3	19.3	8/3	19.4	8/19	13.7	12587.0
Rowdy (low)	18013001	1	1996	1:12	16.2	7/28	18.6	7/27	19.4	7/21	14.2	12587.0
Rowdy (low)	18013001	1	1997	1:12	16.5	8/5	19.1	8/5	19.9	8/7	14.8	12587.0
Rowdy (low)	18013001	1	1999	1:12	15.5	8/27	17.4	7/12	18.1	7/12	13.4	12587.0
Rowdy (low)	18013001	1	2000	1:12	15.6	7/31	17.4	7/30	18.3	8/2	14.3	12587.0
SF Rowdy (low)	18013002	1	1994	1:12	13.7	8/14	14.9	8/15	15.2	9/20	12.5	2573.0
SF Rowdy (low)	18013002	1	1997	1:12	14.3	8/27	15.6	8/5	16.1	8/7	13.3	2573.0
Savoy	18013003	1	1998	0:08	14.0	8/13	15.0	8/13	15.4	9/12	12.9	2573.0
Savoy	18013003	1	1999	1:12	13.7	8/27	14.4	8/25	14.6	8/22	13.1	2573.0
SF Rowdy (high)	18013004	1	1998	0:08	14.1	8/13	15.1	8/13	15.5	9/12	13.2	1552.0
SF Rowdy (high)	18013004	1	1999	1:12	13.7	8/27	14.4	8/24	14.6	7/12	12.1	1552.0
Savoy (high)	18013201	1	1994	1:12	13.1	8/16	13.8	8/15	14.0	8/16	12.5	2264.1
SF Winchuck River (high)	18510101	1	1994	1:12	13.2	8/16	13.8	8/15	14.3	9/21	12.5	1193.1
SF Winchuck River (high)	18510101	1	1999	1:12	13.4	8/25	13.8	8/25	14.4	7/12	12.9	1193.1
Gilbert	18510401	1	1997	1:12	14.6	9/3	15.5	9/2	15.9	9/5	13.9	1506.7
Gilbert	18510401	1	1999	1:12	13.2	8/27	13.8	8/24	14.1	8/21	12.7	1506.7
D2010CD	18511101	2	1996	1:12	12.2	10/9	12.5	10/8	13.1	10/8	12.5	10.5
D2010 CD	18511101	2	1997	1:12	12.3	8/7	12.8	8/7	13.7	8/7	12.8	10.5
D2010 CD	18511101	2	1998	1:12	12.6	9/3	13.1	9/3	13.9	9/3	12.9	10.5
D2010 CD	18511101	2	1999	1:12	12.1	8/25	12.5	8/25	13.4	7/12	11.2	10.5
D2010 CD	18511101	2	2000	1:12	10.9	9/19	11.0	9/19	11.2	9/19	10.9	10.5
D2010CU	18511102	2	1996	1:12	10.9	10/10	10.9	10/7	11.1	9/14	10.8	1.6

Table C5-2 Continued. Summer water temperature monitoring summary, Smith River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
D2010 CU	18511102	2	1997	1:12	11.0	9/14	11.2	9/14	11.4	8/7	11.1	1.6
D2010 CU	18511102	2	1998	1:12	11.9	9/11	12.1	9/11	12.0	9/2	11.9	1.6
D2010 CU	18511102	2	1999	1:12	10.6	8/27	10.7	9/10	10.8	8/26	10.8	1.6
D2010TD	18511103	2	1996	1:12	12.4	10/8	12.7	10/8	13.7	10/8	12.2	37.3
D2010 TD	18511103	2	1997	1:12	13.1	8/6	13.8	8/6	15.2	8/7	13.4	37.3
D2010 TD	18511103	2	1998	1:12	13.4	9/3	14.0	9/3	14.8	9/3	13.7	37.3
D2010 TD	18511103	2	1999	1:12	12.6	8/25	12.9	8/25	13.7	7/12	12.0	37.3
D2010 TD	18511103	2	2000	1:12	13.3	9/18	13.7	9/18	14.5	9/19	14.0	37.3
D2010TU	18511104	2	1996	1:12	11.6	10/10	11.7	10/10	12.0	10/8	12.0	7.3
D2010 TU	18511104	2	1997	1:12	11.6	9/28	11.7	8/8	12.0	8/7	12.0	7.3
D2010 TU	18511104	2	1998	1:12	12.2	9/4	12.4	9/4	12.8	9/3	12.5	7.3
D2010 TU	18511104	2	1999	1:12	11.2	8/26	11.3	8/25	11.7	8/27	11.2	7.3
D2010 TU	18511104	2	2000	1:12	11.9	9/19	12.1	9/19	12.5	9/19	12.3	7.3
D1120TD	18511105	2	1996	1:12	13.0	10/8	13.5	10/8	14.6	10/8	13.1	71.5
D1120 TD	18511105	2	1997	1:12	13.1	9/3	13.5	9/8	14.3	8/8	12.5	71.5
D1120 TD	18511105	2	1998	1:12	13.2	9/12	13.7	9/12	14.3	9/12	14.0	71.5
D1120 TD	18511105	2	1999	1:12	12.5	8/25	12.8	8/25	13.3	8/27	12.5	71.5
D1120 TD	18511105	2	2000	1:12	14.7	9/18	15.5	9/18	16.8	9/19	15.4	71.5
Dominie (high)	18511201	1	1994	1:12	11.9	9/19	12.1	7/10	12.5	9/20	12.0	394.5
D1120TU	18511202	2	1996	1:12	12.2	10/8	12.5	10/7	13.4	10/8	12.0	14.4
D1120 TU	18511202	2	1997	1:12	12.1	9/26	12.5	9/26	13.1	8/7	12.2	14.4
D1120 TU	18511202	2	1998	1:12	12.4	9/4	12.8	9/12	13.3	9/12	12.9	14.4
D1120 TU	18511202	2	1999	1:12	11.9	8/25	12.1	8/25	12.5	8/27	12.0	14.4
D1120 TU	18511202	2	2000	1:12	12.6	9/19	12.9	9/19	13.6	9/19	13.2	14.4
D1120CU	18511203	2	1996	1:12	13.0	10/8	13.4	10/8	14.6	10/8	13.1	17.7
D1120 CU	18511203	2	1997	1:12	12.7	9/8	13.1	8/6	14.0	8/7	13.4	17.7
D1120 CU	18511203	2	1998	1:12	13.4	9/4	13.8	9/12	14.3	9/3	13.9	17.7
D1120 CU	18511203	2	1999	1:12	12.7	8/25	13.1	8/25	13.7	8/26	13.3	17.7
D1120 CU	18511203	2	2000	1:12	14.0	9/18	14.5	9/19	15.6	9/19	14.8	17.7
D1120CD	18511204	2	1996	1:12	12.5	10/10	12.9	8/31	13.7	10/8	12.5	33.7
D1120 CD	18511204	2	1997	1:12	12.9	9/3	13.2	9/3	13.7	8/7	13.1	33.7
D1120 CD	18511204	2	1998	1:12	13.1	9/12	13.7	9/12	14.5	9/12	13.7	33.7
D1120 CD	18511204	2	1999	1:12	12.3	8/25	12.7	8/25	13.1	8/27	12.2	33.7
D1120 CD	18511204	2	2000	1:12	14.1	9/19	14.7	9/19	15.9	9/20	13.2	33.7
Dom Trib. #1	18511401	2	1995	1:12	13.3	7/30	14.0	8/1	14.6	7/31	13.1	40.8
Dom Trib. #2	18511402	2	1995	1:12	14.4	8/2	18.5	8/3	20.7	7/31	12.8	15.2
Dom Trib. #3	18511403	2	1995	1:12	13.9	9/22	14.8	9/21	16.9	9/19	14.8	38.7

Table C5-2 Continued. **Summer water temperature monitoring summary, Smith River HPA.**

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Dom Trib. #4	18511404	2	1995	1:12	13.7	8/1	14.5	8/1	15.8	7/31	13.7	4.7
Dominie trib	18511405	2	1996	1:12	13.3	8/31	14.0	8/31	14.3	8/30	12.5	347.9
Dominie (low)	18512301	1	1994	1:12	14.3	8/15	16.0	8/16	16.2	8/13	13.1	2254.0
Dominie (low)	18512301	1	1997	1:12	14.7	9/3	15.8	9/3	16.4	9/5	13.6	2254.0
Dominie (low)	18512301	1	1998	0:08	14.2	8/13	15.4	8/13	15.7	7/26	13.3	2254.0
SF Winchuck River (low)	19513301	1	1994	1:12	14.5	8/15	16.0	8/16	16.5	8/31	12.2	5891.0
SF Winchuck River (low)	19513301	1	1995	1:12	14.7	8/3	16.1	8/3	16.9	8/19	13.4	5891.0
SF Winchuck River (low)	19513301	1	1996	1:12	14.8	8/31	16.5	8/30	17.5	9/1	13.4	5891.0
SF Winchuck River (low)	19513301	1	1997	1:12	15.5	9/3	16.9	8/5	17.7	8/7	14.5	5891.0
SF Winchuck River (low)	19513301	1	1998	0:08	14.7	8/14	16.6	9/14	18.4	9/11	12.2	5891.0
SF Winchuck River (low)	19513301	1	1999	1:12	14.1	8/16	15.1	8/23	15.8	6/22	12.5	5891.0
SF Winchuck River (low)	19513301	1	2000	1:12	15.3	9/18	16.5	9/18	17.8	9/20	14.1	5891.0

Table C5-3. Summer water temperature monitoring summary, Coastal Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Little Surpur	11020201	1	1996	1:12	15.3	7/28	16.2	7/28	16.4	7/30	14.6	1601.0
Little Surpur	11020201	1	1998	1:12	14.8	7/25	15.3	7/24	15.8	7/26	14.6	1601.0
Little Surpur	11020201	1	1999	1:12	15.1	8/26	16.2	8/26	16.8	9/11	12.2	1601.0
Surpur	11020301	1	1996	0:30	14.6	7/28	15.8	7/28	16.1	7/30	13.9	3236.6
Surpur	11020301	1	1997	1:12	14.4	9/2	15.6	8/6	16.1	8/8	14.0	3236.6
Surpur	11020301	1	1999	1:12	14.0	8/26	14.7	8/25	14.9	8/24	12.8	3236.6
Tectah (old)	11021201	1	1995	0:48	16.2	8/4	18.2	8/3	18.6	8/4	15.1	11413.0
Tectah (low)	11021202	1	1995	1:12	16.3	8/4	18.5	8/3	19.1	8/4	15.2	11413.0
Tectah (low)	11021202	1	1997	1:12	16.3	8/16	16.6	8/15	16.7	8/8	15.9	11413.0
Tectah (low)	11021202	1	1998	1:12	16.9	7/25	18.7	7/25	19.4	7/26	16.1	11413.0
Tectah (low)	11021202	1	1999	1:12	16.2	8/26	17.4	7/11	18.1	7/12	14.6	11413.0
Tectah (low)	11021202	1	2000	1:12	17.3	8/4	18.8	8/1	19.5	6/28	15.4	11413.0
Tectah Trib. (class II)	11021301	2	1996	1:12	13.1	8/26	13.4	8/25	13.9	8/24	13.1	189.5
Tectah (mid)	11023401	1	1995	0:48	15.1	8/4	16.5	8/3	17.0	8/4	14.8	6892.5
Tectah (mid)	11023401	1	1997	1:00	15.6	8/6	17.1	8/6	17.9	8/8	14.9	6892.5
Tectah (mid)	11023401	1	1999	1:00	15.4	8/26	16.5	8/25	16.7	7/13	13.9	6892.5
McGarvey (high)	12010201	1	1994	0:48	12.8	8/17	13.3	8/16	13.4	8/13	12.8	1337.4
NF Ah Pah Trib. (161_up)	12020901	2	1996	1:12	13.3	7/29	13.6	7/28	13.9	7/30	13.1	616.7
NF Ah Pah Trib. (161_up)	12020901	2	1997	1:12	13.4	9/3	13.5	9/1	13.6	8/7	13.3	616.7
Ah Pah (North Fork)	12021601	1	1995	1:12	14.8	8/4	15.8	8/3	16.2	8/4	14.3	670.0
Ah Pah (North Fork)	12021602	1	1996	0:30	15.0	7/28	16.1	7/28	16.4	7/30	14.1	670.0
Ah Pah (North Fork)	12021602	1	1997	0:30	14.7	8/6	15.5	8/6	16.1	8/8	14.2	670.0
NF Ah Pah Trib. (161_lo)	12021603	2	1996	1:12	13.9	7/28	14.5	7/28	14.8	7/30	13.3	669.7
NF Ah Pah Trib. (161_lo)	12021603	2	1997	1:12	14.0	9/3	14.5	8/6	14.8	8/7	13.7	669.7
Ah Pah (South Fork)	12022101	1	1995	1:12	14.2	8/4	15.5	8/2	15.9	8/4	13.4	1501.0
Ah Pah (South Fork)	12022101	1	1996	0:30	14.0	7/30	14.8	7/28	14.9	7/29	13.8	1501.0
Ah Pah (South Fork)	12022101	1	1997	0:30	14.0	9/4	14.2	9/4	14.2	8/7	13.7	1501.0
Ah Pah (South Fork)	12022101	1	1999	1:00	13.4	8/27	13.7	8/27	13.8	8/28	13.5	1501.0
Ah Pah (Middle Fork)	12022103	1	1996	0:30	15.2	7/31	15.6	8/25	15.4	7/29	15.0	3068.0
Ah Pah (Middle Fork)	12022103	1	1997	0:30	15.9	8/6	18.3	8/6	18.7	8/7	14.9	3068.0

Table C5-3 Continued. **Summer water temperature monitoring summary, Coastal Klamath HPA.**

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Bear Creek Trib	12022401	2	1996	1:12	13.4	8/18	13.7	8/18	14.1	8/24	13.1	435.2
Bear (Klamath)	12022501	1	1995	0:48	14.1	8/4	14.9	8/3	15.1	8/4	13.9	2659.0
Bear (Klamath)	12022601	1	1994	0:48	13.7	8/17	14.4	8/16	14.5	8/13	13.3	5343.0
Bear (Klamath)	12022601	1	1995	0:48	14.4	8/4	15.2	8/3	15.4	8/4	13.9	5343.0
Bear (Klamath)	12022601	1	1996	0:30	14.9	7/28	16.0	7/28	16.2	7/29	14.2	5343.0
Bear (Klamath)	12022601	1	1996	1:12	14.9	7/28	15.9	7/28	16.2	7/30	14.2	5343.0
Bear (Klamath)	12022601	1	1997	0:30	14.8	8/9	15.7	8/6	16.0	8/8	14.3	5343.0
Bear (Klamath)	12022601	1	1999	1:00	14.1	8/26	14.8	8/25	15.0	8/26	14.0	5343.0
Bear, South Fork(Klamath)	12033101	1	1995	0:48	13.2	8/4	13.7	8/3	14.0	8/4	13.1	1216.5
Hunter	13010402	1	1995	0:30	12.2	6/23	13.6	6/22	14.2	6/20	11.2	13710.7
Hunter	13010402	1	1999	0:30	12.2	7/14	13.9	7/14	14.3	6/22	10.9	13710.7
McGarvey (low)	13012401	1	1994	0:48	13.4	7/20	14.3	7/19	14.5	7/7	12.8	4808.0
McGarvey (low)	13012501	1	1996	0:30	14.6	7/28	15.7	7/27	15.9	7/29	14.0	4808.0
McGarvey (low)	13012501	1	1999	0:30	15.0	8/26	16.4	8/25	16.8	8/25	14.6	4808.0
Turwar (low)	13020501	1	1994	0:48	17.6	8/16	19.7	8/16	19.9	8/14	16.1	16746.0
Turwar (low)	13020501	1	1995	0:48	16.9	8/4	18.7	8/2	19.1	7/16	15.1	16746.0
Turwar (low)	13020501	1	1996	1:12	17.2	7/28	18.9	7/27	19.3	7/29	15.8	16746.0
Turwar (low)	13020501	1	1997	1:12	17.4	8/6	19.1	8/5	19.6	8/7	16.2	16746.0
Turwar (low)	13020501	1	1998	1:12	17.0	8/15	18.4	8/13	19.0	7/26	15.8	16746.0
Turwar (low)	13020501	1	1999	1:12	16.6	8/25	18.6	7/12	19.1	7/13	14.8	16746.0
Turwar (low)	13020501	1	2000	1:12	17.2	8/1	19.3	8/1	19.7	8/1	16.0	16746.0
Tarup	13022901	1	1996	0:30	13.6	7/28	14.2	7/28	14.3	7/29	13.4	3098.0
Omagaar	13023201	1	1996	0:30	13.5	7/28	14.0	7/28	14.2	7/29	13.1	857.2
Hunter (mid)	14010201	1	1994	0:48	13.5	8/16	14.2	8/16	14.3	8/13	13.3	3197.6
Hunter (mid)	14010201	1	1999	1:00	14.3	8/27	14.9	8/27	15.1	8/22	13.8	3197.6
Hunter (mid)	14010201	1	1999	1:12	14.1	8/27	14.5	8/25	14.6	8/22	13.7	3197.6
Kurowitz	14010202	1	1999	1:12	14.5	8/26	15.3	7/12	15.9	7/12	12.9	864.9
Kurowitz	14010202	1	2000	1:12	15.2	9/19	16.4	7/31	17.0	9/19	15.3	864.9
Hunter Trib.	14011101	2	1996	1:12	13.1	8/30	15.5	8/31	15.9	9/1	11.4	608.2
Hunter (low)	14011102	1	1995	0:48	14.9	8/4	17.0	8/2	17.2	7/31	13.6	5701.0
Hunter (low)	14011102	1	1996	1:12	14.9	7/28	16.8	7/28	17.0	7/29	13.9	5701.0
Hunter (low)	14011102	1	1997	1:12	15.3	8/6	17.6	8/5	18.0	8/7	14.2	5701.0
Hunter (low)	14011102	1	1998	0:08	15.2	8/14	17.0	8/13	17.5	7/26	14.2	5701.0
Hunter (low)	14011102	1	2000	1:12	15.6	7/31	18.2	7/31	18.5	8/1	14.4	5701.0
Mynot	14013501	1	1999	1:12	13.4	8/26	14.0	8/22	14.3	6/22	11.4	516.7
Turwar (high)	14020601	1	1994	0:48	13.2	7/31	13.7	7/31	13.7	7/31	12.8	1317.0
Turwar (high)	14020601	1	1995	0:48	14.4	8/4	14.9	8/3	15.4	8/4	14.6	1317.0

Table C5-3 Continued. Summer water temperature monitoring summary, Coastal Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Turwar Trib. (class II)	14020602	2	1996	1:12	14.3	8/14	14.8	8/14	14.9	8/15	14.0	369.1
Turwar (mid)	14022101	1	1994	0:48	17.0	8/15	19.2	7/16	19.6	7/18	15.0	7606.0
Turwar (mid)	14022101	1	1995	0:48	16.8	8/4	19.1	8/3	19.4	8/4	15.6	7606.0
Turwar	14022102	1	1997	1:00	16.4	8/6	17.9	7/21	18.7	7/19	15.0	8238.0
Turwar	14022102	1	1999	1:00	16.2	8/25	17.9	8/23	18.6	7/12	14.4	8238.0
SF Turwar	14022901	1	2000	1:12	14.4	9/19	15.2	9/19	16.3	9/19	13.9	5091.2
Hunter (high)	15013501	1	1994	0:48	14.0	8/16	14.7	8/16	14.8	7/18	12.6	2163.2
Hunter (high)	15013501	1	1995	0:48	14.3	8/7	15.5	8/7	15.8	8/7	13.1	2163.2

Table C5-4. Summer water temperature monitoring summary, Blue Creek HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Blue (West Fork)	12020101	1	1994	0:48	14.2	8/16	15.8	8/16	15.9	8/13	13.4	8616.0
Blue (West Fork)	12020101	1	1996	1:12	15.3	7/28	16.8	7/28	17.0	7/30	14.5	8616.0
Blue (West Fork)	12020101	1	1997	1:00	15.2	8/6	16.9	8/6	17.3	8/7	14.5	8616.0
Blue (West Fork)	12020101	1	1999	1:00	15.1	8/26	16.5	8/23	16.8	8/22	14.3	8616.0
Blue	12021101	1	1994	0:48	17.2	8/15	19.8	7/16	20.2	7/18	14.8	78520.0
Blue	12021101	1	1995	0:48	17.3	8/4	19.9	8/3	20.6	8/4	15.6	78520.0
Blue	12021101	1	1997	0:30	18.0	8/6	20.9	8/6	21.4	8/7	16.1	78520.0
Blue	12021101	1	1999	0:30	17.1	8/23	19.5	8/23	20.1	8/22	15.6	78520.0
Slide	12021401	1	1997	1:00	14.3	8/6	16.1	8/6	16.4	8/7	13.6	78520.0
Slide	12021401	1	1999	1:00	13.5	8/25	15.4	7/12	16.0	7/12	12.4	78520.0
Nickowitz	12030301	1	1996	0:30	14.4	7/28	15.3	7/28	15.5	7/30	13.6	9693.0
Nickowitz	12030301	1	1997	1:00	14.2	8/7	15.2	8/6	15.7	8/8	13.8	9693.0
Nickowitz	12030301	1	1999	1:00	13.7	8/26	14.4	8/25	14.7	7/13	12.7	9693.0
Coyote (Blue Cr.)	12031701	2	1996	1:12	11.5	8/26	12.0	8/25	12.8	8/24	11.3	435.2
Dandy	13020801	1	2000	1:12	13.3	9/16	13.8	9/16	13.7	9/13	12.9	1244.3
Blue (West Fork)	13022301	1	1994	0:48	12.9	7/19	13.5	7/19	13.7	7/17	12.3	1389.0
Potato Patch (185_up)	13022501	2	1996	1:12	14.1	7/28	15.6	7/28	15.8	7/25	13.1	482.5
Potato Patch (185_up)	13022501	2	1997	1:12	15.1	8/10	16.7	8/9	17.3	8/8	14.6	482.5
Potato Patch	13023601	1	1994	0:48	14.0	8/11	14.5	8/10	14.6	8/9	13.9	1782.0
Potato Patch	13023601	1	2000	1:12	14.5	9/20	14.9	9/19	15.6	9/20	14.3	1782.0
Blue	13033401	1	1996	0:30	17.1	7/28	18.7	7/28	19.0	7/30	15.5	31753.1
Blue	13033401	1	1997	1:00	16.7	8/9	18.2	8/6	18.7	8/8	15.7	31753.1
Blue	13033401	1	1999	1:00	15.5	8/26	16.5	8/25	17.2	7/13	13.8	31753.1
Crescent City Fork	13033402	1	1996	0:30	15.2	7/28	16.5	7/28	16.6	7/28	14.1	14343.1
Crescent City Fork	13033402	1	1997	1:00	14.7	8/9	15.9	8/6	16.5	8/8	14.2	14343.1
Crescent City Fork	13033402	1	1999	1:00	14.2	8/26	15.0	8/25	15.8	7/13	13.0	14343.1
Potato Patch (185_lo)	13033801	2	1996	1:12	15.4	7/28	16.0	7/28	16.2	7/28	15.0	1079.0
Potato Patch (185_lo)	13033801	2	1997	1:12	15.3	8/10	16.0	8/9	16.7	8/8	15.1	1079.0

Table C5-5. Summer water temperature monitoring summary, Interior Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Tully (high)	09030301	1	1999	1:12	14.5	8/26	15.4	8/26	16.2	7/13	13.4	1096.2
Tully	09030301	1	2000	1:12	14.9	8/3	16.1	8/3	16.4	8/2	14.2	1096.2
Pine	09040501	1	1999	1:00	17.4	8/26	18.2	8/25	19.3	7/13	16.3	31200.7
Mettah (high)	10021001	1	1999	1:12	14.0	8/26	14.6	8/26	14.8	8/28	13.9	362.6
Mettah (high)	10021001	1	2000	1:12	14.1	8/5	15.0	8/4	15.3	8/3	13.7	362.6
Roach (upper)	10022501	2	1997	1:12	16.4	8/9	18.6	8/6	19.6	8/8	15.3	10808.1
Cappell	10030301	1	1996	0:30	16.4	7/28	17.3	7/28	17.5	7/28	15.3	5253.1
Roach	10030801	1	1995	0:48	18.2	7/22	20.3	7/22	21.5	7/24	17.8	18613.0
Roach	10030801	1	1996	0:30	20.1	7/28	22.1	7/28	22.4	7/27	19.1	18613.0
Morek	10030901	1	1996	1:12	14.7	7/28	15.5	7/28	15.6	7/26	14.3	2561.9
Waukell (past Tectah)	10032301	2	1996	1:12	12.9	8/13	13.2	8/13	13.4	8/13	12.5	153.9
Tully (low)	10032501	1	1995	0:48	16.1	8/4	17.1	8/4	18.0	8/5	15.3	11085.0
Tully (low)	10032501	1	1997	1:00	16.6	8/9	17.8	8/9	18.5	8/8	16.2	11085.0
Johnson (188_lo)	11022401	2	1996	1:12	13.7	7/28	14.1	7/28	14.2	7/26	13.4	907.7
Johnson (188_lo)	11022401	2	1997	1:12	13.2	8/8	13.7	8/7	14.2	8/8	13.3	907.7
Johnson (188_up)	11022402	2	1996	1:12	13.3	7/28	13.7	7/28	13.9	7/30	12.8	770.9
Johnson (188_up)	11022402	2	1997	1:12	12.8	8/7	13.2	8/7	13.9	8/8	12.9	770.9
Johnson	11022403	1	1997	1:12	13.2	8/27	13.3	8/24	13.4	9/17	13.1	1940.5
Mettah (low)	11023601	1	1996	1:12	16.1	7/28	17.2	7/28	17.3	7/27	15.6	6180.5
Clirliah Trib	11023602	2	1996	1:12	13.6	8/26	13.9	8/25	14.2	8/24	13.6	259.5
Pecwan, West Fork	11030901	1	1995	0:48	12.9	8/4	13.4	8/3	14.2	8/4	13.3	7473.8
Pecwan, West Fork	11030901	1	1999	1:12	12.5	8/26	12.8	8/26	13.1	8/26	12.6	7473.8
Pecwan, West Fork	11030901	1	2000	1:12	12.4	8/3	12.8	6/26	13.3	8/2	12.3	7473.8
Pecwan, East Fork	11031501	1	1995	0:48	12.7	8/4	13.2	8/3	14.2	8/4	12.9	6585.0
Pecwan, East Fork	11031501	1	1999	1:12	12.7	8/26	13.0	8/26	13.4	8/26	12.9	6585.0
Pecwan, East Fork	11031501	1	2000	1:12	12.6	8/4	13.1	6/26	13.6	6/28	12.3	6585.0
Pecwan	11031701	1	1994	0:48	14.1	7/20	15.0	7/19	15.3	7/17	13.3	17594.0
Pecwan	11031701	1	1995	0:48	14.4	8/7	15.6	8/7	17.8	8/4	14.6	17594.0
Pecwan	11031701	1	1996	0:30	15.7	7/28	17.0	7/28	17.3	7/30	14.8	17594.0
Pecwan	11031701	1	1999	1:00	14.7	8/26	16.0	8/26	16.3	8/26	14.4	17594.0

Table C5-6. Summer water temperature monitoring summary, Redwood Creek HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Lake Prairie	05041901	2	1997	2:24	15.4	8/9	17.5	8/9	18.5	8/7	14.6	973.5
Lake Prairie	05041901	2	1998	2:30	15.6	7/21	17.1	7/21	17.8	7/21	14.9	973.5
Lake Prairie	05041901	2	1999	1:12	14.4	8/26	15.9	7/12	17.1	7/13	13.6	973.5
Lake Prairie	05041901	2	2000	0:36	15.1	8/2	16.4	6/26	17.0	6/28	13.5	973.5
Redwood at Miñon	05042001	1	2000	1:12	17.7	8/5	20.6	8/5	20.9	8/3	15.9	18416.6
Pardee	05043201	2	1996	2:24	14.4	7/28	15.0	7/27	15.2	7/27	14.3	1579.0
Pardee	05043201	1	1997	2:24	13.6	8/10	14.1	8/10	14.8	8/8	13.3	1579.0
Pardee	05043201	2	1998	2:30	14.1	7/21	14.7	7/22	15.2	7/22	14.3	1579.0
Pardee	05043201	2	1999	1:12	13.2	8/26	13.6	8/26	14.3	7/13	12.3	1579.0
Pardee	05043201	1	2000	0:36	9.9	5/24	10.7	5/23	11.3	5/21	8.9	1579.0
Lupton	06031501	1	1997	1:12	14.9	8/10	15.9	8/10	16.7	8/8	14.5	2862.0
Lupton	06031501	1	1998	1:12	15.2	9/4	16.0	9/4	16.5	7/26	14.5	2862.0
Lord Ellis	07033301	2	1996	1:12	12.7	8/26	12.9	8/26	13.3	8/24	12.8	371.7
Coyote (Rdwd)	08020201	1	1994	1:12	16.0	8/16	16.9	8/14	17.4	8/13	15.5	5025.0
Coyote (Rdwd)	08020201	1	1999	1:12	17.1	8/26	18.5	7/12	19.7	7/12	15.9	5025.0
Coyote (Rdwd)	08020201	1	2000	1:12	17.8	8/1	19.2	8/1	19.9	6/28	16.1	5025.0
Redwood at Panther	08021301	1	1999	1:12	20.9	7/13	22.9	7/12	24.0	7/12	19.8	15688.1
Redwood at Panther	08021301	1	2000	1:12	22.0	8/1	23.9	8/1	24.7	6/27	19.6	15688.1
Panther (Rdwd)	08021401	1	1998	0:08	14.6	9/4	15.3	7/25	15.9	7/26	14.4	3814.0
Panther (Rdwd)	08021401	1	1999	1:12	14.5	8/26	14.9	8/26	15.0	8/24	13.8	3814.0
Panther (Rdwd)	08021401	1	2000	1:12	14.4	8/4	15.1	8/2	15.4	8/2	14.0	3814.0
Panther (Rdwd)	08021402	1	1994	2:00	13.1	7/22	13.5	7/20	13.6	7/17	12.2	3814.0
Panther (Rdwd)	08021402	1	1995	2:00	14.2	8/4	14.8	8/4	15.2	8/5	13.6	3814.0
Panther Rhva 2	08021601	2	2000	1:12	12.7	9/20	13.0	9/20	13.7	9/20	12.3	58.1
Panther Rhva 3	08022102	2	2000	1:12	12.8	9/20	13.1	9/20	13.8	9/20	12.7	75.5
Panther O-6	08022201	2	1999	1:12	13.6	8/26	13.8	8/26	14.1	8/27	13.7	455.6
North Fork Dolly Varden	08023601	2	1996	2:24	14.5	7/28	14.9	7/28	15.2	7/30	14.0	1069.0
North Fork Dolly Varden	08023601	2	1996	2:24	12.5	10/10	12.7	10/8	13.1	10/8	12.2	1069.0
North Fork Dolly Varden	08023601	2	1997	2:24	13.9	7/27	14.3	7/26	14.6	7/28	13.7	1069.0
North Fork Dolly Varden	08023601	2	1997	2:24	14.4	8/12	14.7	8/12	14.8	8/13	14.0	1069.0
North Fork Dolly Varden	08023601	2	1998	2:30	14.7	8/30	15.1	8/30	15.2	7/22	14.6	1069.0
North Fork Dolly Varden	08023601	2	1999	1:12	13.9	8/26	14.1	8/26	14.3	7/13	13.1	1069.0
South Fork Dolly Varden	08023602	2	1996	2:24	14.8	7/28	15.2	7/28	15.5	7/29	14.6	597.2
South Fork Dolly Varden	08023602	2	1998	2:30	14.7	8/30	15.1	8/30	15.5	7/23	13.7	597.2
South Fork Dolly Varden	08023602	2	1999	1:12	13.9	8/26	14.2	8/26	14.5	7/12	13.6	597.2
South Fork Dolly Varden	08023602	2	2000	1:12	13.8	8/3	14.3	6/27	15.1	6/27	13.8	597.2
Coyote Trib (Redwood Cr.)	09033101	2	1996	1:12	14.4	8/26	15.1	8/25	16.1	8/24	14.2	879.1

Table C5-7. Summer water temperature monitoring summary, Coastal Lagoons HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
WindyTD	08010601	2	1999	1:12	11.8	8/27	11.9	8/26	12.1	8/27	11.9	34.3
WindyTD	08010601	2	2000	1:12	12.1	9/19	12.2	9/19	12.3	9/19	12.1	34.3
WindyCD	08010602	2	1999	1:12	11.7	8/27	11.9	8/27	12.0	8/27	11.9	45.6
WindyCD	08010602	2	2000	1:12	12.2	9/19	12.4	9/18	12.8	9/19	12.5	45.6
WindyTU	08010701	2	1999	1:12	10.6	8/27	10.6	8/27	10.8	8/27	10.6	26.9
WindyTU	08010701	2	2000	1:12	11.3	9/25	11.3	9/23	11.3	9/20	11.2	26.9
WindyCU	08010702	2	1999	1:12	11.9	8/27	12.0	8/27	12.2	8/27	12.0	33.7
WindyCU	08010702	2	2000	1:12	12.4	9/19	12.6	9/20	12.9	9/19	12.7	33.7
Maple (mid)	08010801	1	1994	2:00	15.0	8/19	15.8	8/19	16.1	8/21	14.3	1687.7
Maple (mid)	08010801	1	1996	2:00	14.9	7/28	15.5	7/27	15.8	7/29	14.2	1687.7
Maple (mid)	08010801	1	1999	1:12	15.3	8/26	15.8	8/26	16.1	8/29	14.2	1687.7
Maple (mid)	08010801	1	2000	1:12	15.3	7/31	15.9	7/31	16.5	6/27	14.5	1687.7
M-Line	08010802	1	1994	2:00	13.9	8/19	14.6	8/21	14.6	8/20	13.4	361.7
M-Line	08010802	1	1995	2:00	14.2	8/3	15.5	8/3	15.8	8/1	13.2	361.7
M-Line	08010802	1	1999	1:12	14.1	8/26	14.5	8/26	14.8	8/26	14.2	361.7
M-Line	08010802	1	2000	1:12	13.7	7/31	14.4	7/31	14.8	8/1	13.3	361.7
Maple (high)	08011201	1	1994	2:00	14.0	8/19	14.5	8/21	14.8	8/19	14.0	2639.2
Maple (high)	08011201	1	1995	2:00	14.9	8/4	15.8	8/3	16.2	8/1	14.1	2639.2
Maple (high)	08011201	1	1996	2:00	15.2	7/28	16.1	7/27	16.2	7/27	14.6	2639.2
Clear	08011202	1	1997	2:00	14.1	9/3	14.8	7/25	15.2	8/8	13.7	1864.2
Clear	08011202	1	2000	1:12	14.1	8/1	15.3	7/31	16.0	6/28	13.2	1864.2
Beach	08011501	1	1994	2:00	13.9	8/19	14.6	8/21	14.6	8/19	13.6	469.1
Beach	08011501	1	1995	2:00	14.3	8/4	14.7	8/3	14.9	8/1	14.0	469.1
Beach	08011501	1	1999	1:12	14.8	8/26	15.6	8/25	16.0	8/26	14.7	469.1
Beach	08011501	1	2000	1:12	14.8	8/1	15.6	7/31	15.9	8/1	14.3	469.1
Beach	08011501	1	2000	1:12	14.8	8/1	15.6	7/31	15.9	8/1	14.3	469.1
Luffenholtz	08012901	1	1996	2:00	12.8	7/28	13.3	7/28	13.8	7/29	12.1	1688.9
Luffenholtz	08012901	1	1997	2:00	13.5	9/3	14.0	9/3	14.4	8/27	12.6	1688.9
M1CU	08020701	2	1999	1:12	13.2	8/26	13.5	8/26	14.1	8/27	13.2	179.3
M1CU	08020701	2	2000	1:12	13.5	9/18	13.8	9/18	14.9	9/20	13.0	179.3
M1CD	08020702	2	1999	1:12	13.5	8/26	13.9	8/26	14.8	8/27	13.4	193.3
M1CD	08020702	2	2000	1:12	13.9	9/17	14.4	9/18	15.6	9/20	12.8	193.3
M1TU	08021701	2	1999	1:12	13.2	8/26	13.4	8/26	14.0	8/27	13.3	70.0
M1TU	08021701	2	2000	1:12	13.9	9/18	14.3	9/20	15.2	9/20	13.2	70.0
M1TD	08021702	2	1999	1:12	14.1	8/26	14.6	8/25	15.3	8/27	14.1	79.1
M1TD	08021702	2	2000	1:12	14.7	9/17	15.1	9/18	16.3	6/27	14.2	79.1
M1TU2	08021703	2	1999	1:12	13.6	8/26	13.9	8/26	14.6	8/27	13.6	59.4
M1TU2	08021703	2	2000	1:12	14.2	9/20	14.6	9/20	15.4	9/20	13.8	59.4
M1TD2	08021704	2	1999	1:12	14.0	8/26	14.4	8/26	15.2	8/27	14.1	65.4

