Effects of Sediments from the Redwood National Park Bypass Project (CALTRANS) on the Amphibian Communities in Streams in Prairie Creek State Park

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ABSTRACT

Road construction of the Redwood National Park Bypass (CALTRANS) resulted in a large infusion of fine sediments into pristine streams in Prairie Creek State Park during an October 1989 storm event. We employed a habitat-based, stratified sampling design to assess the impacts of these sediments on the densities of aquatic amphibians in five impacted streams by comparing them with densities in five control streams in summer 1990. These same streams were resampled in summer 1996. In this report we will focus on the current sampling results and discuss changes that have occurred between years. Three amphibian species were sampled in numbers sufficient to be informative: tailed frogs (larvae), Pacific giant salamanders (paedomorphs and larvae), and southern torrent salamanders (adults and larvae). Densities of all three species were significantly lower in the sediment-impacted streams in 1990; in 1996 none of the amphibian species densities were significantly lower in impacted streams; although a shift in habitat use was detected. Sediment levels have changed between sampling periods. Pool sediment measures and overall embeddedness indicate significant increases in sediment deposition, particularly in pools.

INTRODUCTION

Amphibian species comprise a major component of the biomass in stream systems throughout the Pacific Northwest and can exceed fish in numbers and total biomass (Hawkins et al. 1983). Streams of the coastal redwood forests of northwestern California are particularly rich in amphibian life, potentially containing up to nine species during the breeding season (one toad, four frogs, and four salamanders) (Stebbins 1985).

Current research indicates a disturbing, worldwide trend in population declines among amphibians (Wake and Morowitz 1991, Blaustein and Wake 1990). Amphibians are sensitive to habitat perturbations in both terrestrial and aquatic environments because of their dual natural histories, highly specialized physiological adaptations, and specific microhabitat requirements (Wake 1990, Vitt et al. 1990). During their aquatic stage many species of amphibian larvae are very specialized in their uses of stream microhabitats for both foraging and cover. Such specialized adaptations can render them susceptible to even minor environmental changes that alter their ability to seek cover from predation and to forage for phytoplankton, zooplankton, insects, and other invertebrates.

The California Department of Transportation (CALTRANS) Redwood National Park Bypass Area is the site of a large highway construction project adjacent to the eastern border of Prairie Creek Redwoods State Park, Humboldt County, California. This part of Humboldt County received over five inches of precipitation during a major storm event from 20-23 October 1989, which resulted in large infusions of sediments from the ongoing road construction into seven drainages in the Park. The fine sediment layer deposited on affected streambeds measured 0.3 to 5.0 cm (Anon

1991). In this report we provide an analysis of the effects of this sediment infusion on densities of the three most abundant native stream-dwelling amphibian species in five of these streams. Our approach was to examine and compare these densities with those of the same species in five unimpacted (control) streams in the same vicinity. These sets of streams were initially sampled in summer 1990, this report contains results from the resampling of these stream sets in summer 1996 and details differences observed between years.

Background

This research project satisfies requirements by the Regional Water Quality Control Board in their Cleanup and Abatement Order 90-8. The research reported here compliments three other projects already undertaken at the Pacific Southwest Research Station. These earlier projects involved assessing impacts of timber harvesting and road building on amphibian populations inhabiting National Forest lands (Welsh and Lind 1991, Welsh et al. 1991, Welsh et al. 1992, Welsh et al. 1997, Welsh and Ollivier 1998). Two of the previous projects employed **extensive** sampling of aquatic amphibians across a wide geographic range in northwestern California and southwestern Oregon. The current research project repeated a previous project with an **intensive** approach where a large number of samples are taken from a few streams. The former intensive study and the project described herein were undertaken on the same ten streams from Prairie Creek Redwoods State Park and Redwood National Park using the same sampling methodology. In addition to information on the effects of sediments, these two studies greatly increased our knowledge of amphibian community variation within and between streams in a limited geographic area.

Objectives

In this study we proposed to quantify the impacts of a human-caused sedimentation event on stream amphibians in a coastal old-growth redwood forest and to compare results across years. Our objectives were as follows: (1) compare densities of common amphibian species in sediment-impacted and unimpacted streams by stream habitat type using the 1996 data; and (2) compare sediment levels over time (1990 vs. 1996 data). Data from the unimpacted streams also provide a standard by which to compare changes over time as the affected streams flush sediments and amphibian life responds.

METHODS AND ANALYSES

Sampling Methods

We resampled five impacted and five unimpacted streams for amphibian life in order to evaluate and compare the effects of construction induced sediment deposits from a strong 20-23 October 1989 storm event (Anon. 1991) on the species composition and densities of common native, stream -dwelling amphibians. This 'paired sets' study design assumes that amphibian community composition and densities in the unimpacted streams reflects the composition and densities present in the impacted streams prior to the storm induced sedimentation event. These two sets of streams were originally sampled in the summer of 1990 for similar objectives as for this project.

All streams were located in Prairie Creek State Park except one control stream (Little Lost

Man Creek) in adjacent Redwood National Park (Figure 1). These streams were of similar size, physical structure, orientation, and had similar vegetation cover.

Mesohabitat Typing and Mapping of Streams

Our method was derived from fish population sampling methods developed by Hankin (1984, 1986) and Hankin and Reeves (1988), and stream habitat classifications developed by Bisson et al. (1982), and McCain et al. (1990). We modified Hankin's design to subsample within natural habitat units by randomly placing bank-to-bank belts for area-constrained searches (ACS; see Welsh et al. 1997, Welsh 1987, Bury and Corn 1991).

Before sampling for amphibians, each stream was mapped from mouth to headwaters. This mapping included subdivision and classification of each stream at the level of geomorphological reach type (alluvial, braided, or confined) and channel type (Rosgen 1994), if streams varied among these categories. We simultaneously mapped and classified each stream habitat type (e.g., backwater pool, main channel pool, run/glide, riffle, step run, step pool, seep; see Appendix I) (modified from Hawkins et al. [1993]) (Welsh et al. 1997, Figure 2). These stream habitat types were defined as mesohabitats. We inventoried habitat-forming structures within the stream channel (e.g., logs), measured pool sediment (e.g., Lisle and Hilton 1992), and recorded a number of other physical variables (see Appendix I).

Selection of Units for Amphibian Sampling

After streams were mapped, we selected quasi-systematic samples of individual mesohabitat units from within each mesohabitat type. For each mesohabitat type, we sampled a random mesohabitat unit between the first and the fifth. All streams are located east of Highway 101 (Figure 1). We then selected every fifth unit (i.e., k=5) thereafter until all mesohabitat units of a given type were exhausted. This method of selection ensured (a) selection of at least one unit from each identified mesohabitat type, (b) approximately equal sampling effort within each mesohabitat type, (c) good spatial coverage of selected units from among all those units of a given type, and (d) independence of sampling among identified mesohabitat types, thus allowing valid statistical comparisons between different habitat types.

Individual selected mesohabitat units were generally too long to allow complete enumeration of amphibian presence so that subsampling within the selected units was necessary. When complete enumeration of a unit was impossible, we counted numbers of amphibians within two or more 1.0 m wide belts running bank to bank, perpendicular to stream flow (Figure 2). The width of the belts was increased from 0.6 m as sampled in 1990. In mesohabitat units < 5 m long, we sampled the entire unit. In those units between 5 and 10 m long, two non-overlapping belts were placed randomly within the unit. In units > 10 m, we located the first belt at a random distance between 0 m and 10 m from the unit's downstream edge; additional belts were placed every 10 m thereafter until the end of the unit was reached (Figure 2). Adjustments were made in the 1996 sampling, such as requiring a minimum unit of 2 belts per habitat unit or a census of the unit. This change led to a large increase in the number of belts sampled in 1996 over 1990. In 1990, 267 belts were sampled in 10 streams, in 1996 this number grew to 448 belts in the same 10 streams.

Measurements of Habitat Characteristics

We then measured or estimated, numerous biotic and abiotic variables including: microclimate, water velocity, water depth, substrate composition, embeddedness, overstream canopy closure, and available substrate and instream cover (e.g., white water, bedrock ledges, banks, roots, other woody materials, and vegetation). A complete list of all variables measured or estimated, with definitions and descriptions of the method used for each, is provided in Appendix I. All data collection occurred from June through October of 1996.

Enumeration of Amphibians

Belts were thoroughly searched for all amphibians using ACS (Welsh et al. 1997, Welsh 1987, Bury and Corn 1991; see also quadrat sampling Jaeger and Inger 1994, Shaffer et al. 1994). We first scanned the belt area for visible animals and then all cover objects were turned or moved systematically, working upstream and across until the entire belt was searched. Animals were spotted using a plexiglass bottomed box held at the water surface, and then captured with a metal mesh net (e.g., kitchen strainer) or a fabric aquarium net held downstream of the animal. Individual amphibians were identified, sexed, measured, and released. We replaced all cover objects after sampling. Stream width at the belt location and area searched were recorded for later use in calculation of amphibian densities. We assumed that observed captures in a given belt were equal to the true number of amphibians present in that belt in the open or under the first layer of substrate. The chance of missing an amphibian during a thorough search is believed slight (Bury and Corn 1989). It is possible that some animals may escape notice, however. Thus density estimates derived should be considered minimum values.