Table C5-7 Continued. Summer water temperature monitoring summary, Coastal Lagoons HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
M1TD2	08021704	2	2000	1:12	14.7	9/17	15.3	9/18	16.5	9/20	13.4	65.4
Mill Cr. (LP)	08511301	1	1997	2:00	13.8	9/3	14.0	9/3	14.2	9/4	13.7	617.3
McDonald	09010501	1	1996	2:00	12.6	8/29	13.6	8/30	14.0	8/30	12.6	3346.4
McDonald	09010501	1	1997	2:00	14.9	9/3	15.7	9/3	16.0	8/24	14.4	3346.4
McDonald	09010501	1	2000	1:12	13.3	9/17	14.1	9/18	14.7	9/19	12.9	3346.4
Pitcher	09012001	1	1996	2:00	13.6	8/30	15.0	8/31	15.6	8/30	13.6	3358.4
Pitcher	09012001	1	1997	2:00	14.9	9/3	15.9	9/3	16.2	9/1	13.7	3358.4
Pitcher	09012001	1	1999	1:12	13.7	8/27	14.3	8/27	14.7	8/29	13.1	3358.4
Pitcher	09012001	1	2000	1:12	13.3	7/31	13.8	9/17	14.2	9/19	12.6	3358.4
NF Maple Trib Fline	09012701	2	1999	1:12	13.0	8/27	14.5	7/21	13.5	8/27	13.2	249.5
Maple,NF (lower)	09012901	1	1994	2:00	14.4	8/13	15.3	8/5	15.8	8/13	14.5	6467.0
Maple,NF (lower)	09012901	1	1995	2:00	14.9	7/29	16.2	7/29	16.7	7/27	14.2	6467.0
Maple,NF (lower)	09012901	1	1996	2:00	14.8	7/27	15.7	7/27	16.2	7/27	14.4	6467.0
Maple,NF (lower)	09012901	1	1998	1:12	15.0	8/14	15.8	8/14	16.2	7/19	14.3	6467.0
Maple,NF (lower)	09012901	1	1999	1:12	15.3	8/27	16.0	8/23	16.5	8/29	14.8	6467.0
Maple,NF (lower)	09012901	1	2000	1:12	15.1	7/31	15.8	7/31	16.2	8/1	14.6	6467.0
Maple (low)	09012902	1	1994	2:00	15.3	8/16	16.2	8/6	16.7	8/3	15.0	16797.0
Maple (low)	09012902	1	1996	2:00	15.4	7/27	17.0	7/12	17.4	7/13	14.1	16797.0
Maple (low)	09012902	1	1998	1:12	15.8	8/14	17.5	7/16	18.4	7/19	14.9	16797.0
Maple (low)	09012902	1	1999	1:12	16.1	8/24	18.4	8/23	19.1	8/21	14.5	16797.0
Maple (low)	09012902	1	2000	1:12	16.5	7/31	18.7	7/31	19.6	8/1	15.4	16797.0
Maple,NF (upper)	09013401	1	1996	2:00	13.2	7/31	13.3	7/31	13.4	7/29	13.2	3460.3
Maple,NF (upper)	09013401	1	1997	2:00	14.1	8/27	14.7	8/22	15.3	8/7	13.3	3460.3
NF Maple Trib F8	09013401	2	2000	1:12	13.5	9/17	13.8	9/17	14.5	9/19	13.3	273.0
McDonald, NF	10012901	1	1997	2:00	13.5	9/3	13.8	9/3	14.0	8/29	12.5	1273.5
McDonald, NF	10012901	1	2000	1:12	13.3	9/17	13.6	9/17	14.1	9/19	12.7	1273.5

Table C5-8. Summer water temperature monitoring summary, Little River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Freeman	07010301	1	2000	1:12	13.4	7/31	14.1	7/31	14.3	8/1	13.0	1083.0
Little River (low)	07010801	1	1994	2:00	16.8	8/5	18.8	8/5	19.3	8/3	16.2	26011.0
Little River (low)	07010801	1	1995	2:00	16.8	8/4	18.6	7/29	19.3	7/31	15.6	26011.0
Little River (low)	07010801	1	1996	2:00	17.4	7/28	19.4	7/31	20.1	7/28	16.6	26011.0
Little River (low)	07010801	1	1998	1:12	17.0	8/15	19.0	8/15	19.8	7/19	16.2	26011.0
Little River (low)	07010801	1	1999	1:12	16.9	8/23	18.7	7/14	19.6	8/21	16.4	26011.0
Little River (low)	07010801	1	2000	1:12	17.4	7/31	19.2	7/30	20.2	7/28	16.5	26011.0
Carson	07011001	1	1997	1:12	14.9	8/27	15.4	8/27	15.8	7/18	13.7	2440.0
Carson	07011002	1	1998	1:12	14.7	8/14	15.6	7/16	16.2	7/18	14.0	2440.0
Carson	07011002	1	1999	1:12	14.8	8/24	15.3	8/23	16.0	6/22	13.2	2440.0
Carson	07011002	1	2000	1:12	14.8	7/31	15.3	7/31	15.7	7/28	14.4	2440.0
M155CD	07011201	2	1999	1:12	12.4	8/27	12.5	8/27	12.6	8/27	12.5	44.1
M155CD	07011201	2	2000	1:12	12.3	8/2	12.4	8/1	12.6	8/1	12.4	44.1
M155CU	07011202	2	1999	1:12	13.2	8/27	13.5	8/27	13.7	8/26	13.4	34.5
M155CU	07011202	2	2000	1:12	13.9	9/18	14.5	9/18	15.7	9/19	13.8	34.5
MitsuiCU	07011301	2	1996	1:12	12.8	8/30	13.0	8/30	13.4	10/8	12.8	60.4
MitsuiCU	07011301	2	1997	1:12	14.1	9/3	14.2	9/3	14.3	8/26	14.0	60.4
MitsuiCU	07011301	2	1998	1:12	12.8	8/14	13.0	8/14	13.3	7/26	12.8	60.4
MitsuiCU	07011301	2	1999	1:12	12.9	8/26	13.1	8/26	13.3	8/26	12.8	60.4
MitsuiCU	07011301	2	2000	1:12	13.2	9/18	13.5	9/18	14.1	9/19	13.3	60.4
MitsuiCD	07011401	1	1996	1:12	12.4	7/30	12.6	8/30	12.8	7/30	12.2	97.9
MitsuiCD	07011401	1	1997	1:12	13.7	9/3	14.4	9/3	14.6	9/4	13.7	97.9
MitsuiCD	07011401	1	1998	1:12	12.8	8/15	13.0	8/14	13.1	8/15	12.6	97.9
MitsuiCD	07011401	1	1999	1:12	13.1	8/27	13.6	8/26	14.0	8/29	12.8	97.9
MitsuiCD	07011401	1	2000	1:12	12.9	8/1	13.3	7/31	13.6	9/19	12.7	97.9
MitsuiTD	07011402	2	1996	1:12	11.8	8/31	12.0	8/30	12.0	8/27	11.4	63.0
MitsuiTD	07011402	2	1997	1:12	12.9	9/3	13.0	9/5	13.1	9/3	12.8	63.0
MitsuiTD	07011402	2	1998	1:12	12.3	8/16	12.4	8/14	12.5	8/12	12.2	63.0
MitsuiTD	07011402	2	1999	1:12	12.3	8/27	12.4	8/27	12.5	8/26	12.3	63.0
MitsuiTD	07011402	2	2000	1:12	12.5	9/19	12.7	9/19	13.0	9/18	12.7	63.0
MitsuiTU	07011403	2	1996	1:12	12.1	8/30	12.2	8/30	12.5	9/15	12.0	47.0
MitsuiTU	07011403	2	1997	1:12	14.0	8/27	14.1	8/26	14.6	8/26	14.0	47.0
MitsuiTU	07011403	2	1998	1:12	14.3	7/24	14.4	7/23	14.6	7/26	14.3	47.0
MitsuiTU	07011403	2	1999	1:12	13.6	8/27	13.6	8/27	13.7	8/27	13.6	47.0
MitsuiTU	07011403	2	2000	1:12	13.4	8/3	13.4	8/2	13.6	8/1	13.4	47.0

Table C5-8 Continued. Summer water temperature monitoring summary, Little River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Little River, Upper SF	07020601	1	1994	2:00	14.5	8/19	15.9	8/16	16.2	8/3	14.0	3619.0
Little River, Upper SF	07020601	1	1995	2:00	14.7	8/3	16.5	8/3	17.0	7/31	13.7	3619.0
Little River, Upper SF	07020601	1	1998	1:12	15.0	8/14	16.5	7/20	16.8	7/18	13.7	3619.0
Little River, Upper SF	07020601	1	1999	0:36	14.8	8/27	15.2	8/27	15.6	8/29	14.5	3619.0
Little River, Upper SF	07020601	1	2000	1:12	15.3	7/31	16.5	7/31	16.8	8/1	14.6	3619.0
Little River Headwaters	07021401	2	1996	2:24	12.7	7/28	13.3	7/28	13.4	7/28	12.5	468.0
Little River Headwaters	07021401	2	1997	2:24	12.1	8/10	12.4	8/9	12.9	8/7	12.3	468.0
Little River Headwaters	07021401	2	1998	2:30	12.4	8/30	12.7	8/30	13.1	8/29	12.2	468.0
Little River Headwaters	07021401	2	1999	1:12	11.6	8/26	11.8	8/26	12.2	8/26	11.9	468.0
Little River Headwaters	07021401	2	2000	1:12	11.8	8/3	12.1	8/2	12.4	8/2	12.0	468.0
M155TD	07021801	2	1999	1:12	12.2	8/27	12.4	8/27	12.6	8/29	11.4	26.5
M155TD	07021801	2	2000	1:12	12.4	9/18	12.7	9/18	13.3	9/19	12.5	26.5
M155TU	07021802	2	1999	1:12	12.8	8/27	13.2	8/27	13.6	8/27	12.7	21.2
M155TU	07021802	2	2000	1:12	14.1	9/17	14.9	9/17	16.3	9/19	14.0	21.2
Railroad	08013401	1	1994	2:00	14.4	8/19	15.7	8/19	15.9	8/19	13.3	1721.0
Railroad	08013401	1	1995	2:00	14.4	7/29	15.6	7/29	15.9	7/31	13.4	1721.0
Railroad	08013401	1	1998	1:12	14.6	8/14	15.5	8/13	15.9	7/19	14.0	1721.0
Railroad	08013401	1	1999	0:36	15.0	8/27	16.8	8/23	17.5	8/29	14.4	1721.0
Railroad	08013401	1	2000	1:12	15.2	7/31	16.3	7/31	16.6	8/1	14.7	1721.0
Little River, Lower SF	08013601	1	1994	2:00	14.6	7/24	16.3	8/5	16.9	8/3	14.5	3452.0
Little River, Lower SF	08013601	1	1995	2:00	15.2	7/30	16.7	8/3	17.2	8/1	14.0	3452.0
Little River, Lower SF	08013601	1	1998	1:12	15.9	7/23	17.4	7/23	18.1	7/26	15.2	3452.0
Little River, Lower SF	08013601	1	1999	0:36	15.6	8/27	16.5	8/23	17.2	8/22	14.5	3452.0
Little River, Lower SF	08013601	1	2000	1:12	16.1	7/31	18.0	7/31	18.5	8/1	15.2	3452.0
Little River (mid)	08013602	1	1994	1:36	15.2	7/30	16.4	7/29	16.9	7/31	14.4	13176.3
Little River (mid)	08013602	1	1996	2:00	16.0	7/28	17.5	7/28	17.9	7/29	14.8	13176.3
Little River (mid)	08013602	1	1999	1:12	15.5	8/27	16.2	8/27	16.6	8/29	15.3	13176.3
Little River (mid)	08013602	1	2000	1:12	15.8	7/31	17.0	7/31	17.4	8/1	15.0	13176.3
Danielle	08013603	1	2000	1:12	14.2	7/31	16.0	7/31	16.4	8/1	13.4	479.2
Heightman	08013604	1	2000	1:12	13.6	8/1	14.0	7/31	14.3	8/1	13.4	688.3
Little River (upper)	08022901	1	1994	2:00	13.4	8/21	14.2	8/21	14.5	8/19	13.3	8755.0
Little River (upper)	08022901	1	1995	2:00	14.0	8/3	15.2	8/3	15.8	7/31	13.3	8755.0
Little River (upper)	08022901	1	1996	2:00	14.1	7/28	15.3	7/27	15.8	7/30	12.6	8755.0
Little River (upper)	08022901	1	1999	1:12	14.1	8/27	14.7	8/27	15.3	8/29	13.1	8755.0
Little River (upper)	08022901	1	2000	1:12	14.3	9/18	15.1	9/18	16.1	9/19	13.9	8755.0
Little River (up98)	08023101	1	1998	1:12	15.3	8/14	17.0	8/14	17.4	7/26	14.3	9557.0
C-Line	08023201	1	2000	1:12	13.7	9/18	14.1	9/18	15.0	9/19	13.4	788.0

Table C5-9. Summer water temperature monitoring summary, Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Boulder Trib	04030301	2	1996	1:12	13.2	8/31	13.8	8/31	14.7	8/24	12.7	190.0
Boulder	04030501	1	1994	1:12	16.6	8/17	19.6	8/17	20.1	8/14	14.0	11617.1
Boulder	04030501	1	1995	1:12	16.7	8/3	18.3	8/3	18.8	7/16	15.2	11617.1
Boulder	04030501	1	1997	1:12	18.1	8/9	20.8	8/6	21.6	8/7	15.8	11617.1
Boulder	04030501	1	1998	1:12	17.7	7/25	20.1	8/13	21.2	7/26	16.4	11617.1
Boulder	04030501	1	1999	1:12	17.2	8/26	19.6	7/12	20.6	7/13	14.4	11617.1
Goodman Prairie	04032902	1	1999	1:12	15.2	8/26	15.9	8/26	16.3	8/29	14.1	938.0
Goodman Prairie	04032902	1	2000	1:12	15.3	9/18	16.3	9/18	17.0	9/19	15.0	938.0
Graham	04033501	2	1996	1:12	12.6	8/17	13.4	8/17	13.6	8/17	11.4	723.0
Cañon (high)	05020101	1	1994	1:12	14.9	8/17	15.9	8/16	16.2	8/14	14.0	6421.0
Cañon (high)	05020101	1	1999	1:12	16.0	8/26	17.1	8/26	17.8	8/29	14.6	6421.0
Black Dog Treatment/5300	05020701	2	1999	1:12	11.8	8/27	12.0	8/27	12.4	8/27	11.9	92.0
Dry	05020801	1	1994	1:12	12.1	8/17	13.1	6/26	13.4	6/23	10.2	1601.0
Dry	05020801	1	1999	1:12	13.2	8/27	14.2	6/24	15.2	6/22	11.3	1601.0
Black Dog	05020802	2	1996	2:24	13.0	7/28	13.5	7/27	13.7	7/28	12.8	503.0
Black Dog	05020802	2	1997	2:24	14.2	9/3	14.4	9/3	14.6	8/26	14.0	503.0
Black Dog	05020802	2	1998	2:30	13.3	7/19	13.8	7/19	14.0	7/16	12.8	503.0
Black Dog	05020802	2	1999	1:12	13.3	8/26	13.6	8/26	13.9	8/26	13.4	503.0
Cañon (low)	05021001	1	1994	1:12	16.7	8/17	18.5	7/16	19.1	7/18	14.3	9869.0
Cañon (low)	05021001	1	1995	1:12	16.9	8/4	18.4	7/29	19.4	7/16	15.5	9869.0
Cañon (low)	05021001	1	1996	1:12	17.7	7/28	19.4	7/6	19.9	7/6	15.1	9869.0
Cañon (low)	05021001	1	1997	1:12	18.8	7/17	21.6	7/17	22.1	7/15	16.7	9869.0
Cañon (low)	05021001	1	1998	0:08	18.5	7/24	20.6	7/18	21.2	7/19	17.0	9869.0
Cañon (low)	05021001	1	1999	1:12	17.6	8/24	18.9	8/24	20.0	6/22	14.7	9869.0
Cañon (low)	05021001	1	2000	1:12	18.2	8/1	20.0	6/26	21.1	6/27	16.1	9869.0
Cañon (mid)	05021201	1	1994	1:12	15.8	8/17	18.0	8/17	18.4	8/14	13.1	8620.0
Cañon (mid)	05021201	1	1999	1:12	16.8	8/27	17.4	8/24	17.8	7/11	15.3	8620.0
Cañon (mid)	05021201	1	2000	1:12	17.9	8/1	19.6	8/1	20.3	8/1	16.6	8620.0
Simpson	05021401	1	1997	1:12	15.3	9/3	16.4	7/21	17.0	8/7	13.9	226.5
Simpson	05021401	1	1999	1:12	15.1	8/27	16.2	8/26	16.8	8/29	14.0	226.5
Mad River Trib. #1	05021601	2	1995	1:12	12.8	8/3	13.3	8/2	13.7	7/31	12.5	30.5
Mad River Trib. #2	05021602	2	1995	1:12	14.1	7/30	15.1	7/30	15.5	7/16	13.1	38.5
Mad River Trib. #3	05021603	2	1995	1:12	12.6	8/4	12.8	8/2	12.8	7/27	12.2	38.4
Mad River Trib. #4	05021604	2	1995	1:12	14.0	8/3	15.3	7/29	16.5	7/31	13.4	74.5
6001CD	05021605	2	1996	1:12	12.0	7/28	12.1	7/28	12.2	7/28	12.0	26.5
6001CD	05021605	2	1997	1:12	12.8	9/3	13.0	8/27	13.1	8/26	12.8	26.5
6001CD	05021605	2	1998	1:12	12.3	8/14	12.5	7/23	12.8	7/26	12.2	26.5
6001CD	05021605	2	1999	1:12	12.1	8/27	12.3	8/26	12.5	8/26	12.3	26.5

Table C5-9 Continued. Summer water temperature monitoring summary, Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
6001CD	05021605	2	2000	1:12	12.6	9/18	12.8	9/18	13.0	9/18	12.7	26.5
6001CU	05021606	2	1996	1:12	11.6	8/30	11.7	8/30	11.9	7/29	11.6	10.2
6001CU	05021606	2	1997	1:12	12.7	9/3	12.8	9/3	12.8	8/29	12.2	10.2
6001CU	05021606	2	1998	1:12	12.1	9/5	12.2	9/3	12.3	7/26	12.0	10.2
6001CU	05021606	2	1999	1:12	12.5	8/27	12.7	8/27	12.9	8/26	12.6	10.2
6001CU	05021606	2	2000	1:12	12.8	9/18	13.1	9/18	13.7	9/18	13.0	10.2
6001TD	05021607	2	1996	1:12	12.7	7/28	13.5	7/27	13.8	7/28	12.4	62.5
6001TD	05021607	2	1997	1:12	13.3	9/3	13.6	9/1	13.9	8/7	12.4	62.5
6001TD	05021607	2	1998	1:12	13.6	8/14	14.9	8/13	15.4	7/26	13.1	62.5
6001TD	05021607	2	1999	1:12	13.4	8/26	14.1	8/24	14.3	8/22	12.5	62.5
6001TD	05021607	2	2000	1:12	13.4	9/17	14.4	7/31	14.7	6/27	12.2	62.5
5410CU	05021608	2	1996	1:12	12.3	7/28	13.1	8/30	13.3	7/29	11.9	854.9
5410CU	05021608	2	1997	1:12	12.6	9/4	13.1	9/4	13.4	9/6	11.6	854.9
5410CU	05021608	2	1998	1:12	12.0	9/5	12.7	9/5	12.9	9/4	11.4	854.9
5410CU	05021608	2	1999	1:12	11.6	8/27	11.8	8/26	12.0	8/29	11.2	854.9
5410TD	05021701	2	1996	1:12	12.2	8/29	12.5	8/30	12.7	8/30	12.2	365.0
5410TD	05021701	2	1998	1:12	12.9	8/14	13.1	8/4	13.3	7/26	12.6	365.0
5410TD	05021701	2	1999	1:12	12.5	8/27	12.7	8/23	12.8	8/21	12.2	365.0
5410TD	05021701	2	2000	1:12	12.7	9/17	13.0	7/31	13.2	9/19	12.7	365.0
5410TU	05021702	2	1996	1:12	12.5	7/28	12.7	7/28	13.0	7/30	12.0	187.9
5410TU	05021702	2	1997	1:12	13.8	9/3	13.9	9/1	14.1	8/29	13.3	187.9
5410TU	05021702	2	1998	1:12	13.1	9/5	13.5	9/3	14.0	9/3	13.4	187.9
5410TU	05021702	2	1999	1:12	13.4	8/26	14.0	8/26	14.3	8/26	14.0	187.9
5410TU	05021702	2	2000	1:12	13.1	9/18	13.5	6/27	14.3	6/27	13.3	187.9
5410CD	05021703	2	1996	1:12	13.6	7/28	14.3	7/28	14.5	7/28	13.3	885.8
5410CD	05021703	2	1997	1:12	14.0	9/4	14.3	9/4	14.5	8/7	13.4	885.8
5410CD	05021703	2	1998	1:12	13.3	8/14	14.2	7/24	14.5	7/26	12.9	885.8
5410CD	05021703	2	1999	1:12	13.3	8/27	13.6	8/27	13.9	8/29	12.6	885.8
5410CD	05021703	2	2000	1:12	13.3	8/1	14.0	7/31	14.3	8/1	12.9	885.8
6001TU	05022101	2	1996	1:12	12.3	7/31	12.5	8/30	12.7	8/12	12.2	43.9
6001TU	05022101	2	1997	1:12	12.6	9/14	12.7	9/14	12.8	9/17	12.5	43.9
6001TU	05022101	2	1998	1:12	12.9	7/25	12.9	7/22	12.9	7/19	12.8	43.9
6001TU	05022101	2	1999	1:12	12.3	8/26	12.8	8/24	13.3	8/26	12.3	43.9
6001TU	05022101	2	2000	1:12	12.4	9/17	13.2	9/18	14.0	9/18	12.2	43.9
Mad River Trib. #5	05022201	2	1995	1:12	13.7	7/30	14.1	8/2	14.6	8/1	13.1	356.7
Mad River Trib. #6	05022202	2	1995	1:12	13.6	7/30	14.0	7/30	14.3	7/31	13.4	242.4
Mad River Trib. #7	05022203	2	1995	1:12	13.5	7/30	13.9	7/29	14.6	7/31	13.4	149.9
Devil	05022301	1	1997	1:12	14.5	9/6	14.5	9/4	14.6	9/6	14.5	1447.6
Devil	05022301	1	1999	1:12	14.5	8/27	15.0	8/27	15.2	8/26	14.6	1447.6

Table C5-9 Continued. Summer water temperature monitoring summary, Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Cañon (class II)	05030801	2	1996	1:12	12.4	8/17	12.9	8/17	13.3	8/24	12.0	260.9
Maple (Mad)	05033001	1	1994	1:12	14.1	8/17	15.5	8/16	15.9	8/14	12.0	7496.0
Maple (Mad)	05033001	1	1996	1:12	16.8	7/28	20.4	7/28	21.1	7/30	14.5	7496.0
Maple (Mad)	05033001	1	1997	1:12	16.4	8/9	19.0	8/6	19.6	8/7	14.6	7496.0
Maple (Mad)	05033001	1	1999	1:12	15.7	8/26	17.6	8/23	18.4	8/29	14.1	7496.0
Mill Cr (Mck.)	06010401	1	1997	1:12	13.3	8/27	13.7	8/27	14.0	9/17	12.3	704.6
Lindsay	06011101	1	1994	1:12	15.9	8/18	16.8	8/17	17.1	8/19	15.2	8811.0
Lindsay	06011101	1	1995	1:12	15.9	7/29	17.1	7/29	17.8	7/16	15.2	8811.0
Lindsay	06011101	1	1996	1:12	15.9	7/28	16.6	7/28	17.2	7/30	15.0	8811.0
Lindsay	06011101	1	1997	1:12	16.1	7/18	16.9	7/17	17.3	7/7	14.8	8811.0
Lindsay	06011101	1	1998	0:08	15.8	8/14	16.8	8/13	17.4	7/15	14.5	8811.0
Lindsay	06011101	1	1999	1:12	15.3	8/24	16.2	8/24	17.1	8/29	14.9	8811.0
Lindsay	06011101	1	2000	1:12	15.4	7/31	16.0	7/31	16.3	8/1	15.0	8811.0

Table C5-10. Summer water temperature monitoring summary, North Fork Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Mule	06020301	1	1996	1:12	13.0	8/30	15.1	8/31	15.6	8/31	11.6	338.7
Mule	06020301	1	2000	1:12	13.4	8/2	14.0	8/1	14.3	8/1	12.9	338.7
Jackson	06020302	1	1998	1:12	13.6	7/25	13.9	7/25	14.2	7/26	13.6	511.0
Jackson	06020302	1	1999	1:12	13.2	8/27	13.6	8/26	13.7	8/29	12.5	511.0
Denman	06020303	1	2000	1:12	15.3	9/17	17.0	9/17	18.1	9/18	14.2	878.2
Long Prairie	06021101	1	1994	1:12	14.2	8/17	15.4	8/17	15.5	8/14	12.8	6231.0
Long Prairie	06021101	1	1998	0:08	14.9	7/24	16.0	7/24	16.6	7/26	14.4	6231.0
Long Prairie	06021101	1	1999	1:12	14.8	8/26	15.6	8/26	16.1	8/29	13.4	6231.0
Gossinta	06021102	1	2000	1:12	15.0	9/17	17.9	9/19	19.7	9/18	13.8	730.9
Pollock	06021401	1	1996	2:24	13.7	7/29	14.0	7/28	14.3	7/29	13.7	1060.3
Pollock	06021401	1	1997	2:24	14.5	9/3	14.9	9/3	15.1	8/7	13.7	1060.3
Pollock	06021401	1	1998	2:30	13.9	8/10	14.6	8/9	15.2	7/23	13.4	1060.3
Pollock	06021401	1	1999	1:12	13.6	8/27	13.8	8/27	13.9	8/29	13.3	1060.3
Bald Mountain	06021402	1	1999	1:12	14.2	8/26	14.7	8/26	14.9	8/29	13.4	3008
Poverty	06021501	1	1999	1:12	14.1	8/26	15.7	8/26	16.4	8/29	10.5	404.4
Jiggs	06022201	2	1996	2:24	12.9	7/28	13.3	7/27	13.4	7/25	12.2	664.9
Jiggs	06022201	2	1997	2:24	13.7	9/3	14.5	8/5	14.6	8/7	13.1	664.9
Jiggs	06022201	2	1998	2:30	12.9	8/10	13.4	7/19	13.7	7/22	12.5	664.9
Jiggs	06022201	2	1999	1:12	13.4	8/26	14.5	8/25	14.8	8/26	13.1	664.9
NF Mad (middle)	06022301	1	1994	1:12	17.1	8/17	18.7	8/17	18.8	8/14	15.5	23462.9
NF Mad (middle)	06022301	1	1999	1:12	17.3	8/26	19.0	7/12	19.6	7/13	14.5	23462.9
NF Mad (middle)	06022301	1	2000	1:12	17.3	8/1	19.8	7/31	20.2	8/1	15.6	23462.9
Jiggs Upper	06022601	2	1999	1:12	12.9	8/26	13.1	8/26	13.7	8/27	13.0	421.5
Sullivan Gulch	06022801	1	1997	1:12	15.2	9/3	15.9	7/16	16.3	7/18	13.9	1536.0
Sullivan Gulch	06022801	1	1999	1:12	14.6	8/27	15.1	8/24	15.9	6/22	12.5	1536.0
Sullivan Gulch	06022801	1	2000	1:12	14.9	7/31	15.6	6/17	16.2	6/27	13.5	1536.0
NF Mad (lower)	06022802	1	1994	1:12	17.7	7/17	20.3	8/16	20.5	8/14	15.5	27634.0
NF Mad (lower)	06022802	1	1995	1:12	18.4	8/3	20.7	8/2	21.5	7/16	16.2	27634.0
NF Mad (lower)	06022802	1	1996	1:12	19.7	7/28	21.4	7/28	21.9	7/30	18.1	27634.0
NF Mad (lower)	06022802	1	1997	1:12	19.5	7/17	22.4	8/5	23.2	8/7	17.2	27634.0
NF Mad (lower)	06022802	1	1998	1:12	18.9	7/23	21.8	8/13	22.6	8/13	17.4	27634.0
NF Mad (lower)	06022802	1	1999	1:12	17.8	8/23	20.4	7/14	21.2	8/22	16.2	27634.0
NF Mad (lower)	06022802	1	2000	1:12	19.0	8/1	21.1	8/1	22.0	6/27	16.5	27634.0

Table C5-10 Continued. Summer water temperature monitoring summary, North Fork Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
NF Mad (site 1a)	06022803	1	1998	1:12	18.8	8/14	22.3	8/13	23.1	8/12	16.6	26613.0
Watek	06023201	1	1996	1:12	12.1	8/28	13.4	8/30	13.7	8/31	10.5	615.8
East Fork North Fork Mad	07022201	2	1996	2:24	14.1	7/28	14.4	7/28	14.6	7/30	13.7	153.7
Canyon (class II)	07022601	2	1996	2:24	14.6	7/28	15.1	7/28	15.2	7/14	14.0	847.5
Canyon (class II)	07022601	2	1997	2:24	14.0	8/10	14.4	8/10	14.8	8/8	13.9	847.5
Canyon (class II)	07022601	2	1998	2:30	14.2	8/30	14.6	7/21	15.5	7/22	13.7	847.5
Canyon (class II)	07022601	2	1999	1:12	13.8	8/26	14.2	8/26	14.6	7/13	12.3	847.5
Canyon (class II)	07022601	2	2000	1:12	13.8	6/27	14.7	6/27	15.8	6/28	13.2	847.5
Canyon RHVA 1	07022701	2	2000	1:12	13.4	9/19	13.7	9/19	14.4	9/20	13.1	28.1
Canyon RHVA 2	07022702	2	2000	1:12	12.0	9/20	12.3	9/20	12.9	9/20	11.8	95.4
NF Mad (upper)	07022801	1	1994	1:12	13.9	8/17	14.9	8/17	15.2	8/20	12.5	5252.6
NF Mad (upper)	07022801	1	1999	1:12	14.5	8/26	15.2	8/26	15.6	8/29	13.3	5252.6
NF Mad (upper)	07022801	1	2000	1:12	14.7	8/1	15.3	8/1	15.9	6/28	13.7	5252.6
Canyon	07022802	1	1997	1:12	14.4	8/9	15.3	8/9	15.9	8/7	13.7	1870.2
Canyon	07022802	1	1998	1:12	14.0	9/4	15.1	9/3	15.6	9/3	13.4	1870.2
Canyon	07022802	1	1999	1:12	14.0	8/26	14.6	8/26	14.8	8/26	14.0	1870.2
East Fork of North Fork	07022803	1	2000	1:12	13.5	8/2	14.1	8/1	14.6	6/28	12.8	1276.6
Krueger	07023401	1	2000	1:12	14.1	8/2	14.5	8/1	14.8	8/1	13.8	708.9
Railroad (NF Mad)	07023402	1	2000	1:12	13.4	8/3	13.5	8/1	13.7	8/1	13.3	545.3

Table C5-11. Summer water temperature monitoring summary, Humboldt Bay HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Salmon (high)	03011801	1	1994	1:12	13.7	8/19	14.1	8/16	14.3	8/19	13.4	3294.3
Salmon (high)	03011801	1	1999	1:12	15.2	8/24	16.7	8/24	17.3	8/22	14.3	3294.3
Salmon (high)	03011801	1	2000	1:12	15.1	7/31	16.0	7/31	16.7	6/27	13.7	3294.3
Salmon (low)	03510901	1	1995	1:12	16.0	7/16	17.2	7/15	18.1	7/16	15.5	7858.0
Salmon (low)	03510901	1	1996	1:12	16.1	7/28	17.3	7/28	17.7	7/28	15.9	7858.0
Salmon (low)	03510901	1	1997	1:12	18.1	8/5	20.5	8/5	20.9	8/3	15.9	7858.0
Salmon (low)	03510901	1	1998	0:08	17.4	7/17	19.3	7/16	20.1	7/15	16.0	7858.0
Salmon (low)	03510901	1	1999	1:12	16.6	8/23	17.7	8/22	18.5	6/22	14.6	7858.0
Salmon (low)	03510901	1	2000	1:12	16.7	7/29	17.8	7/29	18.9	6/27	15.2	7858.0
Salmon (mid)	03511001	1	1994	1:12	15.8	8/19	16.4	8/17	16.8	8/14	14.6	6979.0
Ryan (upper)	04011801	1	1994	2:00	14.7	8/19	15.7	8/20	16.1	8/19	14.6	1154.6
Ryan (upper)	04011801	1	1995	2:00	15.2	7/26	15.9	7/16	16.8	7/27	15.2	1154.6
Ryan (upper)	04011801	1	1997	2:00	14.9	7/18	15.3	7/22	15.9	7/24	13.7	1154.6
Ryan (upper)	04011801	1	1999	1:12	14.9	8/24	15.4	8/24	15.8	8/22	14.3	1154.6
Henderson	04510101	1	1997	2:00	14.2	9/4	14.5	9/4	14.8	9/5	14.2	922.6
Henderson	04510101	1	1999	1:12	13.5	8/27	13.7	8/26	14.0	8/26	13.3	922.6
Henderson	04510101	1	2000	1:12	13.4	7/31	13.5	7/31	13.8	7/31	13.5	922.6
Guptil	04511201	1	1997	2:00	14.9	9/3	15.5	8/26	16.1	8/25	14.9	1146.2
Guptil	04511201	1	1999	1:12	14.2	8/27	14.4	8/27	14.8	8/29	13.9	1146.2
Guptil	04511201	1	2000	1:12	13.8	7/31	14.2	7/31	14.3	7/31	13.9	1146.2
Bear Ryan	04511202	1	1999	1:12	13.5	8/27	13.7	8/27	14.0	8/29	13.3	719.2
Bear Ryan	04511202	1	2000	1:12	13.5	7/31	13.8	7/31	14.1	8/1	13.1	719.2
Ryan, SF	04511302	1	1997	2:00	14.9	7/18	16.5	7/20	17.0	7/18	13.4	1799.2
Ryan, SF	04511302	1	2000	1:12	14.6	7/31	15.1	7/31	15.4	8/1	14.1	1799.2
Morrison	05011401	1	1997	1:12	14.8	9/3	15.3	9/3	15.8	7/18	13.1	575.0
Morrison	05011401	1	1998	1:12	14.1	8/14	15.2	8/13	15.4	8/12	13.6	575.0
Morrison	05011401	1	1999	1:12	13.9	8/27	14.4	8/23	14.6	8/22	13.1	575.0
Morrison	05011401	1	2000	1:12	13.8	9/18	14.3	7/31	14.7	8/1	13.5	575.0
Rocky	05011501	1	1999	1:12	12.4	8/27	12.4	8/27	12.5	8/22	12.2	465.8
Jacoby (low)	05012401	1	1994	1:12	13.5	8/17	14.7	8/16	14.9	8/14	11.7	4345.0
Jacoby (high)	05023001	1	1994	1:12	12.4	8/17	13.9	8/16	14.3	8/14	10.8	1128.4
Ryan (low)	05513601	1	1997	2:00	15.6	9/3	16.3	9/3	16.9	9/5	14.8	7341.1
Ryan (low)	05513601	1	1998	1:12	15.1	8/13	15.7	8/13	16.1	8/13	14.5	7341.1
Ryan (low)	05513601	1	1999	1:12	14.8	8/24	15.3	8/24	15.9	8/22	14.3	7341.1
Ryan (low)	05513601	1	2000	1:12	14.8	7/31	15.3	7/31	15.6	7/28	15.0	7341.1

Table C5-12. Summer water temperature monitoring summary, Eel River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Slater	01510101	1	1997	1:12	14.4	7/8	14.9	7/8	15.3	7/8	14.5	1133.0
Slater	01510101	1	1998	0:08	14.9	7/20	15.5	7/18	15.8	7/18	14.7	1133.0
Slater	01510101	1	1999	1:12	14.1	8/27	14.5	8/12	14.8	8/29	13.4	1133.0
Wilson (VanD)	02012301	1	1997	1:12	15.3	9/3	15.9	9/3	16.3	9/4	15.3	686.2
Wilson (VanD)	02012301	1	1998	0:08	14.3	8/13	15.1	8/13	15.5	8/12	13.6	686.2
Wilson (VanD)	02012301	1	1999	1:12	14.3	8/24	15.0	8/23	15.6	8/29	13.3	686.2
Cuddeback	02012302	1	1998	0:08	14.2	8/13	14.9	8/13	15.2	8/12	13.8	558.0
Cuddeback	02012302	1	1999	1:12	14.0	8/27	14.4	8/23	14.9	8/29	13.4	558.0
Fielder	02012501	1	1999	1:12	14.0	8/26	14.4	8/23	14.6	8/22	13.7	109.5
Fielder	02012501	1	2000	1:12	13.8	7/31	14.2	7/31	14.3	7/29	13.7	109.5
Stevens	02023501	1	1999	1:12	16.4	8/26	19.3	8/23	20.5	8/29	14.6	506.9
Stevens	02023501	1	2000	1:12	16.6	7/31	18.4	7/31	19.0	7/29	15.9	506.9

The nomenclature changed as well. The term MWAT (Mean Weekly Average Temperature) is a specific threshold determined for a particular life stage and species (Armour 1991). MWAT is a fixed value for a specific species, not a field measurement that varies by stream. The more appropriate term is 7DMAVG (Seven-Day Moving Average) which is the highest average temperature out of all possible seven consecutive days. The 7DMAVG may or may not include the absolute maximum temperature or the 7DMMX recorded during the season. The maximum temperature often occurs later in the fall during low flow conditions that coincide with the loss of deciduous canopy and a reduced coastal marine layer influence. During this time of year the daily peaks may be high but the daily average, due to overnight cooling, will be less than the mid summer peaks.

C5.1.2.2 Methods

Simpson continues to use Onset Computer Corporation's temperature data loggers although the HOBO® models are being phased out for a variety of reasons. The reliability of the HOBO® models came into question when calibration of the units began to occur annually. Even with regular maintenance and battery exchanges the thermographs failed more frequently as they aged. Advances in memory capacity and battery life provided for a new model know as a TidbiT®. The TidbiT® has the same accuracy as the HOBO® HTI –05/37^{o}C, 3 years more battery life, almost 18 times more memory and it is water proof. Every thermograph is calibrated (see Appendix D) to confirm its reliability. Individual recorders with identical measurements are used in Paired Watershed BACI experiments (see Objectives and Methods-Class II Paired Watershed Streams below). With the introduction of the TidbiT® the length of deployment became less of a concern yet the primary monitoring window remained from July through September. Early attempts at modifying the recording interval to capture as much data as the thermograph was capable of only produced huge files that were difficult to analyze. For instance a Tidbit® launched at 8-minute intervals (0.13 hours) will record 180 records per day and last 180 days before the memory is full. Analysis again confirmed that an interval of 1.2 hours would capture the necessary details of the diurnal extremes. The recording interval was kept at 1.2 hours.

In addition to the Class I monitoring Simpson began a program of Class II monitoring in headwater streams known to have populations of Tailed Frogs or Torrent Salamanders. All of the methods apply to both classes of streams with a few exceptions. Due to the small size of many of the Class II watercourses the actual placement of the recorders tended to be in deeper water in order to avoid the possibility of late summer dewatering. Also, the Class II sites were frequently associated with other biological monitoring and thus are not necessarily at the lowest point in the sub-watershed.

Other site-specific variables are collected at every temperature-monitoring site or measured from maps, aerial photos or GIS. The inclusion of specific variables will help in the interpretation of the thermograph data. These variables currently include canopy closure, stream aspect, channel dimensions, flow and watershed area. Simpson has cooperated extensively during this period with the Forest Science Project's "*Regional Assessment of Stream Temperatures Across Northern California and Their Relationship to Various Landscape – Level and Site – Specific Attributes*". The previous list of variables and more were collected for and contributed to the FSP for inclusion in the regional temperature analysis.

Simpson has also acquired temperature profiles from other agencies and landowners that have worked within or near the HPAs. Louisiana Pacific (LP) monitored temperature in several Class I watercourses across their ownership in Humboldt County. When Simpson purchased the LP property in 1998, it also acquired these data files along with site location maps dating back to 1994. Simpson and LP were active participants in the Fish, Farm, and Forest Community effort to establish standardized monitoring methods in order to conduct regional temperature evaluations such as the FSP's "*Regional Assessment of Stream Temperatures Across Northern California and Their Relationship to Various Landscape – Level and Site – Specific Attributes*". LP's methods were comparable to Simpson's and as a result their historic data has been assimilated into the database. Many of the LP sites have become some of Simpson's annual monitoring stations. The Yurok Tribal Fisheries Program (YTFP) has extensively monitored the tributaries as well as the main stem of the lower Klamath River. This is a coordinated effort to make the best use of respective resources and avoid repetitive monitoring of specific sites. The YTFP and Simpson share the same monitoring methods and thus resulting data files for the Klamath area. Several agencies such as the California Conservation Corp, California Department of Fish and Game, National Park Service, the United States Fish and Wildlife Service and the US Forest Service have all monitored stream temperature on or near Simpson Property. Unfortunately most of these monitoring efforts are not coordinated with Simpson or potentially have different methods and protocols. As a result these data must be evaluated on a case by case basis as to whether or not to include them in the database.

C5.1.2.3 Results

At the end of the year 2000, Simpson has recorded and/or collected 400 temperature profiles in approximately 108 Class I watercourses and 210 temperature profiles in approximately 70 Class II watercourses. All of these profiles have been processed to calculate the 7DMAVG, 7DMMX, absolute maximum, and the minimum following the maximum temperatures as well as the associated dates of occurrence. Various attributes have been collected for many of these monitoring stations, specifically watershed area. Temperature monitoring data are summarized and shown in Tables C5-2 through 12.

C5.1.2.4 Discussion

The monitoring window from mid-June through mid-September generally captures the seasonal peak 7DMAVG temperature. On occasion 7DMAVG temperatures in late September and early October were documented. In several stream reaches, maximum water temperatures occurred in late September (upper Dominie Creek, lower Savoy Creek, and Upper South Fork Winchuck River) [Smith River HPA]. These late occurring maximum temperatures were probably affected in part by diminishing stream flow, since the photoperiod of maximum daylight hours and sun angle had occurred two months earlier. Also, the geography of coastal northern California may promote the late occurrence of maximum stream temperatures. A dense band of marine fog that often extends up coastal stream courses is common during June and July. By mid-August this marine layer starts to break up, and the rest of the late summer/early fall is generally clear and warm prior to the onset of fall and winter rains. Finally, the deciduous habit of alders and willows in riparian areas may influence late peak temperatures.

Of the 400 Class I records for the period 1994 to 2000, 375 (93.8%) were at or below the "Inter-agency Matrix" suggested MWAT threshold of 17.4°C. Simpson believes that the single MWAT threshold value fails to account for natural variations in water temperature due to geographic location, climatic factors and drainage area of the monitored sub-basin. Also, depending on the method used to test the upper incipient lethal temperature of juvenile salmonids, a critical MWAT can range from 16.8°C to 18.4°C (Armour 1991; Thomas et al. 1986; Becker and Genoway 1979). Stream and watershed specific factors create a wide variation in processes that affect water temperatures (Beschta et al. 1987). The relationship of water temperature and watershed area was examined to help account for the observed natural variation in water temperature. The data suggests that water temperature was positively associated with watershed area and was relatively predictable for watershed areas up to 10,000 acres. Above 10,000 acres, the temperature variation increased probably in response to the complex interacting physical factors (Beschta et al. 1987).

Of the 25 records that were above the suggested MWAT threshold, 17 had watershed areas of more than 10,000 acres above the monitoring site. The 8 records that exceeded the 17.4°C threshold and had watershed areas less than 10,000 acres occurred in 6 different streams. The higher temperatures appear to be caused by either variations in climatic factors or by a flood event that set back the riparian vegetation. For example, in the winter of 1995/1996 Cañon Creek experienced a flood that removed the riparian canopy in the lower reaches of the stream. Prior to the flood in 1994 and 1995 this reach had 7DMAVG temperatures of 16.7°C and 16.9°C, respectively. For the last 5 years following the flood, the 7DMAVG temperatures have exceeded 17.4°C. With the loss of the streamside vegetation, there was a greater proportion of the stream surface exposed to direct solar radiation. Low discharge in this lower reach also exacerbates the high stream temperatures. However, the general trend since the flood has been a gradual recovery of the riparian canopy and a decrease of the highest 7DMAVG stream temperatures.

C5.1.2.5 Conclusions

Simpson believes that a single threshold value fails to accurately represent the natural variation found in water temperature between sites. For this reason, future water temperatures will be evaluated based on the yellow and red light thresholds described in Section 6.3. The expected temperature for a site will be based on its watershed size rather than a generic threshold value applied equally to all streams.

C5.1.3 References

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C5.2 CLASS II PAIRED WATERSHED TEMPERATURE MONITORING

C5.2.1 Retrospective Study

C5.2.1.1 Objectives and Methods

The first study was a retrospective study of water temperature conducted during the summer of 1995. For this study, groups of small headwater streams in close proximity with similar flow, aspect, and geology were selected. One group of streams were direct tributaries of the Mad River, while the other streams within Simpson's ownership were tributaries of Rowdy and Dominie Creeks in the Smith River watershed (Table C5-13). The streams differed in that some flowed through areas that had been recently harvested by clearcutting (cut) on both sides of the stream with Simpson's riparian buffers (standard state regulated widths but minimum 70% total canopy retention) left along the streams, while the other streams (uncut) were located in intact stands of second growth. One stream had only been harvested on one side (1/2 cut), but it was included with the cut group for analysis. In an attempt to see if there was a coastal effect in the results, Simpson also collaborated with the Hoopa Tribal Forestry to conduct the same type of study on similar sized streams within the Hoopa Reservation. A wide variety of silvicultural practices and riparian buffers have been implemented on the Hoopa Reservation over the years, so they selected sites that most resembled Simpson's silviculture and riparian leave standards. HOBO thermographs were placed in a total of 11 cut streams and 10 uncut streams. However, two of the HOBOs in cut streams were placed in reaches that went dry during the study, and one of the HOBOs in an uncut stream was removed by some unknown person during the study. The restrictions of finding comparable sites within the Hoopa Reservation limited the interior area to only three cut and two uncut streams (Table C5-13).

Table C5-13. List of uncut and cut tributaries with watershed area (acres), stream orientation (aspect in ^{o}), adjacent stand age (years for uncut, feet for cut), and cover type (RW=redwood, DF=Douglas-fir), mean and mean maximum water temperature (^{o}C) with standard deviations.[1]

Uncut	Area	Aspect	Adjacent Stand	Mean Temp	Mean Max.
MR #4	74	46	70, RW	13.2 (1.05)	14.7 (0.73)
MR #5	338	19	70, RW	12.8 (0.60)	13.7 (0.38)
MR #7	160	344	70, RW	12.5 (0.63)	13.6 (0.46)
Rowdy #2	28	291	35-40, RW	12.7 (0.39)	13.1 (0.50)
Rowdy #3	78	159	35-40, RW	12.1 (0.45)	12.6 (0.55)
Dominie #3	46	345	45-50, RW	12.9 (0.91)	14.4 (1.01)
Dominie #4	7	210	45-50, RW	12.9 (0.79)	14.0 (1.00)
Hoopa #1	28	30	35-40, DF	13.5 (0.57)	14.1 (0.82)
Hoopa #6	338	100	*10-15/OG, DF	12.2 (1.23)	13.3 (1.46)
Cut	**Area**	**Aspect**	**Adjacent Stand**	**Mean Temp**	**Mean Max.**
MR #1	28	39	1400	12.4 (0.42)	13.0 (0.31)
MR #2	46	24	1900	13.2 (0.73)	14.7 (0.44)
MR #3	38	15	**1100/70, DF	12.2 (0.23)	12.6 (0.21)
MR #6	234	6	2700	12.8 (0.56)	13.7 (0.33)
Rowdy #1	22	255	1200	12.5 (0.64)	13.4 (0.83)
Dominie #1	37	298	1000	12.5 (0.62)	13.3 (0.74)
Hoopa #2	46	22	1500	13.3 (1.45)	14.6 (1.82)
Hoopa #3	38	107	1000	11.8 (1.01)	12.9 (1.14)
Hoopa #5	234	80	600	11.1 (0.55)	11.6 (0.70)

Notes

1 For cut tributaries, all variables are the same except that the adjacent stand description is replaced with the length (feet) of clearcut on both sides of the stream. Cover types of the riparian buffers of the cut tributaries were presumed to be the same as the corresponding uncut tributaries.

* West side was 10-15 year old second growth and the east side was old growth.

** West side was clearcut and the east side had 70 year old second growth.

In all cases, HOBOs were placed at the lower end of the cut unit, or in the same respective location on the uncut streams. Prior to placement, the HOBOs for each region were tested in a water bath to insure that they were all giving readings that were within the manufactures specified limits (plus or minus 0.2^0 C) relative to each other. However, they were not calibrated to a known standard (ice bath) to insure that the readings were accurate. For each region, the seven consecutive warmest days of the season were selected and the mean maximum and overall mean water temperatures for the period were calculated. Differences between means and variances of the two groups of streams were tested using a two-sample t-test (NCSS 1997).

C5.2.1.2 Results

Visual inspection of HOBO data output from the two groups of streams did not reveal any consistent trends. The coldest streams with the least daily variation appeared to be Mad River #3 (1/2 cut), Rowdy #3 (uncut) and Hoopa #5 (cut). The warmest streams with the greatest daily extremes in temperature were Mad River #4 (uncut), Dominie #3 (uncut), Dominie #4 (uncut) and Hoopa # 2 (cut). In general, a visual ranking of all of the streams would indicate that prior timber harvesting did not correlate well with either the mean values or amount of variation in stream temperatures. Analysis of the data also indicated that there was no significant difference between the mean maximum (t = 0.74, d.f. =16, P = 0.471) or overall mean (t = 1.34, d.f. = 16, P = 0.199) temperatures for the cut and uncut groups (see below).

Stream Groups	N	Mean Temp (^{0}C)	S.E.	Mean Max. (^{0}C)	S.E.
Uncut	9	13.51	0.192	14.19	0.283
Cut	9	13.11	0.227	13.85	0.352

There were too few streams available to make a meaningful comparison of uncut and cut streams in the more interior Hoopa Reservation, but a comparison was made between all coastal and all interior (Hoopa) streams. The temperatures of the five Hoopa streams (mean max. = 14.25; overall mean = 13.33) were similar to the 13 coastal streams (mean max. = 13.93; overall mean = 13.30), with no significant difference (mean max: t = 0.68, d.f. = 16, P = 0.508; overall mean: t = 0.94, d.f. = 16, P = 0.363).

This retrospective comparison of stream temperatures in cut versus uncut streams provided evidence that timber harvest was not having a substantial impact on stream temperature. Increasing the sample size of the two groups would have increased confidence in the conclusion that as a group, streams with riparian buffers on Simpson's ownership were not warmer than streams that were flowing through uncut areas. However, it did not permit a comparison of more subtle changes in stream temperature following timber harvesting. Since the inherent differences in stream temperatures between the two groups of streams was not known prior to harvesting, it was not possible to directly assess the changes that might have occurred. Due to the fundamental limitations of a retrospective study, Simpson concluded that continuing these comparisons between cut and uncut streams would provide little additional information and discontinued the study.

C5.2.2 Before-After-Control-Impact (BACI) Water Temperature Study

C5.2.2.1 Objectives and Methods

In summer 1996, Simpson initiated a monitoring program in non-fish bearing (Class II) watercourses to assess the adequacy of riparian buffers in maintaining water temperatures following timber harvest. Streams in areas where timber harvest was planned were identified and paired with streams in close proximity that had similar size, aspect, and streambed geology. The objective of this study was to examine the impact of timber harvest on water temperature in small Class II watercourses by comparing maximum temperature differentials between fixed upper and lower points of selected stream reaches. These temperature differentials were measured on matched pairs of streams, one member of which was scheduled for timber harvest, while the other was to be left undisturbed. The paired stream design was adopted to control for confounding factors that can influence water temperature such as ground water inputs and microclimatic factors. Measurements were initiated in both streams of a pair at least one year prior to timber harvest. These data represent a before-after-control-impact (BACI) (Green 1979; Stewart-Oaten et al. 1986; Skalski and Robson 1992) observational study. While observational studies cannot infer cause and effect relationships, BACI studies represent the best available setup for detecting changes after disturbance. Monitoring of the stream pairs is scheduled to continue at least three years after harvest, or until the temperature profile of the two streams return to the pre-treatment pattern. However, the data reported here only represent a preliminary assessment of data collected from 1996-1998. Analysis of 1999 and 2000 data is currently in progress.

For each pair of streams, the stream located in a future harvest unit was designated as the “treatment” stream, while the other stream was designated as the “control” stream. Two remote temperature data loggers were placed in the treatment stream at the upstream and downstream edges of the harvest unit. Another pair of temperature recording devices was placed in the control stream at locations that were similar in stream spacing (distance apart) and watershed position relative to the treatment stream. Treatments consisted of clearcuts placed on both sides of the stream with standard forest practice buffer widths (50-75 feet) and 70% total canopy retention. Each steam pair is referred to as a *site.*

The five sites selected in 1996 include:

- One pair in the headwaters of Dominie Creek (D1120) in the Smith River HPA ;
- One pair of tributaries to the South Fork Winchuck River (D1120 in the Smith River HPA ;
- One pair in the headwater tributaries of the Little River (Mitsui) in the Little River HPA;
- One pair off the mainstem Mad River in the Mad River HPA; and
- One pair in the headwater tributaries of Dominie Creek in the Mad River HPA.

In 1999, three pairs were added to the study:

- Two pairs of tributaries to Maple Creek (Windy Point and M1) in the Mad River HPA; and
- One pair of tributaries to the Lower South Fork Little River (M155) in the Little River HPA.

Timber harvest at Mitsui and D2010 took place in winter 1996/1997. Timber harvest at 6001 and 5410 took place in winter 1997/1998. As of winter 1999/2000, timber harvest had not yet occurred at D1120. Timber harvest at Mitsui and D2010 took place in winter 1996/1997. Timber harvest at 6001 and 5410 took place in winter 1997/1998. The Maple Creek units where harvested in winter 1999/2000. As of winter 1999/2000, timber harvest had not occurred at D1120 or the Lower South Fork unit.

The study is still in its data collection phase on pairs where the treatment site was harvested after 1999 or has yet to be harvested. However, a preliminary analysis has been conducted of data from the four pairs harvested before 1999 (Mitsui, D2010, 6001, and 5410).

As indicated in Table C5-14, mean length of control and treatment reaches on Mitsui, D2010, 6001, and 5410 was 1069.2 feet (SE = 515.71) and 1210.2 feet (SE = 650.63), respectively. Mean percent canopy closure following timber harvest was 79.8 (SE = 5.27) and 75.2 (SE = 3.70) for control and treatment reaches, respectively, but the difference was not statistically significantly ($P < 0.05$) using a one-tailed paired t-test ($t = 1.73$, d.f. = 3, $P = 0.091$).

The upstream and downstream placement of temperature recording devices allowed measurement of the temperature differential across the treatment area and an assessment of the extent to which water temperature changed as it flowed through the treatment area. Interest was primarily in quantifying increases in water temperature as it flowed through the treatment area compared to similar measurements in the control stream reach.

Temperature recording devices were calibrated prior to deployment. For calibration, all data loggers (mostly HOBOs initially and later TidbiTs) were placed in an ice bath and temperature readings were taken after three hours. Pairs of data recorders for upstream and downstream deployment on the same stream were formed by pairing instruments with identical readings after three hours in the ice bath. The manufacturer's specification limit was 0.2 ^{o}C . All recorders were programmed to record temperature (^{o}C) every 1.2 hours or 20 times every 24 hours. For this analysis, data were recorded on five pairs of streams.

Table C5-14. Initial five pairs in the Class II BACI study, with stream reach length, mean canopy closure throughout the reach, and aspect.

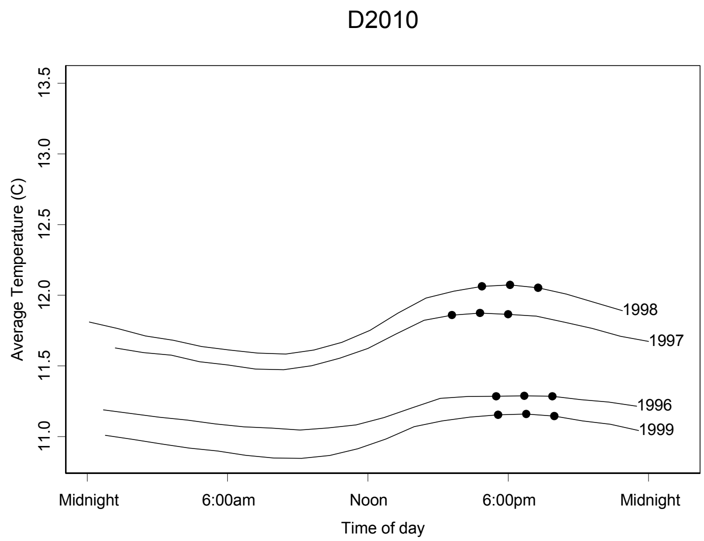
Stream (Drainage)	Type of Treatment	Reach Length (ft)	Canopy Closure (%)	Aspect (°)
5410 (Dry Creek)-	Control	1755	81	320
5410 (Dry Creek)-	Harvested	2090	73	0
6001 (Mad River)+	Control	541	74	10
6001 (Mad River)+	Harvested	764	69	55
Mitsui (Little River)-	Control	856	70	285
Mitsui (Little River)-	Harvested	1312	73	330
D1120 (Dominie Creek)	Control	1605	95	185
D1120 (Dominie Creek)	Scheduled for harvest	*1625	95	200
D2010 (SF Winchuck)+	Control	1125	94	345
D2010 (SF Winchuck)+	Harvested	675	86	350

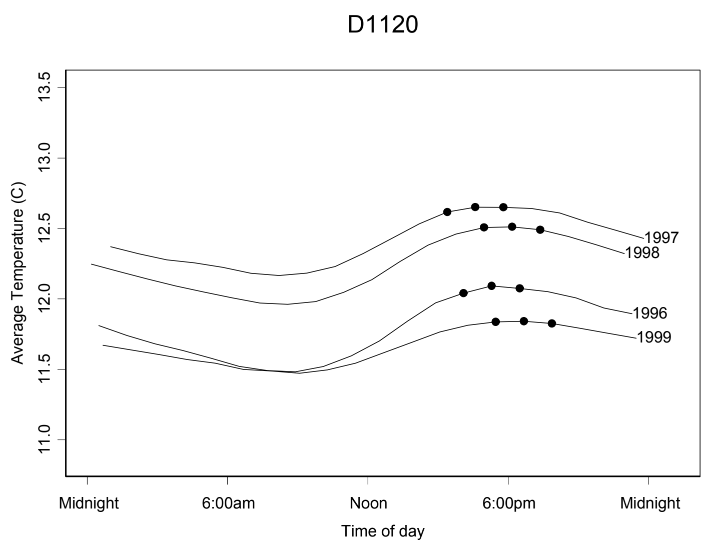
Note
*Asterisks on the reach length for the D1120 indicate the expected length of stream that will be adjacent to the scheduled harvest.

Data loggers were deployed in all streams by early summer each year and collected after 15 September. For analysis, attention was restricted to the time period 1 August to 15 September. This time period is generally the warmest time of year in Northern California. Upstream and downstream temperatures collected on a single stream were matched according to the time of day they were recorded and the difference between them (downstream - upstream) was calculated every 1.2 hours. To identify a response variable that quantified the amount of heat gain produced in the treatment area, intra-day temperature profiles were computed that identified the warmest time of day for each stream each year. The three temperature readings closest to the warmest time of day for each stream were defined to be the *maximum temperature window*. The intra-day temperature profiles used to define the maximum temperature window and, consequently, the daily maximum temperature differences appear in Figure C5-1. In Figure C5-1, values from all four temperature probes (i.e., the upstream and downstream probes on both the treatment and control streams) were averaged every 1.2 hours to arrive at an estimate of overall average water temperature. The three readings that defined the maximum temperature window for each stream each year have been plotted as circles in Figure C5-1. Across streams and years, the maximum temperature window varied from 2:00 pm to 9:07 pm. The warmest time of day for the five study sites was, on average, 5:45 pm.

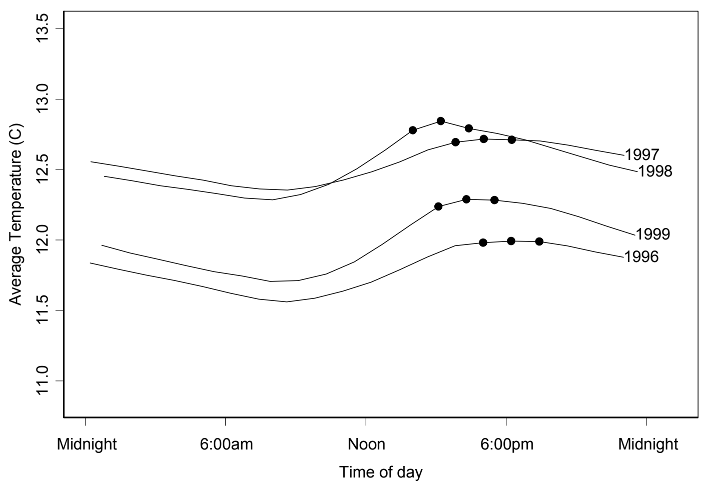
The maximum downstream – upstream temperature difference that occurred within the daily maximum temperature window was computed and used as the response variable in the BACI analysis. For example, suppose that the three temperature readings nearest to the warmest time of day at a stream occurred at 5:00 pm, 6:12 pm, and 7:24 pm. For each day between 1 August and 15 September, the difference between the downstream and upstream probe at 5:00 pm, 6:12 pm, and 7:24 pm was computed. The maximum of these three differences was used as the response variable in the BACI analysis for that particular day. One maximum difference was computed for each day.

Figure C5-1. **Initial five study sites shown below with smoothed daily water temperature profiles computed from the mean of all four temperature probes (i.e. upstream and downstream from the treatment and control streams). Dots show recordings defining the daily maximum temperature window for each site.**

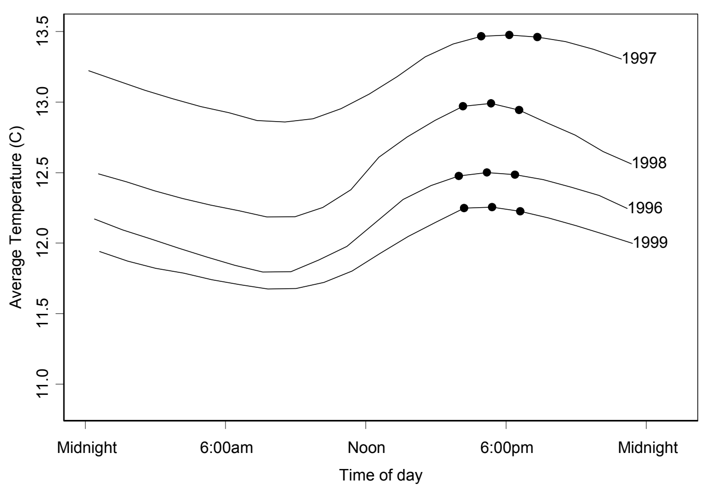


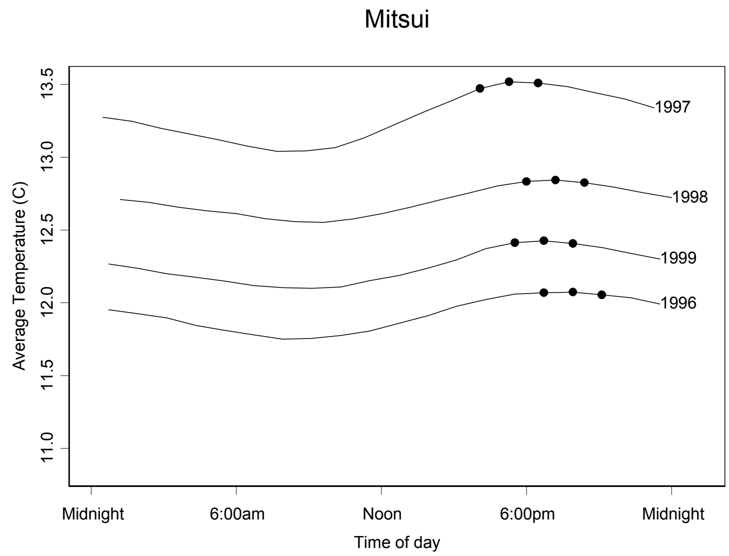


6001


13.5
13.0
12.5
12.0
11.5
11.0
Average Temperature (C)
1997
1998
1999
1996
Midnight
6:00am
Noon
6:00pm
Midnight
Time of day

5410


13.5
13.0
12.5
12.0
11.5
11.0
Average Temperature (C)
1997
1998
1996
1999
Midnight
6:00am
Noon
6:00pm
Midnight
Time of day



Given the serial nature of the daily temperature recordings, the data were assessed for temporal auto-correlation. Significant auto-correlation existed in the yearly time series of maximum temperature differentials at each site. Where significant auto-correlation was found, error estimates were adjusted to correct for the estimated auto-correlations. (See Attachment A below for details.)

The statistical analysis used to assess harvest impacts was a modified BACI analysis. The modification was made necessary due to the estimated auto-correlations in the daily temperature recordings. BACI analyses assess the lack of parallelness in response profiles through time. This lack of parallelness was measured by the treatment by time (year) interaction from an analysis of variance (ANOVA) with time as one factor and treatment as the other. The BACI analysis allows the overall level of responses to be different between control and treated sites both before and after treatment, but requires the after treatment *difference* in control and treated responses to be the same as the before treatment *difference* in control and treated responses. If the after treatment difference in responses is different from the before treatment difference in responses, the BACI analysis will show that there was significant change in treatment areas after application. Differences between sites in the direction and magnitude of temperature changes after harvest became apparent upon plotting of the data. Given the variability in which individual streams responded to the treatment, each site was analyzed separately and no statistical inference to other sites was possible. Additional information on the use of ANOVA in the BACI estimation process can be found in McDonald et al. (2000). Additional details specific to this study can be found in Attachment A below.

C5.2.2.2 Results of Preliminary Analysis

Significant auto-correlation existed in the yearly time series of maximum temperature differential at each site. Estimated correlation of maximum temperature differential values that were one day apart ranged from 0.49 at D1120 to 0.81 at 5410. Auto-correlation at D2010, D1120, and 6001 was negligible between values separated by more than 5 days. Auto-correlation at 5410 and Mitsui was negligible between values separated by 13 or more days.

Table C5-15 contains estimated mean maximum temperature difference and standard errors between the downstream and upstream temperature probes for all streams each year of the study. Means and standard errors in Table C5-15 were estimated from the BACI model adjusting for auto-correlation. Positive values indicate that the average maximum downstream temperature was warmer than the upstream temperature, while negative numbers indicate the reverse. Average heating or cooling between the upstream and downstream probes was variable.

Table C5-16 contains estimated average maximum temperature differences before and after timber harvest. (D1120 is missing from Table C5-16, because it had not yet been harvested.) After harvest, D2010 and 6001 experienced an increase in the maximum temperature differential, while Mitsui and 5410 experienced a decrease relative to their control streams. The 95% confidence intervals for the increases at D2010 and 6001, and decreases at Mitsui and 5410 did not include zero and therefore should be considered "significantly" different from zero.

D1120 was not harvested during the course of data collection and provided a check of the appropriateness of BACI analysis. Under similar conditions, the BACI analysis hypothesizes that the profile of temperature responses through time on the treatment and control streams should, within statistical error, be parallel to one another. Figure C5-2 plots the estimated profile of average maximum temperature differential across years for D1120. Assuming a hypothetical harvest occurred in winter 1996/1997, the estimated change in maximum temperature differential on the hypothetical treatment stream was 0.013^{o}C with approximate 95% confidence interval of -0.149^{o}C to 0.175^{o}C. Applying the same hypothetical treatment to the following year, the estimated change in maximum temperature differential on the hypothetical treatment stream was -0.082^{o}C with approximate 95% confidence interval of -0.223^{o}C to 0.058^{o}C. The profiles plotted in Figure C5-2 are parallel within the limits of statistical error, because the associated confidence intervals contain zero.

Plots of the estimated mean maximum downstream-upstream differences from Table C5-15 were plotted in Figure C5-3 below along with the average maximum temperature differential expected by the BACI analysis had there been no harvest. With no treatment effect, the expected mean treatment profiles were parallel to the control stream profile.

Table C5- 15. Yearly estimated mean maximum downstream-upstream temperature differences of the initial five sites. [1]

Mean Maximum Downstream-Upstream Temperature Difference, °C			
Stream	**Year**	**Treatment Stream (SE)**	**Control Stream (SE)**
D2010	1996	0.839 (0.101)	0.991 (0.101)
	1997	1.601 (0.101)	1.436 (0.101)
	1998	1.705 (0.101)	1.029 (0.101)
	1999	1.288 (0.101)	1.234 (0.101)
D1120	1996	0.952 (0.051)	0.175 (0.051)
	1997	1.300 (0.051)	0.393 (0.051)
	1998	0.977 (0.051)	0.136 (0.051)
	1999	0.764 (0.051)	0.176 (0.051)
6001	1996	0.392 (0.087)	0.240 (0.087)
	1997	0.787 (0.087)	0.293 (0.083)
	1998	1.484 (0.087)	0.226 (0.083)
	1999	1.227 (0.088)	-0.243 (0.088)
5410	1996	0.316 (0.099)	1.227 (0.099)
	1998	-0.026 (0.095)	1.423 (0.095)
	1999	-0.041 (0.101)	1.480 (0.101)
Mitsui	1996	-0.146 (0.125)	-0.071 (0.125)
	1997	-0.928 (0.125)	0.135 (0.125)
	1998	-1.294 (0.125)	0.007 (0.125)

Note
1 All measurements in Celsius. Standard errors estimated from BACI model.

Table C5-16. Estimated average maximum temperature differences before and after harvest on four sites where harvesting occurred prior to 1999.[1]

Stream	Harvest Period	Estimated Average Maximum Temperature Difference, °C		Estimated Change After Harvest, °C (SE)	Approximate 95% Confidence Interval on Increase
		Treatment (SE)	Control (SE)		
D2010	Before	0.756 (0.098)	0.898 (0.098)	0.497 (0.16)	0.182 to 0.811
	After	1.515 (0.057)	1.16 (0.057)		
6001	Before	0.535 (0.061)	0.139 (0.061)	1.044 (0.123)	0.803 to 1.286
	After	1.323 (0.062)	-0.117 (0.062)		
5410	Before	0.178 (0.139)	0.486 (0.139)	-1.372 (0.239)	-1.84 to -0.904
	After	-0.368 (0.096)	1.312 (0.096)		
Mitsui	Before	-0.214 (0.129)	-0.222 (0.129)	-1.31 (0.224)	-1.748 to -0.871
	After	-1.28 (0.091)	0.022 (0.091)		

Note
1 Values of change after harvest (Column 5) quantify the lack of parallelism in temperature differential profiles and are equal to the interaction effects in the BACI ANOVA. For example, at D2010 estimated change after harvest equaled 0.497 = (1.515-0.756)-(1.16-0.898). Positive numbers for change after harvest indicate heating of the treatment section after harvest relative to the control section. Negative numbers indicate cooling of the treatment section after harvest relative to the control section.

Figure C5-2. Estimated means at D1120 where no harvest has occurred. Hollow circles and dashed line indicate perfect parallelness between treatment and control streams. Filled circles show actual estimates.