Comparisons of Unimpacted versus Impacted Streams

Stream Macrohabitat Variables

These variables were included as descriptors of each stream sampled. Log counts within stream channels are presented per kilometer of stream (Appendix II). In order to rule out the possibility that differences in reach or channel composition may have influenced the amphibian species composition or densities between sets of streams, we tested for differences in these categories. The composition of each stream, by reach type (alluvial, braided, or confined), and channel type (Rosgen 1985), was determined and presented as a proportion of stream length (Appendix II). We tested for differences using student t tests (Zar 1994). The alpha level for these tests was set at <0.05 with a bonferroni adjustment applied for all tests (Stevens 1986).

Stream Habitat Composition

We sampled approximately 7 % of the available mesohabitat area of each stream using the spacing and belt size described above. Length of stream surveyed and percent of stream length by mesohabitat type are presented in Table 1. In order to insure that differences in amphibian densities detected between our unimpacted and impacted streams could not be attributed to differences in mesohabitat composition, we tested for differences in structural composition between unimpacted and impacted sets of streams. An additional habitat type has been added since the 1990 sampling, we sampled seeps in addition to the other stream mesohabitats. We then performed unpaired student's t tests (Zar 1998) using the mean proportions of these five mesohabitat types for each set of streams. The alpha level was set at <0.05 with a Bonferroni

Table 1. Distribution of mesohabitats within 10 streams sampled for aquatic amphibians in Prairie Creek Redwoods State Park and Redwood National Park, California in 1996. Kilometers surveyed and percent of stream length by mesohabitat type are reported for seven composite habitat types after Hawkins et al. (1994). Means, standard errors (in parentheses), and comparisons of impacted and unimpacted streams using student's t are reported.

Stream	Km surveyed	Backwater Pool	Main Channel Pool	Glide /Run	Riffle	Step Run	Step Pool	Seep
Unimpacted								
Corkscrew	0.336	1.70	8.27	40.95	5.80	17.74	19.91	5.54
Good	1.040	1.60	54.85	9.11	6.52	75.56	0	1.58
Little Lost Man	1.342	4.96	9.43	6.39	4.23	45.65	18.14	11.21
S. Fk. Big Tree	0.083	0	5.64	0	14.53	32.41	44.42	3.00
Sweet	1.277	0.52	3.57	3.82	2.17	70.36	0.99	18.60
$\bar{X}$	0.816	1.756	16.352	12.054	6.650	48.344	16.692	7.986
	(0.256)	(1.930)	(21.642)	(16.499)	(4.711)	(24.613)	(18.073)	(6.981)
Impacted								
Big Tree	1.318	3.05	10.34	8.67	8.20	32.53	29.02	8.14
Boyes	0.559	3.92	27.11	13.94	13.56	41.48	0	0
Brown	1.412	40.16	16.83	13.09	3.41	49.51	8.12	5.00
N. Fk. Big Tree	0.799	1.06	4.37	4.11	3.45	20.88	62.58	3.53
Ten Tapo	0.476	3.47	4.22	35.25	2.63	0	48.17	6.28
$\bar{X}$	0.913	10.332	12.574	15.012	4.610	28.880	29.578	4.584
	(0.193)	(16.710)	(9.637)	(11.974)	(4.644)	(19.337)	(26.317)	(3.077)
t[1]	0.303	1.140	-0.357	0.325	-0.135	-1.391	0.903	-0.997
P	0.770	0.316	0.731	0.754	0.896	0.202	0.393	0.348
Power	0.058	0.145	0.062	0.060	0.052	0.233	0.126	0.143

[1] Probability values for significant t were interpreted using the Bonferroni method to maintain an overall significance level of 0.10, which requires that each individual test after the first have a $P \leq 0.016$ (Stevens 1983). High variability caused the backwater pool comparison to be calculated using the unpooled method at 4.11 degrees of freedom, all others were compared at 8 df.

adjustment applied for all tests (Stevens 1986; alpha level for each test therefore equaled <0.01).

Pool Sediment Measurements

We used the pool mesohabitats in order to get a direct measurement of the sediment load in each stream (Lisle and Hilton 1992). The percent of the pool tail embedded was visually estimated for each mesohabitat unit (Appendix I), and sediment depths (in cm) were taken at three locations in each pool bowl: at the top, middle, and bottom (Figure 3). The three sediment measurements were averaged prior to analysis. We then performed unpaired student's t tests (Zar 1998) using the mean pool bowl sediment depth and the mean percent embedded for each set of streams.

Analysis of Amphibian Abundance

We present capture summaries by habitat type for the three most abundant species: larval tailed frogs (*Ascaphus truei*) (Stejneger 1899), larval and paedomorphic (animals with larval morphology and sexual maturity) Pacific giant salamanders (*Dicamptodon tenebrosus*) (Good 1989), and larval and adult southern torrent salamanders (*Rhyacotriton variegatus*) (Good and Wake 1992) (Table 2). These species were detected and sampled in numbers sufficient for statistical analyses. We made no effort to distinguish larval and paedomorphic Pacific giant salamanders and they are lumped in our analysis. Only 7 adult tailed frogs were captured; because of this small sample and their terrestrial habitat associations compared with the aquatic larvae, they were omitted from the analysis. Fifteen adult Torrent salamanders were found, however, they occurred in the same aquatic habitats as the larvae so the two age classes were lumped for our analyses (Table 2).

Table 2. Total numbers captured of three amphibian species sampled in 10 streams in Prairie Creek Redwoods State Park and Redwood National Park in summer 1990 (a) and 1996 (b). Numbers of adults shown in parentheses.

Mesohabitat type	# Units surveyed	# Units sampled	*Ascaphus truei* TAILED	*Dicamptodon tenebrosus* PGS	*Rhyacotriton variegatus* STS
a) 1990 Sampling:					
Riffle	185	47	62 (2)	63	18
Run/Glide	35	15	4	14	0
Step run	84	27	95 (2)	77	9
Step pool	98	31	32 (1)	72 (3)	12
Main channel pool	116	47	9 (1)	58 (2)	0
Backwater pool	16	12	3	12	0
Total (1990):	**534**	**179**	**205 (6)**	**196 (5)**	**39**
b) 1996 Sampling:					
Riffle	106	31	53	130 (1)	7 (1)
Run/Glide	141	33	15	187 (1)	1
Step run	240	62	145 (6)	507 (5)	20 (6)
Step pool	86	25	35	156 (2)	5
Main channel pool	164	40	13	186 (3)	0
Backwater pool	82	25	1 (1)	57 (2)	0
Seep	50	17	4	18 (1)	23 (8)
Total (1996):	**869**	**233**	**266 (7)**	**1241 (15)**	**56 (15)**

The individual mesohabitat units were the units for our analysis of abundance; each unit was considered an independent sample. In the case of multiple belts in a habitat unit, the capture densities for the belts were averaged. Tailed frog larvae were seldom found in backwater or main channel pools (when they did occur in these types it was at the pool tail crest-riffle head interface). Southern torrent salamanders were never found in backwater pools or main channel pools. Pacific Giant salamanders were found in all mesohabitat types. In all, we sampled 448 belts in 233 mesohabitat units in the 1996 sampling (Table 2).

Captures from all belts in a particular stream were totaled and divided by the belt area searched to provide densities (captures/m^2) by mesohabitat type and overall for each species . Species capture data were averaged, by mesohabitat type and unit, for the impacted and unimpacted sets of streams. We used a two-factor analysis of variance (ANOVA, Type I; Milliken and Johnson 1984) to examine the relative effects of sediment impacts and mesohabitat structure on the densities of these amphibians; this procedure was followed by Tukey pairwise comparisons. Dependent variables were transformed to meet the assumption of normality.

In contrast to the stricter alpha level of 0.05 used in testing for differences in the composition of the streams by mesohabitat, reach, channel type, and pool sediment levels, we set the alpha level at $P < 0.10$ for the ANOVA. This moderate alpha level provides a criterion more appropriate for the detection of ecological trends (Toft and Shea 1983, Toft 1991). In addition, this moderate level reduces the chances of type II errors (i.e. not detecting a real difference that is present). In applied situations related to management of ecological systems, making a type II error is often more tangible and costly than making a type I error (Toft and Shea 1983).

Variable Transformations

All dependent variables were examined using normal probability plots and measures of skewness and kurtosis (SAS 1990). All non-normal variables (amphibian densities) were transformed to meet the assumption of normality required for parametric statistical analysis. These continuous data were transformed using the natural logarithm (Zar 1998).

RESULTS AND DISCUSSION

Comparisons of Forest and Stream Habitat Structure between Impacted and Unimpacted Streams

Forest Macrohabitat Structure

We surveyed approximately the same amount of stream as in the 1990 sampling; with approximately 4.6 km of impacted streams and 4.1 km of unimpacted streams in Prairie Creek Redwoods State Park and Redwood National Park (Little Lost Man Creek). Complete results of the analysis of forest structure and composition along these streams are summarized Welsh and Ollivier (1990, Table 2). These streams traverse a mixed coniferous-hardwood forest dominated by large, mature coast redwoods. This forest is also characterized by high volumes of woody debris (logs and snags). Both the size and number of large trees, and the abundance of logs and snags, are characteristic of advanced successional or old-growth forest ecosystems in the Pacific Northwest (see Franklin and Hemstrom 1981).