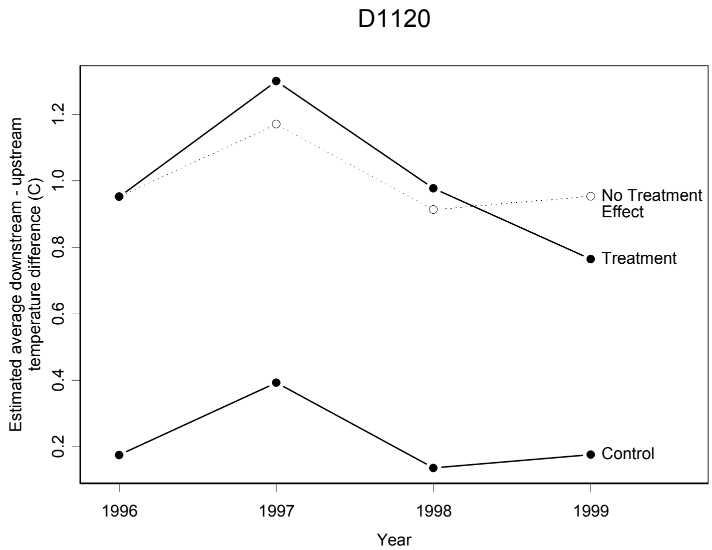
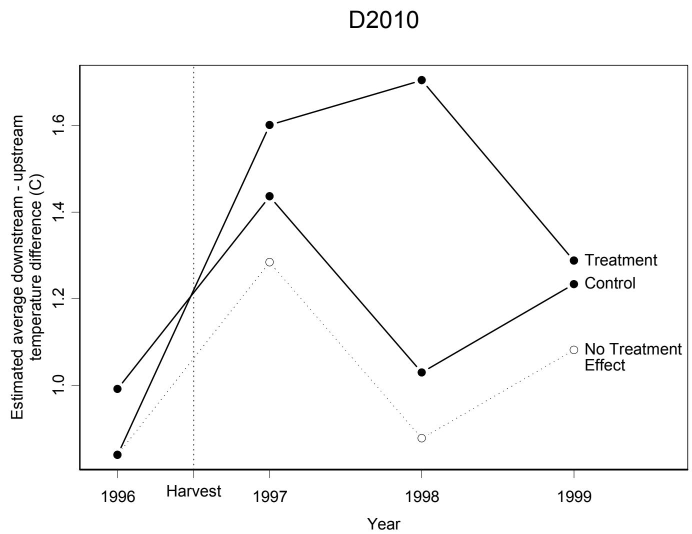
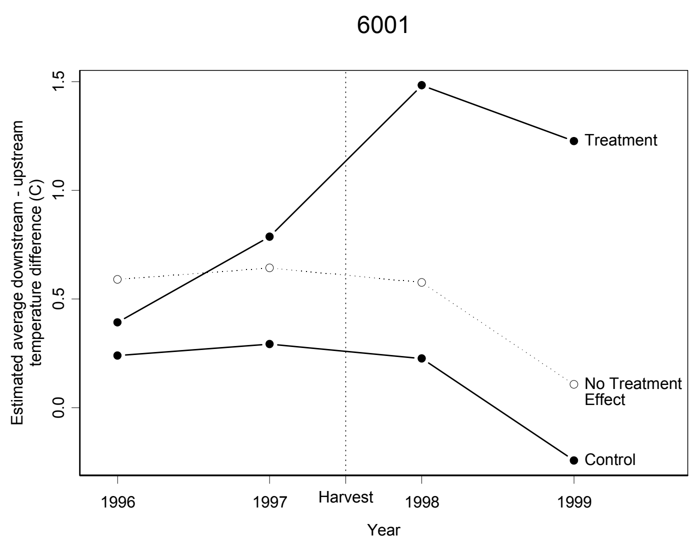
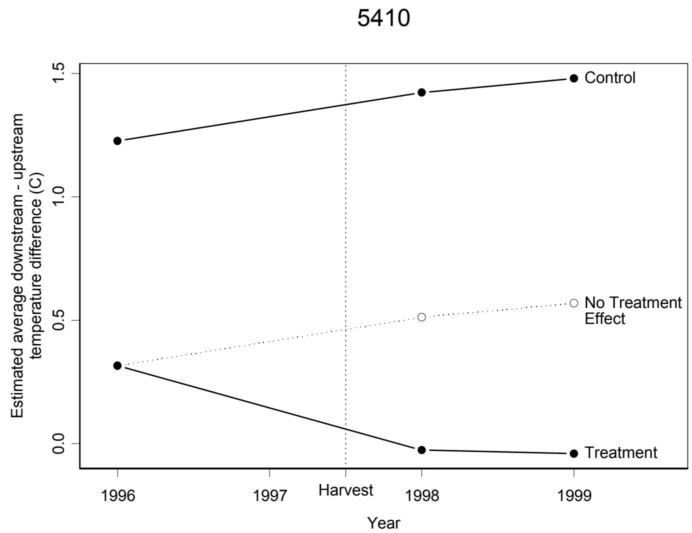
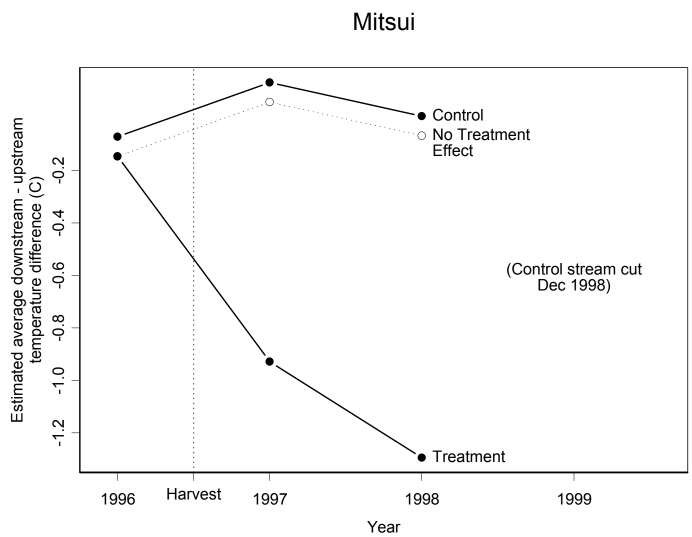


Figure C5-3. Estimated means before and after harvest from the BACI model adjusted for auto-correlation. Filled circles show actual estimates, while hollow circles show locations of treatment means under the hypothesis of no treatment effect. Monitoring stopped in 1998 at Mitsui, because timber surrounding the control stream was harvested during winter 1998.






5410
Estimated average downstream - upstream temperature difference (C)
0.0
0.5
1.0
1.5
Control
No Treatment Effect
Treatment
1996
1997
Harvest
1998
1999
Year


Mitsui
Estimated average downstream - upstream temperature difference (C)
-0.2
-0.4
-0.6
-0.8
-1.0
-1.2
Control
No Treatment Effect
(Control stream cut Dec 1998)
Treatment
1996
Harvest
1997
1998
1999
Year

C5.2.2.3 Discussion

The impacts of timber harvest on water temperature on small Class II watercourses were assessed at the warmest time of day during the warmest time of the year. This was done to insure the maximum test of the effectiveness of riparian buffers in mitigating the potential impacts of increased water temperatures following clearcut timber harvest adjacent to a watercourse. In addition, the assessment was focused on the warmest time of the year, since it is believed that the Covered Species are most likely to be impacted by increases in water temperature that may cause water temperature to exceed some biological threshold. It is also important to note that the retention standards on the riparian buffers were significantly less than what is being proposed in the AHCP. The riparian buffers all followed standard forest practice widths, but with Simpson's minimum 70% total canopy retention (retention standard created by Simpson's NSO HCP).

Empirical data and theoretical models of water temperature profiles indicate that water temperature generally increases in most watersheds as water flows downstream during the warmest times of the year (Beschta et al. 1987). Increases in the water temperature are the result of multiple factors, but typically most of the increased thermal energy of the water results from the air temperature being elevated relative to the water temperature. The rate of increase is largely a function of the temperature differential between air and water. Therefore, if air temperature increases in the riparian areas following timber harvest, one would predict an increase in the rate at which water temperature warms as it flows downstream through the harvested area.

The thermal profiles of the monitored streams indicated that the changes in water temperature as it flowed downstream was a rather complex process and did not always fit the pattern of increased warming as water flowed downstream. As noted in Table C5-16, mean water temperature decreased rather than increased as it flowed downstream during at least one year in four of the ten streams. Monitoring reaches were selected to insure that tributaries did not enter within the sample reach, so these decreases were most likely due to ground water inputs or changes in the microclimate within the stream reach.

Fortunately, this study was designed using a BACI approach, which controlled for unexpected patterns in the thermal profiles of either the treatment or control streams. All that was necessary for a valid experiment was for the relationship between treatment and control streams to remain constant through time minus a treatment effect. The results from the D1120 (Figure C5-2) provided support that this assumption was valid.

The data from this study are preliminary, but already it is apparent that the response of water temperature to timber harvest in small headwater streams is complex. All of the treatment streams showed a significant change in water temperature relative to the controls streams following timber harvest, but in two of the sites, the treatment streams were warmer while the other two were colder. There are no other data to help provide clues as to why these sites responded in opposite directions to timber harvest, but Simpson speculates that it may be due to altered hydrology. Clearcutting adjacent to a stream should increase the amount of water that is retained in the soil for a few years following harvest primarily due to a reduction of evapotranspiration water losses. If some treatment streams had groundwater inputs while others did not, it would be possible that the increased groundwater could result in relatively cooler water temperatures following harvest in those treatment streams with groundwater inputs. Those treatment streams without significant groundwater inputs would have the greater potential to experience increases in water temperature following harvest. If this pattern persists in additional monitored sites, one would conclude that the cumulative effect of timber harvest on water temperature in small Class II watercourses within a watershed should net to zero.

The retrospective study of water temperature did not allow us to assess changes in water temperature following timber harvest, but the results were consistent with the observations of the BACI study. Cut and uncut streams varied in terms of which streams were colder and there was no statistical difference in the mean values for the streams.

It is also important to note that the magnitude of the differences following harvest, regardless their direction, were quite small (about 0.5 to 1.4^{o}C) even thought the streams were being analyzed during the annual extremes in elevated water temperatures. In addition, the peaks in water temperature only lasted a few hours in the late afternoon and early evening. Simpson believes that it is unlikely that the magnitude of these temperatures would have a biological impact on any of the Covered Species given the 7DMMX reported for most of the Class II watercourses within the Plan Area. (See Water Temperature Monitoring above.)

C5.2.2.4 Conclusions

The Class II water temperature monitoring is in the early phases of a long term study that will include additional sites along with additional post-harvest monitoring on the existing sites. As such, these data should be considered preliminary. However, pre-AHCP mitigation measures associated with small Class II watercourses appear to prevent large magnitude changes in water temperature following timber harvest. Presumably, the increased protection measures afforded Class II watercourses in the AHCP will further reduce the likelihood of temperature impacts due to timber harvest. Simpson believes that the small magnitude and reversed direction of the temperature changes following timber harvest will not result in any direct or cumulative biological impacts on any of the Covered Species.

C5.2.3 References

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C5.2.4 Attachment A to BACI Class II Temperature Monitoring

This attachment describes estimation of the BACI model and correction for auto-correlation in the data. The analysis is described in three steps; 1) ordinary least squares parameter estimation, 2) auto-correlation modeling, and 3) weighted linear regression.

C5.2.4.1 Ordinary Least Squares Parameter Estimation

Step one of the analysis fit a Normal theory regression model to indicator variables delineating treatment and control observations and before and after observations. Let x_{ti} be an indicator variable whose value was 1 if observation i came from the treatment stream, 0 otherwise. Let x_{97i} be an indicator variable whose value was 1 if observation i was collected in 1997, 0 otherwise. Similarly, let x_{98i} be an indicator function with value 1 if observation i was collected in 1998 and let x_{99i} be an indicator function with value 1 if observation i was collected in 1999. Step one of the analysis fit the regression model,

$$\mathbf{E[y_i]} = \boldsymbol{\beta_0} + \boldsymbol{\beta_1}\,\mathbf{x_{ti}} + \boldsymbol{\beta_2}\,\mathbf{x}_{97i} + \boldsymbol{\beta}_3\,\mathbf{x}_{98i} + \boldsymbol{\beta}_4\,\mathbf{x}_{99i} + \boldsymbol{\beta}_5\,\mathbf{x_{ti}}\,\mathbf{x}_{97i} + \boldsymbol{\beta}_6\,\mathbf{x_{ti}}\,\mathbf{x}_{98i} + \boldsymbol{\beta}_7\,\mathbf{x_{ti}}\,\mathbf{x}_{99i}$$

where y_i was the maximum difference between downstream and upstream temperature readings on day i that occurred during the maximum temperature window.

Estimates of the overall before-after control-impact interaction (i.e., the difference of differences in means) were computed using contrasts of coefficients in the model (McDonald et al., 2000). For example, the overall BACI contrast for a pair of streams harvested in winter 1996/1997 was,

$$\mathbf{BACI}_{96} = -1/3\boldsymbol{\beta}_5 - 1/3\boldsymbol{\beta}_6 - 1/3\boldsymbol{\beta}_7$$

The overall BACI contrast for a pair of streams harvested in winter 1997/1998 was,

$$\mathbf{BACI}_{97} = 1/2\boldsymbol{\beta}_5 - 1/2\boldsymbol{\beta}_6 - 1/2\boldsymbol{\beta}_7$$

Let μ_{BT} be the mean response on the treatment stream before treatment. Let μ_{BC} be the mean response on the control stream before treatment. Let μ_{AT} be the mean response on the treatment stream after treatment, and let μ_{AC} be the mean response on the control stream after treatment. The BACI contrasts listed above both estimate,

$$(\mu_{BT} - \mu_{BC}) - (\mu_{AT} - \mu_{AC})$$

The negative of these BACI contrasts appear in column 5 of Table C5-16 above.

C5.2.4.2 Auto-correlation Modeling

Step two of the analysis assessed and modeled auto-correlations among residuals of the regression fit during step one. No auto-correlations were checked among residuals from different streams or different years. Auto-correlations among residuals from different stream or years were assumed to be zero. If significant auto-correlation were found in the residuals of the regression model, a non-linear variance model was fit to the correlations and an estimated residual variance-covariance matrix was constructed. The variance model used at this step was of such a form that non-singularity of the resulting variance-covariance matrix was assured.

The significance of auto-correlations among residuals of the original model were assessed using Moran's I (Moran, 1950) statistic at various separations in time (time lags). If a (Bonferronni corrected) 95% confidence interval surrounding Moran's I did not overlap zero, the auto-correlation was deemed significant.

Provided significant auto-correlations existed, a *spherical* correlation model was fit to observed correlations. The spherical variance model was fit by forming all possible pairs of residuals and calculating the statistics $z_{ij} = (r_i - \mu_r)(r_j - \mu_r)/s_r^2$, where r_i was the model residual from the *i*-th observation and s_r^2 the sample variance of the residuals. The z_{ij} were then plotted against the time between observation *i* and observation *j* to form a correlation scatter gram. The correlation scatter gram was then smoothed using a Gaussian kernel smoother (Venables and Ripley, 1994; Statistical Sciences, 1995). The spherical correlation model was fit to the smoothed correlation scatter gram using non-linear least squares techniques. The spherical correlation model contained two parameters and had the form,

$$v(d_{ij}) = \left\{ \begin{array}{c} c_1\left(1 - \frac{3}{2}\frac{d_{ij}}{h_0} + \frac{1}{2}\left(\frac{d_{ij}}{h_0}\right)^3\right) \text{ if } 0 \le \mathrm{d_{ij}} \le \mathrm{h_0} \\ 0 \text{ if } \mathrm{d_{ij}} > \mathrm{h_0} \end{array} \right\}$$

where d_{ij} was the time between observation i and j. Based on the significance of auto-correlations beyond 20 days, only d_{ij} less than 20 days were considered when fitting the spherical model.

C5.2.4.3 Weighted Linear Regression

Step three of the analysis used the estimated residual variance-covariance matrix from step 2 as a weight matrix to recompute the coefficients of the regression model obtained at step one. Standard errors for coefficients and contrasts were also recomputed using elements of the estimated variance-covariance matrix as weights. Specifically, if **X** was the design matrix containing the indicator variables used in the regression model at step one, **Y** was the vector of responses, and **V** was the estimated residual variance-covariance matrix obtained at step two, then the recomputed vector of coefficients, $\hat{\beta}$, and variances were,

$$\hat{\beta} = (X'V^{-1}X)^{-1}X'V^{-1}Y$$
$$\mathrm{var}(\hat{\beta}) = (X'V^{-1}X)^{-1}.$$

In this study, interest was in the BACI contrasts defined above. Variance of the BACI contrasts were computed as,

$$\mathrm{var(BACI)} = \mathrm{var}(x\hat{\beta}) = x(X'V^{-1}X)^{-1}x'$$

where **x** was the vector of constants defining the contrast.

Appendix C6. Fish Presence/Absence Surveys

CONTENTS

C6.1 INTRODUCTION AND PURPOSE

Fish presence/absence surveys are ongoing across the Plan Area. The purpose of the presence / absence (P/A) survey is to positively identify a stream reach of interest as a Class I (fish bearing) or Class II (non-fish bearing) watercourse. These surveys are primarily employed in association with a proposed Timber Harvesting Plan (THP) and are intended to assist the RPF with a proper identification of watercourse reaches within the proposed THP. However, the P/A Survey may on occasion be used to identify watercourse reaches not associated with a THP. Both situations will serve to help Simpson to better understand and manage for the public trust resources located within the Plan Area.

A key assumption of these surveys is that it is specifically understood that only the presence of fish species can be absolutely proven. Absence of fish can only be inferred from a lack of presence.

C6.2 METHODOLOGY

C6.2.1 Materials

- Appropriate Safety Equipment
- Backpack Electrofisher
- Dip Nets
- Maps and/or aerial photos of area

C6.2.2 Methods

The watercourse reach of interest shall be searched in an upstream direction whenever reasonable. The electrofisher settings shall be adjusted to the least harmful, yet effective setting possible (begin with P-16). Electrofishing will occur in appropriate salmonid habitat such as slower water and pools.

If fish are observed; capture the first few fish in order to identify to species and then release immediately. Continue working upstream, once fish are observed in a pool discontinue shocking and proceed to the next appropriate salmonid habitat. Continue until the reach of interest is covered or 1000' past the last observed salmonid.

If no fish are observed; confirm that the electrofisher unit is working properly. Search for an amphibian species, usually a Pacific giant salamander (*Dicamptodon ensatus*), and observe its behavior during shocking (shock the water within 3 feet of the amphibian, not the organism itself). If the amphibian responds to the electrofishing, then continue working upstream searching for fish. If the organism does not respond, double-check the settings and all connections on the electrofisher unit. Confirm that the warning beeper is working. Re-shock the pool and observe the amphibian. If there is still no response, increase the electrofisher units' settings to I-5 at 300 volts. Re-shock. If there is still no response, discontinue electrofishing and troubleshoot the electrofishing unit. If the amphibian responds, continue working upstream searching for salmonids until the reach of interest has been covered or 1000' past the last observed salmonid or known Class I watercourse.

C6.2.3 Follow-up

Once presence or absence has been determined this information will be reported to the Plan Coordinator. A map showing the exact location of electrofishing activities and a summary of field notes describing what was found during the survey will be provided to the Plan Coordinator. All information will also be recorded on the Fish and Herp base maps to update the map records.

C6.3 RESULTS AND DISCUSSION

The presence/absence survey information will be entered into Forest Resources Information System (FRIS) database and the results appropriately incorporated into the Timber Harvest Plan (THP). A series of GIS based (FRIS) maps will be continuously updated with information obtained from the presence/absence surveys. The maps and database provides current information on the distribution of fish on a property wide basis. The current fish distribution maps and tables for each HPA are presented in Section 7.

C6.4 CONCLUSIONS

A presence/absence survey is a valuable technique to establish Class I watercourse determinations and fish species distributions across the Plan Area on a site-specific basis. The extent of anadromy for streams is generally known across the Plan Area with the exception of the actual extent for each individual species. The presence/absence surveys are primarily used to delineate the extent of resident populations in low order Class I watercourses.

Appendix C7. Summer Juvenile Salmonid Population Estimates

CONTENTS

Tables

C7.1 INTRODUCTION

In 1995, data collection on the summer populations of juvenile coho salmon and 1+ and older steelhead was initiated in three Plan Area streams: South Fork of the Winchuck River (Smith River HPA), Wilson Creek (Smith River HPA), and Cañon Creek (Mad River HPA). Since 1995, data collection has occurred annually on these three original creeks for chinook salmon, and cutthroat trout in addition to coho salmon and steelhead. Four more creeks were added in 1998: Hunter Creek (Coastal Klamath HPA); Lower South Fork Little River, Railroad Creek, and Upper South Fork Little River (all Little River HPA). Sullivan Gulch (North Fork Mad River HPA) was added to the program in 1999. The purpose of these population surveys is to estimate and monitor summer populations of young-of-the-year coho salmon, chinook salmon, steelhead and cutthroat trout. Dive counts estimate salmonid population size during summer low flow periods (August-September). These fish represent the population of juvenile salmonids that will be shortly out-migrating or over-wintering in Plan Area streams.

C7.2 METHODS

The 1995 effort was part of a pilot study to test and refine a sampling methodology developed by Drs. Scott Overton and David Hankin in conjunction with funding through the Fish, Farm and Forest Communities Forum (FFFC). Juvenile salmonid population sampling has evolved since the program's inception in 1995. The population estimate methodology was based on the Hankin and Reeves (1988) two-phase survey design, with the most recent modifications being incorporated from Hankin (1999). These changes have been adopted to improve statistical validity, reduce variance, increase efficiency in the field, and reduce electrofishing effort. The current protocol is especially appropriate for small streams containing special status species where injury and mortality are a concern from a federal Endangered Species Act "take" stand-point.

The current protocol allows for increased use of diver counts for estimating the abundance of juvenile salmonids in streams. This approach reduces the need for electrofishing and related possible mortality of special status species (e.g. coho salmon).

The first phase of the current sampling design classifies habitat units into riffles, runs, pools, and deep pools, measures dimensions of each unit, and then randomly selects a fraction of units in each habitat class for phase 1 sampling (employing the Adaptive Sequential Independent Sampling [ASIS] method [Hankin 1999]). ASIS is used in first and second phase unit selection permitting habitat mapping and unit selection decisions to be made in the field. Phase 1 sampling consists of diving each selected unit to obtain an initial count of salmonids within the sampling unit. Riffle segments are electrofished as diving cannot be conducted in riffles. A subset of the sampled units is then randomly selected for calibration using the ASIS method. The mode of calibration (2nd phase sampling) is determined by the number of individuals counted in each unit. If the initial dive count is less than 20 individuals (of a given species), calibration is conducted by Method of Bounded Counts (Robson and Whitlock 1964). The Method of Bounded Counts (MBC) is utilized to calibrate dive counts when the unit population size is small ($n<20$), producing a substantial reduction in electrofishing effort. If the initial dive count of the target species exceeds 20 fish, calibration is made by four-pass removal electrofishing method. Calibration within deep-pool stratums is made only by MBC, as electrofishing is inefficient in this habitat stratum. In riffles selected for calibration, a 2 to 3 pass-removal electrofishing method is the mode of calibration.

If the method of bounded counts is the mode of calibration the 3 additional diver counts are made immediately following the 1 phase dive counts. If the 2nd phase sampling is conducted by the 4 pass-removal electrofishing method the electrofishing is conducted within no more than 2 days following phase 1 sampling. The methods employed for sample selection and estimation, the ASIS methodology, and phase 2 calibration methods are those of Hankin (1999). Additional discussion of the applicability and assumptions of the population estimation methodology employed by Simpson are found in Hankin (1999).

This protocol has also been slightly modified from previous years to provide more consistency between individual crews and from year to year. In the past, the difference between a deep pool and a shallow pool was based on processional judgment on whether or not the habitat mapping crew thought it possible to effectively electrofish a particular unit. If a pool was considered to be too complex; i.e. too much large woody debris (LWD), small woody debris (SWD), or deep undercut banks, it was classified as a deep pool and only calibrated by repeated dive counts.

Since 1999, pools less than 1.1 meters in depth are considered shallow pools and pools greater than or equal to 1.1 meters in depth are considered deep pools regardless of cover. This provided better consistency between crews, allowing comparisons of population estimates between different streams, crews, and property owners. The reduction in total number of deep pools and the corresponding increase in shallow pools is a result of this protocol change and not in the quality or quantity of available habitat. Simpson believes that this change to the protocol has also provided a much better estimate due to the increased number of calibrated shallow pools. The complexity of the pool does not appear to influence the ability to effectively electrofish those units.

C7.3 RESULTS

The summarized results of the summer juvenile population estimates for the 8 Plan Area streams are presented in Tables C7-1 through C7-4. The summer juvenile population estimates and the (+/-) 95% confidence interval (C.I.) for coho salmon for the years 1995 through 2000 are shown in Table C7-1. Table C7-2 summarizes the summer juvenile population estimates and (+/-) C.I.s for steelhead for the years 1995 through 2000. Tables C7-3 and C7-4 provide summaries of juvenile summer population estimates and corresponding (+/-) 95% C.I.s for cutthroat trout and chinook salmon respectively, for the years 1996 through 2000.

C7.4 DISCUSSION

C7.4.1 Methodology Effectiveness

The modified Hankin and Reeves juvenile sampling protocol has worked well for estimating juvenile coho salmon and 1+ steelhead populations. Consideration early in the development of the protocol was also given to cutthroat and chinook. Including cutthroat and chinook as species accounted for in the survey methodology has presented some complications, which are apparent looking at data collected from 1995 to 2000.

Table C7-1. Summer juvenile coho population estimates in eight Plan Areas streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	Unable to be estimated	
	1996	DP	32	23
		SP, Run, Riffle	4*	n/a
			Total 36	
	1997	DP	156*	n/a
		SP, Run, Riffle	331	140
			Total 487	
	1998	DP	33	7
		SP, Run, Riffle	0	0
			Total 33	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Wilson Creek	1995	DP, SP, Run, Riffle	1370†	212
	1996	DP	357	116
		SP, Run, Riffle	164	123
			Total 521	
	1997	DP	209*	n/a
		SP, Run, Riffle	27*	n/a
			Total 236	
	1998	DP	355	108
		SP, Run, Riffle	25	22
			Total 380	
	1999	DP	0	0
		SP, Run, Riffle	19	21
			Total 19	
	2000	DP	21	18
		SP, Run, Riffle	23	23
			Total 44	
Hunter Creek	1998	DP	317	122
		SP, Run, Riffle	81	88
			Total 398	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Railroad Creek (Little River)	1998	DP	85	34
		SP, Run, Riffle	164	84
			Total 249	
	1999	DP	0	0
		SP, Run, Riffle	339	64
			Total 339	
	2000	DP	14*	n/a
		SP, Run, Riffle	162	79
			Total 176	

Table C7-1 Continued. **Summer juvenile coho population estimates in eight Plan Areas streams, 1995-2000.**

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	2,397	282
		SP, Run, Riffle	1,213	312
			Total 3,610	
	1999	DP	1,774	253
		SP, Run, Riffle	6,129	883
			Total 7,903	
	2000	DP	1,403	232
		SP, Run, Riffle	3,364	761
			Total 4,767	
Upper SF Little River	1998	DP	265	101
		SP, Run, Riffle	473	186
			Total 738	
	1999	DP	182	134
		SP, Run, Riffle	1,048	484
			Total 1,230	
	2000	DP	68	89
		SP, Run, Riffle	275	83
			Total 343	
Sullivan Gulch	1999	DP	147	30
		SP, Run, Riffle	636	265
			Total 783	
	2000	DP	10*	n/a
		SP, Run, Riffle	41	37
			Total 51	
Cañon Creek	1995	DP, SP, Run, Riffle	919†	377
	1996	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1997	DP	20*	n/a
		SP, Run, Riffle	23	36
			Total 43	
	1998		Not Estimate Made	
	1999	DP	231	101
		SP, Run, Riffle	179	89
			Total 410	
	2000	DP	160	47
		SP, Run, Riffle	123	38
			Total 283	

Notes
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.
† Estimate from Chris Moyer's thesis work.

Table C7-2. Summer juvenile steelhead population estimates in eight Plan Area streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	932†	332
	1996	DP	1,092	145
		SP, Run, Riffle	822	150
			Total 1,914	
	1997	DP	237*	n/a
		SP, Run, Riffle	619	230
			Total 856	
	1998	DP	1,459	189
		SP, Run, Riffle	1,069	206
			Total 2,528	
	1999	DP	327	71
		SP, Run, Riffle	768	101
			Total 1,095	
	2000	DP	1,205	175
		SP, Run, Riffle	2,028	463
			Total 3,233	
Wilson Creek	1995	DP, SP, Run, Riffle	1,041†	253
	1996	DP	909	189
		SP, Run, Riffle	960	348
			Total 1,869	
	1997	DP	146*	n/a
		SP, Run, Riffle	100	21
			Total 246	
	1998	DP	875	177
		SP, Run, Riffle	544	96
			Total 1,419	
	1999	DP	331	153
		SP, Run, Riffle	410	124
			Total 741	
	2000	DP	365	149
		SP, Run, Riffle	932	148
			Total 1,297	
Hunter Creek	1998	DP	1,012	351
		SP, Run, Riffle	790	154
			Total 1,802	
	1999	DP	130	42
		SP, Run, Riffle	745	123
			Total 875	
	2000	DP	815	270
		SP, Run, Riffle	1,206	394
			Total 2,021	
Railroad Creek (Little River)	1998	DP	35	54
		SP, Run, Riffle	80	44
			Total 115	
	1999	DP	12	9
		SP, Run, Riffle	64	24
			Total 76	
	2000	DP	5*	n/a
		SP, Run, Riffle	72	35
			Total 77	

Table C7-2 Continued. Summer juvenile steelhead population estimates in eight Plan Areas streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	176	61
		SP, Run, Riffle	54	31
			Total 230	
	1999	DP	56	20
		SP, Run, Riffle	157	42
			Total 213	
	2000	DP	23	19
		SP, Run, Riffle	39	17
			Total 62	
Upper SF Little River	1998	DP	132	28
		SP, Run, Riffle	218	55
			Total 350	
	1999	DP	50	11
		SP, Run, Riffle	168	66
			Total 218	
	2000	DP	16	28
		SP, Run, Riffle	236	55
			Total 252	
Sullivan Gulch	1999	DP	10	4
		SP, Run, Riffle	7	8
			Total 17	
	2000	DP	2*	n/a
		SP, Run, Riffle	55	21
			Total 57	
Cañon Creek	1995	DP, SP, Run, Riffle	1,041†	253
	1996	DP	359	99
		SP, Run, Riffle	317	69
			Total 676	
	1997	DP	90	n/a
		SP, Run, Riffle	508	106
			Total 598	
	1998		No Estimate made	
	1999	DP	197	53
		SP, Run, Riffle	375	121
			Total 572	
	2000	DP	348	70
		SP, Run, Riffle	585	93
			Total 933	

Notes
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.
† Estimate from Chris Moyer's thesis work.

Table C7-3. Summer juvenile coastal cutthroat trout population estimates in eight Plan Area streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	299	56
		SP, Run, Riffle	131	25
			Total 430	
	1997	DP	56*	n/a
		SP, Run, Riffle	331	140
			Total 487	
	1998	DP	283	67
		SP, Run, Riffle	194	39
			Total 477	
	1999	DP	115	32
		SP, Run, Riffle	265	66
			Total 380	
	2000	DP	172	50
		SP, Run, Riffle	302	123
			Total 474	
Wilson Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	120	47
		SP, Run, Riffle	38	16
			Total 158	
	1997	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1998	DP	27	19
		SP, Run, Riffle	3	4
			Total 30	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	15	15
		SP, Run, Riffle	0	0
			Total 15	
Hunter Creek	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	35	25
		SP, Run, Riffle	15	10
			Total 50	
Railroad Creek (Little River)	1998	DP	0	0
		SP, Run, Riffle	10	6
			Total 10	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	

Table C7-3 Continued. Summer juvenile coastal cutthroat trout population estimates in eight Plan Areas streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	82	22
			Total 82	
	2000	DP	1*	n/a
		SP, Run, Riffle	18†	17
			Total 19	
Upper SF Little River	1998	DP	1*	n/a
		SP, Run, Riffle	6	7
			Total 7	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	4	13
			Total 4	
Sullivan Gulch	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Cañon Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	13	13
		SP, Run, Riffle	0	0
			Total 13	
	1997	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1998		No Estimate Made	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	17	11
		SP, Run, Riffle	4	4
			Total 21	

Notes
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.
† Estimate made using data from electro-fishing

Table C7-4. Summer juvenile chinook population estimates in eight Plan Area streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	313	101
		SP, Run, Riffle	35	13
			Total 348	
	1997	DP	12*	n/a
		SP, Run, Riffle	85	17
			Total 97	
	1998	DP	688	232
		SP, Run, Riffle	220	163
			Total 908	
	1999	DP	496	208
		SP, Run, Riffle	899	156
			Total 1,395	
	2000	DP	66	26
		SP, Run, Riffle	42	30
			Total 108	
Wilson Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1997	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1998	DP	3*	n/a
		SP, Run, Riffle	8	13
			Total 11	
	1999	DP	1*	n/a
		SP, Run, Riffle	0	0
			Total 1	
	2000	DP	0	0
		SP, Run, Riffle	1*	n/a
			Total 1	
Hunter Creek	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	30	37
		SP, Run, Riffle	26	34
			Total 56	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Railroad Creek (Little River)	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	

Table C7-4 Continued. Summer juvenile chinook population estimates in eight Plan Areas streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	4*	n/a
		SP, Run, Riffle	0	0
			Total 4	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Upper SF Little River	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	2*	n/a
			Total 2	
	2000	DP	0	0
		SP, Run, Riffle	6	19
			Total 6	
Sullivan Gulch	1999	DP	2	2
		SP, Run, Riffle	1*	n/a
			Total 3	
	2000	DP	4*	n/a
		SP, Run, Riffle	8	10
			Total 12	
Cañon Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	23	37
		SP, Run, Riffle	0	0
			Total 23	
	1997	DP	8*	n/a
		SP, Run, Riffle	8	18
			Total 16	
	1998		No Estimate Made	
	1999	DP	249	208
		SP, Run, Riffle	89	48
			Total 338	
	2000	DP	28	15
		SP, Run, Riffle	44	46
			Total 72	

Note
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.

Juvenile population estimates within Plan Area streams continue to include estimates for juvenile chinook (0+) and 1+ cutthroat. Chinook population estimates are relatively small compared to coho and steelhead. In the Plan Area, the majority of the chinook out-migrate before summer low flow conditions are reached, making it difficult to sample a closed population.

Cutthroat greater than 1+ years of age are included in the population estimate, although small populations and species migration patterns may complicate the estimation methodology. Both cutthroat and steelhead can sometimes be difficult to distinguish as young of the year or 1+ fish. Generally, when cutthroat reach a size greater than 120mm, they are easily distinguished from steelhead. By inaccurately distinguishing between “trout” life history stages, the methodology may underscore year class population size and may potentially underestimate or overestimate steelhead and/or cutthroat populations within Plan Area streams that contain sizeable runs of either species. A second concern for estimating cutthroat populations can be drawn from juvenile out-migration trapping results obtained from the Little River drainage. As seen during juvenile out-migrant trapping, a large number of parr and pre-smolting cutthroat are observed moving through the traps during late winter and fall. Steelhead of similar age classes are also observed moving through the traps. The summer population estimates, only include those cutthroat or steelhead that remain in the streams throughout the year. It is possible that the “trout” population is underestimated because a large proportion of the population left the system during winter and fall prior to conducting the summer population estimate. A third concern when applying this methodology to “trout” is the approachability of the species through diver observation. Unlike coho salmon, “trout” are skittish and hide as a diver approaches, making counts difficult and identification sometimes impossible. During Phase 2 calibration, this can affect MBC, which relies on a surveyor’s ability to observe the same fish on subsequent dives.

C7.4.2 Population Size

Juvenile coho population estimates from the Plan Area vary from stream to stream and year to year. In data sets that span a period of five years, juvenile coho population estimates vary widely; increasing in some streams and decreasing in others. Overall, Plan Area streams north of Redwood Creek show a downward progression in coho populations (Table C7-1). Data collected from streams south of Redwood Creek show relatively stable or increasing populations. Studies within these streams have not occurred long enough to infer trends; however, factors such as low winter flows and poor ocean conditions can contribute to poor adult escapement. This observation is supported by spawning surveys that occur within Plan Area streams, which documented little to no returning adult coho. These observations do not always hold true as is discussed under the Spawning Survey section of Appendix C, however, it can help to explain population estimates that observed no coho salmon in some north Plan Area streams (S.F. Winchuck and Hunter Creek).

Steelhead estimates indicate stable or increasing populations both north and south of Redwood Creek (Table C7-2). Juvenile populations within streams north of Redwood Creek tend to show the highest population estimates. Within these streams, habitat conditions may be more suited for this species that has behaviors adapted for swift flowing, higher gradient watercourses, with reduced velocity refuge.

Juvenile cutthroat populations tend to show very limited numbers within Plan Area streams, other than the SF Winchuck. However, presence/absence surveys indicate that cutthroat are widely dispersed across the Plan Area. Cutthroat trout populations tend to decrease south of Redwood Creek and disappear from state records south of the Eel River (Gerstung 1997). Populations of cutthroat trout that often prefer low velocity habitats, may out compete coho within areas like the S.F. Winchuck.

Juvenile chinook salmon tend to out-migrate from Plan Area streams prior to June. The juvenile dive counts take place in the months of August and September during summer low flow. Residual populations of chinook salmon counted during the summer dives demonstrate species presence, but cannot be used for population estimates due to their early season out-migration patterns.

C7.4.3 Summer Habitat Preference

During summer low flows, pool habitat is the preferred habitat type for all species (Tables C7-1 through C7-4), specifically deep pools. Species competition within this habitat type becomes apparent in high production years or in small streams with limited pool habitat available. Other habitat types such as runs and shallow pools are well utilized by all species. Depending on the amount of available habitat during high production years, juvenile coho salmon can be found distributed in all habitat types including riffles. This is likely a result of fully seeded habitats, where intraspecific competition causes redistribution among available habitat types even into “less desirable” rearing habitats such as riffles. In lower production years, such as 2000, coho salmon may be out competed by steelhead or cutthroat trout for deep pool habitat.

C7.5 CONCLUSIONS

Using this protocol to estimate juvenile chinook populations is not recommended, but may work for more northern populations (British Columbia and Alaska) that over-winter in freshwater. It is also not well suited for cutthroat trout due to their limited numbers within Plan Area streams and their tendency to move downstream of survey reaches prior to summer low flows. Overall, juvenile population sampling using the modified Hankin and Reeves survey methodology is very useful for estimating juvenile coho populations, and appears to be well suited for 1+ steelhead trout, although significant numbers of steelhead can be observed moving downstream prior to summer surveys. Juvenile coho are generally unafraid of divers and are very approachable. Identification is simple, using both physical attributes and their distinct behavior as key identifiers. Steelhead are skittish and not often seen during subsequent Phase 2 calibration dives, never-the-less 95% C.I. indicate limited variation among population estimates for this species.

Juvenile coho populations within the Little River watershed appear stable and well seeded in all three-survey years, and in the majority of Little River tributaries. Population estimates north of Little River may reflect habitat conditions more suitable for steelhead, however many other factors including adult escapement and interspecific competition could account for the observed estimates. Steelhead 1+ juveniles appear to be distributed in sizable numbers in all surveyed Plan Area streams. While changes (positive or negative) in summer population estimates is clearly of interest, it remains unclear what, if any, changes can be related to management. Currently, population trends cannot be inferred from available data for any of the species, however these estimates may help determine relationships between coho populations in different streams throughout the Plan Area, and the climactic and/or habitat conditions which affect summer population size, when combined with other monitoring efforts.