Stream Macrohabitat structure

Counts of logs in stream channels (Appendix II; logs/km) indicated that the downed woody debris from the forest contributes an important component of the habitat structure within these streams, forming many pools and step runs. There was no significant difference found in log abundance between impacted and unimpacted stream sets (Appendix II). Comparisons of mean proportions of streams by reach type and channel type indicated no significant differences in overall composition in these categories between impacted and unimpacted sets of streams (Appendix II).

Stream Habitat Composition

Analysis of the mesohabitats along these streams indicated a predominance of step runs, followed by step pool, glide/run, main channel pool, riffle, seep and backwater pool habitats (Table 1). Our tests of mean proportions of each mesohabitat type for the two sets of streams revealed that they were essentially identical in mesohabitat composition (Table 1). This result allowed us to assume that any differences in amphibian densities detected between the unimpacted and impacted streams are not attributable to differences in overall mesohabitat composition.

Pool Sedimentation

We found mean sediment depths in the impacted pools ranged from 0 to 42 cm compared with 0 to 27 cm in the unimpacted pools. Average sediment depths in the pool bowl have increased over time in both stream sets. Percent of pool tail embedded ranged from 0 to 100% on the impacted streams and from 0 to 100% on the unimpacted streams. Pool tail embeddedness increased over time in Impacted streams while decreasing in Unimpacted streams. Comparisons of both mean pool bowl sediment depth and pool tail percent embedded between the sets of streams showed significantly greater sediment effects in the impacted streams (Table 3). A significant decrease in percent substrate embeddedness did occur between the 1990 and 1996 sampling periods (Figure 4). Percent fines in the impacted streams has remained at the same levels seen in 1990 in the pool habitats. In the fast-water habitats % fines appears to have declined significantly. Levels of sand sediment have remained constant since the 1990 sampling.

Comparisons of Amphibian Abundance Between Impacted and Unimpacted Streams

We conducted area-constrained searches (ACS) on 229 belts in impacted streams and 219 belts in unimpacted streams. We captured a total of 1570 amphibians, with larval and paedomorphic individuals of the Pacific giant salamander being the most common, followed by larval tailed frogs, and larval and adult southern torrent salamanders (Tables 2 & 4). We subset our capture densities for each species by mesohabitat type for each set of streams in order to assess the relative roles of sediment impact and mesohabitat use (Table 5). We used a 2-factor ANOVA, blocking for impact (sediment) and mesohabitat type. Sedimentation had no significant influence on densities of the three species (Table 6a). The Pacific giant salamander, tailed frog and southern torrent salamander showed significant differences in abundance by mesohabitat type. These results differ from the 1990 ANOVA result that showed significant differences with respect to Impact (sediment) and significant interaction effects between Impact and habitat type (Welsh and Ollivier 1998). - NOT CITED - WRONG DATE?

Following the ANOVA, pairwise comparisons among mesohabitat types were conducted for each species (Table 6b). These tests yielded significant results for all three species. Pacific giant

Table 3. Comparisons of sediment depths and pool tail embeddedness estimates in impacted and unimpacted streams of Prairie Creek Redwoods State Park and Redwood National Park, Humboldt County, California, summer 1990 (a) and 1996 (b). Mean ( ) sediment depths were calculated using measures of sediment depth (cm) at the top, middle, and bottom of the pool bowls.

	Impacted $\bar{X}$ (SD)[1] n = 5	Unimpacted $\bar{X}$ (SD) n = 5	df[2]	t[3]	P[4]
a) 1990 Sampling:					
Sediment depth pool bowl	1.522 (0.81)	0.306 (0.19)	4	3.28	0.0300
Pool tail percent embedded	62.6 (8.7)	44.2 (7.2)	8	3.64	0.0067
b) 1996 Sampling:					
Sediment depth pool bowl	7.09 (3.29)	4.20 (1.60)	8	1.77	0.1100
Pool tail percent embedded	47.4 (8.1)	24.6 (11.3)	8	3.67	0.0064

[1] SD = standard deviation
[2] df = degrees of freedom
[3] t = student's t test
[4] P = probability of result by chance alone, set at $P \leq 0.05$.

Table 4. Capture totals for three amphibian species (larvae only) sampled in 10 streams in Prairie Creek Redwoods State Park and Redwood National Park in a) summer 1990 and b) 1996.

Stream	# Belts	# Belts with captures	*Ascaphus truei*	*Dicamptodon tenebrosus*	*Rhyacotriton variegatus*
a) 1990 Sampling:					
Unimpacted					
Corkscrew	17	13	10	15	9
Good	37	26	32	33	7
Little Lost Man	37	29	65	60	0
S. Fk. Big Tree	7	6	7	9	2
Sweet	32	22	23	35	12
Total (Unimpacted):	**130**	**96**	**137**	**152**	**30**
Impacted					
Big Tree	39	30	33	45	4
Boyes	17	6	2	7	0
Brown	42	32	27	69	1
N. Fk. Big Tree	25	13	2	17	3
Ten Tapo	14	8	4	6	1
Total (Impacted):	**137**	**89**	**68**	**144**	**9**
Total (1990):	**267**	**185**	**205**	**296**	**39**
b) 1996 Sampling:					
Unimpacted					
Corkscrew	25	22	1	55	6
Good	48	47	15	169	4
Little Lost Man	60	57	201	233	6
S. Fk. Big Tree	21	18	1	32	5
Sweet	65	64	4	194	8
Total (Unimpacted):	**219**	**208**	**222**	**683**	**29**
Impacted					
Big Tree	79	72	9	189	7
Boyes	29	4	0	6	0
Brown	61	58	29	226	0
N. Fk. Big Tree	39	38	3	115	13
Ten Tapo	21	18	3	22	6
Total (Impacted):	**229**	**190**	**44**	**558**	**26**
Total (1996):	**448**	**398**	**266**	**1241**	**55**

Table 5. Densities of three amphibian species (larvae only) sampled in 10 streams in Prairie Creek Redwoods State Park and Redwood National Park in summer 1996. Average density, standard deviation, range and number of habitat units are reported by mesohabitat type.

Mesohabitat type	*Ascaphus truei*		*Dicamptodon tenebrosus*		*Rhyacotriton variegatus*	
	Impacted	Unimpacted	Impacted	Unimpacted	Impacted	Unimpacted
Backwater pool	0	0.01 (0.03)	0.57 (0.81)	1.17 (0.88)	0	0
	0	0-0.12	0-2.48	0-2.60	0	0
	13	12	13	12	13	12
Main channel pool	0	0.10 (0.38)	0.62 (0.47)	1.39 (1.40)	0	0
	0	0-1.46	0-1.59	0-5.45	0	0
	24	15	24	15	24	15
Riffle	0.34 (1.07)	0.44 (0.73)	1.25 (1.56)	1.93 (1.35)	0.08 (0.30)	0.07 (0.26)
	0-4.55	0-1.85	0-6.06	0.07-4.51	0-1.26	0-0.95
	18	13	18	13	18	13
Run/Glide	0.06 (0.22)	0.13 (0.35)	0.94 (1.03)	2.30 (2.21)	0.04 (0.14)	0
	0-0.83	0-1.34	0-2.78	0-9.02	0-0.56	0
	15	19	15	19	15	19
Step pool	0.03 (0.09)	0.39 (0.60)	1.29 (1.13)	1.97 (1.86)	0.03 (0.11)	0.10 (0.14)
	0-0.37	0-1.60	0-4.29	0.27-5.79	0-0.48	0-0.31
	18	7	18	7	18	7
Step run	0.27 (0.54)	0.48 (0.81)	2.66 (1.93)	2.02 (1.29)	0.11 (0.26)	0.11 (0.24)
	0-2.31	0-3.52	0-8.00	0-4.55	0-0.98	0-1.08
	29	33	29	33	29	33
Seep	0.05 (0.16)	0.01 (0.04)	0.35 (0.43)	0.27 (0.42)	0.72 (1.09)	0.44 (0.71)
	0-0.49	0-0.10	0-1.11	0-1.06	0-3.33	0-1.67
	10	7	10	7	10	7

Table 6. Partial hierarchical analysis of variance, using Type I Sums of Squares, of three aquatic amphibians by impact (presence or absence of sediment), stream number, and composite habitat type, followed by Tukey pairwise comparisons of habitat types. We sampled 233 habitat units in Prairie Creek Redwoods State Park and Redwood National Park, Humboldt County, California, in summer 1996.

Factor	DF	MSE	*F*	Pr. > *F*	Result
a) ANOVA results					
Dependent: Pacific giant salamander					
Overall model	63, 232	0.170	3.91	0.0001	
Tests					
Impact	1, 8	2.822	1.47	0.2594	
Habitat type	6, 42	2.016	8.49	0.0001	See Tukey result
Impact * Habitat type	6, 42	0.269	1.13	0.3607	
Dependent: Tailed frog					
Overall model	63, 232	0.058	2.59	0.0001	
Tests					
Impact	1, 8	0.316	0.63	0.4514	
Habitat type	6, 42	0.334	4.90	0.0007	See Tukey result
Impact * Habitat type	6, 42	0.030	0.43	0.8516	
Dependent: Southern torrent salamander					
Overall model	63, 232	0.030	2.04	0.0002	
Tests					
Impact	1, 8	0.006	0.07	0.7921	
Habitat type	6, 42	0.286	8.82	0.0001	See Tukey result
Impact * Habitat type	6, 42	0.014	0.43	0.8567	

b) Tukey pairwise comparison results[1] (All comparisons Habitat type irrespective of Impact)

Pacific giant salamander

Seep Backwater Pool Main Pool Riffle Step Pool Glide/Run Step Run

Tailed frog

Backwater Pool Main Pool Seep Glide/Run Step pool Riffle Step Run

Southern torrent salamander[2]

Backwater Pool Main Pool Glide/Run Step Pool Riffle Step Run Seep

[1] Amphibian mean density increases left to right; lines indicate non-rejecting subsets.

[2] No Southern torrent salamanders were found in backwater or main channel pools.

salamanders were more abundant in step runs (average density of 1.079 salamanders / m^2) and significantly less abundant in seeps (mean = 0.233 salamanders / m^2) as compared to other habitat types. Tailed frog larvae were more abundant in step runs and riffles (mean = 0.238 and 0.211 salamanders / m^2, respectively) as compared with other mesohabitat types in these streams. Southern torrent salamanders were significantly more abundant in seeps (mean = 0.354 salamanders / m^2) as compared to all other mesohabitat types. These results do not agree with the strong negative influences related to the sediment deposits from the October 1989 storm events seen in the 1990 analysis results. Immediately following the sediment deposition negative influences on the populations of all three species in the affected streams were seen. Results we reported here for the 1996 sampling (Table 6), indicated that habitat type was probably the primary influence in the impacted streams.

The southern torrent salamander showed the most mesohabitat specificity, being absent from main channel and backwater pools, and occurring predominately in seeps (Table 6b). When they were detected in association with a main channel habitat type these were typically found in the margins of step runs, riffles and step pools. The specific habitat associations could reflect a response to lower sediment loads in these mesohabitats, but the lack of an interaction effect (Table 6) suggests that this is probably an indication of an evolutionary adaptation to these specific mesohabitats. As such, this species may be able to compensate somewhat for the negative effects of heavy sedimentation by its choice of mesohabitats with greater flow or using those that are not subject to heavy sediment effects from the main channel (e.g., seeps).

Tailed frog larvae remained very specific in mesohabitat use, showing a strong association with step runs and riffles versus all other mesohabitats. These faster-water mesohabitats are less prone to trapping sediment due to the higher, more uniform velocity of water (Lisle and Hilton 1992). However, results for the tailed frog showed no significant interaction effect between sediment impact and mesohabitat type for the 1996 sampling (Table 6). In the Welsh and Ollivier (1998) habitat type and impact were shown to have a significant interaction suggesting that tailed frog larvae were adversely impacted even in those mesohabitats that were likely to have lower sediment loads immediately after the sediment deposition event.

Levels of substrate embeddedness have significantly declined in the impacted streams in the years between sampling passes (Figure 5). Substrate embeddedness can be seen as a surrogate for the availability of interstitial spaces among the larger substrate items for use by stream amphibians. It is probable that the significant effects identified in Welsh and Ollivier (1998) have been alleviated by the decline in embeddedness, and thus a decline in the filling of interstitial spaces used by the larvae and adults of the tailed frog. As in Welsh and Ollivier (1998), adult tailed frogs were poorly sampled by our methods because of their predominantly terrestrial habits, and hence were not included in our analyses. However, we have observed them to use interstitial spaces at the bottoms of streams for diurnal cover during warm weather (H. Welsh, pers. obs.). It is probable that sedimentation has a negative impact on adult tailed frogs by reducing this critical summer, dry season cover, particularly at more interior sites in northwestern California.

The Pacific giant salamander was the least mesohabitat specific, showing a relatively stronger association with step runs than other main channel habitat types and a lack of use of seeps (Table 6). As a habitat generalist, then, this species is affected by sedimentation across the range of mesohabitats, but probably more so in pool mesohabitats where fine sediment accumulation is greatest (Lisle and Hilton 1992). However, because the giant salamander used more of the available mesohabitats, it is probably best able of the three species to cope with habitat loss resulting from any sedimentation. It should be noted that this species shows a shift in habitat use between the two time periods. In the 1990 sampling, the salamanders still used pools quite significantly. In the 1996 sampling, they had moved out of the pools into water with greater velocity and thus potentially cleaner substrates with more interstitial spaces for them to use. Further support of this is the decrease in fine sediment noted in the fast water habitats, while the pool habitat sediment levels have remained the same with respect to fines. The fines are continuing to fill interstitial spaces used by Pacific giant salamanders in the pool and slow water habitats. The shift in habitat use may be a response to availability of cover for this sit and wait predator.

Bury and Corn (1988) discussed the potential negative impacts of mass wasting events on stream amphibians of the Pacific Northwest. Welsh and Ollivier (1998) documented significant impacts to the Prairie Creek system due to the sediment deposition event associated with a highway bypass. Such impacts have been documented for similar stream systems in connection with logging activities and associated road building (Beschta 1978, Burns 1972, Reid and Dunne 1984, Rice et al. 1979). Corn and Bury (1989) documented differences in amphibian species richness and in the density and biomass of southern Torrent salamanders, tailed frog larvae, and Pacific giant salamanders in logged versus unlogged streams in southern Oregon. They attributed these declines to loss of critical microhabitat due to infusions of fine sediments. Populations of stream amphibians can be particularly sensitive to increased siltation because they frequent interstitial spaces among the loose coarse substrates that comprise the matrix of most natural streambeds of the Pacific Northwest (Bury and Corn 1988, Corn and Bury 1989, and this study). Sedimentation fills these spaces, reducing available cover and foraging area and undoubtedly having similar impacts on the prey base these amphibians depend upon.

Newcombe and MacDonald (1991) reviewed the literature on the effects of sediments on aquatic ecosystems. They concluded that both concentration and duration of exposure are important factors in assessing the magnitude of such impacts on the biota. The October 1989 storm event that triggered the sediment infusion into our study streams from the highway bypass project deposited a 0.3 to 5.0 cm deep layer of fine sediment on streambed surfaces (Anon 1991). As much as three quarters of the average annual load of sediment may have been transported during that storm and subsequent runoff (Anon 1991). As of this writing (December 1998) we have no information of when these sediments may finally be flushed from the streams. We have reported in this current study a reduction in substrate embeddedness, potentially indicating a trend toward reduction of sediment across the habitat types sampled. Unfortunately, we have also reported an increase in pool bowl sediment depths and pool tail embeddedness, indicating that the sediment may only be moving to habitat types with low flows that have a high potential for sediment deposition and storage. It would, therefore, be prudent to continue monitoring these streams to determine the long-term impacts on amphibian populations.

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LITERATURE CITED

Alabaster, J. S., and R. Lloyd. 1982. Finely divided solids. Pp. 1-20, *in*: J. S. Alabaster and R. Lloyd, editors. Water quality criteria for freshwater fish. Second edition. Butterworth, London, UK.

Anderson, J. D. 1960. Cannibalism in *Dicamptodon ensatus*. Herpetologica 16: 260.

Anderson, J. D. 1968. *Rhyacotriton* and *R. olympicus*. Catalogue of American Amphibians and Reptiles 68: 1-68.2.

Anonymous. 1991. Monitoring the impacts and persistence of fine sediment in the Prairie Creek watershed 1989-1990. USDI NPS Redwood National Park Report. March 1991, 123 pp.

Beschta, R. L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research 14: 1011-1016.

Bingham, B. B., and J. O. Sawyer, Jr. 1991. Distinctive features and definitions of young, mature, and old-growth Douglas-fir/hardwood forests. Pp. 362-377, *in*: L. F. Ruggiero, K. B. Aubry, A. B. Carey, and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. USDA Forest Service, Pacific Northwest Station, General Technical Report PNW-GTR-285, Portland, OR.

Bisson, P. A., J. L. Nielson, R.A. Palmason, and L.E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization during low streamflow. Pp. 62-73, *in*: N. B. Armantrout, editor. Acquisition and utilization of aquatic habitat inventory information. Bethesda, MD: American Fisheries Society Western Division.

Blaustein, A. R., and D. B. Wake. 1990. Declining amphibian populations: a global phenomenon? Trends in Ecology and Evolution 5(7): 203-204.

Brattstrom, B. H. 1963. A preliminary review of the thermal requirements of amphibians. Ecology 44: 238-255.

Brown, H. A. 1975. Temperature and development of the tailed frog, *Ascaphus truei*. Comparative Biochemistry and Physiology 1975(50A): 397-405.

Burns, J. W. 1972. Some effects of logging and associated road construction on northern California streams. Transactions of the American Fisheries Society 101: 1-17.

Bury, B. R. and P. S. Corn. 1988. Responses of aquatic and streamside amphibians to timber harvest: a review. Pp. 165-181, *in*: K. J. Raedeke, editor. Streamside management: riparian wildlife and forestry interactions, College of Forest Resources, University of Washington, Seattle, WA.