C7.6 REFERENCES

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Appendix C8. Out-Migrant Smolt Trapping

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Figures

Tables

C8.1 INTRODUCTION

Juvenile salmonid out-migrant (emigrant) smolt trapping has been conducted on several Plan Area streams since 1999. The out-migrant trapping project is designed to monitor the abundance, size, and timing of out-migrating smolts, and to look for long term trends in any or all of these variables. This trapping program is conducted to obtain annual population estimates on emigrating salmonid smolts (coho salmon, chinook salmon, steelhead trout and coastal cutthroat trout). The results of the out-migrant trapping are used in conjunction with the summer population monitoring to estimate overwinter survival in those streams monitored. The juvenile out-migrant trapping also helps to identify factors affecting smolt emigration timing, and establish baseline and long-term trend data on the abundance of juvenile salmonid populations in the watersheds monitored.

During March through July, 1999 Simpson conducted juvenile out-migrant trapping for salmonids on the Lower South Fork of the Little River (LSFLR), Upper South Fork of the Little River (USFLR) and Railroad Creek (RRC). These three creeks are all located in the Little River drainage and in the Little River HPA. During March through June, 2000 Simpson again conducted juvenile out-migrant trapping for salmonids on the LSFLR, USFLR and RRC as well as adding Carson Creek (CC) to the monitoring program. Like the other three creeks, Carson Creek is located in the Little River drainage.

C8.2 METHODS

C8.2.1 Trapping

Trapping was conducted using a V-notch weir, pipe, and a live-box to capture the juvenile salmonids (Figure C8-1). A second box was attached to the primary box to reduce in-trap predation. Fine mesh screen separated the entrances between the two boxes to serve as a barrier to separate larger fish from the smaller fish. Additional rock cover was provided within the live boxes to serve as refugia for young of the year (YOY) fish. The weirs were constructed with fence posts and wooden pallets. A weir overflow was constructed to provide adult fish passage upstream. The pipe emptied out onto a McBane ramp that dissipated the velocity of the water and gently guided the fish into the box trap. Inside the trap there is a V-shaped panel which creates a large slack water area in the box. This provides an area where the fish can be protected from the stream's current. In 1999, the trap was operated 24 hours a day and checked daily each morning. In 2000, the traps were checked twice daily, in the morning and in the evening. During periods when significant numbers of out-migrants were captured, the trap was checked more frequently as needed. The captured juvenile fish were anesthetized with Alka Seltzer™, identified, measured (fork length) and most were immediately released below the weir.

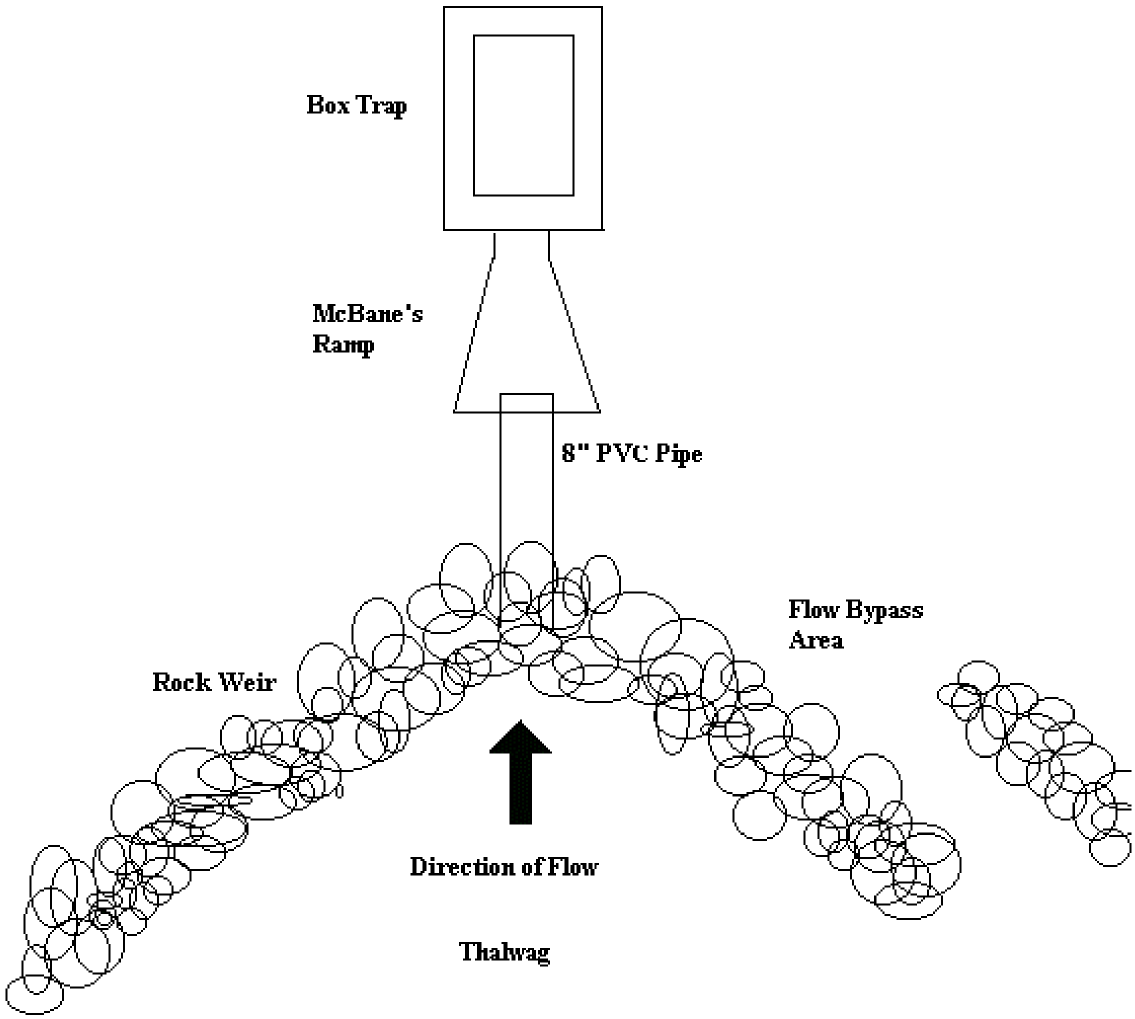

Figure C8-1. Out-migrant fish trapping system (not shown to scale),

Steelhead and cutthroat trout one year or older had their stomachs pumped (gastric lavage) to determine if predation had possibly occurred in the live-box. A subsample of all smolted salmonids were fin clipped (caudal) and released upstream of the weir to determine the trap efficiency. The fin clipped smolts were held in separate live box to determine any possible mortality associated with handling and marking the fish. The smolts were released during the evening trap check period. Recaptured fish from the trap efficiency tests were not used again for subsequent efficiency tests. All caudal fin clip samples from juvenile coho salmon were collected and stored in individual coin envelopes. The samples were air-dried on filter paper and sent to the Bodega Marine Laboratory, University of California. That institution is conducting a study on the genetic variation and population structure of coho salmon in California. Simpson is also sending tissue samples from coho carcasses collected during adult escapement surveys to the UC Bodega Bay Marine Laboratory for genetic analyses.

Trap efficiency was calculated by using only species that were actively leaving the drainage on their seaward migration (defined as smolts). Smolts received a fin clip. Four different clips were used throughout the trapping season to test trap efficiency. The easiest clips to identify are caudal fin clips. They were released upstream of the weir in the evening. This allowed the fish ample recovery time and allows for checking for possible mortality from the clipping and handling of them.

C8.2.2 Stomach Pumping (Gastric Lavage)

1+ and older cutthroat and steelhead underwent a stomach pumping procedure to determine predation in the live box. No adult run-back steelhead underwent the pumping procedure. In 1999 the size of the fish that under went gastric lavage ranged from 62-341mm in length. In 2000 the size of the fish ranged from 62-332mm in length. Anesthetized fish were pumped by inserting a small tube down their throat and into their stomach. Water was then pumped into their stomach through the tube. Once the stomach is filled with water, the stomach contents spill out. The contents were then processed. Items were identified to species if possible. After identification the contents were stored in zip-lock bags and preserved with isopropyl alcohol. The pumped fish were placed in a recovery bucket and monitored for approximately one hour prior to their release downstream of the weir. Any stomach pumped smolts were held in a live trap and released during the evening trap check.

C8.3 RESULTS

C8.3.1 Drainage Area and Length of Streams Trapped

A summary of the 1999 project stream drainage area and lengths of utilized habitat above the traps is provided in Table C8-1. In 1999, Simpson also quantified habitat conditions in these three streams to assess the survival of juvenile populations in varying freshwater habitats. A summary of the stream drainage and length of utilized habitat above the out-migrant trap during the 2000 out-migrant-trapping project is provided in Table C8-2.

Table C8-1. Drainage area and length of utilized habitat above the trap location for each creek in the 1999 out-migrant trapping study.

	USFLR	LSFLR	Railroad Creek
Drainage area (sq. miles)	5.70	5.31	2.75
Length of available habitat (miles)	1.50	2.16	1.21

Table C8-2. Drainage area and length of utilized habitat above the trap location for each creek in the 2000 out-migrant trapping study.

	USFLR	LSFLR	Railroad Creek	Carson Creek
Drainage area (sq. miles)	5.70	5.31	2.75	3.81
Length of available habitat (miles)	1.1	2.2	0.5	≈2.0

C8.3.2 Population Estimates

Out-migrant smolt population estimates were generated using a preliminary version of software for analysis of stratified mark-recapture data (Bjorkstedt, 2000). The summary of the smolt out-migrant population estimates and their 95% confidence intervals are shown in Table C8-3.

Table C8-3. Summary of the out-migrant population estimated for the years 1999 and 2000.

Coho	**1999**		**2000**	
	Estimate	95% CI	Estimate	95% CI
LSFLR	293	± 21	1,682	± 60
USFLR	27	± 13	147	± 25
Carson Ck	Did Not Trap		1,802	± 30
Railroad Ck	21	± 4	68	± 1
Steelhead	**1999**		**2000**	
	Estimate	95% CI	Estimate	95% CI
LSFLR	103	± 27	46	± 43
USFLR	50	± 7	72	± 3
Carson Ck	Did Not Trap		12	± 3
Railroad Ck	46	± 16	14	± 1
Cutthroat	**1999**		**2000**	
	Estimate	95% CI	Estimate	95% CI
LSFLR	108	± 28	22	± 4
USFLR	35	± 10	13	± 7
Carson Ck	Did Not Trap		60	± 6
Railroad Ck	50	± 5	23	± 1

C8.3.3 Over Wintering Survival

Overwintering survival is calculated by dividing the winter population by the summer population. One of the key assumptions with overwintering survival is that none of the fish in the summer population estimate migrate prior to the downstream migrant trapping being installed. The summer and winter population estimates are shown in Table C8-4.

Table C8-4. Summary of the summer and winter population estimates for the tributaries of the Little River for 1998-1999 and 1999-2000.

Stream	Coho (YOY) Summer Population	Coho Smolts Winter Population	Overwintering Survival Estimate	Drainage Area (Sq. miles)	Miles of Habitat	Summer Fish/Mile	Winter Fish/Mile
	1998	**1999**					
USFLR	738	27	3.7%	5.70	1.5	492	18
LSFLR	3,610	293	8.1%	5.31	2.2	1,641	133
RR Ck	249	21	8.4%	2.75	1.2	208	18
	1999	**2000**					
USFLR	1,230	147	12.0%	5.70	1.1	1,118	134
LSFLR	7,903	1,682	21.3%	5.31	2.2	3,592	765
RR Ck	339	69	20.4%	2.75	0.5	678	138
Carson Ck	NA	1,802	NA	3.81	≈2.0	NA	901

C8.3.4 Species Composition

In 1999 juvenile out-migration trapping captured several different fish (and amphibian) species within the Little River drainage (Table C8-5). The majority of the fish captured were in the genus *Oncorhynchus*. However, there was incidental capture of non-target species mostly lamprey and amphibians. Table C8-6 summarizes the total number of salmonid smolts that were captured and recaptured for all streams in 1999. From these results the Lower South Fork Little River was the most productive coho stream of those trapped in 1999. Trapped fish were identified to species when possible. Due to the similarities between YOY steelhead and YOY cutthroat trout these were grouped into the trout category. All coho, chinook and trout (YOY cutthroat and steelhead) were YOY fish, while all steelhead and cutthroat trout were 1+ fish or older. Some of the 1+ steelhead were determined to be run-back steelhead returning to the ocean. The total numbers of all salmonids trapped in 1999 are summarized below (Table C8-7). The USFLR and LSFLR produced significant numbers of trout and chinook in addition to coho salmon. Table C8-8 summarizes the total number of salmonid smolts that were captured and recaptured for all streams in 2000. From these results Carson Creek was the most productive coho stream trapped.

In 2000 adult cutthroat trout were defined as fish >200mm and not showing signs of smoltification. The total numbers of all salmonids captured in 2000 are shown in Table C8-9. The Lower South Fork Little River and Carson Creek were the most productive coho streams, while both the Upper South Fork Little River and Lower South Fork Little River produced significant numbers of trout and chinook.

There were some mortalities associated with the trapping process. The summary of the trapping mortality in 1999 and 2000 are provided in Tables C8-10 and C8-11 respectively. This summary also includes the mortalities associated with the stomach pumping (gastric lavage) procedure Tables C8-12 and C8-13. Improvements in trap design and trapping procedures were made throughout the trapping season in an effort to reduce these mortalities.

Table C8-5. Species captures during out-migrant trapping in the Little River drainage in 1999 and 2000.

Common Name	Scientific Name	1999	2000
Coho Salmon	*Oncorhynchus kisutch*	X	X
Chinook Salmon	*Oncorhynchus tshawytscha*	X	X
Steelhead	*Oncorhynchus mykiss*	X	X
Coastal Cutthroat Trout	*Oncorhynchus clarki clarki*	X	X
Pacific Lamprey	*Lamperta tridentata*	X	X
Western Brook Lamprey	*Lamperta richardsoni*	X	X
Pacific Giant Salamander	*Dicamptodon ensatus*	X	X
Tailed Frog	*Ascaphus truei*	X	X
Prickly Sculpin	*Cottus asper*		X
Three-Spined Stickleback	*Gasterosteus aculeatus*		X

Table C8-6. Trapping totals for clipped and recaptured smolts in 1999.

Stream	Clipped Smolts			Recaptured Smolts		
	Coho	Steelhead	Cutthroat	Coho	Steelhead	Cutthroat
LSFLR	220	36	40	187	13	19
USFLR	15	30	18	9	20	10
Railroad Ck	15	18	35	12	10	25
Total	250	84	93	208	43	54

Table C8-7. Trapping totals for unclipped fish in 1999.

Stream	Unclipped Fish				
	Coho	Steelhead	Cutthroat	Trout	Chinook
LSFLR	3,543	454	57	10,435	5,812
USFLR	599	778	112	14,503	4,133
Railroad Creek	422	281	88	4,131	0
Total	4,564	1,513	257	29,069	9,945

Table C8-8. Trapping totals for captured and recaptured smolts in 2000.

Stream	Captured Smolts			Recaptured Smolts		
	Coho	Steelhead	Cutthroat	Coho	Steelhead	Cutthroat
LSFLR	1,104	34	15	522	19	13
USFLR	100	57	7	72	42	5
Carson Ck	1,408	7	43	612	4	31
Railroad Ck	63	12	21	56	10	16
Total	2,675	110	86	1,262	75	65

Table C8-9. Trapping totals for unclipped fish in 2000.

Stream	Unclipped Fish				
	Coho (YOY)	Steelhead	Cutthroat	Trout (YOY)	Chinook (YOY)
LSFLR	1,911	509[a]	50[e]	4,911	3,680
USFLR	140	960[b]	31[f]	5,451	5,277
Carson Ck	131	504[c]	296[g]	1,481	874
RRC	763	850[d]	44[h]	1,228	0
Total	2,945	2,823	421	13,071	9,831

Notes
[a] 16 of steelhead were adult
[b] 17 of steelhead were adult
[c] 2 of steelhead were adult
[d] 6 of steelhead were adult
[e] 7 of cutthroat were adult
[f] 4 of cutthroat were adult
[g] 23 of cutthroat were adult
[h] 6 of cutthroat were adult

Table C8-10. 1999 in-trap mortality.

Stream	In-Trap Mortality				
	Coho (YOY)	Steelhead (1+)	Cutthroat (1+)	Trout (YOY)	Chinook (YOY)
LSFLR	23	4	0	155	56
USFLR	3	1	1[a]	318	58
Railroad Creek	3	2	1[a]	157	0
Total	29	7	2	630	114

Note
[a] These fish were killed in the gastric lavage procedure

Table C8-11. 2000 in-trap mortality.

Stream	In-Trap Mortality					
	Coho Smolts	Coho (YOY)	Steelhead (1+)	Cutthroat (1+)	Trout (YOY)	Chinook (YOY)
LSFLR	4	7	5	0	77	23
USFLR	1	1	5	0	105	74
CC	8	7	4	1	46	19
RRC	1	3	2	0	24	0
Total	14	18	16	1	252	116

Table C8-12. 1999 predation mortality determined from gut contents from stomach pumping.

Stream	Predation Mortality				
	Coho (YOY)	Steelhead (1+)	Trout (YOY)	Chinook (YOY)	Salmonids (YOY)
LSFLR	112	6	934	361	105
USFLR	30	3	1,731	329	119
Railroad Creek	82	1	1,162	0	50
Total	224	10	3,827	690	274

Table C8-13. 2000 predation mortality determined from gut contents from stomach pumping.

Stream	Predation Mortality					
	Coho (Smolt)	Coho (YOY)	Steelhead (1+)	Trout (YOY)	Chinook (YOY)	Salmonids (YOY)
LSFLR	9	89	5	157	133	84
USFLR	1	4	2	578	265	153
Carson Ck	15	11	9	141	244	49
Rail Road Ck	2	39	7	212	0	44
Total	27	143	23	1,088	642	330

Contents from stomach pumping conducted during the 1999 and 2000 trapping program were identified to species if possible (Tables C8-12 and C8-13). Some of the items were digested to a point to which species could not be determined but fish were positively identified as juvenile salmonids. All preyed on coho, chinook, trout and salmonids were young of the year (YOY) fish. The preyed on steelhead were 1+ fish. Some of the other contents identified from stomach pumping from the 1999 trapping included: aquatic invertebrates, salmonid eggs, Pacific giant salamanders, tailed frog tadpoles and one mouse.

C8.3.5 Size and Condition

Salmonid growth increases at varying rates depending on the abundance of aquatic insects and plant life during critical rearing periods (Murphy and Meehan 1991). Size can also be influenced by density related competition. The fork lengths of the first 20 YOY coho (fork length) were measured to the nearest mm. The remaining individuals were counted but not measured. All smolts were measured. Table C8-14 shows the range of coho lengths measured in 1999 as well as their average length by age for each stream. All fish handled appeared to be in good condition and length of YOY fish increased steadily as the trapping season progressed. Table C8-15 shows the range of coho lengths and weights as well as their average length by age for each stream as measured in 2000. All fish handled appeared to be in good condition and lengths and weights of YOY fish increased steadily as the trapping season progressed.

Table C8-14. Average and range of lengths (mm) of coho salmon in USFLR, LSFLR, and Railroad Creel in 1999.

Length	USFLR		LSFLR		Railroad Ck	
	Smolts	YOY	Smolts	YOY	Smolts	YOY
Range (mm)	96-114	34-50	81-136	32-58	98-124	31-49
Avg. Length (mm)	102.9	37.5	104.6	39.3	110.6	37.9

Table C8-15. Average range of lengths and weights of coho salmon trapped in USFLR, LSFLR, Carson Creek, and Railroad Creek in 2000.

	USFLR		LSFLR		Carson Creek		Railroad Ck	
	Smolts	YOY	Smolts	YOY	Smolts	YOY	Smolts	YOY
Length Range (mm)	80-120	34-59	65-139	29-64	68-135	28-51	78-115	31-69
Avg. Length (mm)	103.0	46.9	94.3	42.5	97.9	40.5	96.0	45.5
Weight Range (gms)	5.8-22.4	0.4-2.1	3.3-27.7	0.1-2.9	3.4-24.0	0.4-1.6	5.3-16.3	0.2-2.3
Avg. Weight (gms)	12.7	1.1	9.9	0.7	10.3	0.7	9.8	0.9

C8.3.6 Migration Timing

In 1999 the migration of coho smolts began in April and continued into June (Figure C8-2). Factors that affect the timing of migration include the size of the fish, flow conditions, water temperature, dissolved oxygen levels, day length, and availability of food (Shapovalov and Taft 1954). The peak days of migration within USFLR and LSFLR were determined to be May 19, 1999 and April 24 within Railroad Creek. The peak period of migration lasted from the last week of April to the end of May. Migration tapered off after approximately May 3rd. This slow down coincided with a rain storm event. Figure C8-3 shows the flow of the Little River during the period of smolt migration.

In 2000, migration of coho smolts began in March and continued into June (Figure C8-4). The migration peak for Carson Creek and LSFLR occurred on April 4th 2000 and on April 14th 2000 respectively. The LSFLR had an additional peak on April 26th 2000. There were no significant peaks on Railroad Creek and USFLR in 2000. There were two periods approximately April 17th and May 11th when migration tapered off, coinciding with a storm event. Figure C8-5 shows the flow of the Little River during the period of coho smolt out-migration.

C8.4 DISCUSSION

Lower trapping efficiency is experienced during peak flow events. As shown on Figure C8-2 reduced numbers of fish are trapped during peak flow events (Figure C8-3). In 1999 a large number of smolts were trapped just prior to a peak event on May 3rd and large number were again trapped a few days after that peak event. Simpson believes that there are a large number of fish emigrating from the streams during these peak events.

During 1999 there was some mortality associated with the trapping. These losses were reduced by continually improving the trapping methodology and trapping equipment throughout the trapping season. It was determined that on trapping days where there was high volumes of debris loading into the traps there was a corresponding higher trap mortality. To reduce this mortality, an extra screen to catch and filter out debris was added to the traps. Traps were checked and debris was cleaned out in the evening as well as mornings on rainy or windy days. This effort helped to reduce mortality and was continued in 2000. During 1999 some mortalities were observed when fish were stranded onto dry portions of the McBane ramp. A plastic splash shield was installed that immediately solved this stranding problem.

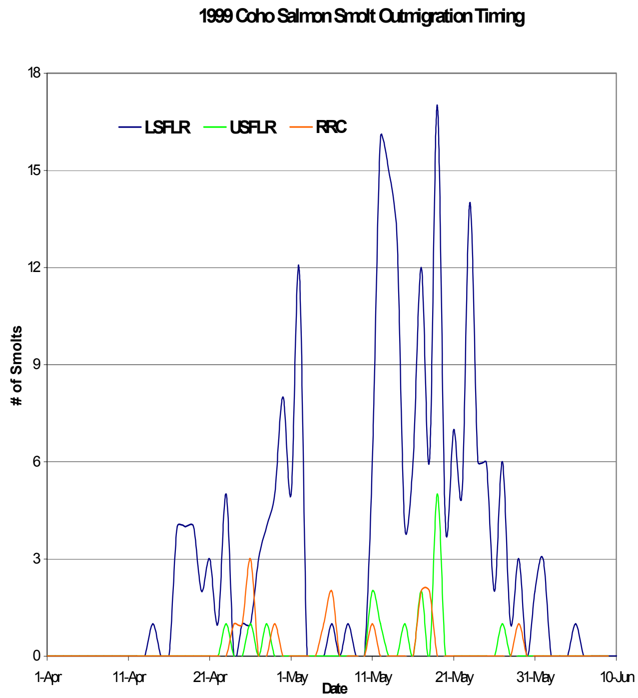

Figure C8-2. **Migration timing for smolts for the 1999 trapping study in Little River.**

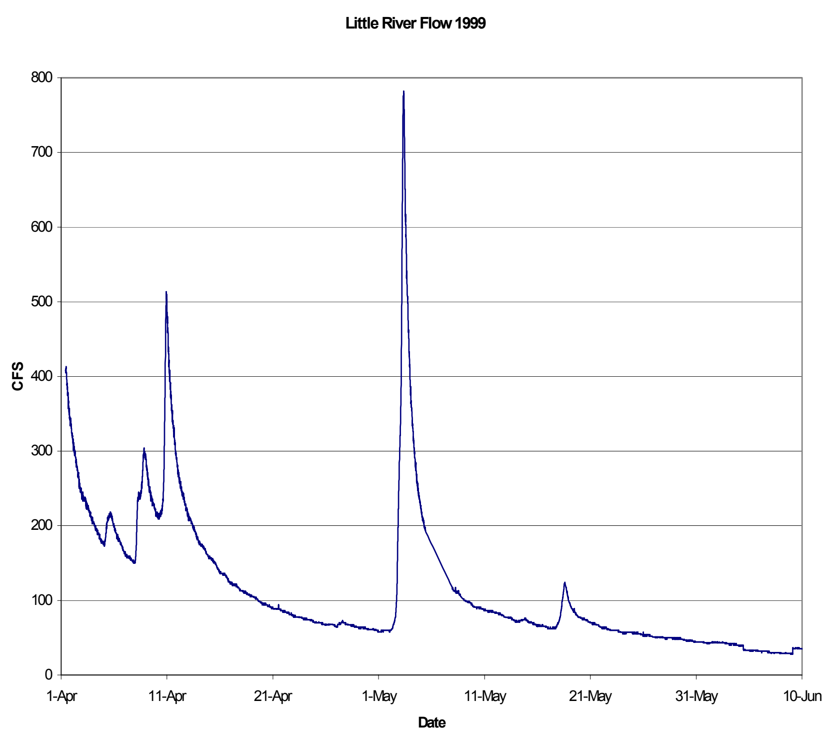

Figure C8-3. **Little River flow (CFS) during 1999 peak smolt out-migration.**

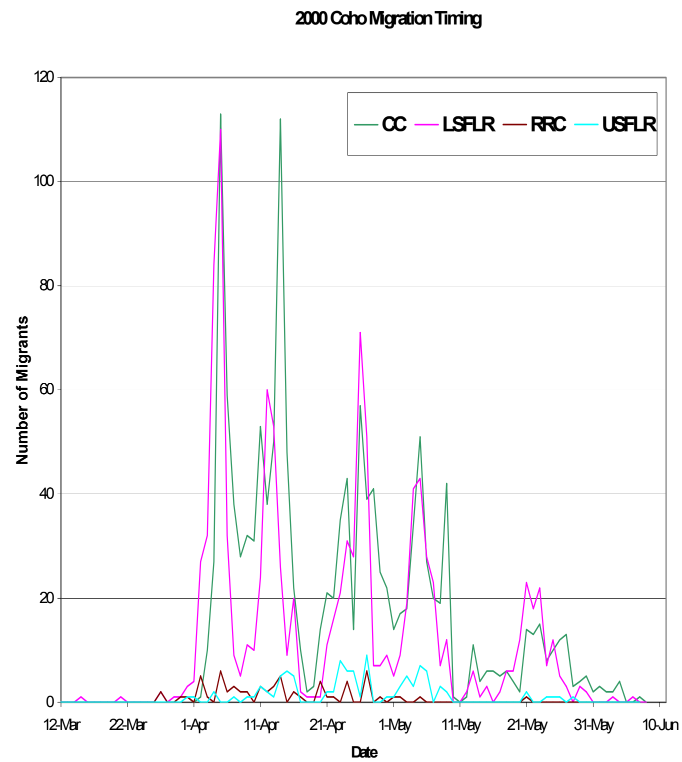

Figure C8-4. Migration timing for smolts for the 2000 trapping study in the Little River.

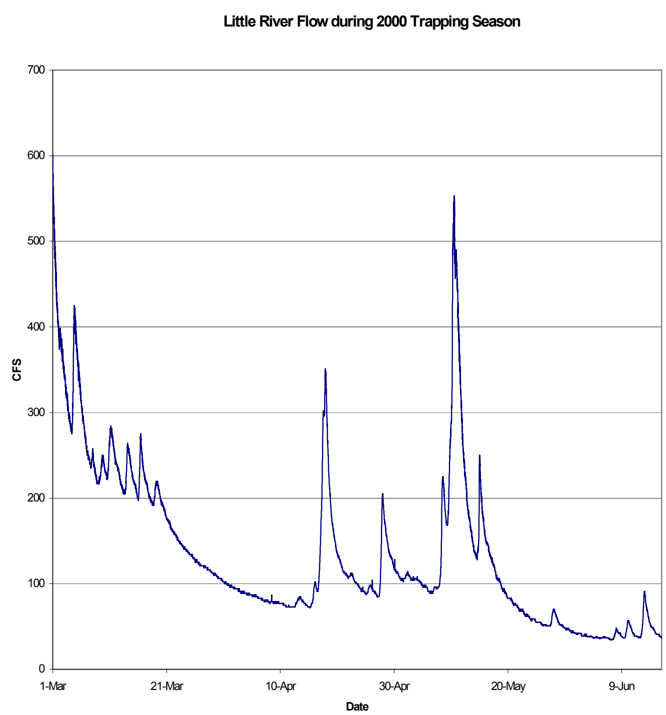

Figure C8-5. Little River flows (CFS) during peak out-migration in 2000.

Most of the trapping mortality was from loss of YOY fish. No smolts of any species were lost during trapping in 1999. There were only 2 mortalities from the stomach pumping procedure in 1999 (Table C8-12). Both of these mortalities were from improper insertion of the pumping tube while a new employee was learning the procedure.

During the 2000 trapping project lower sampling efficiency during peak flow events were again noted. Reduced numbers of fish (Figure C8-4) illustrates this during peak flow events (Figure C8-5). A good number of smolts were trapped just prior to peak events and a few days after the peak event. It is likely that a large number of fish leave the systems on these peak events because the creeks are confined channels with no flood plains. During these events the fish may be flushed out when the flows and velocities are high and the traps are relatively inefficient.

Over wintering survival rates were better in 2000 than in 1999. The increased survival rates may be higher due to the smolts leaving during the peak flow events. To determine an over-winter survival rate from the summer and winter population estimates it must be assumed that there is a closed population. This is not necessarily true. The first few days of trapping in 2000, in the LSFLR, several pre-smolt coho salmon were captured. From this observation, it appears that a portion of the coho began emigrating prior to the installation of the weir. Early pre-smolt migration violates the closed population assumption prior to pre-weir installation. In the future, trapping weirs will be installed earlier in the season to determine if a significant portion of the population begins emigration prior to the completion of smoltification. If a large number of coho are actively dispersing downstream during the winter rather than actively emigrating following smoltification during the spring, this would account for a relatively low over-winter survival rate. At the present time the survival rate of fish that disperse downstream as pre-smolts during the winter months is unknown.

In 2000, there were large numbers of mortalities associated with predation during trapping. In an effort to minimize predation during trapping, extensive refuge (cover) was provided for the YOY fish as they moved to the rear trapbox. The provision of this cover will exclude the predatory fish while provide refuge to the smaller YOY fish. Simpson is continuing its effort to reduce in trap predation by working with a graduate student from Humboldt State University (HSU) to develop an improved live trap box design. The student is conducting experiments to see if differently designed boxes that have different mesh separating devises help reduce predation mortality. Students from HSU are also looking at predation outside of the live boxes to determine how many of the prey items were eaten prior to being trapped. At the present time it is unknown whether the fish that are evaluated for predation are consuming their prey prior to entering the trap or while in the trap. In many cases the advanced stages of deterioration of the material within the stomach of the predatory fish indicates that it is likely that the preyed upon fish were consumed prior to being trapped.

There was also some continuing mortality associated with the trap design in 2000. Improvements were incorporated throughout the trapping season. On days where there were high amounts of organic debris loading in the traps, an increased mortality is expected. There was also some continuing mortality of fish stranding themselves onto a dry portion of the McBane ramp or into a debris deposit after coming out of the pipe. To reduce this mortality two new design elements were developed. An extra screen, to catch and filter out debris, and plastic sheeting on the McBane ramp, which prevented debris accumulation during lower flow conditions, were added to each trap. Also, the traps were checked in the evening on rainy or windy days to clean out the debris on the filter screen and inside the box trap. This combination of efforts significantly reduced mortality of YOY fish and thus will be continued and fine tuned in the future.

In 2000, from the stomach analysis, it was determined that the most common prey was YOY trout. This was followed, in order, by: YOY chinook salmon, YOY un-identified salmonids, YOY coho salmon, coho salmon smolts and 1+ steelhead. The coho smolts and 1+ steelhead were eaten by large predatory cutthroat trout. It was determined that during trapping, prey consumption followed the same order as fish abundance. The most abundant fish (YOY trout) were also most commonly recovered from the stomachs of the fish that were pumped. The only exception to this was the unidentifiable YOY salmonids. This finding suggests that there was no prey item preferred and actively selected over another.

C8.5 CONCLUSION

The use of out-migrant trapping is an excellent tool for collecting downstream migrants and is Simpson's best opportunity to collect information pertaining to coho production in the Little River drainage. The use of a box trap, McBane ramp, pipes and weir trapping system efficiently trap streams during low and normal flow. The out-migrant trapping program is in its preliminary stages and it is too early to determine population trends.

C8.6 REFERENCES

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Murphy M.L. and W.R. Meehan. 1991. Stream Ecosystems. American Fisheries Society Special Publication 19:139-179.

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Appendix C9. Spawning Surveys

CONTENTS

C9.1 METHODS

Simpson's staff does not attempt to generate any form of formal population or escapement estimates from the spawning surveys conducted. Due to the limitations of time, water conditions, and weather these surveys tend to be opportunistic rather than at fixed time intervals or fixed reaches. The purpose of these spawning surveys is to determine habitat use and relative numbers of spawners of all species as well as watershed conditions during the winter months. In general, the entire anadromous reach accessible to coho salmon is surveyed. In long anadromous reaches within one stream, the survey may be broken up into sub-reaches that tend to be based on accessibility and/or time available for the survey. Because of these constraints the surveys are somewhat inconsistent from year to year. Sub-reaches within one watershed may or may not be surveyed on the same day or by the same crew. Within each HPA a general description of the sub-reaches for each stream for which spawner surveys have been conduct are provided.

The following list indicates all streams by their Hydrographic Planning Area (HPA) for which spawning surveys have been conducted since 1995:

Stream	HPA
• Maple Creek	Coastal Lagoons
• North Fork Maple Creek	Coastal Lagoons
• Pitcher Creek	Coastal Lagoons
• Cañon Creek	Mad River
• Carson Creek	Little River
• Danielle Creek	Little River
• Little River	Little River
• Upper South Fork Little River	Little River
• Lower South Fork Little River	Little River
• North Fork Mad River	North Fork Mad River
• Railroad Creek	Little River
• Rowdy Creek	Smith River
• Salmon Creek	Humboldt Bay
• Savoy Creek	Smith River
• South Fork Rowdy Creek	Smith River
• South Fork Winchuck River	Smith River
• Sullivan Gulch	North Fork Mad River
• Wilson Creek	Smith River

C9.2 RESULTS

C9.2.1 Smith River HPA

Spawning surveys have been conducted on five streams within the Smith River HPA during the period of 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.1.1 South Fork Winchuck River

The survey reach extends from the confluence of South Fork and mainstem Winchuck upstream approximately four miles to the end of the W1100 road.

C9.2.1.1.1 1998-1999 Spawning Surveys

Two spawning surveys were conducted on South Fork Winchuck River during 1998-1999: December 10, 1998 and January 8, 1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
46 Chinook	21 Chinook	7 Chinook
1 Steelhead	29 Unknown	
2 Unknown		

C9.2.1.2 Rowdy Creek

The two Rowdy Creek spawning survey reaches extend from the county bridge on Rowdy Creek Road upstream 13,000 feet to the R1400 bridge and then an additional 7,600 feet upstream to the confluence of Rowdy and Copper Creeks.

C9.2.1.2.1 1998-1999 Spawning Surveys

One spawning survey was conducted on December 15, 1998 on Rowdy Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
11 Chinook	4 Chinook	None Observed
	3 Unknown	

C9.2.1.3 Savoy Creek

The spawning reach extends from the confluence with South Fork Rowdy upstream 3,100 feet to the anadromous barrier.

C9.2.1.3.1 1999-2000 Spawning Surveys

Two spawning surveys were conducted on December 3rd and 21st, 1999 on Savoy Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
55 Chinook	27 Chinook	18 Chinook
	13 Unknown	

C9.2.1.3.2 1998-1999 Spawning Surveys

One spawning surveys was conducted on December 16, 1999 on Savoy Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
20 Chinook	13 Chinook	1Chinook
	3 Unknown	

C9.2.1.4 South Fork Rowdy Creek

The survey reach extends from the confluence with Rowdy Creek upstream 4,000 feet to the confluence with Savoy Creek. It continues upstream from Savoy Creek an additional 3,500 feet to the anadromous barrier.

C9.2.1.4.1 1999-2000 Spawning Surveys:

Two spawning surveys were conducted on December 7th and 21st, 1999 on South Fork Rowdy Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
53 Chinook	20 Chinook	15 Chinook
2 Unknown	18 Unknown	

C9.2.1.4.2 1998-1999 Spawning Surveys

One spawning surveys was conducted on December 16, 1999 on South Fork Rowdy Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
20 Chinook	11 Chinook	4 Chinook
	5 Unknown	1 Unknown

C9.2.1.5 Wilson Creek

The survey reach extends from the Pacific Ocean upstream 5,000 feet to the 1st W10 bridge and then 23,000 feet up to the last W10 bridge.