Bury, B. R. and P. S. Corn. 1991. Sampling methods for amphibians in streams in the Pacific Northwest. USDA FS Pacific Northwest Research Station, General Technical Report GTR-275. 29 pp.

Bury, R. B.., P. S. Corn, K. B. Aubry, F. F. Gilbert, and L. L. C. Jones. 1991. Aquatic amphibian communities in Oregon and Washington. Pp. 353-362, *in*: L. F. Ruggiero, K. B. Aubry, A. B. Carey, and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. USDA, Forest Service, General Technical Report PNW-GTR-285, Pacific Northwest Station, Portland, OR.

Chamberlin, T. W., R. D. Harr, and F. H. Everest. 1991. Timber harvesting, silviculture, and watershed processes. Pp. 181-205, *in*: W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19, Bethesda, MD.

Claussen, D. L. 1973. The thermal relations of the tailed frog, *Ascaphus truei*, and the Pacific treefrog, *Hyla regilla*. Comparative Biochemistry and Physiology 1973(44A): 137-171.

Costanza, M. C., and A. A. Afifi. 1979. Comparison of stopping rules in forward stepwise discriminant analysis. Journal of the American Statistical Association 74: 777-785.

Corn, P. S. and R. B. Bury. 1989. Logging in Western Oregon: Responses of headwater habitats and stream amphibians. Forest Ecology and Management 29(1989): 39-57.

Corruccini, R. S. 1975. Multivariate analysis in biological anthropology: some considerations. Journal of Human Evolution 4: 1-19.

Daugherty, C. H., and A. L. Sheldon. 1982. Age-determination, growth, and life history of a Montana population of the tailed frog (*Ascaphus truei*). Herpetologica 38(4): 461-468.

deVlaming, V. L., and R. B. Bury. 1970. Thermal selection in tadpoles of the tailed frog, *Ascaphus truei*. Journal of Herpetology 4(3-4): 179-189.

Franklin, J. F., and M. A. Hemstrom. 1981. Aspects of succession in the coniferous forests of the Pacific Northwest. Pp. 222-229, *in*: D. C. West, H. H. Shugart, D. B. Botkin, editors. Forest Succession. Springer-Verlag, New York, NY.

Furniss, M. J., T. D. Roelofs, and C. E. Yee. 1991. Road construction and maintenance. Pp. 297-323, *in*: W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19, Bethesda, MD.

Good, D. A. 1990. Hybridization and cryptic species in *Dicamptodon* (Caudata: Dicamptodontidae). Evolution 43(4): 728-744.

Good, D. A. and D. B. Wake. 1992. Geographic variation and speciation in the torrent salamanders of the genus *Rhyacotriton* (Caudata; Rhyacotritonidae). University of California Publications in Zoology, University of California Press, Berkeley, CA. 91 pp.

Gradwell, N. 1971. *Ascaphus* tadpole: experiments on the suction and gill irrigation mechanisms. Canadian Journal of Zoology 49: 307-332.

Hawkins, C. P., M. L. Murphy, N. H. Anderson, and M. A. Wilzbach. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Canadian Journal of Fisheries and Aquatic Sciences. 40: 1173-1185.

Hawkins, C. P., L. J. Gottschalk, and S. S. Brown. 1988. Densities and habitat of tailed frog tadpoles in small streams near Mt. St. Helens following the 1980 eruption. Journal of the North American Benthological Society 7(3): 246-252.

Jennings, M. R. 1996. *Ambystoma californiense* (California tiger salamander) burrowing ability. Herpetological Review 27(4): 194.

Lisle, T. 1989. Sediment transport and resulting deposition in spawning gravels, north coastal California. Water Resources Research 25(6): 1303-1319.

Lisle, T., and S. Hilton. 1992. The volume of fine sediment in pools: an index of sediment supply in gravel-bed streams. Water Resources Bulletin 28(2): 371-383.

Lisle, T., and J. Lewis. 1992. Effects of sediment transport on survival of salmonid embryos in a natural stream: a simulation approach. Canadian Journal of Fisheries and Aquatic Sciences 49: 2337-2344.

McCain, M., D. Fuller, L. Decker, and K. Overton. 1990. Stream habitat classification and inventory procedures for Northern California. FHR Currents, USDA Forest Service, Six Rivers National Forest, Eureka, CA. No. 1, 15 pp.

Metter, D. E. 1963. Stomach contents of Idaho larval *Dicamptodon*. Copeia 1962(2): 435-436.

Metter, D. E. 1968. The influence of floods on population structure of *Ascaphus truei* Stejneger. Journal of Herpetology 1: 105-106.

Milliken, G. A., and D. E. Johnson. 1984. Analysis of messy data. Vol 1: designing experiments. Van Nostrand Reinhold Co., New York, NY.

Murphy, M. L., and J. D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Sciences 38(4): 137-145.

Murphy, M. L., C. P. Hawkins, and N. H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Transactions of the American Fisheries Society 110(4): 469-478.

Neff, N. A. and L. F. Marcus. 1980. A survey of multivariate methods for systematics. American Museum of Natural History, New York, NY: privately published. 243 pp.

Newcombe, C. P., and D. D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal Fisheries Management 11(1): 72-82.

Nussbaum, R. A. 1969. The nest of the Olympic salamander, *Rhyacotriton olympicus.* Herpetologica 25(4): 277-278.

Nussbaum, R. A., and G. W. Clothier. 1973. Population structure, growth, and size of larval Dicamptodon ensatus (Eschscholtz). Northwest Science 47(4): 218-227.

Nussbaum, R. D. and C. A. Tait. 1977. Aspects of the life history and ecology of the Olympic salamander, *Rhyacotriton olympicus* (Gaige). American Midland Naturalist 98(1): 176-199.

Nussbaum, R. A., E. D. Brodie, and R. M. Storm. 1983. Amphibians and reptiles of the Northwest. University of Idaho Press, Moscow, ID. 332 pp.

Parker, M.S. 1991. Relationship between cover availability and larval pacific giant salamander density. Journal of Herpetology 25(3): 355-357.

Platt, W. S., W. F. Megahan, and G. W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. USDA Forest Service, Intermountain Experiment Station, General Technical Report GTR INT-138. Ogden, UT. 70 pp.

Reid, L. M., and T. Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research 20: 1753-1761.

Rencher, A. C. 1988. On the use of correlations to interpret canonical functions. Biometricka 75(2): 363-365.

Rice, J. C., R. D. Ohmart, and B. W. Anderson. 1986. Limits in a data-rich model: modeling experience with habitat management on the Colorado River. Pp. 79-86, *in*: J. Verner, M. J. Morrison, and C. J. Ralph, editors. Wildlife 2000: modeling habitat relationships of terrestrial vertebrates. University of Wisconsin Press, Madison, WI.

Rice, R. M., F. B. Tilley, P. A. Datzman. 1979. A watershed's response to logging and roads: South Fork of Caspar Creek, California, 1967-1976. USDA, Forest Service, Pacific Southwest Research Station Research Paper SW-146. Berkeley, CA. 12 pp.

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22: 169-199.

Rotenberry, J. T. 1986. Habitat relationships of shrub steppe birds: even good models cannot predict the future. Pp. 217-221, *in*: J. Verner, M. J. Morrison, and C. J. Ralph, editors. Wildlife 2000: modeling habitat relationships of terrestrial vertebrates. University of Wisconsin Press, Madison, WI.

SAS Institute, Inc. 1990. SAS User's Guide, Release 6.03 edition. Cary, NC. 1028 pp.

Stebbins, R. C. 1953. Southern occurrence of the Olympic salamander, Rhyacotriton olympicus. Herpetologica 11(4): 238-239.

Stebbins, R. C. 1985. A field guide to western reptiles and amphibians. Houghton Mifflin Company, Boston, MA. 336 pp.

Stejneger, L. H. 1899. Description of a new genus and species of discoglossid toad from North America. Proceedings of the United States National Museum 21: 899-901.

Stevens, J. 1986. Applied multivariate statistics for the social sciences. L. Erlbaum Assoc. Hillsdale, NJ 515 pp.

Titus, K., J. A. Mosher, and B. K. Williams. 1984. Chance-corrected classification for use in discriminant analysis: ecological applications. American Midland Naturalist 111(1): 1-7.

Toft, C. A. 1991. Reply to Seaman and Jaeger: an appeal to common sense. *In*: points of view: a controversy in statistical ecology. Herpetologica 46(3): 357-361.

Toft, C. A., and P. J. Shea. 1983. Detecting community-wide patterns: estimating power strengthens statistical inference. American Naturalist 122(5): 618-625.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47(4): 893-901.

Vitt, L. J., J. P. Caldwell, H. M. Wilbur, and D. C. Smith. 1990. Amphibians as harbingers of decay. Bioscience 40(6): 418.

Wake, D. B. 1990. Declining amphibian populations. Science 253: 860.

Wake, D. B., and H. J. Morowitz. 1991. Declining amphibian populations-a global phenomenon? Findings and recommendations. Alytes 9(2): 33-42.

Waters, T. F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society Monograph 7. Bethesda, MD.

Welsh, H. H. Jr. 1987. Monitoring herpetofauna in woodland habitats of northwestern California and southwestern Oregon: a comprehensive strategy. Pp. 203-213, *in*: T. R. Plumb and N. H. Pillsbury, editors. Multiple-use Management of California's Hardwood Resources. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW-100. Berkeley, CA.

Welsh, H. H., Jr. 1990. Relictual amphibians and old growth forests. Conservation Biology 4: 309-319.

Welsh, H. H. Jr. 1993. A hierarchical analysis of the niche relationships of four amphibians from forested habitats of Northwestern California. Dissertation. University of California, Berkeley, CA.

Welsh, H.H., Jr. and A .J. Lind. 1991. The structure of the herpetofaunal assemblage of the Douglas-fir forests of Northwestern California and Southwestern Oregon. Pp. 394-413, *in*: L. F. Ruggiero, K. B. Aubry, A. B. Carey, and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. USDA, Forest Service, Pacific Northwest Station, General Technical Report PNW-GTR-285, Portland, OR.

Welsh, H.H., Jr. and L. M. Ollivier. 1992. Effects of sediments from the Redwood National Park Bypass Project (CALTRANS) on the amphibian communities in streams in Prairie Creek State Park. Final report to the California Department of Transportation. December 1992. 63 pp.

Welsh, H.H., Jr. and A. J. Lind. 1992. Population ecology of two relictual salamanders from the Klamath Mountains of northwestern California. Pp. 419-437, *in*: D. McCullough, and R. Barrett, editors. Wildlife 2001: Populations. Elsevier Science Publishers, Ltd. London, England.

Welsh, H. H., Jr. and A. J. Lind 1996. Habitat correlates of the southern torrent salamander, *Rhyacotriton variegatus* (Caudata: Rhyacotritonidae), in Northwestern California. Journal of Herpetology 30(3): 385-398.

Wiens, J. A. 1989. Spatial scale in ecology. Functional Ecology 1989(3): 385-397.

Wiens, J. A., J. F. Addicott, T. Case, and J. Diamond. 1986. Overview: the importance of spatial and temporal scale in ecological investigations. Pp. 145-153, *in*: J. Diamond and T. J. Case, editors, Community Ecology. Harper and Row, New York, NY.

Zar, J. H. 1998. Biostatistical analysis. Fourth edition. Prentice-Hall, Inc., Englewood Cliffs, NJ 663 pp + appendices.

Figure Captions

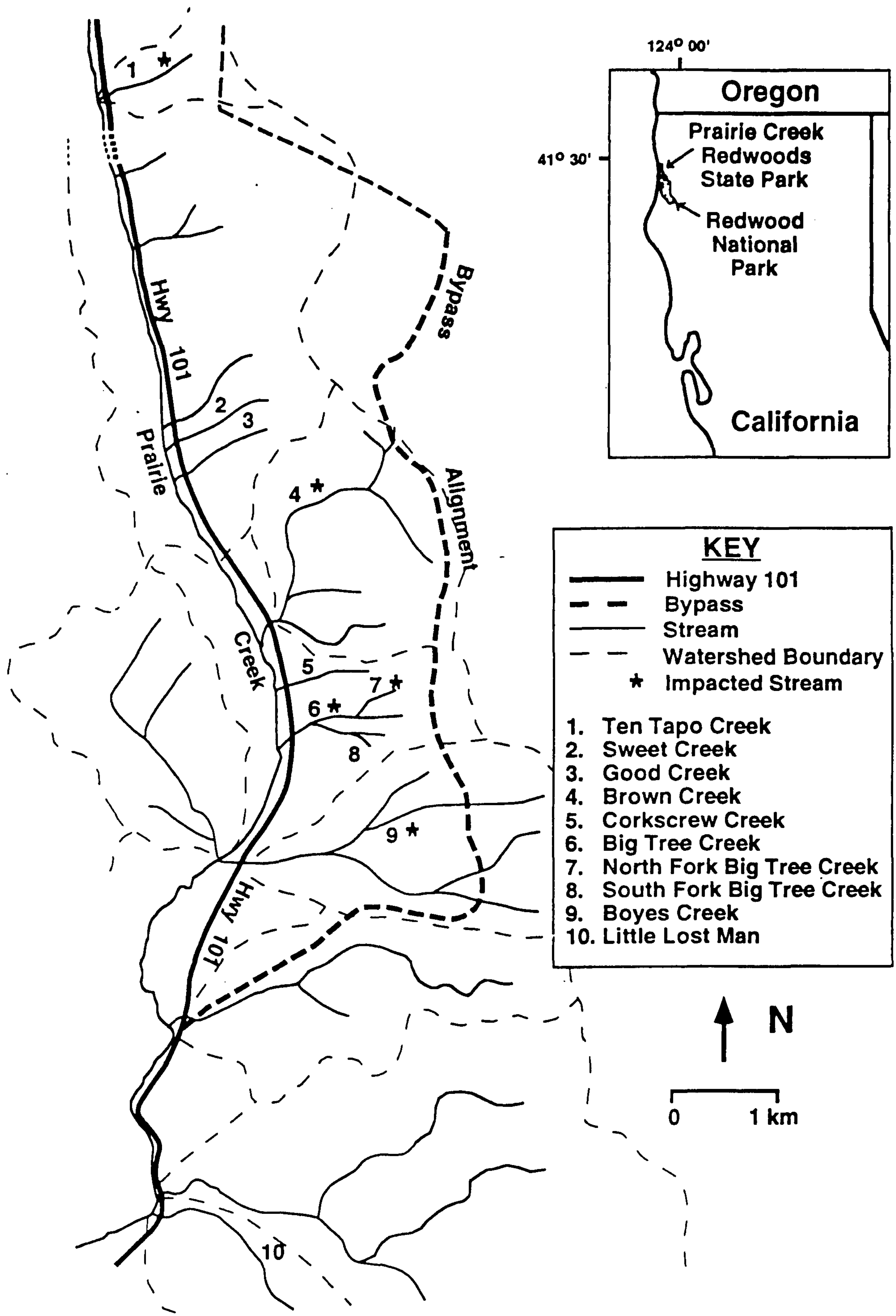
Figure 1. Relationships of impacted and unimpacted drainages in Prairie Creek State Park and Redwood National Park, Humboldt County, California. All drainages were sampled to determine amphibian population densities from June through August, 1990. Stars indicate impacted streams (see text for explanation).

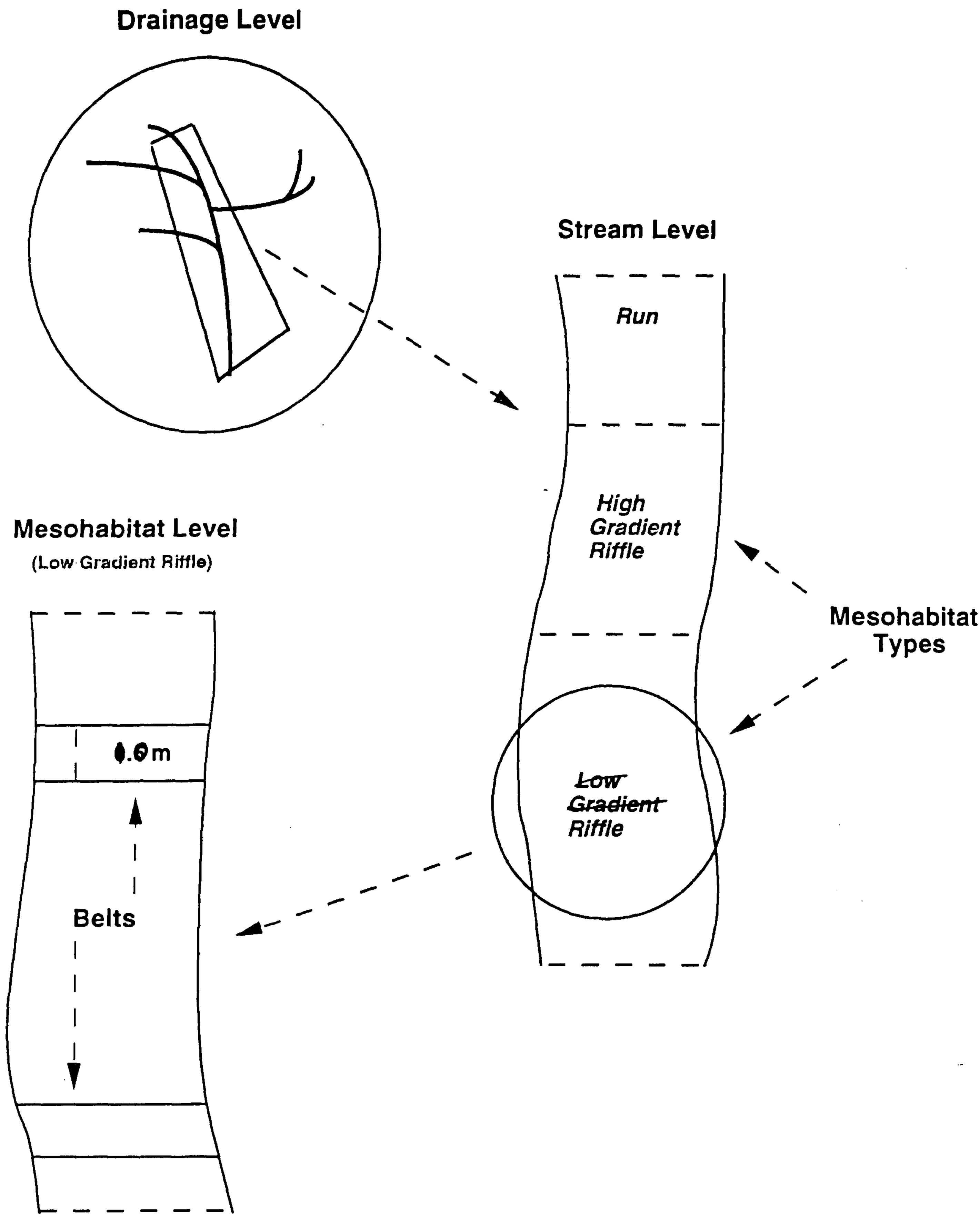
Figure 2. Schematic representation of random systematic sampling design based on habitat structure of streams. See text and Appendix I for details on variables measured at each sample site.