C9.2.1.5.1 1999-2000 Spawning Surveys:

One spawning surveys was conducted on December 16, 1999 on Wilson Creek during 1999-2000. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	1 Unknown	None Observed

C9.2.2 Coastal Lagoons HPA

Spawning surveys have been conducted on three streams within the Coastal Lagoons HPA during the period 0f 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.2.1 Maple Creek

The spawning survey reach extends from the confluence with North Fork Maple Creek to the gauging station for 4,500 feet. The reach continues for an additional 12,000 feet upstream of the gauging station.

C9.2.2.1.1 <u>1999-2000 Spawning Surveys</u>

One spawning survey was conducted on February 9, 2000 on Maple Creek, tributary to Big Lagoon during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.2.1.2 <u>1998-1999 Spawning Surveys</u>

Two spawning surveys were conducted on December 16, 1999 and January 8, 2000 on Maple Creek during 1998-1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.2.2 North Fork Maple Creek

The survey reach extents from the confluence with Maple Creek to the F-4 bridge, approximately 4,500 feet. It continues upstream an additional 2,600 feet to the anadromous barrier.

C9.2.2.2.1 <u>1999-2000 Spawning Surveys</u>

One spawning survey was conducted on February 9, 2000 on North Fork Maple Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	4 Unknown	None Observed

C9.2.2.2.2 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted on December 16, 1999 and January 8, 2000 on North Fork Maple Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.2.3 Pitcher Creek

Pitcher Creek is surveyed from the confluence with Maple Creek upstream to the anadromous barrier, just past the F-2 road bridge, for a total distance of 4,200 feet.

C9.2.2.3.1 <u>1999-2000 Spawning Surveys</u>

One spawning survey was conducted on April 10, 2000 on Pitcher Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	12 Unknown	None Observed

C9.2.2.3.2 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted on January 8, 1999 on Pitcher Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.3 Little River HPA

Spawning surveys have been conducted on six streams within Little River HPA during the period of 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.3.1 Carson Creek

Carson Creek is surveyed from its confluence with mainstem Little River to the bridge on the M-140 road, a total of 5,000 feet.

C9.2.3.1.1 <u>1998-1999 Spawning Surveys</u>

Two spawning surveys were conducted on December 17, 1998 and January 8, 1999 on Carson Creek, during 1998-1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	6 Unknown	1 Chinook 2 Unknown

C9.2.3.2 Danielle Creek

The survey reach extends from the confluence with mainstem Little River upstream approximately 2,500 feet.

C9.2.3.2.1 1998-1999 Spawning Surveys

One spawning survey was conducted on December 9, 1998 on Danielle Creek during 1998-1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.3.3 Little River

Due to the length and depth of Little River, only two reaches totaling approximately 15,500 feet have been regularly surveyed. This reach extends from the confluence of Carson Creek to the mouth of Railroad Creek for a distance of 7,500 feet and from the mouth of Lower South Fork Little River to the mouth of Upper South Fork Little River for a distance of an additional 8,000 feet.

C9.2.3.3.1 1999-2000 Spawning Surveys

Seven spawning surveys were conducted on December 16th, 20th, 30th, 1999 and February 7th. March 3rd and 17th, and April 2, 2000 on Little River during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
45 Chinook	15 Chinook	21 Chinook
21 Steelhead	8 Steelhead	1 Steelhead
	106 Unknown	1 Coho
		2 Unknown

C9.2.3.3.2 1998-1999 Spawning Surveys

One spawning survey was conducted during December 29 through 30, 1998 on Little River during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
66 Chinook	39 Chinook	17 Chinook
1 Coho	15 Unknown	1 Unknown
6 Unknown		

C9.2.3.4 Upper South Fork Little River

The spawning survey reach extends from the confluence with mainstem Little River upstream 5,000 feet to the V-Line bridge and then continues upstream an additional 2,300 feet to the anadromous barrier.

C9.2.3.4.1 1999-2000 Spawning Surveys

Two spawning surveys were conducted on December 13, 1999 and February 7, 2000 on Upper South Fork Little River during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	4 Unknown	4 Chinook

C9.2.3.4.2 1998-1999 Spawning Surveys

Two spawning surveys were conducted on December 9, 1998 and January 29, 1999 on Upper South Fork Little River during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
13 Chinook	2 Chinook	None Observed
4 Unknown	2 Unknown	

C9.2.3.5 Lower South Fork Little River

The spawning survey reach on Lower South Fork Little River extends from the confluence with mainstem Little River upstream 9,400 feet to the anadromous barrier.

C9.2.3.5.1 1999-2000 Spawning Surveys

Three spawning surveys were conducted on December 16, 1999, February 4th and March 24th, 2000 on Lower South Fork Little River during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
1 Chinook	51 Unknown	6 Chinook
1 Steelhead		2 Coho

C9.2.3.5.2 1998-1999 Spawning Surveys

Two spawning surveys were conducted on December 17, 1998 and January 29, 1999 on Lower South Fork Little River during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
6 Chinook	3 Chinook	1 Unknown
18 Coho	12 Coho	
2 Steelhead	1 Steelhead	
4 Unknown	48 Unknown	

C9.2.3.6 Railroad Creek

The spawning reach extends from the confluence with mainstem Little River upstream to the anadromous barrier approximately for a total of approximately 5,000 feet in length.

C9.2.3.6.1 <u>1999-2000 Spawning Surveys</u>

One spawning survey was conducted on February 7, 2000 on Railroad Creek during 1999-2000. The result of this survey is shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	9 Unknown	None Observed

C9.2.3.6.2 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted on December 9, 1998 on Railroad Creek during 1998-1999. The result of this survey is shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.4 Mad River HPA

Spawning surveys have been conducted on one stream, Cañon Creek within the Mad River HPA during the period of 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.4.1 Cañon Creek

The spawning survey reach for Cañon Creek extends from the confluence with the Mad River upstream 9,200 feet to the 4000 bridge. It then continues the 4000 bridge to the anadromous barrier, an additional 6,000 feet.

C9.2.4.1.1 <u>1999-2000 Spawning Surveys</u>

A total of nine surveys were conducted during the winter of 1999-2000. The dates of the surveys are November 11th, 19th, 22nd, and 30th, December 6th, 15th, and 27th, 1999; January 5th and February 8th, 2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
202 Chinook	73 Chinook	66 Chinook
1 Coho	3 Steelhead	1 Coho
12 Steelhead	65 Unknown	10 Steelhead
4 Unknown		2 Unknown

C9.2.4.1.2 <u>1998-1999 Spawning Survey</u>

Two surveys were conducted during the winter of 1998-1999. These were December 12th, 1998 and January 4th, 1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
66 Chinook	32 Chinook	6 Chinook
	30 Unknown	

C9.2.4.1.3 <u>1997-1998 Spawning Survey</u>

Two surveys were conducted during the winter of 1997-1998. These were conducted on December 6th and 29th, 1997. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
30 Chinook	20 Chinook	22 Chinook
3 Steelhead	2 Steelhead	1 Coho
2 Unknown	81 Unknown	

C9.2.4.1.4 <u>1996-1997 Spawning Survey</u>

One survey was conducted during the winter of 1996-1997. This survey was conducted during December 17th through 19th, 1996. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
110 Chinook	42 Chinook	7 Chinook
4 Coho	1 Coho	1 Coho
3 Unknown	4 Unknown	1 Unknown

C9.2.4.1.5 <u>1995-1996 Spawning Survey</u>

One survey was conducted during the winter of 1995-1996, on December 10th, 1995. The results of these surveys are shown below

Live Fish Observed	Redds Observed	Carcasses Observed
73 Chinook	27 Chinook	4 Chinook
3 Coho	1 Coho	
	3 Unknown	

C9.2.5 North Fork Mad River HPA

Spawning surveys have been conducted on two streams, North Fork Mad River and Sullivan Gulch within the North Fork Mad River HPA during the period of 1996 through 2000.

C9.2.5.1 North Fork Mad River

The spawning survey reach of NF Mad River extends form the confluence with Mad River upstream 11,500 feet to the county bridge at Korbel. The reach continues upstream from the county bridge at Korbel upstream 9,600 feet to the anadromous barrier, just downstream of the first bridge on the K&K road.

C9.2.5.1.1 Spawning Survey 1999-2000

One spawning survey was conducted on NF Mad River during the winter of 1999-2000. The survey date was December 29, 1999. The summaries of the results of this survey follow.

Live Fish Observed	Redds Observed	Carcasses Observed
76 Chinook	42 Chinook	21 Chinook
3 Steelhead	65 Unknown	7 Unknown
3 Unknown		

C9.2.5.1.2 Spawning Survey 1998-1999

Two spawning surveys were conducted on NF Mad River during the winter of 1998-1999. These survey dates were December 11th and 21st, 1998. The summaries of the results of these surveys follow.

Live Fish observed	Redds Observed	Carcasses Observed
42 Chinook	15 Chinook	28 Chinook
1 Steelhead	47 Unknown	5 Unknown
4 Unknown		

C9.2.5.1.3 Spawning Survey 1997-1998

Two spawning surveys were conducted on NF Mad River during the winter of 1997-1998. The survey dates were December 5th and 31st, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
121 Chinook	65 Chinook	61 Chinook
3 Coho	18 Unknown	1 Unknown
4 Unknown		

C9.2.5.1.4 Spawning Survey 1996-1997

Two spawning surveys were conducted on the NF Mad River during the winter of 1996-1997. The survey dates were December 2, 1996 and January 16, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
214 Chinook	213 Chinook	293 Chinook
5 Unknown	7 Unknown	2 Steelhead
		20 Unknown

C9.2.5.2 Sullivan Gulch

The spawning survey reach on Sullivan Gulch extends from the confluence with North Fork of the Mad River upstream to the anadromous barrier. This is a total distance of approximately 2,600 feet.

C9.2.5.2.1 *Spawning Survey 1999-2000*

Four spawning surveys were conducted on Sullivan during the winter of 1999-2000. The survey dates were December 10th and 15th, 1999, January 21st, and February 2nd, 2000. The summaries of the results of this survey follow.

Live Fish Observed	Redds Observed	Carcasses Observed
25 Chinook	9 Chinook	4 Chinook
	13 Unknown	2 Coho
		1 Unknown

C9.2.5.2.2 *Spawning Survey 1998-1999*

Two spawning surveys were conducted on Sullivan Gulch during the winter of 1998-1999. These survey dates were December 11th and 28th, 1998. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
12 Chinook	7 Chinook	None Observed
1 Coho	14 Unknown	None Observed

C9.2.5.2.3 *Spawning Survey 1997-1998*

One spawning survey was conducted on Sullivan Gulch during the winter of 1997-1998. The survey date was December 21st, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
1 Coho	1 Coho	None Observed
1 Unknown	10 Unknown	

C9.2.5.2.4 *Spawning Survey 1996-1997*

One spawning survey was conducted on Sullivan Gulch during the winter of 1996-1997. The survey date was January 9, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
220 Chinook	108 Chinook	102 Chinook
5 Steelhead	2 Steelhead	18 Unknown
1 Coho		

C9.2.6 Humboldt Bay HPA

Spawning surveys have been conducted on one stream, Salmon Creek, within the Humboldt Bay HPA once during the period of 1995 through 2000.

C9.2.6.1 Salmon Creek

Spawning surveys were conducted from the County Bridge on Tompkins Hill Road upstream 8,000 feet to the second temperature recording station, just downstream of the road F-1400 bridge. Additional spot checks were made near the Walsh bridge approximately 14,000 feet upstream.

C9.2.6.1.1 Spawning Survey 1998-1999

One spawning survey was conducted on Salmon Creek during the winter of 1998-1999. The survey date was January 12, 1999. The summaries of the results of this survey follow.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	7 Unknown	None Observed

C9.3 DISCUSSION

Chinook and coho relative abundance within the HPAs have fluctuated since monitoring began in 1995. The Smith River HPA, which includes South Fork Winchuck River, Rowdy Creek, Savoy Creek, South Fork Rowdy Creek and Wilson Creek, has been monitored for adult returns since 1998. Spawning surveys within these streams is sporadic, and often only conducted once in a season. Based on observed returns, no coho were seen during surveys in this HPA. Chinook were fairly common and easily distinguished during surveys. Based on late season results, it appears an adequate number of adult chinook annually escape in this HPA. Although spawning surveys have not detected adult coho, juvenile dive counts and electrofishing within these streams frequently find coho. Their numbers, however, are very low, which may factor into low observed escapement numbers. Steelhead are often seen during late winter surveys in small numbers, however juvenile population estimates within this HPA indicate that adult escapement may be much higher.

The Coastal Lagoon HPA which includes spawning survey reaches on North Fork Maple Creek, Maple Creek and Pitcher Creek are streams that are subject to irregular entry by returning salmonids. These systems are regulated by high flow events that allow for the breaching of the sand spit, which would otherwise block the entry of salmonids into their natal streams. Based on spawning survey results since 1998, it is unclear whether adequate adult escapement is received in these streams due to the timing of when the lagoon breaches. Numerous adult cutthroat trout were incidentally observed in the lower reaches of Maple Creek during a training session of the summer population estimate protocol in 1999. It is not known if the adult cutthroat were either anadromous or “lagoon run”. “Lagoon run” fish may utilize the lagoon in the same way anadromous fish utilize the ocean. Age 0+ and 1+ chinook as well as two 18-inch chinook (also possibly “lagoon run” chinook) were observed during the training session. Age 1+ coho were seen in Pitcher Creek during summer 1999, however no 0+ coho were observed in the system. This indicates that the timing of when the lagoon breaches plays an important role in determining if, when or what species enter the Maple Creek system. The absence of 0+ coho during the summer of 1999 indicates that Big Lagoon did not breach during the 1998/1999 coho run, but the presence of 1+ coho indicates that adults were able to enter during the 1997/1998 spawning season. During the formal spawning surveys only redds of unknown species have been found late in the survey season. It is likely these redds where created by anadromous or “lagoon run” cutthroat or by steelhead that were able to enter the lagoon during high winter flow. All four covered salmonid species have been observed in the Coastal Lagoon HPA, however cutthroat is the only species that have been seen in the adult form.

The Little River HPA is currently the most actively surveyed HPA for adult escapement. Surveys are conducted on six streams, which include Carson Creek, Danielle Creek, main stem Little River, Upper South Fork Little River, Lower South Fork Little River and Railroad Creek. Surveys on these streams have only been conducted since 1998, since the acquisition of the Louisiana Pacific land holdings. The main stem Little River has the highest totals of both redds, live fish and carcasses. The second largest counts have been observed on Lower South Fork Little River. The majority of spawning activity appears to be by chinook, however coho and steelhead are occasionally observed during surveys. Although these surveys would indicate very little spawning activity by these species, they are extremely abundant during summer juvenile dive counts and out-migrant trapping, indicating a fair number of adults are not observed during escapement surveys. This is often a result of survey limitations due to high flows, which often reduce visibility and flush carcasses. Survey frequency and timing are important, but even with the increased surveys adult salmonids will be missed, making it very difficult to rely on adult counts as an intricate component of the monitoring program.

Cañon Creek is currently the only stream surveyed in the Mad River HPA. Survey frequency, spacing and duration have helped to make it the most well monitored creek for adult escapement. Chinook are the most common species observed, followed by steelhead and coho salmon, respectively.

The North Fork Mad River HPA consists of two survey streams, Sullivan Gulch and North Fork Mad River. Chinook are the most frequently recorded species in North Fork Mad River, followed by steelhead and coho, respectively. Chinook salmon escapement appears robust, with only one to two surveys each season recording large adult returns. Steelhead are fairly common in early winter surveys, but the majority of survey dates in late December are probably too early to record significant numbers. Coho are infrequently observed; however, this is likely a factor of water visibility and survey timing. Sullivan Gulch, has been surveyed since 1996. Limited numbers of chinook, coho and steelhead have been observed. Chinook are the most frequently recorded salmonid, but steelhead may also make up a significant component of the survey if conducted later in the year. Based on juvenile population estimates, however, coho also make up a significant portion of the adult run, although they are rarely observed during spawning surveys.

Spawning surveys in the Humboldt Bay HPA are only conducted in Salmon Creek. Surveys were first conducted in 1998, with only seven redds being identified. Limited winter access into the watershed and visibility generally prevents effective survey coverage of the stream. Also, near the mouth of Salmon Creek, a tide gate may limit upstream migration into the watershed.

C9.4 CONCLUSIONS

Salmonid escapement surveys have helped to show that returning adult populations are using the majority of anadromous habitat available in monitored HPA streams. Opportunistic surveys looking at chinook and coho escapement may be helpful in examining age structure, sex ratios, migration timing and hatchery infiltration, however the number of HPA streams, high flows and water visibility limit the utility of the surveys to draw definitive conclusions for adult escapement estimates. Similar information would be helpful for cutthroat and steelhead adults within Plan Area streams, but only limited data can be collected on these species due to variations in their life history patterns, high flows, water conditions and the basic behaviors of the adult fish. Other survey methods such as summer juvenile fish population monitoring and out-migrant trapping are more reliable and consistent approaches to monitor population trends. The spawning surveys may help develop an understanding marine survival, however a much more intensive survey methodology would need to be employed such as adult traps installed across the ownership which would also be best combined with the monitoring of other freshwater life history stages.

Appendix C10. Mad River Steelhead Surveys

CONTENTS

C10.1 METHODS

Comprehensive dive counts of adult summer steelhead in the Mad River were conducted since 1994. These surveys were made in response to sharp declines in summer steelhead counts within index reaches surveyed annually by U.S. Forest Service personnel upstream of Simpson's Mad River property. Counts of both adult (over 16") and half-pounder (12"-16") sized summer steelhead were made. If possible, the presence or absence of an adipose fin was noted on all adult summer steelhead because all summer steelhead produced by Mad River Hatchery have an adipose fin clip.

The snorkel surveys were organized by California Trout and were a cooperative effort involving personnel from California Trout, Simpson, CDFG, USFWS, U.S.D.A. Forest Service, Coastal Stream Restoration, Trinity River Associates, Douglas Parkinson and Associates, Natural Resources Management Corporation, and Redwood Community Action Agency.

The portion of the survey identified as the Simpson reach extends from Deer Creek to the Department of Fish and Game's Mad River Hatchery. This segment consists of eight reaches for a total of approximately 36 miles of the Mad River:

- Reach 1: Deer Creek to Humbug Creek, 4.0 miles
- Reach 2: Humbug Creek to Big Bend, 4.6 miles
- Reach 3: Big Bend to Goodman Prairie, 4.3 miles
- Reach 4: Goodman Prairie to Church Camp, 3.7 miles
- Reach 5: Church Camp to Butler Valley Ranch, 5.8 miles
- Reach 6: Butler Valley Ranch to 4510, 3.7 miles
- Reach 7: 4510 road crossing to 4090 road crossing, 5.0 miles
- Reach 8: 4090 road crossing to Mad River Fish Hatchery, 4.7 miles

Since 1982 the U.S. Forest Service has surveyed 2 Index reaches upstream of the Simpson property from Ruth Dam downstream to Deer Creek. Since 1994 CDFG has surveyed the following reaches of the Mad River upstream of the Simpson property:

- Reach 1:
  (1994-1998): Deer Creek to Humbug Creek, 4.0 miles
- Reach 2:
  (1994, 1997, 1998): Humbug to Big Bend, 4.6 miles
  (1995) Humbug to Wilson, 2.8 miles
  (1995): Humbug to Swing Bridge, 6.2 miles

California Department of Fish and Game annually surveys the Mad River in the reach downstream of Simpson property from the Mad River Hatchery to Kadle Hole near the Highway 299 bridge.

C10.2 RESULTS

The 1994 snorkel surveys were conducted on August 26th and September 27-28, 1994 and covered a total of 59.8 miles of channel between Nelson Flat and the Mad River Hatchery. A total of 306 adult steelhead (265 with adipose fins, 3 with adipose clips, and 38 unknowns) and 172 half-pounder (67 with adipose fins, 0 with adipose clips, and 105 unknowns) were observed (Table C10-1). Nearly half the adult summer steelhead (141) were congregated in two pools. These pools were located below large falls (10-15 feet) over boulders that were probably low flow barriers and most of the steelhead observed below these falls were scarred and bruised. These barriers probably influenced the low fish counts in the Forest Service index reach (only 19 adult summer steelhead in 24 miles of channel) and illustrated the need for more complete surveys.

Table C10-1. Total number of summer steelhead observed in snorkeling dives on the Mad River, 1994-2000.

	Forest Service Reaches	Simpson Reaches		CDFG Reaches		All Reaches
	(Ruth Dam to Deer Creek)	(Deer Creek to Hatchery)		(Hatchery to Kadle Hole)		
Year	(in consistent or in-complete)	Adults (clips)	½ pounders (clips)	Adults (clips)	½pounders (clips)	Total (clips)
1994	18	287 (3)	172 (0)	0 (0)	0 (0)	477 (3)
1995	41	501 (6)	10 (0)	27 (0)	11 (0)	552 (6)
1996	5	422 (41)	26 (0)	88 (0)	0 (0)	541 (41)
1997	5	225 (2)	12 (0)	54 (2)	0 (0)	296 (4)
1998	13	176 (0)	12 (0)	12 (0)	8 (0)	221 (0)
1999	No Survey	78 (0)	15 (0)	7 (0)	10 (0)	110 (0)
2000	No Survey	80 (0)	54 (0)	45 (15)	7 (0)	186 (15)

The 1995 snorkel surveys were conducted between August 24th and 26th and covered a total of 72.8 miles of channel, from Matthews Dam downstream to the Highway 299 bridge. A total of 569 adult steelhead (400 with adipose fins, 6 with adipose clips, and 163 unknowns) and 21 half-pounders (4 with adipose fins, 0 with adipose clips, and 17 unknowns) were observed (Table C10-1). Most of the adult summer steelhead (479 fish) was congregated in the upper two reaches, with large numbers of fish in several pools immediately below the upper falls. Only 40 adult summer steelhead were observed in the nearly 30 miles of channel surveyed above the upper falls.

The 1996 snorkel surveys were primarily conducted on August 26th-27th (with reach #4 completed on September 3rd) and covered the entire river from Matthews Dam downstream to the Highway 299 bridge. A total of 515 adult steelhead (408 with adipose fins, 41 with adipose clips, and 66 unknowns) and 26 half-pounders (12 with adipose fins and 14 unknowns) were observed (Table C10-1). Most of the adult summer steelhead (305 fish) was congregated in the two reaches downstream of the falls, with large numbers of fish in several pools immediately below the falls. Only five adult summer steelhead were observed in the nearly 30 miles of channel surveyed above the falls.

The 1997 snorkel surveys determined that a total of only 288 adult steelhead (284 with adipose fins, 4 with adipose clips) and 12 half-pounders (none with adipose fin clips) were observed (Table C10-1). The 1998 snorkel surveys resulted in steelhead counts of 201 adults (87 with unclipped adipose fins and at least 89 unknown) and 20 half-pounders (all with unclipped adipose fins).

In 1999, the US Forest Service reaches were not surveyed so the total number of steelhead observed were from Deer Creek to Kadle Hole and included the Simpson and CDFG reaches. The Simpson reaches were snorkeled on August 25th (reaches 1-5) and 26th (reaches 6-8). In 1999 only a total 85 adult steelhead were observed within the surveyed area. Of these 85, only seven adults were confirmed have been adipose fin clipped. In addition 25 half-pounders were observed within these reaches (Table C10-1), none of which were confirmed to have been ad fin clipped.

In 2000, the US Forest Service reaches were not surveyed so the total number of steelhead observed were from Deer Creek to Kadle Hole and included the Simpson and CDFG reaches. The Simpson reaches were snorkeled on August 31st (reaches 1-5, 8) and September 1st (reaches 6-7). The CDFG reaches were surveyed on August 25th. In 2000 only a total 80 adult steelhead were observed within the surveyed area. Sixteen of these adults were unknown as to whether they were adipose fin clipped or not and 15 were observed with adipose clips. An additional 54 half-pounders were observed within these reaches (Table C10-1).

C10.3 DISCUSSION

The Mad River summer steelhead dives revealed the importance of conducting complete surveys, as opposed to making basin-wide estimates from index reaches. Prior to 1994 information about Mad River summer steelhead was derived solely from the numbers observed within the Forest Service index reach (above the falls). Until recently some biologists considered the Mad River wild summer steelhead population in danger of extinction. However the 1994-2000 results indicate that the Mad River sustains one of the larger known populations in California, especially considering that dive surveys actually provide a minimal count of only the fish actually observed by divers.

Figure C10-1 summarizes the total numbers of summer steelhead observed in the Mad River for the years 1994-2000. From this information it appears that there is a trend that, since the 1996 dive survey, there has been a decline in the total number of summer steelhead in the Mad River (Figure C10-1). This maybe a result of many factors including differing water-year types, habitat conditions, spawning and rearing success as well as ocean and climatic conditions in the years prior to these surveys.

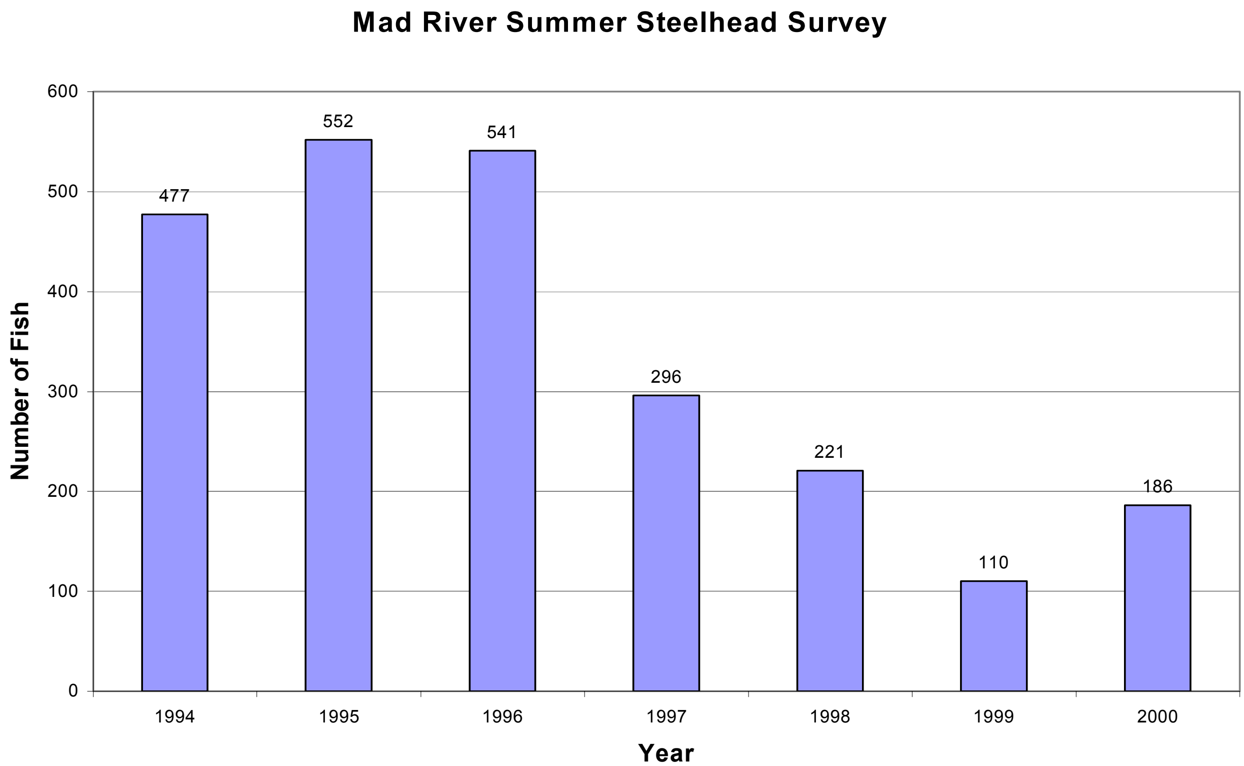

Figure C10- 1. Summary of the total number of Mad River summer steelhead observed (1994-2000).

C10.4 CONCLUSIONS

By conducting the 100 percent surveys annually, the best data for tracking long-term population trends of Mad River summer steelhead will be obtained. The survey results have already resulted in changes in steelhead management by CDFG to better protect this population of wild summer steelhead. Fishing regulations were recently modified to reduce potential impacts from sport fishing by extending the catch-and-release section and prohibiting all fishing within the channel reach where most of the adults are observed. CDFG also terminated its summer steelhead program at the Mad River Hatchery to eliminate the potential for genetic and/or disease exchange from the non-native hatchery fish to the native population.

Appendix C11. Headwater Amphibian Studies and Monitoring

CONTENTS

Tables

C11.1 STUDIES PUBLISHED IN "JOURNAL OF HERPETOLOGY"

- **Distribution and Habitat of *Rhyacotriton variegatus* in Managed, Young Growth Forests in North Coastal California**
- **Distribution and Habitat of *Ascaphus truei* in Streams in Managed, Young Growth Forests in North Coastal California (manuscript as it appeared in the *Journal of Herpetology*)**

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Distribution and Habitat of *Rhyacotriton variegatus* in Managed, Young Growth Forests in North Coastal California

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ABSTRACT.—We examined the distribution and habitat of *Rhyacotriton variegatus* in streams of managed forests in north coastal California. We found 1475 salamanders from 220 streams from 1990-1994 through surveys of randomly selected first and second order streams and incidental searches. Of 71 headwater streams randomly selected to relate landscape variables to the presence/absence of *R. variegatus*, 57 (80.3%) contained salamanders. Geological formation was the only landscape variable that predicted the presence of *R. variegatus* in a stepwise logistic regression model. A second survey was conducted to determine which habitat variables of stream reaches were related to the presence/absence of *R. variegatus*. Thirty-one of 64 stream reaches contained salamanders and stream slope (gradient) was the only variable of 20 measured that entered a stepwise logistic regression model to predict the presence of *R. variegatus*. Pairwise comparisons indicated that reaches with salamanders had significantly higher slope, more small boulders, and less sand. No other variables, including canopy closure and water temperatures, were significant. An additional survey to further define the microhabitat for *R. variegatus* showed that abundance was positively related to stream slope and that this species was found more often than expected in high gradient riffles. The preferred substrate was gravel with smaller amounts of silt/clay, sand, and cobble. We discuss the past and future impacts of timber harvest on this species in north coastal California.

Results of studies in Douglas-fir (*Pseudotsuga menziesii*) dominated forests in the Pacific Northwest suggest that some amphibians are associated with old growth forests (Carey, 1989; Welsh, 1990; Welsh and Lind, 1991) and are sensitive to timber harvest (Bury and Corn, 1988a; Welsh and Lind, 1988; Corn and Bury, 1989; Bury et al., 1991). Torrent salamanders (*Rhyacotriton* spp.) are among stream amphibians that have been reported to be most at risk in the Douglas-fir zone. It has been suggested that local extinction can occur after clearcutting (Bury and Corn, 1988b; Corn and Bury, 1989) and that recolonization may take decades because torrent salamanders have limited dispersal abilities, small home ranges (Nussbaum and Tait, 1977), and are closely tied to cool headwaters and seeps (Nussbaum et al., 1983; Stebbins, 1985).

The southern torrent salamander (*Rhyacotriton variegatus*) is the most southerly distributed of the four species of the genus (Good and Wake, 1992). These salamanders have an aquatic larval stage, lasting perhaps 2-4 yr (Nussbaum and Tait, 1977). Transformed individuals live in the same microhabitats as the larvae. Subadults and adults are occasionally found under objects adjacent to streams and splash zones, but seldom more that 1 m from flowing water (Nussbaum and Tait, 1977). *Rhyacotriton* spp. are sensitive to timber harvest presumably because they require cool microhabitats with clean gravel and cobble (Nussbaum et al., 1983; Corn and Bury, 1989). Timber harvesting may increase deposition of fine sediments and remove canopy cover resulting in elevated temperatures.

Only one study has focused on the relationships between amphibians and logging in the redwood (*Sequioa sempervirens*) zone of north coastal California. Bury (1983) compared one clearcut and one old growth site on four study areas in western Humboldt and Del Norte Counties. He found a slight reduction in the number of species, number of individuals, and the biomass of salamanders in logged compared to old growth sites. The southern torrent salamander (*Rhyacotriton variegatus*; = *R. olympicus* of Bury) was found only in old growth sites, suggesting they are sensitive to timber harvest in this region. However, only one rivulet per site was searched and a total of two specimens of the species was captured.

We conducted a more detailed study at three hierarchical levels of survey to determine the distribution and relative abundance of *R. variegatus* in relation to major landscape variables, to correlate the presence/absence of this species with stream reach habitat variables, and to determine selected microhabitat components associated with sites utilized by *R. variegatus*.

MATERIALS AND METHODS

Study Area.—Our study was conducted on about 1500 km^2 of private timber lands located west of the crest of the Coast Range in western

Del Norte and Humboldt Counties, northwestern California. Most of this property lies within 32 km of the coast, but extends up to 85 km inland in places. The study area is located mostly within the north coast redwood zone (Mayer, 1988) where fog is common. Near the coast, mean summer and winter temperatures are about 18 C and 5 C, respectively, whereas extremes of 38 C in summer and −1 C in winter are not uncommon 48 km from the coast. Precipitation ranges from 102 to 254 cm annually, with 90% falling from October through April (Elford, 1974).

Coast redwood and Douglas-fir are the codominant conifers over most of the study area, but Douglas-fir is more prevalent at higher, drier locations. Hardwoods, such as tanoak (*Lithocarpus densiflorus*), red alder (*Alnus rubra*), Pacific madrone (*Arbutus menziesii*), and California Bay (*Umbelluaria californica*) also are major stand components. Common species along the watercourses surveyed include red alder, big leaf maple (*Acer macrophyllum*), and willows (*Salix* spp.).

Timber harvesting in the north coast area began in the late 1800s when entire drainages were clearcut in a continuum of operations that migrated inland from the coast. In the 1940s, virgin stands in our study area were selectively cut 1–4 times to remove the best redwood and Douglas-fir. Since the late 1960s, even-aged management has been used that involves relatively small clearcuts (average about 24 ha) followed by prompt artificial regeneration. About 97% of the study area consists of 0–80 yr old second and third growth forests, with the following stand age distribution: seedling/shrub (0–9 yr), 13%; sapling/poletimber (10–20 yr), 16%; small sawtimber (21–60 yr), 60%; and large sawtimber (61 + yr), 11%. Prior to 1973, no prescribed protection was given to streams in areas being harvested. Since 1973, California law has required leaving variable-width buffers along streams supporting fish or other aquatic life.

Landscape Surveys.—In 1992 we began systematic surveys of amphibians on the study area by using a stratified random sampling design to select up to four sections per township from U.S. Geological Survey maps. The number of sample sections per township was reduced if the study area was not located in the entire township. Sampling was designed to insure selection of one section per ¼ township (9 sections). Each section chosen had to include at least one half study area and have road access.

We selected 71 sections for a presence/absence survey of *Rhyacotriton variegatus*. We sampled the first headwater stream encountered along the major road through the section. Based on aerial photographs and direct observations of stream flow, the starting point for each survey was selected to ensure a minimum of 200 m of searchable length of stream but no more than about 500 m to the beginning of the wetted channel. If no *R. variegatus* was found, the entire stream reach to the beginning of the wetted channel was walked and all suitable habitat searched, with the greatest effort expended in the best habitat. If *R. variegatus* was found, we recorded distance from the starting point (m, with a hip chain), and the search was continued for a measured distance of 20–30 m to get an estimate of relative abundance. Life history category (larva or transformed) and sex of adults (inspection of cloacal lips, enlarged and squared in males) were recorded for a portion of the animals collected. Forest age of the stream drainage, cover type, stream aspect, elevation, and stream protection history were taken from a G.I.S. data base, aerial photographs (1:12,000 scale), and U.S. Geological Survey topographic maps (1:48,000 scale). Cover types were grouped into redwood, Douglas-fir, redwood/Douglas-fir mix, and hardwoods. Stream protection history was determined by the year of logging and grouped into early (pre-1974 California Forest Protection Act), intermediate (1975 through 1989), or current (1990 to present). The geological formation in which the watershed occurred was taken from U.S. Geological Survey topographic maps overlain by State of California, Department of Forestry geology maps and photographic interpretations (O. Huber, pers. comm.). Thirteen geological formations were identified but grouped into two categories, consolidated and unconsolidated, based on formation age and particle type formed following decomposition. The geologically younger category included unconsolidated marine deposits that decompose into silt and sand whereas the other group was composed of older consolidated formations that form boulders, cobbles, and gravel.

Stream Reach Surveys.—To determine which stream reach habitat variables predict presence of *Rhyacotriton variegatus*, we used the same sampling protocol to select an additional 37 first-order and 27 second-order streams with flows less than 10,500 cm^3/s within the contiguous portion of the study area.