Figure 3. Schematic representation of random-systematic belt placement within selected mesohabitats. Modified from Welsh et al. (1997).

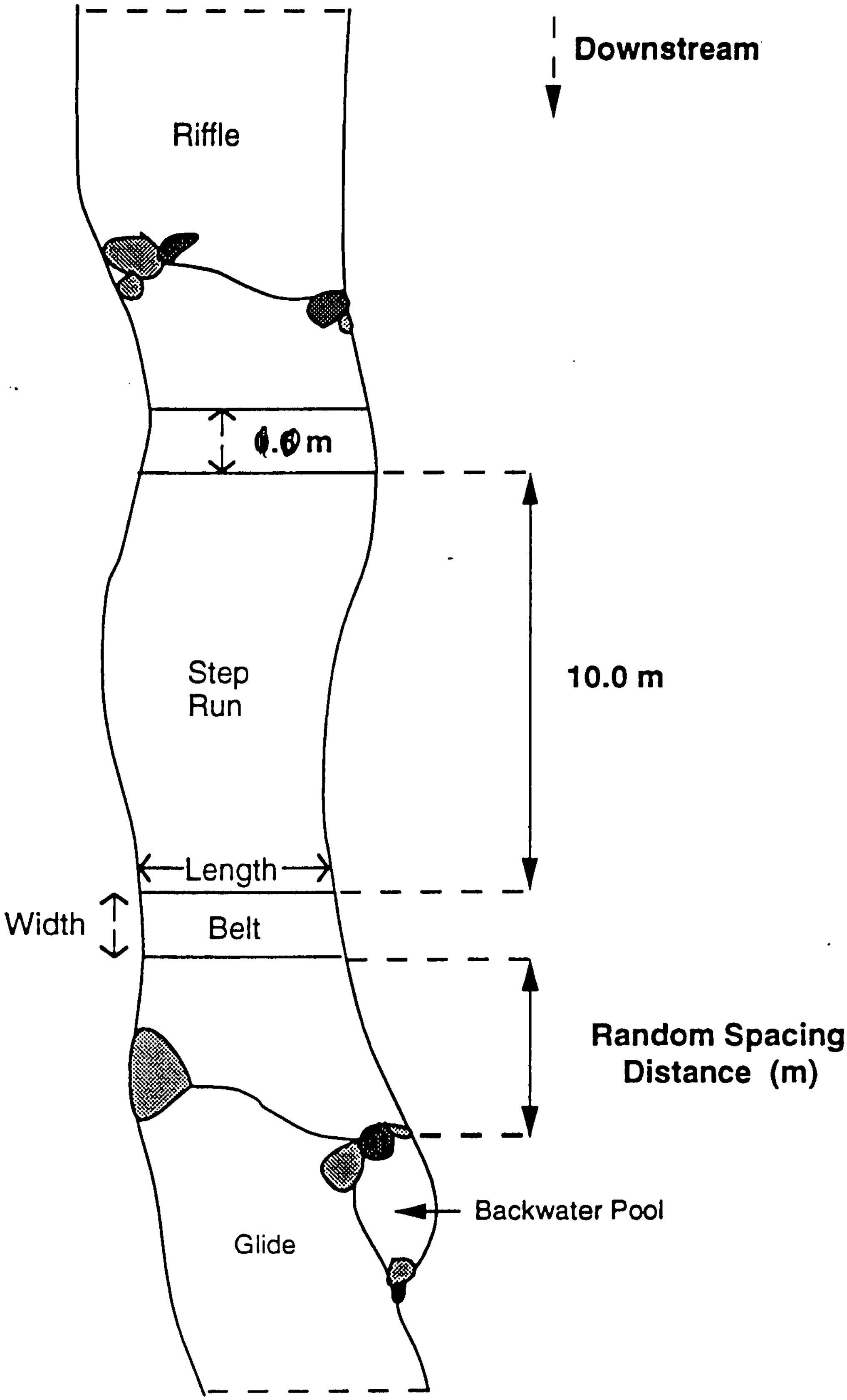
Figure 4. Schematic representation of a generalized pool habitat. The locations where sediment depths were measured (cm) are indicated as bottom, middle and top. The depths were measured on the center line of the pool bowl (Appendix I).

Figure 5. Comparison of substrate embeddedness (visual estimate) from 1990 and 1996 sampling periods. Mean percent embedded, standard error and student's t result are reported. All data are from Prairie Creek Redwoods State Park and Redwood National Park.

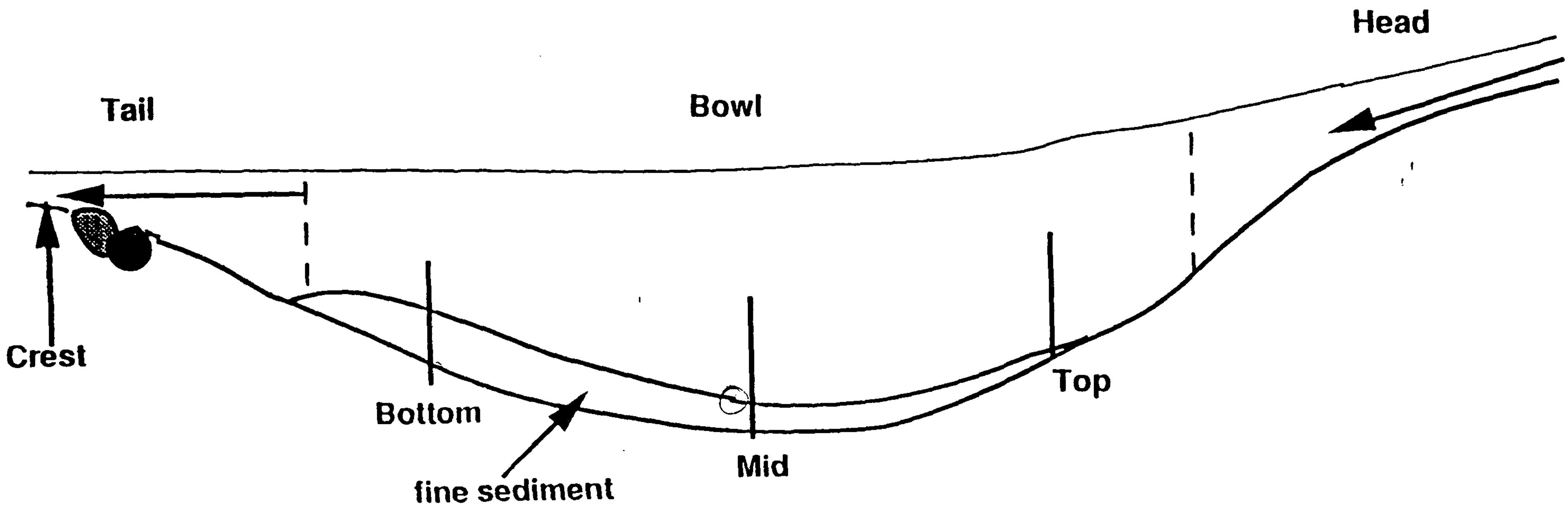

124° 00'
Oregon
Prairie Creek Redwoods State Park
41° 30'
Redwood National Park
California
Hwy 101
Bypass
Prairie
Creek
Alignment
1
2
3
4
5
6
7
8
9
10
KEY
Highway 101
Bypass
Stream
Watershed Boundary
Impacted Stream
1. Ten Tapo Creek
2. Sweet Creek
3. Good Creek
4. Brown Creek
5. Corkscrew Creek
6. Big Tree Creek
7. North Fork Big Tree Creek
8. South Fork Big Tree Creek
9. Boyes Creek
10. Little Lost Man
N
0
1 km



Drainage Level
Stream Level
Run
High Gradient Riffle
~~Low~~ ~~Gradient~~ Riffle
Mesohabitat Types
Mesohabitat Level
(Low Gradient Riffle)
Belts



Downstream
Riffle
Step
Run
10.0 m
Length
Width
Belt
Random Spacing
Distance (m)
Backwater Pool
Glide


Head
Tail
Bowl
Crest
Bottom
Top
Mid
fine sediment

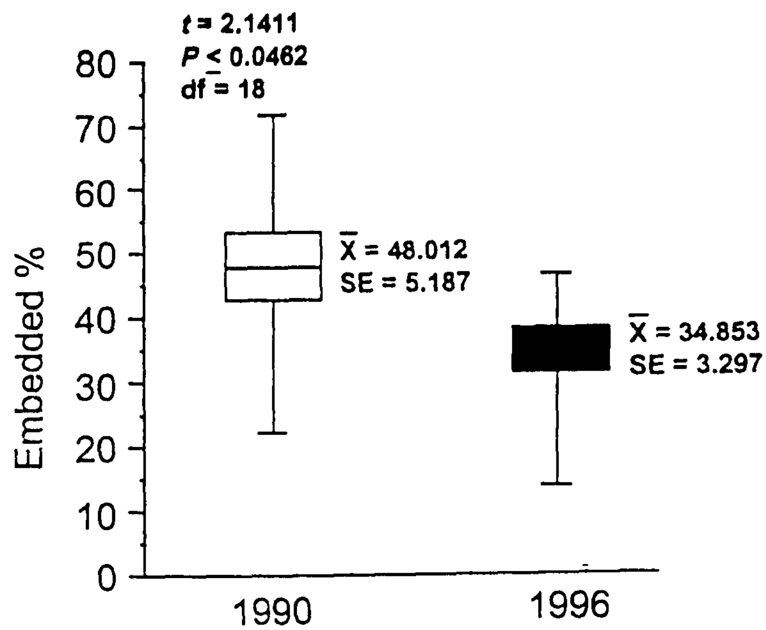

t = 2.1411
P < 0.0462
df = 18
80
70
60
50
40
30
20
10
0
Embedded %
X̄ = 48.012
SE = 5.187
X̄ = 34.853
SE = 3.297
1990
1996

FIG. 5

Appendix I. Definitions of primary mesohabitat types, pool sediment measures, and microhabitat attributes measured or estimated in association with belt samples.

Measure	Definition
a) Mesohabitat attributes:	
1) Primary mesohabitat types[1]:	
All pools	Reaches with water depths from shallow to deep with evidence of scour. Cause of scour may be an obstruction, blockage, merging of flows, or constriction. This type includes main channel, lateral, backwater, and secondary channel pools. Flow velocities range from very low to swift. Substrate size is highly variable.
Run/Glide	Wide shallow reaches flowing smoothly, with little surface agitation and no major flow obstructions. Velocities are low to moderate. These often appear as flooded riffles. Typical substrates are gravel, cobble, and boulders.
Riffle	Shallow to moderately deep, swift, turbulent water. Amount of exposed substrate will vary. Substrates are usually cobble or boulder dominated.
Step Run	A sequence of runs separated by short riffle steps. Substrates are usually cobble and boulder dominated.
Step Pools	A sequence of pools separated by short riffle steps. Substrates are usually cobble and boulder dominated.
Seep	A small spring or habitat where shallow water flows over and through gravel and larger substrates. Habitat unit may or may not be entirely contained within bank full channel.
2) Pool sediment measures:	
Pool tail embedded	Visual estimate (%) of vertical surfaces of large substrates buried in fines and/or sand in pool tail.
Pool bowl sediment depth	Depth of sediment to the nearest tenth of a cm is taken at three points along the midline of the pool bowl. These measures are then averaged.

[1] Modified from Hawkins et al. (1993).
[2] Variable is transformed using arcsine to meet assumptions of normality.

Appendix I. (cont'd)

Measure	Definition
b) Microhabitat attributes:	Measures and estimates of microhabitat attributes taken in association with amphibian sampling.
1) Aquatic conditions:	
Proportion margin[2]	Visual estimate (%) of channel composed of margin flow (%).
Proportion intermediate	Visual estimate (%) of channel composed of intermediate flow.
Proportion thalweg	Visual estimate (%) of channel flow composed of thalweg flow.
Flow margin	Flow rate in channel margin measured with a flowmeter in cm/sec.
Flow intermediate	Flow rate in intermediate channel flow measured with a flowmeter in cm/sec.
Flow thalweg	Flow rate in channel thalweg measured with a flowmeter in cm/sec.
Water depth	Average and maximum depth in cm.
Stream width	Width of wetted channel at belt location (m).
Belt area[3]	Area searched for amphibians. Includes only wetted channel.
Canopy open[2]	Measured by densiometer at center of the belt (%).
Water temperature	Measured by thermometer (C°).
Density of other amphibians[3]	Density (captures/m^2) of the two other species of amphibians present in the belt.
2) Cover estimates:	Visual estimate of instream cover (%) in a series of categories.
Undercut banks[2]	Overhang of stream banks, within 30 cm of water surface.
Woody debris[2]	Woody debris of any size, including leaf litter overhanging water surface or underwater.
Riparian vegetation[2]	Vegetation growing on the banks or in the stream. Must overhang within 30 cm of the water surface.

[3] Variable is transformed using natural log to meet assumptions of normality.

Appendix I. (cont'd)

Measure	Definition
2) Cover estimates (cont'd):	
Large rock[2]	Comprised of boulders and bedrock ledges. Only those portions that provide an overhang capable of hiding an amphibian are counted in this estimate.
Without cover[2]	Portion of the belt lacking any of the above cover types.
3) Coarse aquatic substrates[4]:	Visual estimate of belt surface area comprised of coarse substrates (%) in the following categories:
Gravel	2.0-32.0 mm in diameter;
Pebble	32.0-64.0 mm in diameter;
Cobble	64.0-256.0 mm in diameter;
Large rock	> 256.0 mm in diameter and bedrock.
Woody debris[2]	Woody debris of any size and leaf litter. Must be in or surrounded by water.
Fine gravel proportion	Proportion of weight of sediment sample taken at each belt (2.0 - 16.0 mm diameter).
Coarse gravel proportion	Proportion of weight of sediment sample taken at each belt (16.0 - 32.0 mm diameter).
4) Fine aquatic substrates[4]:	
Embedded	Visual estimate (%) of vertical surfaces of large substrates buried in fines and/or sand in the belt.
Fines[2]	Visual estimate (%) of belt surface area comprised of substrates <0.06 mm diameter.
Sand[2]	Visual estimate (%) of belt surface area comprised of substrates 0.06 - 2.0 mm diam.
Silt volume[2]	Proportion of weight of sediment sample taken at each belt (samples are dried before sifting and weighing; <0.063 mm diam).

[4] Particle size based on Platts et al. 1983.

Appendix I. (cont'd)

Measure	Definition
4) Fine aquatic substrates (cont'd):	
Sand volume[2]	Proportion of weight of sediment sample taken at each belt (0.063 - 2.0 mm diameter.).
Nonfilamentous algae	Visual estimate (%) of belt substrates covered by nonfilamentous algae growth.

Appendix II. Reach and channel types present on 10 streams sampled for aquatic amphibians in Prairie Creek Redwoods State Park and Redwood National Park in summer 1996. Reach and channel types are presented as percent of stream length. Means and standard errors (in parentheses) are reported. Logs per kilometer are counts irrespective of species or size. Comparisons of impacted and unimpacted stream groups using student's t are shown below.

Stream	Km Surveyed	Mesohabitat units/km	Logs/km	Reach Types: Alluvial	Braided	Confined	Channel Types[1]: A2	B2	B3
Unimpacted									
Corkscrew	0.336	137.03	715.91	0	44.09	55.84	0	33.30	66.64
Good	1.041	122.06	301.27	0	69.14	30.91	0	100.00	0
Little Lost Man	1.342	89.41	445.68	8.37	78.13	13.52	0	100.00	0
S. Fk. Big Tree	0.083	144.06	108.43	0	0	100.00	100.00	0	0
Sweet	1.277	79.85	194.99	50.95	23.43	25.59	0	100.00	0
$\bar{X}$	0.816	114.48	353.26	11.86	42.96	45.17		66.66	
	(0.256)	(12.78)	(106.71)	(9.91)	(14.41)	(15.34)		(21.08)	
Impacted									
Big Tree	1.318	132.83	273.55	6.86	58.76	34.40	45.75	54.27	0
Boyes	0.559	89.45	78.71	42.80	57.21	0	0	100.00	0
Brown	1.412	82.16	250.17	40.29	19.57	40.14	0	100.00	0
N. Fk. Big Tree	0.799	91.39	544.21	0	10.46	89.48	0	100.00	0
Ten Tapo	0.476	71.41	316.06	84.08	15.85	0	0	100.00	0
$\bar{X}$	0.913	93.44	292.54	34.81	32.37	32.80		90.85	
	(0.193)	(10.45)	(74.76)	(15.02)	(10.56)	(16.46)		(9.15)	
t	0.303	-0.466	1.275	-0.593	-0.550	-0.493		-1.000	
P	0.770	0.654	0.238	0.570	0.598	0.635		0.347	
df	8	8	8	8	8	8		8	
power	0.058	0.070	0.203	0.082	0.078	0.072		0.143	

[1] Channel types as described by Rosgen (1994). The A2 and B3 channel types did not occur often enough to permit comparison using t.