Fixed stream reaches were located 10 m above the roadway or culvert. We established cross-stream transects (Platts et al., 1983) at 5 m intervals starting 2.5 m above the lower end of the transect. We established 10 transects in each sample stream unless physical features (waterfalls, log jams, stream going underground) prevented this. In one stream, only 25 m (5 transects) was sampled, but in all other streams a minimum of 30 m (6 transects) was sampled. We searched streams with a viewing box, where possible, and turned the substrate in search of animals. For each salamander captured, the distance from the lower end of the stream reach and the substrate type where the animal was found (boulder, gravel, cobble, sand) were recorded.

Habitat variables measured (in cm) at each transect included the amount of living vegetative overhang (total linear length up to chest height), small organic debris (SOD, total linear length of leaves, twigs, and sticks <10.2 cm in diameter on the substrate), and logs (total linear length of dead woody material >10.2 cm in diameter occurring over or in the stream but not on the stream bottom). Substrate was classified by measuring along each transect the amount of mud/silt, sand, gravel, cobble, small boulder, large boulder, and large organic debris in cm (Platts et al., 1983). The slope of the stream at each transect was measured by placing the center of a 1 m rod on the transect line parallel to the flow of water and at the stream surface and recording the slope, in degrees, from a clinometer. Canopy closure was estimated by taking readings with a densiometer in each cardinal direction in the center of the stream at the first, fifth, and tenth transects and converting to percent canopy cover.

To reduce the effects of seasonal variation, we estimated stream flow and measured temperature, pH, and conductivity during August and September. Stream flow (cm^3/sec) was estimated by measuring stream depth (cm) at ¼, ½, and ¾ intervals across the stream (Platts et al., 1983), stream width (cm) at this point, and timing the surface speed of a small floating object for three trials. Temperature was taken to the nearest 0.1 C with a Schultheis quick recording thermometer. Conductivity (ms) and pH were taken by an Oakton water test kit. Cover type and aspect were determined from maps.

Microhabitat Surveys.—We conducted a final study to further investigate the relationship between stream slope and presence of *Rhyacotriton variegatus* and to better quantify the microhabitat of this species. For this study, 14 streams (not sampled above) known to have salamanders from incidental sightings were randomly selected for sampling. The headwater portions of these streams were partitioned into low (0–5°), medium (6–10°), and high gradient (>15°) reaches. If available, two 10 meter reaches of each slope were sampled. Due to obstacles in the stream, some reaches had to be shortened to a minimum of 5 m and not all slope categories were available in all streams. The length (m, hip chain) and slope (1 m rod and clinometer, in degrees) of each reach were recorded as noted above. Aspect (compass) and stream temperature (Schultheis quick recording thermometer, 0.1 C) were recorded in the field. A water sample was taken to the laboratory and pH determined with a Beckman 40 pH meter and recorded to the nearest 0.01. Canopy closure was estimated for each reach with a densiometer read at the four cardinal directions and converted to percent canopy cover. For each reach sampled, at least five habitat point samples were taken. The point samples were collected where *R. variegatus* was located or, if no animals were found, at the best available habitat at 2 m intervals, starting at a randomly selected point. Each sample point was assigned to one of four habitat types; cascade, high gradient riffle, low gradient riffle or pool (modified from Platts et al., 1983), because they were the only habitat types readily distinguished in a headwater stream. The dimensions of the habitat type were measured and area (cm^2) recorded. Surface substrate composition was estimated by placing a wire 15 × 15 cm grid with 5 cm mesh on the stream bottom centered on the sample point. At each mesh intersection (12), the substrate type (boulder/bedrock, cobble, gravel, sand, or silt/clay) covered by the intersection was recorded (Cazier, 1993). Vegetative overhang was recorded as the amount overhanging the mesh screen, in percent. The life history stage (larva, transformed) and sex if an adult (inspection of cloacal lips) were recorded for all *R. variegatus* captured.

Data Analysis.—In our analysis of the relationships of landscape variables to the presence and relative abundance of *Rhyacotriton variegatus*, elevation (m) and forest age (0–80 yr) were considered independent continuous variables. Aspect was measured as a continuous variable (0–360°), but grouped into eight 45° octants and treated as a categorical variable. All other variables were treated as categorical variables. We used a stepwise logistic regression analysis, SLR (BMDP Version 7.0; Dixon, 1992) with 10 iterations and *P* values of 0.10 and 0.15 to enter and exit the model, respectively, to determine which of the landscape variables best predicted the presence of *R. variegatus*. We then used Chi-square tests (NCSS, Version 6.0; Hintze, 1995) on the variables 'aspect' and 'geology' to see which category was related to presence of *R. variegatus*. We divided the study area into a northern and southern region for further analysis because of a decreasing gradient of rainfall from north to south (Diller and Wallace, 1994). The Mann-Whitney U test (NCSS, Version 6.0) was used to compare the forest age of stands in the drainage to streams with and without *R. variegatus*.

For the stream reach data, we considered stream order and cover type as categorical variables; all others were treated as continuous variables. We used SLR (BMDP Version 7.0; Dixon, 1992) with 20 iterations and *P* values of 0.10 and 0.15 to enter and exit the model, respectively, to determine which of the reach habitat variables best predicted the presence of *R. variegatus* in the sample reaches. A Chi-square test (NCSS, Version 6.0) was then used on 'aspect' (divided into eight equal octants) to see if there was a relationship to presence of *R. variegatus.* Pooled t-tests were used to compare the continuous variables (slope, canopy closure, and the average amounts of each substrate type) at sites with or without *R. variegatus.*

In our study of microhabitat use, habitat type (high and low gradient riffle, cascade, and pool) was considered a categorical variable; all others were considered continuous variables. We used a logistic regression analysis (NCSS, Version 6.0) with 20 iterations and *P* values of 0.10 and 0.15 to enter and exit the model, respectively, to determine which microhabitat variables were related to the presence of *Rhyacotriton variegatus.* A Chi-square test was then used to determine the relationships between the three categories of habitat type and the presence of *R. variegatus.* (Cascade and high gradient riffle were combined into high gradient habitat, because only five observations were made in cascades.) ANOVA (NCSS, Version 6.0) was used to determine the relationship between relative abundance of *R. variegatus* and the three slope categories. We used a Mann-Whitney U test (NCSS, Version 6.0) to determine if differences existed between average percent surface substrate composition of microsites with and without *R. variegatus.* A significance level of 0.05 was set for all analyses.

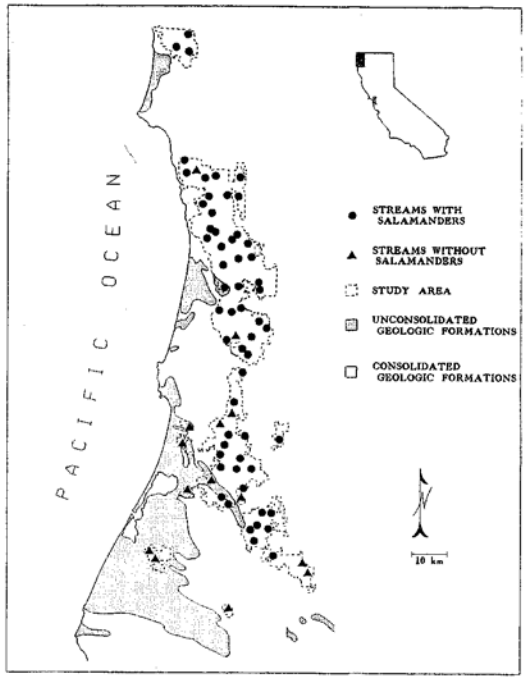

FIG. 1. Map of study area showing consolidated and unconsolidated formations, and streams with and without the southern torrent salamander (*Rhyacotriton variegatus*), north coastal California.

RESULTS

From 1990 through 1994, we found 1475 *Rhyacotriton variegatus* from 220 different streams across the study area, including 410 animals that were found incidentally at 107 sites while conducting other field work. The remaining 1065 individuals (including 72 found in a pilot survey in 1992 that were not included in further analysis) were located at 113 sites from stream surveys. A sample of these animals contained 415 transformed individuals and 498 larvae. The sex ratio of 252 adults was nearly 1:1 (121 females, 131 males).

Landscape Surveys.—We recorded 694 salamanders from 57 of the 71 streams (80.3%) randomly selected from across the study area. The SLR analysis with six independent landscape variables showed that only geology (improvement $\chi^2 = 16.53$, df = 1, $P < 0.001$) and forest age (improvement $\chi^2 = 4.01$, df = 1, $P = 0.045$) entered the model to predict the presence of *Rhyacotriton variegatus.* A greater percentage of streams flowing through the consolidated geologic materials contained *R. variegatus* than those flowing through the younger, unconsolidated materials ($\chi^2 = 21.37$, df = 1, $P < 0.001$). Only one of seven (14.3%) located in the unconsolidated geologic formation contained *R. variegatus* compared to 56 of 64 streams (87.5%) located in the consolidated geologic formations (Fig. 1). Forest age differed significantly among sites with and without *R. variegatus* (Mann-Whitney U test: Z = 2.66, $P < 0.007$). The average age of stands surrounding streams with and without *R. variegatus* was 38.6 years (SD = 30.35, N = 57) and 63.1 years (SD = 42.95, N = 14), respectively. A greater proportion of streams with a northerly aspect (34 of 36) had *R. variegatus* compared to those with a southerly aspect (10 of 18; $\chi^2 = 12.05$, df = 1, $P < 0.001$), and there was a greater proportion of streams with *R. variegatus* in the northern (37 of 39) compared to the southern portion of the study area (20 of 32; $\chi^2 = 11.64$, df = 1, $P < 0.001$). *Rhyacotriton variegatus* was found from 49 to 1219 m in elevation and relative abundance varied from 0.008–1.12 *R. variegatus*/linear m searched (overall average = 0.15/linear m).

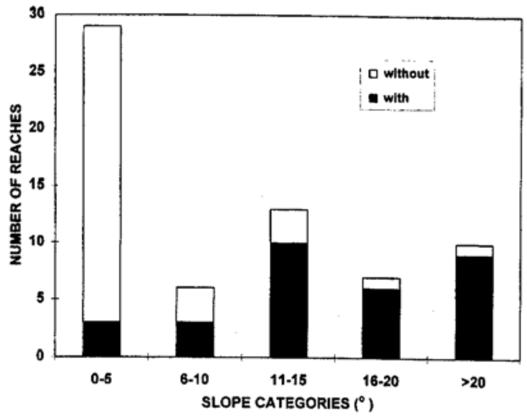

FIG. 2. Number of stream reaches with and without *Rhyacotriton variegatus* in five categories of stream slope (gradient), north coastal California.

TABLE 2. Comparison of the percent surface substrate composition of sites with and without *Rhyacotriton variegatus*. Significance based on Mann-Whitney U Tests.

Substrate composition	Sites with (N = 111)	Sites without (N = 216)	*P*
Boulder/bedrock	2.9	1.7	0.035
Cobble	11.0	19.4	<0.001
Gravel	54.3	46.3	0.004
Sand	10.2	16.2	NS
Silt-clay	21.6	16.4	<0.001

Stream Reach Surveys.—We recorded 109 *Rhyacotriton variegatus* from 31 of 64 stream reaches (48.4%) surveyed. A SLR with 16 continuous and two categorical variables determined that stream slope was the only significant variable to enter a model predicting the presence of *R. variegatus* (improvement $\chi^2 = 24.7$, df = 1, $P < 0.001$). The model with the single variable of slope provided a 82.5% correct classification of stream reaches. The strong relationship between stream slope and salamander presence is illustrated in Fig. 2. A Chi-square analysis of aspect (divided into 8 equal octants) was not significant. Pairwise comparisons indicated that reaches with *R. variegatus* had significantly greater slope, more small boulders, and less sand than those without *R. variegatus* (Table 1). All other comparisons were not significant. Canopy closure was greater and water temperature lower in reaches with *R. variegatus*, but these differences were not significant (Table 1). Estimates of salamander densities at sampling sites varied from 0.014 to 1.26 *R. variegatus*/m² from the 31 stream reaches of first and second order streams (overall average = 0.118/m²).

Microhabitat Surveys.—We collected 190 individuals from the 14 study streams known to have *Rhyacotriton variegatus*. Microhabitat sampling showed that abundance of *R. variegatus* was positively related to slope (ANOVA, F = 20.43, df = 2, $P < 0.001$). A logistic regression analysis without slope but with six continuous variables and one categorical variable showed that habitat type was the most important variable ($\chi^2 = 20.55$, df = 2, $P < 0.001$). Average percent overhang also was significant ($\chi^2 = 4.79$, df = 1, $P = 0.029$) with greater percent overhang at microsites with compared to those without salamanders. *Rhyacotriton variegatus* was found more often than expected in high gradient habitats (cascades and high gradient riffles), and less often than expected in low gradient riffles and pools ($\chi^2 = 53.64$, df = 2, $P < 0.001$). Of 147 microsites with *R. variegatus*, 89.8% were in high gradient habitats, 8.8% in low gradient riffles, and 1.4% in pools. In comparison, 53.3% of 212 mircosites without *R. variegatus* were in high gradient riffles, 36.8% in low gradient riffles, and 9.9% were in pools. The surface substrate in which *R. variegatus* was found was composed mostly of gravel (54%) and the overall ratio of substrate categories was significantly different from sites without *R. variegatus* ($\chi^2 = 141.29$, df = 4, $P < 0.001$; Table 2). Salamander densities at sampling sites varied from 0.09 to 5.0 *R. variegatus*/m² (overall average = 0.28/m²) in 14

TABLE 1. Continuous variables measured at stream reaches with and without *Rhyacotriton variegatus*, north coastal California, 1993. Significance based on pooled t-tests.

Habitat variable	Sites with (N = 31)		Sites without (N = 33)		*P*
	$\bar{x}$	SD	$\bar{x}$	SD	
Slope (°)	17.64	10.20	5.05	6.01	<0.001
Sand (%)	3.29	7.23	10.04	13.60	0.017
Small boulder (%)	11.74	13.65	5.79	8.23	0.037
Canopy cover (%)	85.94	28.78	72.13	37.33	NS
Water temp. (°)	12.51	1.40	12.87	1.27	NS

streams known to have populations of *R. variegatus*. Restricting the analysis to high gradient riffles of these same streams, densities varied from 0.18 to 5.5/m² (overall average = 0.83/m²).

Discussion

Rhyacotriton variegatus is widespread throughout most of the study area at the landscape level and was found in 80.3% of headwater streams surveyed. However, its presence was closely tied to the geological formation of the stream drainage. The small proportion of streams where this species was not found in the consolidated geologic region were typically in areas that had a high proportion of unconsolidated materials even though the site fell within a consolidated geologic type. When our search was confined to a randomly selected stream reach of fixed length (stream reach survey), *R. variegatus* was found only in 48.4% of the reaches. This illustrates that presence of *R. variegatus* was not effectively determined by a sampling methodology that was restricted to a relatively short (30–50 m) randomly selected sample reach.

Data are not available to make direct comparisons of our presence data of *Rhyacotriton variegatus* within headwater streams to other studies because different sampling procedures were employed. However, estimates of the proportion of streams with *R. variegatus* have varied from 28.5% in young forests to 86.4% in old growth areas (Carey, 1989; Corn and Bury, 1989; Welsh et al., unpubl. data).

We found an inverse relationship between presence of *Rhyacotriton variegatus* and forest age rather than a direct relationship as is often reported for *Rhyacotriton* spp. (Welsh and Lind, 1988; Carey, 1989; Welsh, 1990; Welsh et al., unpubl. data). However, this probably is a stasitical artifact produced by a secondary correlation with historical timber harvest patterns; we do not believe that *R. variegatus* favors landscapes dominated by young forests. Historically, coastal forests, where unconsolidated geologic formations were more likely to be encountered, were harvested first (late 1800s to early 1900s) and now have the oldest second growth forests. The more interior areas with steeper topography and shallower soils associated with consolidated geologic formations have been harvested within the last 30 yr. Therefore, we believe the strong association of *R. variegatus* with certain geologic formations and the history of harvesting in our study area produced a spurious association between forest age and presence of *R. variegatus*. The higher proportion of streams with this species in the northern portion of the study area relative to the southern region also is best explained by the fact that only one of the sample streams occurred in an unconsolidated geological formation in the northern portion of our study area. There also is a precipitation gradient with decreasing rainfall from north to south, which may create more favorable conditions for *R. variegatus* in the northern portion of the study area.

The strong association between the presence of *Rhyacotriton variegatus* and steep slopes suggests that this species prefers microhabitats with relatively loose gravel and cobble, open interstices, and minimal fine sediments. We believe that high gradient reaches are important because they are transport areas where finer sediments do not accumulate and gravel and cobble do not become embedded. Good and Wake (1992) also noted that *Rhyacotriton* is associated with areas of "considerable relief" and is generally absent from areas with low relief.

Rhyacotriton requires cold water (Nussbaum et al., 1983; Corn and Bury, 1989) and both aspect and canopy influence water temperature (Beschta et al., 1987; Bury and Corn, 1991). We believe that the positive association between the presence of *R. variegatus* and northerly aspects at the landscape level indicates that water temperature may be limiting to *R. variegatus* in some southerly exposures in our study area.

The lack of a correlation of aspect and canopy closure to presence of *Rhyacotriton variegatus* at the stream reach level would suggest that these variables should be measured over a larger area. We also believe these variables tend to have a lesser impact in our study area because of the influence of the coastal climate. Cool summer temperatures and coastal fog moderate the impacts of variation in aspect and canopy closure on water temperatures. The narrow range of water temperatures measured in all streams (10–16 C) would suggest that the climate of the area moderates impacts on water temperature.

At the level of the microhabitat survey, there was a positive relationship between higher stream gradients and abundance of *Rhyacotriton variegatus*. Corn and Bury (1989) found a similar relationship for streams flowing through forests logged between 14 to 40 yr prior to their study. They noted that this relationship might be suspect because only three streams of 20 contained *Rhyacotriton*, but the species was absent from all logged streams with gradients <11%. They found no relationship between abundance of *Rhyacotriton* and stream gradient in streams flowing through uncut forests.

We found *Rhyacotriton variegatus* significantly more often in high gradient habitats compared to other habitat types, which would be expected given the relationship between abundance and stream slope. This further suggests that *Rhyacotriton* prefers microhabitats where sand is not deposited and interstices remain open. However, *R. variegatus* apparently was selecting for specific microsites within the high gradient riffles where there was more gravel but also more of the finest of sediments. This same type of relationship was noted by Welsh et al., (unpubl. data). They hypothesized that the relationship may be due to the finest sediments being composed of organic material that is important to many aquatic invertebrates and thus may be linked to potential prey for the salamanders (Welsh et al., unpubl. data).

Our surveys were not designed to provide estimates of population densities. In addition, searches of headwater streams often were incomplete, because of large amounts of debris left from past logging. However, our data on relative abundance and salamander densities at sampling sites do provide useful information about the patterns of abundance. Although most headwater streams had *R. variegatus*, their abundance was highly variable from stream to stream (0.014 to 5.0 animals/m^2), a pattern similar to that reported by Welsh and Lind (1992). In addition, the species was patchily distributed within streams. Usually, the best habitat and most *R. variegatus* were located near the upper most portion of the wetted channel, although there was likely some bias in this observation because it was easier to locate animals where there was only minimal flow.

Estimates of salamander densities reported from other studies ranged from 0.01 to 6.7 *Rhyacotriton* spp./m^2 (Bury, 1988; Corn and Bury, 1989; Welsh and Lind, 1992). However, the highest densities reported from single isolated localities are 14–22 individuals/m^2 in a seep (Welsh and Lind, 1992) and 27.6–41.2 individuals/m^2 in a small Oregon headwater stream (Nussbaum and Tait, 1977). Direct comparisons of estimates of salamander densities from this or previous studies are not appropriate because of differences in study designs, and because none of the studies were designed to estimate population densities.

Comparisons between undisturbed and disturbed streams were not possible because virtually all of our study area has been harvested at least once. Consequently, it is difficult to assess the extent to which past timber harvest impacted populations of *Rhyacotriton variegatus*. We believe that in most streams in our study area, habitat probably existed further downstream in lower gradient reaches prior to timber harvest and was reduced or eliminated due to the accumulation of sediments. High gradient reaches were probably less impacted by timber harvest. We do not know how isolated springs and seeps may have been impacted because our surveys were restricted to continuous stream channels. However, incidental observations indicate that some of the highest densities of *R. variegatus* occur in these habitat types within our study area. We conclude that previous unregulated timber harvest practices caused a reduction in the number of individuals in most headwater streams in consolidated geologic areas, but probably did not often cause the total extinction of populations in a stream because virtually all streams in our study area have some high gradient reaches. Our data also suggest that *R. variegatus* is not tied to old growth per se; however, the specific microhabitats required by this species are more likely to exist in undisturbed areas.

Continued survival of this species in our study area cannot directly be assessed. However, several factors suggest that habitat for the species will be maintained and possibly improved. The mean age of forests surrounding streams with *Rhyacotriton variegatus* was 39 yr. Therefore, most stands immediately adjacent to streams with *R. variegatus* will continue to grow for decades. Current timber harvest regulations in California mandate protection for all streams with *R. variegatus* or their habitat. Whereas little or no protection was provided to headwater streams in the past, protection of streams now includes equipment exclusion zones and tree retention standards ranging from 15–30 m on each side of the stream. With these protection zones, better road construction, and improved logging practices (i.e., cable logging), current and future impacts of timber harvest will be significantly less relative to those of the unregulated past.

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Distribution and Habitat of *Ascaphus truei* in Streams on Managed, Young Growth Forests in North Coastal California

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ABSTRACT.—We studied the distribution and habitat of larval *Ascaphus truei* in first and second order streams of managed forests in north coastal California from 1993–1996. Of 72 streams randomly selected to relate landscape variables to the presence of *A. truei*, 54 (75%) contained larvae. Geologic formation was the only landscape variable that predicted the presence of *A. truei* in a stepwise logistic regression (SLR) model. A second survey was conducted to determine which habitat variables of stream reaches were related to the presence of *A. truei*. Larvae were found in 18 (37%) of 49 stream reaches with flows >1500 cm³/sec, and percent fines (negative association with frog presence), stream gradient (positive association), and water temperature (negative association) were the only habitat variables of 21 measured that entered a SLR model to predict the presence of *A. truei*. Only stream gradient differed significantly between reaches with and without tailed frogs; canopy cover, temperature, and forest age were not significantly different. A final survey to further define the microhabitat used by larval *A. truei* showed that larvae were found more often than expected in high gradient riffles and less often than expected in pools and runs. Occurrence of larvae was positively associated with cobble, boulder, and gravel substrates with lower embeddedness, and negatively associated with fine substrates. We discuss the comparative habitat requirements and sensitivities to land management activities of the two amphibian headwater stream inhabitants, *A. truei* and *Rhyacotriton variegatus*, in our study area.

Ascaphus truei, the tailed frog, is unique among North American anurans because it is highly specialized for life in cold, clear, mountain streams (Nussbaum et al., 1983). The larval stage lasts from two to five years (Metter, 1964; Brown, 1990), and tadpoles have an enlarged oral disc modified into an adhesive, sucker-like structure enabling individuals to adhere to rocks in swift current. Tadpoles feed almost exclusively on diatoms which are scraped off rocks (Metter, 1964). Transformed individuals can be found under objects in streams or near the stream margins in daytime. At night, under appropriate conditions of temperature and humidity, they are found on top of objects along the stream and up to 20–30 m from the stream feeding on insects and other invertebrates (Nussbaum et al., 1983). The species is found from southern British Columbia south to northwestern California from the Cascade Mountains west to the coast (Metter, 1968). It also occurs inland as disjunct populations in the Blue Mountains of southeastern Washington and northeastern Oregon, and in the northern Rocky Mountains of northern Idaho and western Montana (Nussbaum et al., 1983).

Ascaphus truei is one of the stream amphibians reported to be at risk in the Douglas-fir (*Pseudotsuga menziesii*) zone and it has been suggested that local extinctions of this species will occur after clearcutting these forests (Bury and Corn, 1988a; Corn and Bury, 1989). These authors have speculated that recolonization may take decades because *A. truei* has limited dispersal abilities and adults tend to breed in their natal stream. They also stated there is a need to assess the effects of logging in streamside and upland forests on headwater and small stream amphibians, such as *Ascaphus* (Bury and Corn, 1988b). No studies have been conducted on the habitat requirements of this species in the redwood (*Sequoia sempervirens*) zone of northwestern California, where a mild coastal climate has been shown to modify its life history patterns (Wallace and Diller, 1998) and may also modify its distribution patterns and habitat requirements.

In 1993, we began an extensive sampling program across the study area to determine the distribution and habitat associations of *Ascaphus truei* at three hierarchical levels of survey. Our study focused on larval *A. truei* because we believe the larval stage, which is restricted to streams, is the most sensitive to the impacts of timber harvest. The objectives of this study were to determine the distribution and relative abundance of this species in relation to major landscape variables, to correlate the presence/absence of the species with stream reach variables, and to determine the specific microhabitat components associated with sites utilized by *A. truei*.

Materials and Methods

Study Area.—Our study area encompassed 1500 km² of private timber lands located west of the crest of the Coast Range in western Del Norte, Humboldt and Trinity counties, northwestern California. Most of the property is within 32 km of the coast, but extends up to 85 km inland in places. The study area is located mostly within the north coast redwood zone (Mayer, 1988) where fog is common. Near the coast, mean summer and winter air temperatures are about 18 C and 5 C, respectively, but extremes of 38 C in summer and −1 C in winter are not uncommon 48 km from the coast. Precipitation varies from 102 to 254 cm annually, with 90% falling from October through April (Elford, 1974).

Coast redwood and Douglas-fir are the codominant conifers over most of the study area, with Douglas-fir becoming more prevalent at higher, drier locations. Hardwoods, such as tanoak (*Lithocarpus densiflorus*), red alder (*Alnus rubra*), Pacific madrone (*Arbutus menziesii*), and California bay (*Umbellularia californica*) also are major stand components. Common species along the watercourses surveyed include red alder, big leaf maple (*Acer macrophyllum*), and willows (*Salix* spp.).

Three major types of logging have occurred in the north coast area in the past; clearcutting entire drainages, selective logging, and—since the late 1960s—even-aged management with small clearcuts and prompt artificial regeneration. As a result of this logging history, the study area mostly consists of 0–80 yr old second and third growth forests with a stand age distribution of: 0–9 yr, 13%; 10–20 yr, 16%; 21–60 yr, 60%; and 61+ yr, 11%. Before 1973, streams were not protected in areas being harvested. Since 1973, state law has required leaving variable-width forest buffers along streams supporting fish or other aquatic life.

Landscape Surveys.—In 1993, we began surveys of amphibians on the study area by using a stratified random sampling design to select up to four sections per township from U.S. Geological Survey maps. The number of sample sections per township was reduced if the study area was not located in the entire township. Sampling was designed to insure selection of one section per 1/4 township (9 sections). Each section chosen had to include at least one half study area and have road access.

We selected 72 sections for a presence/absence survey of *A. truei*. We sampled the first second-order stream encountered along the major road through the section that had at least 1000 m of channel with flowing water. Tailed frogs were surveyed by searching for larvae attached to rocks on the stream bottom. A glass-bottomed viewing box was used to search for larvae across the entire streambed. Each stream was searched for 1000 m or until presence was documented. Once the first *Ascaphus* was found, an additional 20 m was searched to establish relative abundance for that particular stream. Search effort for all streams was concentrated in the best available habitat. Life history category (larvae, juvenile, adult) and sex of adults (presence of tail in males) were recorded for all *Ascaphus* collected. Forest age of the stand adjacent to each stream, stream aspect, and elevation were taken from a geographic information system data base and aerial photographs (1:12,000 scale). Cover types were grouped into redwood, Douglas-fir, redwood/Douglas fir mix, and hardwoods. The geological formation in which the watershed occurred was taken from U.S. Geological Survey topographic maps overlain by State of California, Department of Forestry geology maps and photographic interpretations (O. Huber, pers. comm.). Thirteen geological formations were identified but grouped into two categories, consolidated and unconsolidated, based on formation age and particle type formed following decomposition. The consolidated geologic group was composed of older formations that form boulders, cobbles, and gravel during decomposition into fine sediments, whereas the unconsolidated category included younger marine deposits that decompose directly into silt and sand.

Stream Reach Surveys.—To determine which stream reach habitat variables predict the presence of *A. truei*, we used the same sampling protocol to select an additional 13 first-order and 41 second-order streams with flows greater than

1500 cm^3/sec within the contiguous portion of the study area. Fixed stream reaches were located 10 m above the roadway or culvert. We placed cross-stream transects (Platts et al., 1983) at 5 m intervals starting 2.5 m above the lower end of the reach and established 10 transects in each sample stream unless physical features (waterfalls, log jams, stream going underground) prevented this. In one stream, only 25 m (five transects) were sampled, but in all other streams at least 30 m (six transects) were sampled. We searched streams with a viewing box and turned the substrate in search of animals.

Habitat variables measured (in cm) at each transect included amount of living vegetative overhang (total linear length up to chest height), small organic debris (sod, total linear length of leaves, twigs, and sticks <10.2 cm in diameter on the substrate), and logs (total linear length of dead woody material >10.2 cm in diameter occurring over or in the stream but not on the stream bottom). Substrate was classified by measuring along each transect the amount of mud/silt, sand, gravel, cobble, small boulder, large boulder, and large organic debris in cm (Platts et al., 1983). Stream gradient at each transect was measured by placing the center of a 1 m rod on the transect line parallel to the water flow and at the stream surface and recording the gradient, in degrees, from a clinometer. Percent canopy closure was estimated by taking readings with a densiometer in each cardinal direction in the center of the stream at the first, fifth, and tenth transects.

We estimated stream flow and measured temperature, pH, and conductivity during August and September to reduce the effects of seasonal variation. Stream flow (cm^3/sec) was estimated by measuring stream depth (cm) at 1/4, 1/2, and 3/4 intervals across the stream and dividing by four to get mean depth (Platts et al., 1983), measuring stream width (cm) at this point, and timing the surface speed of a small floating object for three trials. Temperature was taken to the nearest 0.1 C with a Schultheis quick recording thermometer. Conductivity (ms) and pH were estimated by an Oakton water test kit. Cover type, forest age of the stand surrounding the stream, and stream aspect were determined from maps.

Microhabitat Surveys.—We conducted a final study to better quantify the microhabitat associations of *A. truei*. For this study, 17 streams were subsampled from the 54 streams of the landscape survey known to have *A. truei*, using a stratified, random design. We first conducted a stream layout by walking the stream and identifying reaches in each of three gradient classes, 0–5%, 6–10%, and >10%. A reach was recorded if it was at least 20–30 m long, allowing for the placement of two or more sampling belts within that gradient class. We continued upstream until two reaches in each gradient class were identified, or 300 m, whichever was less.

Sampling belts were started 10 m upstream from the road, or beyond the influence of the road, whichever distance was greatest. We randomly placed the first sample belt 0–5 m upstream from the start of the sample reach. Sampling belts were 1.5 m long and assigned to a habitat type (pool, run, low-gradient riffle, or high-gradient riffle). Additional belts were systematically placed at 10 m intervals with a maximum of 15 belts per gradient class. If one gradient class exceeded 150 m (more than 15 belts), we increased the distance between belts to systematically sample over the entire length of the gradient class. If placement of the belt occurred on an unsearchable portion of the stream or between two habitat units, we adjusted the placement of the belt upstream to include a single habitat unit.

Before quantifying microhabitat, the surface of the substrate was visually searched for *Ascaphus* using a viewing box. Five cross-stream transects were then placed within each belt by laying a measuring rod perpendicular to the stream channel at 3 dm intervals beginning and ending 1.5 dm from the lower and upper limits of the belt. We recorded the substrate particle (fines, sand, gravel, small cobble, large cobble, small boulder, large boulder; Platts et al., 1983; and sod or lod) at each 2 dm point. Average water depth (at the midpoint of each sample belt), vegetative overhang, and gradient of the belt were measured as noted above (stream reach survey). Canopy closure was estimated (as above) at the mid-point of each belt and the upstream distance to the nearest log or log jam was measured (directly if 10 m or less, estimated if from 10–30 m, and not recorded if >30 m). Embeddedness of cobbles was visually estimated and assigned to one of four categories (0–25, 26–50, 51–75, and 76–100%) for each sampling belt. Each belt was searched for *Ascaphus* by working upstream and removing all loose objects from the channel while holding an aquarium net downstream of the object. After all loose objects were removed from the channel, the entire belt was searched again with the viewing box. If *Ascaphus* was found, we recorded the following life history data: larva or transformed; snout to tail length and limb measurements of larvae; and snout to vent length and sex of adults or transformed *Ascaphus*.

Data Analysis.—In our analysis of the relationships of landscape variables to the presence of *A. truei*, elevation (m) and forest age (0–117 yr) were considered independent continuous variables. Aspect was measured as a continuous variable (0–360°) but grouped into eight 45° octants and treated as a categorical variable. Geologic formation and cover type were treated as categorical variables. We used a stepwise logistic regression analysis, SLR (NCSS, Version 6.0; Hintze, 1995), with 20 iterations and a P value of 0.20 to enter the model to determine which of the five landscape variables best predicted the presence of *A. truei*. We then used Chi-square analysis (NCSS, Version 6.0) to test for association with the presence of *A. truei* to the variables 'aspect' (divided into four quadrants) and cover type. We divided the study area into a northern and southern region for further analysis because of a decreasing rainfall gradient from north to south (Diller and Wallace, 1994). The Mann-Whitney U test (NCSS, Version 6.0) was used to compare the forest age of stands adjacent to streams with and without *A. truei*.

For the stream reach survey, five of the original 54 streams sampled were omitted from the analysis because of missing data. We considered stream order and cover type as categorical variables; all others were treated as continuous variables. We used SLR (NCSS, Version 6.0) with 20 iterations and a P value of 0.10 to enter the model to determine which of the stream reach variables best predicted the presence of *A. truei* in the sample reaches. Mann-Whitney U tests were used to compare the continuous variables slope, canopy closure, temperature, sod, forest age, and the average amounts of each substrate type of reaches with and without *A. truei*.

In our study of microhabitat use, habitat type (pool, run, low gradient riffle, and high gradient riffle) was considered a categorical variable. For each sample belt, average substrate composition, stream width (dm) and depth (cm), stream gradient (%), distance to the nearest log (m), vegetative overhang (%), and canopy closure (%) were calculated and considered independent continuous microhabitat variables. We used SLR (NCSS, Version 6.0) with 20 iterations, and a P of 0.20 to enter the model to determine which of the microhabitat variables best predicted the presence of *A. truei* in the sample belts. Because substrate particle size is associated with different habitat types (Rosgen, 1996), we ran a second SLR without habitat type as one of the independent variables. We did not use abundance of larval *A. truei* as a dependent variable because tadpoles in some streams were metamorphosing during the survey period and the larval population was declining throughout the survey. Mann-Whitney U tests were used to compare microhabitat variables of sample belts with and without *A. truei*. A Chi-square analysis then was used to compare habitat types and the presence of *A. truei*. Alpha for all analyses was 0.05.

Results

From 1993 through 1996, we recorded 725 *Ascaphus truei* from the study area; 693 were larvae and 32 were transformed juveniles, subadults, or adults. The statistical analyses reported here are based on the 693 larvae.

Landscape Surveys.—We found 443 *A. truei* in 54 (75%) of 72 streams randomly selected from the study area. The SLR analysis with five independent landscape variables showed that only geologic formation (improvement χ^2 = 12.11, df = 1, $P < 0.001$) and forest age (negative association, improvement χ^2 = 7.68, df = 1, $P < 0.01$) entered the model to predict the presence of *A. truei*. The model correctly classified 86% of the streams sampled. A greater percentage of streams flowing through consolidated geologic materials (54 of 67, 81%) contained *A. truei* than those flowing through the younger, unconsolidated materials (zero of five; Fig. 1). There was a significant difference in forest age of stands surrounding sites with and without *A. truei* (Mann-Whitney U test: Z = 1.95, df = 1, P = 0.051), with mean stand age greater at sites without (median = 39.5 yrs, range = 109, N = 18) compared to sites with *A. truei* (median = 32, range = 84, N = 54). There was no significant difference in the proportion of streams with a northerly aspect having *A. truei* compared to those with a southerly aspect (χ^2 = 5.47, df = 3, P = 0.140). However, there was a significantly greater proportion of streams with *A. truei* in the northern (29 of 30, 97%) compared to the southern area (25 of 42, 60%; χ^2 = 12.88, df = 1, $P < 0.001$). Only two cover types, redwood and Douglas-fir, were recorded in drainages of the sample streams and there was no significant difference (P = 0.682) in cover type between streams with and without *A. truei*. Relative abundance of *A. truei* varied greatly among streams. In nine streams, only one to three animals were found within 200–1500 m of stream that was covered in search of suitable habitat (90–450 m actually surveyed), while in four other streams, 24–56 animals were found in 30 to 50 m surveyed. *Ascaphus truei* was found from 24 to 1038 m in elevation. Of the 443 animals captured during the landscape survey, only five were transformed individuals (one juvenile, four adults).

Stream Reach Survey.—We recorded 63 larval *A. truei* from 18 (37%) of 49 stream reaches. The SLR with 18 continuous and three categorical variables determined that percent fines (negative association), stream gradient (positive association), and water temperature (negative association) were the only variables to enter a model predicting the presence of larval *A. truei* (improvement χ^2 = 3.82, df = 1, P = 0.051; χ^2

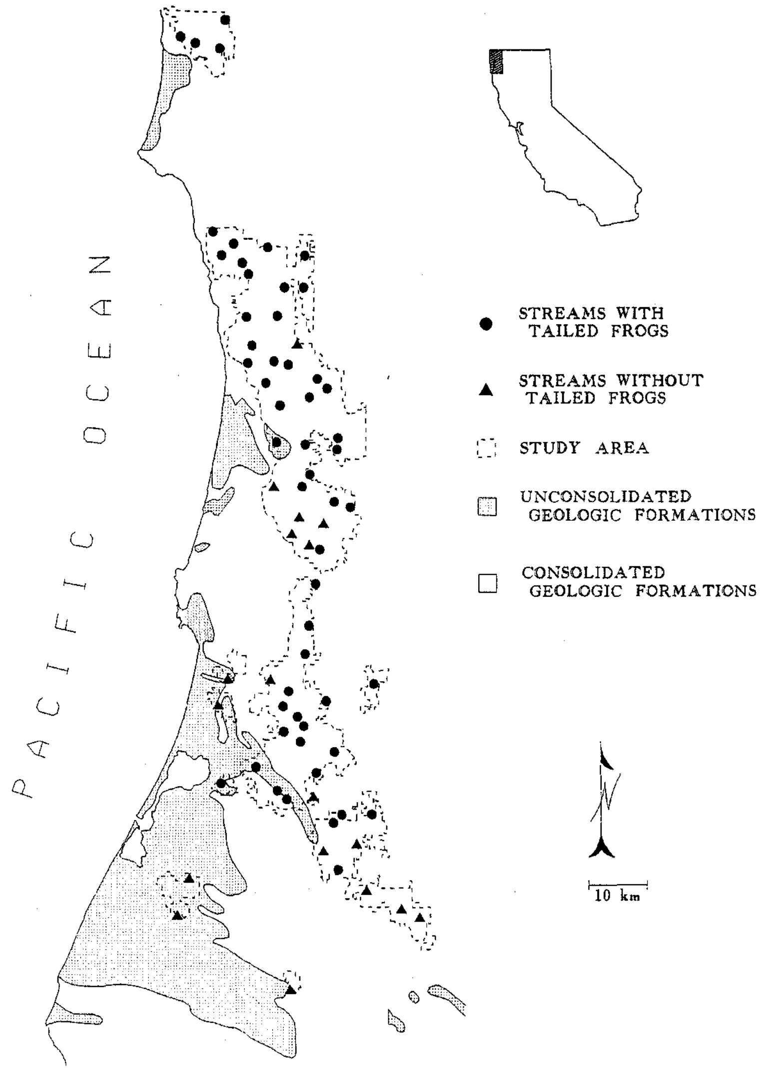

FIG. 1. Map of study area showing consolidated and unconsolidated geological formations, and streams with and without *Ascaphus truei*, north coastal California. Data obtained from landscape survey.

$= 3.79$, df $= 1$, $P = 0.051$; and $\chi^2 = 2.99$, df $= 1$, $P = 0.084$, respectively, but none of these variables were statistically significant, assuming a strict interpretation of the alpha level). The model correctly classified 78% of the stream reaches sampled. Only gradient differed significantly between reaches with and without *A. truei* (Mann-Whitney U test: $Z = 2.45$, df $= 1$, $P = 0.014$) with tadpoles more likely to be found in higher gradient reaches. Canopy cover, temperature, forest age, and aspect did not differ significantly between reaches (Table 1). There were no significant differences in percent substrate composition between reaches with and without *A. truei*. In streams with *Ascaphus* larvae, relative abundance in sample reaches varied from 0.02–0.24 larvae/linear m (overall mean = 0.11).

Microhabitat Survey.—We recorded 192 larval *A. truei* from 17 streams surveyed to determine microhabitat associations of this species. A total of 349 1.5 m-belts was sampled, of which 82 (23%) had *A. truei*. A SLR analysis with one categorical and 15 continuous variables showed that the high gradient riffle habitat type was the first variable to enter the model (positive association, improvement $\chi^2 = 43.80$, df $= 1$, $P < 0.001$). The next three variables entering the model with a significant improvement χ^2 were percent small cobble (positive association, $\chi^2 = 25.06$, df $= 1$, $P < 0.001$), low gradient riffle

TABLE 1. Comparison of selected habitat variables between stream reaches with (N = 18) and without (N = 31) *Ascaphus truei*. Significance based on Mann-Whitney U Tests. * nonsignificant results.

Variable	Sites with *Ascaphus truei* Median	Range	$\bar{x}$ (SD)	Sites without *Ascaphus truei* Median	Range	$\bar{x}$ (SD)	P
Stream gradient (%)	7.1	18.1	9.1 (6.00)	3.6	28.2	5.9 (6.29)	0.014
Canopy cover (%)	100.0	99.3	81.3 (30.67)	100.0	93.5	87.6 (25.95)	0.385*
Water temp (C)	12.0	8.0	12.2 (1.71)	12.5	6.3	12.8 (1.33)	0.124*
Forest age (yrs)	21.5	46.0	23.0 (11.88)	22.0	81.0	25.8 (21.14)	0.884*
Substrate fines (%)	0.0	14.2	2.3 (4.04)	1.5	49.2	8.1 (13.81)	0.286*

(positive association, χ^2 = 11.90, df = 1, $P <$ 0.001), and percent fines (negative association, χ^2 = 6.21, df = 1, P = 0.013). The first model with just one independent variable (high gradient riffle) provided 77% correct classification, while the final model with all four variables only increased the correct classification to 81%. A second SLR analysis, omitting habitat type as an independent variable, found that percent fines was the first variable entering the model (negative association, improvement χ^2 = 41.95, df = 1, $P <$ 0.001), followed by small cobble (positive association), water depth (negative association), and large boulder (positive association) (improvement χ^2 = 21.39, df = 1, $P <$ 0.001; χ^2 = 7.40, df = 1, $P <$ 0.001; and χ^2 = 4.48, df = 1, P = 0.034, respectively). The model provided 78% correct classification. *Ascaphus truei* was found more often than expected in high gradient riffles and less often than expected in pools and runs (χ^2 = 52.37, df = 3, $P <$ 0.001). Of 90 belts with *A. truei*, 81.1% were in high gradient riffles, 15.6% in low gradient riffles, and 3.3% in pools and runs. Sample belts with *A. truei* contained cobble with significantly lower embeddedness, higher stream gradient, and less mean depth (Table 2) than belts without the species. Belts with *A. truei* also had significantly less fines, more gravel, and more cobble. Average densities of larvae in the sampling belts varied from 0.04–0.73 individuals/m^2 (overall average = 0.24/m^2) among the 17 streams sampled. However, if only high gradient riffles were considered, where most larvae were found, the average density varied from 0.20–7.25 larvae/m^2 (overall average = 1.23/m^2).

DISCUSSION

Ascaphus truei was widespread at the landscape level and was found in 75% of the streams sampled. However, its presence was closely tied to the geological formation of the stream drainage. No tailed frogs were found in five streams identified as being in an unconsolidated geologic region of the study area. Several of the remaining 13 streams without *A. truei* appeared during sampling to have a high proportion of unconsolidated geologic material influencing the stream sediments, even though they were identified from maps at the landscape scale as being in consolidated geologic regions. Therefore, we believe that a site specific quantification of the geology of streams sampled would further strengthen our conclusion that geologic formation of the stream basin was an important factor in predicting the occurrence of tailed frogs, due to the influence that it has on the composition of the stream substrate.

It is difficult to directly compare the proportion of streams sampled with *A. truei* in this study relative to other studies, because different sampling procedures were used. However, 75%

TABLE 2. Comparison of selected microhabitat variables between belts sampled with (N = 82) and without (N = 267) *Ascaphus truei*. Significance based on Mann-Whitney U tests. Embeddedness (N = 72 with and N = 151 without) based on a rating system where 1 = 0–25%, 2 = 26–50%, 3 = 51–75%, and 4 = 76–100% embedded.

Variable	Sites with *Ascaphus truei* Median	Range	$\bar{x}$ (SD)	Sites without *Ascaphus truei* Median	Range	$\bar{x}$ (SD)	P
Embeddedness score	2.0	2.0	1.99 (0.54)	3.0	3.0	2.85 (0.82)	<0.001
Fines (%)	5.1	43.3	7.11 (8.10)	15.0	94.6	21.88 (22.63)	<0.001
Gravel (%)	21.8	63.9	21.95 (11.92)	16.7	87.5	19.77 (15.91)	0.029
Small cobble (%)	17.0	56.1	18.91 (10.49)	8.7	43.3	10.04 (8.43)	<0.001
Large cobble (%)	19.1	46.7	20.46 (10.61)	13.6	100.0	15.29 (11.96)	<0.001
Gradient (%)	8.0	54.0	11.40 (9.70)	3.0	60.0	5.72 (8.44)	<0.001
Depth (cm)	5.1	12.0	5.32 (2.27)	6.8	45.3	8.46 (5.95)	<0.001

of the streams with *A. truei* in our study is intermediate to other studies where estimates varied from 35% in young forests to 96% in old growth areas (Corn and Bury, 1989; Welsh, 1990; Bull and Carter, 1996).

We found an inverse relationship between presence of *Ascaphus* and forest age rather than a direct relationship as is often reported for this species (Carey, 1989; Corn and Bury, 1989; Welsh, 1990). However, this probably is a correlation that resulted from past timber harvest patterns; we do not believe *Ascaphus* favors landscapes dominated by young forests. Historically, coastal forests, where unconsolidated geologic formations were more likely to be encountered, were harvested first (late 1800s to early 1900s) and now have the oldest second growth forests. The more interior sites with steeper topography and shallow, rocky soils associated with consolidated geologic formations have been harvested within the last 30 yr. Therefore, we believe that geologic formation has such a profound influence on stream substrate condition that it negates the potential impact of stand age on the occurrence of *Ascaphus* in our study area. The higher proportion of streams with this species in the northern portion of the study area relative to the southern region also is best explained by geology. All of the streams sampled that were in the unconsolidated geologic formation occurred in the southern portion of the study area. There also is a precipitation gradient with decreasing rainfall from north to south, which may create more favorable conditions for *A. truei* in the northern portion of the study area.

At the level of the stream reach, gradient was the only variable that was significantly different in reaches with versus without *A. truei*, and the same variable, with a positive association, entered the SLR model to predict the occurrence of this species. Percent fines, with a negative association, also entered the SLR model to predict the occurrence of *A. truei* within stream reaches. This is likely due in part to the association between stream gradient and substrate, where higher gradient reaches are typically transport areas that do not accumulate fine sediments (Rosgen, 1996). We suspect that the lack of any other significant results with substrate variables was due, in part, to the stream reach being too large of a scale for attempting to quantify variables that correlate best with stream habitat units. As noted below, we did observe significant differences in substrate variables among habitat units with and without frogs at the microhabitat scale.

Ascaphus requires cold water to complete larval development (Brattstrom, 1963; de Vlanning and Bury, 1970; Brown, 1975), and increased water temperature is thought to be one of the short-term impacts from timber harvest that may negatively affect populations of *A. truei* (Bury and Corn, 1988a). Our data provides no direct evidence that water temperature influenced the occurrence of *Ascaphus* in this study area. Water temperature, with a negative association, did enter the SLR model to predict the occurrence of the species, but the variable was not significant at the traditional alpha level of 0.05. In addition, the difference in mean temperatures of stream reaches with and without *A. truei* was small and not significant (12.2 versus 12.8 C, respectively). The minimal impact of temperature on the occurrence of *A. truei* in our study area probably was best explained by the ameliorating influence of the cool coastal climate of this region, which reduces the magnitude of the increase in water temperatures that could occur following timber harvest. We make this suggestion because the range of water temperatures recorded during the stream reach survey only varied from 7.5 to 15.7 C. Furthermore, there was no significant correlation between water temperature and aspect or canopy closure, even though both of these factors are known to influence water temperature (Beschta et al., 1987; Bury and Corn, 1991).

The association of *A. truei* with different substrate types was best seen at the level of the microhabitat survey. There was a consistent pattern of larval *A. truei* being associated with higher gradient riffles and substrate types such as small cobble and large boulder, while being less likely found in pools and runs, and habitat units with greater embeddedness and fine sediment. These findings are similar to those of Corn and Bury (1989) and Bury et al. (1991), who noted that *A. truei* preferred rocky substrates with cobble-sized rocks and was most commonly found in riffles. Hawkins et al. (1988) also found that higher density of larvae was associated with higher water velocities, lower embeddedness, and cobble-sized substrate (10–30 cm). There are a variety of possible reasons why larval *A. truei* might be associated with high gradient reaches, which have higher water velocities (e.g., increased oxygen and reduced predation). However, we believe that the strong association with high gradient riffles was at least partly due to larvae seeking out the habitat type that was less likely to have substrates embedded with fine sediment. This conclusion was reached because we observed that larvae could be found in low gradient riffles or runs when the substrate was not embedded. Unfortunately, this phenomenon did not occur with sufficient regularity to allow quantification of the relationship. The influence of water velocity on habitat selection in larval *A. truei* is largely unknown, and could not be readily elucidated without an experimental design in a controlled environment.

Of the three hierarchical levels of study, the microhabitat study came the closest to providing a density estimate for *A. truei*. We found that mean abundance of larvae in sample belts varied among streams from 0.04 to 0.73 larvae/m² (overall mean = 0.24/m²). In addition to variation among streams, *A. truei* was often patchily distributed within streams, usually dependent on the distribution of appropriate habitat and substrate type. The upstream limit of tadpole distribution within streams was typically restricted to flows greater than 1500 cm³/sec, but incidental observations indicate that subadult and adult frogs often can be found in small headwater portions of streams. The abundance of *A. truei* reported in our study suggests a lower density of larvae compared to uncut and logged streams in the Coast Range of western Oregon (Corn and Bury, 1989). They found a mean abundance of 0.76 *A. truei*/m² (23 uncut streams) and 0.37/m² (20 logged streams), and Hawkins et al. (1988) estimated mean densities of 0.58 to 4.40 larvae/m² in three different classes of watersheds in the Mt. St. Helens region of Washington. However, direct comparisons are not possible since in the first case these authors reported they reconnoitered the stream and then selected a "typical" section to sample a 10 m reach, and in the second case, two larval cohorts occurred in two of the three streams sampled. In the current study, most of our streams contained only one larval cohort (Wallace and Diller, 1998).

Rhyacotriton variegatus, the southern torrent salamander, is a stream species whose distribution overlaps that of *Ascaphus truei* in upper portions of streams. Both species are generally thought to be sensitive to the impacts of land management activities that either increase sediment delivery to the stream or increase water temperature (Bury and Corn, 1988a; Corn and Bury, 1989). Overall, the distribution of *A. truei* mimicked that of *R. variegatus* in our study area, both being associated with consolidated geologic formations and were found in a similar proportion of streams surveyed (Diller and Wallace, 1996). We have recorded *R. variegatus* at a greater number of sites within the study area compared to *A. truei* (304 versus 126, respectively; L. V. D., unpubl. data). However, sites with *A. truei* were generally larger in size relative to sites with *R. variegatus* (10s of m of stream length for *R. variegatus* versus 100s of m for *A. truei*).

At the level of the stream reach, both species showed a positive association with stream gradient, but the association was much stronger for *R. variegatus* compared to *A. truei* (mean gradient 31.8% and 9.1% for reaches with *R. variegatus* and *A. truei*, respectively; Diller and Wallace, 1996). Data from the microhabitat surveys further support the conclusion that both species are less likely to be found in areas with higher levels of fine sediments, although *A. truei* larvae are generally associated with larger substrate compared to *R. variegatus* (cobble versus gravel).

Both species are sensitive to the same types of impacts (increased sediment inputs that result in a higher proportion of fine sediments and embeddedness of the stream substrate, and to a lesser extent, increases in water temperature). However, the results of our studies provide no direct evidence for which species may be the most sensitive to these changes in the physical environment of the stream. In spite of this, we believe that it is possible to predict that based on their occurrence within a watershed, *R. variegatus*, being in the uppermost headwater areas, is more sensitive to direct impacts of land management activities, while *A. truei* is more likely to be influenced by indirect cumulative effects of these activities.

Comparisons between undisturbed and disturbed streams were not possible in our study area because virtually all areas have been disturbed at least once. Consequently, it is difficult to assess the extent to which past timber harvest impacted populations of *Ascaphus truei*. We believe that in most streams in our study area at least some habitat was eliminated due to the accumulation of sediments, with high gradient reaches being less impacted by land management activities. Our data also suggest that *A. truei* is not tied to old growth habitats per se; however, the specific microhabitats required by this species are more likely to exist in undisturbed areas.

Continued survival of this species in our study area cannot be directly assessed, but we believe that stream and riparian habitat conditions should be improving for *Ascaphus truei*. Whereas most streams in our study area with *A. truei* were logged at least once with little or no protection in the past, protection of these streams now includes equipment exclusion zones and tree retention zones from 15–30 m on each side of the stream. With these protection zones, better road construction, and improved logging practices (i.e., cable logging), current and future impacts of timber harvest will be significantly less relative to those of the unregulated past.

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C11.2 MONITORING OF SOUTHERN TORRENT SALAMANDER POPULATIONS

C11.2.1 Introduction

Torrent salamanders are generally found in springs, seeps and the most extreme headwater reaches of streams (Nussbaum et al. 1983; Stebbins 1985). They are a small salamander that appears to spend most of its time within the interstices of the stream's substrate, which make them difficult to locate and capture without disturbing their habitat. The larvae have gills and are restricted to flowing water while adults also appear to spend most of their time in the water, but are capable of movements out of the water. They are thought to have limited dispersal abilities and small home ranges so that recolonization of extirpated sites may take decades (Nussbaum and Tait 1977; Welsh and Lind 1992; Nijhuis and Kaplan 1998). Given the highly disjunct nature of their habitat, individuals at a given site (sub-population) are likely to be isolated from other adjacent sub-populations. The degree of isolation of these sub-populations probably varies depending on the distance and habitat that separates them so that torrent salamanders could be best described as existing as a meta-population.

Although there is some evidence for cumulative effects of sediment input in certain sites, torrent salamanders are primarily vulnerable to potential direct impacts from timber harvest (Diller and Wallace 1996). Direct impacts could include activities such as excessive canopy removal at the site leading to elevated water temperature, operating heavy equipment in the site, or destabilizing soil leading to excessive sediment deposits at the site. Past observations have indicated that these direct impacts can lead to extinction of the sub-population at the site. Due to the survey difficulties noted above, an attempt to get a statistically rigorous estimate of the number of individuals at monitored sites would be impractical. In spite of this, an index of the number of individuals at each site and record the life history stage of each individual captured will be determined. However, given the unreliability of the index of sub-population size, the persistence of individual sub-populations will be used as the primary response variable for the torrent salamander monitoring.

Concerns could be raised that there are too few sub-populations in the meta-population of torrent salamanders to expect to see significant changes over time, or that any loss in sub-populations would threaten the long-term persistence of torrent salamanders within the Plan Area. However, 598 torrent salamander sites (sub-populations) already have been located across Simpson's ownership in the HPAs, and it is estimated that no more than 25-30% of the total potential habitat has been surveyed. In addition, without a formal monitoring protocol, the apparent extinction and re-colonization of several torrent salamander sites have been documented. This would indicate that the meta-population concept does appear to apply to torrent salamanders in this region.

C11.2.2 Objectives

The primary monitoring approach for southern torrent salamanders will employ a paired sub-basin design. Changes in the persistence of sub-populations will be compared in randomly selected sites in watersheds with (treatment) and without (control) timber harvest. In some cases, control sub-basins will not be available in which case changes in sub-populations will be compared to the amount of timber harvest. In either case, the objective will be to determine if timber harvest activities have a measurable impact on the persistence of sub-populations. Therefore, the objective for torrent salamander monitoring will be to determine if their is a difference in the persistence rate for treatment and control sub-populations, and to document any apparent changes in the habitat conditions or index of sub-population size at each site. The monitoring reaches within each sub-basin will be sampled at least one year prior to operations that could influence the treatment sites and every year thereafter. New sub-basins will be added across the ownership until there are 12-15 paired sites well distributed across the Plan Area. Depending on the schedule of harvesting in the treatment sub-basins, it will likely be necessary to monitor a site for more than 10 years to determine if a treatment effect has occurred. (Refer to Appendix D for full details of the field protocol.)

A secondary monitoring objective will be to document long-term changes in torrent salamander populations across Simpson's ownership. Previous studies done within the Plan Area estimated that 80% of all surveyed streams (almost 90% excluding geologically unsuitable areas) had torrent salamander populations (Diller and Wallace 1996). Given that this occurrence rate is near the highest reported for the species even in pristine conditions (Carey 1989; Corn and Bury 1989; Welsh et al. 1992), an additional objective is to sustain the occupancy of torrent salamander populations in streams across the ownership at a minimum of 80% through time. To determine if this objective is being met, the landscape-level survey previously completed (Diller and Wallace 1996) will be repeated at 10-year intervals.

C11.2.3 Thresholds/Triggers

The extinction of a sub-population of torrent salamanders is a stochastic event that will not be likely to occur on a regular basis. As such it will not provide a responsive trigger to incremental changes in habitat conditions for torrent salamanders. However, any extinction of a sub-population will trigger a first phase (yellow light) evaluation to determine if the extinction was likely to be related to management activities. The apparent decline in the index of sub-population size in treatment sites compared to control sites would also trigger a first phase evaluation, but Simpson does not believe these data could be used to determine a reliable estimate of a population trend. Any significant increase in the extinction of treatment sub-populations relative to control streams would initiate a second stage review, but it is likely that this could be documented only after many years of monitoring.

The yellow light thresholds will be:

- any extinction of a sub-population, or
- an apparent decline in the average index of sub-population size in treatment sites compared to control sites.

The red light thresholds will be:

- a statistically significant increase in the extinction of treatment sub-populations relative to control streams, or
- a significant increase in the net rate of extinctions over the landscapes.

The change in the occurrence of torrent salamander populations across the ownership would not be suitable to use as a trigger to initiate management review due to the extended time-lag between successive data points. However, the occurrence of torrent salamanders in streams across the Plan Area would serve as corroborative evidence to support the findings of the meta-population monitoring, and a significant decrease in the occurrence rate would initiate a review of the probable cause of the decline.

C11.2.4 Temporal Scale

Based on previous monitoring of torrent salamander sites, the extinction of a site will likely be due to a catastrophic event (natural or anthropogenic). This will be detected during the first survey season following the event. Therefore, yellow light conditions will trigger an evaluation in a single year. As noted above, the torrent salamander monitoring is not well suited for a red light threshold, because the temporal scale would likely be too long for effective use in adaptive management.

C11.2.5 Spatial Scale

The zone of monitoring influence for a specific site will be determined on a case-by-case basis. Given that torrent salamanders are most likely to be impacted by direct site impacts, assessment of yellow conditions will include a field inspection of the affected site to determine likely causes. Results from all sites will be examined to determine if extirpations or declines are localized, area-wide, or associated with specific management activities, geologies, climatic variations, or other variables. Potential adaptive management changes could occur within a HPA, across the Plan Area, or in all areas with similar geology, for example, depending on the nature of the monitoring results.

C11.2.6 Feedback to Management

As noted above, the extinction of a sub-population of torrent salamanders due to management activities will most likely be caused by the direct impacts of timber harvest. Simpson believes that most of these impacts can be avoided by the proper identification of the site as a Class II watercourse. Ongoing training of the forestry staff will be designed to insure that improper watercourse classification does not occur. However, if it does occur, additional corrective measures such as only utilizing trained biologists to determine watercourse classification on small headwater streams will be employed. Extinctions or apparent declines in numbers that occur for more subtle reasons will be evaluated using habitat data collected at each site such as monitoring water temperature, canopy closure and substrate composition. If the apparent cause is management related, the appropriate adjustments will be made to mitigate future impacts.

C11.2.7 Results to Date

Eight paired sub-basins have already been selected for monitoring southern torrent salamanders including one sub-basin (Poverty Creek) that will serve as a control for two treatment sub-basins (Jiggs and Pollock Creeks). Five were initiated in 1998, two in 1999 and one additional paired sub-basin was selected in 2000 (Table C11-1).

Table C11-1. Summary of southern torrent salamander monitoring sites, 1998-2000.[1]

			Salamanders		
Paired Monitoring Sub-basin	**Site**	**Type**	**1998**	**1999**	**2000**
Blackdog Creek	BD 5400 A	C	6	4	4
Blackdog Creek	BD 5400 B	C	9	27	12
Blackdog Creek	BD 5300 A	T	8	3	5
Blackdog Creek	BD 5300 B	T	18	2	1
Lower NF Mad	Poverty A	C	13	27	18
Lower NF Mad	Poverty B	C	63	87	79
Lower NF Mad	Jiggs A	T	7	6	7
Lower NF Mad	Jiggs B	T	6	5	5
Lower NF Mad	Pollock A	T	9	3	1
Lower NF Mad	Pollock B	T	4	5	11
Upper NF Mad	Canyon A	C	20	21	20
Upper NF Mad	Canyon B	C	8	3	18
Upper NF Mad	Mule A	T	9	9	11
Upper NF Mad	Mule B	T	6	7	2
Panther Creek	O-5 A	C/h	4	6	5
Panther Creek	O-5 B	C/h	8	23	23
Panther Creek	O-6 A	T	8	6	3
Panther Creek	O-6 B	T	3	1	2
Rowdy Creek	R-1700 A	C/h		7	7
Rowdy Creek	R-1700 B	C/h		5	13
Rowdy Creek	R-1000 A	T		13	10
Rowdy Creek	R-1000 B	T		7	3
NF Maple Creek	B (F-10)	C/h		3	3
NF Maple Creek	C (F11.5-1)	C/h		2	2
NF Maple Creek	D (F11.5)	T		5	3
NF Maple Creek	A (F-13)	T		4	6
Surpur Creek	B700A	C			9
Surpur Creek	A400A	C			9
Surpur Creek	B1042B	T			4
Surpur Creek	A400B	T			24
Totals			209	291	320

Note
1 "C" indicates a control site with no timber harvest, C/h represents a control site that will have some limited timber harvesting and "T" indicates treatment sites that will have extensive timber harvesting.

C11.2.8 Discussion

This study has only been going on for three years and there has been no timber harvesting immediately adjacent to any of the torrent salamander monitoring sites. Unlike the tailed frog monitoring protocol (see Appendix D), the torrent salamander protocol is based on the persistence of sites as the primary response variable and not on estimates of abundance of individuals in monitoring reaches. However, the protocol does specify consistent collecting effort over the same sample reach each year so that comparisons of relative abundance of individuals at each site can be made. In spite of the less precise estimate of abundance relative to tailed frogs, there was little annual variation in the number of torrent salamanders collected at monitoring reaches. The mean number of individuals captured per year from 1998-2000 for the 18 sites that were monitored over the entire three years was 11.6, 13.6, and 12.6, respectively. If this pattern persists, it could lend support for using relative abundance as the primary response variable, which would provide much greater sensitivity to the treatment effects for this monitoring approach. Recently, Simpson experimented with marking individual salamanders with a fluorescent elastomer and the initial results have been promising. If this technique proves to be reliable, it will be used to obtain mark-recapture estimates of salamander abundance which will allow tracking of changes in abundance over time.

C11.2.9 Conclusion

This study is in its preliminary stages and it is too early to determine if there were any effects of timber harvest on the persistence of the sites by torrent salamanders. However, most sites seemed to have relatively constant numbers among years and there was no evidence of any local extinction.

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C11.3 MONITORING OF TAILED FROG POPULATIONS

C11.3.1 Introduction

Tailed frog habitat has been characterized as perennial, cold, fast flowing mountain streams with dense vegetation cover (Bury 1968; Nussbaum et al. 1983). To support larval tailed frogs, streams must have suitable gravel and cobble for attachment sites and diatoms for food (Bury and Corn 1988). Streams supporting tailed frogs have been found primarily in mature (Bury and Corn 1988; Welsh 1990) and old growth coniferous forests (Bury 1983; Welsh 1990). Bury and Corn (1988) reported that the frogs seem to be absent from clearcut areas and managed young forests (Welsh 1990). Although these authors did not establish a cause and effect relationship, it is hypothesized that tailed frog populations could be effected by both direct and indirect impacts of timber management. Direct impacts could include activities such as excessive canopy removal at the site leading to elevated water temperature, or destabilizing soil leading to direct sediment inputs at the site. However, tailed frogs may be vulnerable to cumulative impacts from the upper reaches of watersheds that result in elevated water temperatures or excessive sediment loads. In this regard they are similar to the salmonid species except that such cumulative impacts could effect tailed frog populations before the impacts were manifest in the lower fish-bearing reaches of the watershed.

The primary focus of the tailed frog monitoring will be on the larval population. While the adults can move between the stream and adjacent riparian vegetation, the larvae respire with gills and are tied to the stream environment. They require a minimum of one year to reach metamorphosis (Wallace and Diller 1998), which necessitates over-wintering in the streams. They feed on diatoms while clinging to the substrate with sucker-like mouth parts (Metter 1964) and have limited swimming ability. This makes them potentially vulnerable to excessive bed movement of the stream during high flows, which previously have been documented to drastically reduce the larval cohort. As a result of their life history requirements, the larvae provide the most immediate and direct response to changes in stream. In addition, larval tailed frogs can be captured with ease while causing minimal disturbance to the site. Ongoing studies have allowed us to develop a protocol that has been shown to be highly effective in estimating larval populations. Adults can also be captured with minimal disturbance to the site, but in contrast to the larvae, their population size can not be readily estimated. As a result of all the factors discussed above, the primary response variable for the tailed frog monitoring will be the size of the larval population.

C11.3.2 Objectives

The primary monitoring approach will employ a paired sub-basin design. Changes in larval populations of tailed frogs will be compared in randomly selected streams in watersheds with (treatment) and without (control) timber harvest. In some cases, control sub-basins will not be available in which case changes in larval populations will be compared to the amount of timber harvest. In either case, the objective will be to determine if timber harvest activities have a measurable impact on larval populations. The monitoring reaches within each sub-basin will be sampled at least one year prior to operations that could influence the treatment sites and every year thereafter. New sub-basins will be added across the ownership until there are 12-15 paired sites well distributed across the Plan Area. Depending on the schedule of harvesting in the treatment sub-basins, it will likely be necessary to monitor a site for more than 10 years to determine if a treatment effect has occurred. (Refer to Appendix D for full details of the field protocol.)

A secondary monitoring objective will be to document long-term changes in tailed frog populations across Simpson's ownership. Previous studies done within the Plan Area determined that 75% of all surveyed streams (80% excluding geologically unsuitable areas) had tailed frog populations (Diller and Wallace 1999). Given that this occurrence rate is not much lower than the highest reported for the species even in pristine conditions (Corn and Bury 1989; Welsh 1990; Bull and Carter 1996), a secondary objective is to sustain the occupancy of tailed frog populations in streams across the ownership at a minimum of 75% through time. To determine if this objective is being met, the landscape study previously completed (Diller and Wallace 1999) will be repeated at 10-year intervals.

C11.3.3 Thresholds/Triggers

The change in larval tailed frog populations can be used as a trigger to initiate both first and second stage review of management activities. Any significant decrease in the larval populations of treatment streams relative to control streams would initiate a first stage (yellow light) review. A significant decline in treatment streams relative to control streams over a three year period would initiate a second stage (red light) review.

The yellow light thresholds will be:

- any statistically significant decrease in the larval populations of treatment streams relative to control streams, <u>or</u>

- a statistically significant downward trend in both treatment and control streams.

The red light thresholds are:

- a statistically significant decline in larval populations in treatment streams relative to control streams in >50% of the monitored sub-basins in a single year;
- a statistically significant decline in treatment vs. control sites continuing over a three year period within a single sub-basin or;
- a statistically significant downward trend in both treatment and control streams that continues for three years or more.

The change in the occurrence of tailed frog populations across the ownership would not be suitable to use as a trigger to initiate management review due to the extended time-lag between successive data points. However, the occurrence of tailed frogs in streams across the ownership would serve as corroborative evidence to support the findings of the larval population monitoring, and a significant decrease in the occurrence rate would initiate a review of the probable cause of the decline.

C11.3.4 Temporal Scale

If a significant change occurs in the larval populations of treatment streams relative to controls, it will most likely occur during winter high flow events. This change would then be detected during the summer survey season immediately following the winter event. Therefore, the yellow light threshold for adaptive management could be initiated in a single year. The red light threshold would require three years to be initiated.

C11.3.5 Spatial Scale

The spatial scale over which results from an individual monitoring site should apply, (the zone of monitoring influence), will be analyzed on a case-by-case basis. The inherent variability associated with monitoring of a biological indicator necessitates this approach. If a yellow or red light condition is detected, results from all sites across the Plan Area will be examined carefully to determine if the observed population decline(s) appear to be associated with management activity, if they are localized or area wide, and if they appear to be correlated with other factors such as underlying geology or annual climate variation. Field inspection of the problem site(s) will also attempt to identify potential causes of the decline. Because populations in both treatment and control streams could decline for reasons beyond control that may not be related to habitat (e.g. stochastic disease outbreaks), it is essential to examine the results from all monitoring sites to look for patterns in the observed decline. The spatial scale of any resulting adaptive management changes will depend on the particular results. Potential management changes could occur within a HPA, across the Plan Area, or in all areas with similar geology, for example, depending on the nature of the monitoring results.

C11.3.6 Feedback to Management

A decline in tailed frog populations could be caused by a number of factors including elevated water temperatures, change in the algal community due to an increase in insolation or increase in sediment inputs. However, previous research and monitoring of tailed frogs indicated that they were most likely to be impacted by increases in sediment inputs. Given that water temperature, canopy closure, and substrate composition along with the larval populations will be monitored, Simpson believes that the likely cause of a future decline will be determined. If for example some future decline is attributed to sediment inputs, the source of the sediment can be determined, and if it is management related, the appropriate adjustments will be made.

C11.3.7 Results to Date

Eight paired sub-basins have already been selected for monitoring tailed frogs including one sub-basin (Poverty Creek) that will serve as a control for two treatment sub-basins (Jiggs and Pollock Creeks). Five were initiated in 1997, one in 1998, two more in 1999 and one additional paired sub-basin was selected in 2000 (Table C11-2).

Table C11- 2. Summary of tailed frog monitoring sites, 1997-2000.[1]

Paired Monitoring Sub-basin	Site	Type	Tailed Frog Larvae			
			1997	1998	1999	2000
Blackdog Creek	BD 5400	C	86	140	183	30
Blackdog Creek	BD 5300	T	25	76	290	99
Upper NF Mad	Canyon	C	88	103	370	98
Upper NF Mad	Mule	T	79	41	83	78
Lower NF Mad	Jiggs	T	127	136	389	106
Lower NF Mad	Pollock	T	148	272	242	159
Lower NF Mad	Poverty	C		53	90	50
Panther Creek	O5	C/h		107	182	36
Panther Creek	O6	T		122	311	58
Rowdy Creek	R1700	C/h			39	40
Rowdy Creek	R1000	T			153	75
NF Maple Creek	F-8	C/h			121	44
NF Maple Creek	F-line	T			65	30
Surpur Creek	West Fork	C/h				190
Surpur Creek	South Fork	T				27
Totals			553	1050	2518	1120

Note
1 "C" indicates a control site with no timber harvest, C/h represents a control site that will have some limited timber harvesting and "T" indicates treatment sites that will have extensive timber harvesting.

C11.3.8 Discussion

Only one treatment monitoring reach (Jiggs in 1998) has had any significant harvesting to date. In spite of this, the results to date indicate that there is considerable annual variation within monitoring stream reaches for both control and treatment streams. It also appears that the different sites were somewhat in synchrony such that there were generally good and bad years for tailed frog reproduction. For example, the mean number of tailed frog larvae captured per year from 1997-2000 for the 6 sites that were monitored over the entire four years was 92.2, 129.7, 259.5 and 95, respectively. There were almost three times as many larvae produced in 1999 compared to both 1997 and 2000. This may be the result of differential annual reproductive effort by the adult population or differences in larval survival among years. Currently, little is known about the adult population in terms of its size or life history characteristics so that it is difficult speculate as to the cause of these annual fluctuations. In spite of the annual fluctuations in the larval populations, the BACI experimental design that was incorporated in this monitoring program will still allow for the detection of treatment effects since the analysis will be based on a treatment by time interaction. However, these fluctuations will increase the variance in the analysis and therefore decrease the statistically power. As a result, Simpson intends to implement additional studies of the adult population to determine if the effects of annual variation can be removed from the analysis through the inclusion of one or more additional covariates. Simpson currently is experimenting with capturing and marking the adult frogs to determine the feasibility of estimating the size of the adult population. If this proves successful, it would be possible to estimate annual fecundity rates, and subsequently over winter survival rates of the larvae. Having several response variables to monitor would greatly increase the chances of isolating the life history stage that is most sensitive to management activities.

C11.3.9 Conclusion

This study is in its preliminary stages and there has been very little harvesting in any of the treatment sub-basins to date. Therefore, it would be premature to attempt to analyze the data to determine if there were any effects of timber harvest on larval tailed frog populations. However, the data do suggest that there was substantial annual variation in both control and treatment sites, which if not explained through future studies of the adult population, may reduce the statistical power of this monitoring approach.

C11.3.10 Literature Cited

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