Project Completion Report

Responses of fishes and salamanders to instream restoration efforts
in western Oregon and Washington

by Phil Roni

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Funded by
Bureau of Land Management
Oregon State Office
1515 S.W. 5th St.
Portland, OR 97208
(Interagency Agreement No. 1422H952-A98-3007)

and

Environmental Conservation Division
Northwest Fisheries Science Center
National Marine Fisheries Service
2725 Montlake Blvd. E.
Seattle, WA 98112

January 2001

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EXECUTIVE SUMMARY

Placement of large woody debris (LWD) into the active channel is one of the most common techniques for restoring and enhancing streams in the Pacific Northwest. However, the effectiveness of this technique at increasing fish and salamander abundance has not been consistently demonstrated. Thirty streams in western Washington and northwest Oregon were sampled during summer and winter to determine the responses of juvenile salmonids, juvenile lamprey (*Entosphenus tridentatus* and *Lampetra spp.*), sculpin (*Cottus spp.*) and giant salamanders (*Dicamptodon* spp.) to artificial LWD placement and to examine their habitat preferences. In addition, to examine the effects of habitat modification on fish movements, I monitored the monthly movements of marked juvenile coho salmon (*Oncorhynchus kisutch*), steelhead (*O. mykiss*) and cutthroat trout (*O. clarki*) between a reach that had been "restored" (with placed wood) and a reference reach (no wood placement) in Shuwah Creek, Washington from September 1998 to April, 1999. I also examined the size, growth and movements of individually marked coho salmon among habitats in two artificial channels: one with and one without woody debris.

Total pool area, pool number, LWD loading, and LWD forming pools were significantly greater in treatment (LWD placement) than paired reference reaches nearby during both summer and winter. Juvenile coho salmon densities were 1.8 and 3.2 times higher in treated reaches compared to reference reaches during summer and winter, respectively. The response ($\log_{10}$(treatment/reference)) of coho density to artificial LWD placement was correlated with the number of pieces of LWD forming pools during summer and total pool area during winter months. Densities of age 1+ cutthroat trout (*O.*

clarki) and steelhead (*O. mykiss*) did not differ between treatment and reference reaches during summer but were 1.7 times higher in treatment reaches during winter. Age 1+ steelhead density response to treatment was negatively correlated with increases in pool area during summer, but not winter. Trout fry (age 0+ cutthroat and steelhead) densities did not differ between reaches, but trout fry response to treatment was negatively correlated with pool area during winter. This research indicates that artificial LWD placement can lead to higher densities of juvenile coho during summer and winter and cutthroat and steelhead during winter.

In contrast to salmonids, no significant difference was detected between densities or mean lengths in treatment and reference reaches for giant salamanders, reticulate (*C. perplexus*) or torrent sculpin (*C. rhotheus*), or larval lamprey. However, lamprey response to LWD placement (treatment-reference) was positively correlated with LWD forming pools (treatment-reference). Difference (treatment – reference) in length of age 1+ reticulate sculpin was positively correlated with difference in LWD within the wetted channel. Species richness and dominance, two community diversity measures, did not differ between treatment and reference reaches. These results indicate that artificial LWD placement may benefit age 1+ reticulate sculpin and Pacific lamprey, two species known to prefer pools, but have little effect on other torrent sculpin or giant salamanders.

Habitat use patterns for each species were examined at the scale of both individual habitat units and reaches. In the summer, densities of coho salmon, cutthroat trout, and larval lamprey were significantly higher in pools than riffles, whereas densities of age 0 torrent sculpin (*C. rhotheus*) were higher in riffles than pools. In winter, densities of coho salmon, cutthroat and steelhead trout, and young of the year trout fry

were higher in pools than riffles. Cutthroat, steelhead, Pacific giant salamanders, and torrent sculpin found in pools were larger than those found in riffles. Multiple regression analysis indicated that physical variables (e.g., pool depth, cover, large woody debris, etc.) explained 10% or less of the variation in densities among pools. Reach-scale physical variables (e.g., elevation, drainage area, precipitation, stream gradient, percent pool area) explained from 22% to 63% of the variation of species density among streams. This suggests that reach-scale physical variables may be better predictors of fish densities among streams than variables measured within individual habitat units.

Monthly surveys in Shuwah Creek to examine salmonid movement indicated that 0 to 33% (0 to 4 fish) of the marked trout or coho salmon observed on a given date moved between the restored and reference reach. However, the rapid decline in both marked and unmarked fish in late fall and the increasing proportion of unmarked fish over the course of the study indicated considerable migration to and from the study reaches. In the artificial channels, fewer fish moved in the simple (no wood) than the complex (with wood) channels (22% versus 37%, respectively), and the mean distance moved was shorter in the complex than the simple channel (4.4 versus 6.7 habitat units). In the simple channel, the fish that moved exceeded those that did not move in length, weight, and growth rate. We conclude that movement may facilitate increased growth in stream reaches with little woody debris and that the placement of woody debris may lead to more frequent and shorter movements. Movements of juvenile salmonids among stream reaches and individual habitats are common and need to be considered when evaluating restoration projects.

CHAPTER 1: OVERALL INTRODUCTION

The listing of many Pacific Northwest salmon populations as threatened or endangered under the Endangered Species Act has lead to large efforts to restore salmon habitat. Watershed and instream restoration are considered key components of most salmon recovery efforts including the Snake River Recovery Plan, the Northwest Forest Plan, and the Oregon Salmon Plan (USFS and BLM 1994, NMFS 1995). Restoration[1] and enhancement of stream fish habitats has been occurring for more than 50 years in North America (Meehan 1991, Reeves et al. 1991a). The placement of wood and boulders in streams in an effort to restore or enhance fish habitat and increase fish numbers has been occurring in North America since at least the 1930s (Tarzwell 1934; Meehan 1991; Reeves et al. 1991a). Interest in habitat restoration in the Pacific Northwest increased dramatically in the 1980s, when the importance of large woody debris (LWD) in creating and providing fish habitat became widely accepted (Sedell et al. 1984, Murphy et al. 1986, Bisson et al. 1987). Prior to this time, the removal of woody debris from streams (stream cleaning) was a common practice on commercial forest lands (Bilby 1984, House and Boehne 1987). Instream LWD placement has become one of the most common techniques to improve fish habitat in an effort to compensate for the reductions in LWD following several decades of land use practices (Kauffman et al. 1997). Federal, state, county, and municipal governments spend millions of dollars on watershed and instream restoration in the Pacific Northwest. Unfortunately, little

[1] Gore (1985) and Koski (1992) defined restoration as returning a stream to some undisturbed state. However, the terms habitat restoration and enhancement have been used interchangeably throughout the literature and I will use the term restoration to mean habitat enhancement or improvement.

research and monitoring has occurred to determine the effectiveness of these and other restoration efforts (Reeves et al. 1991a, Kondolf 1995, Kauffman et al. 1997). Much of the federal and state funds for stream restoration are distributed to regional enhancement groups to encourage restoration at the local level. Few funds are set aside for monitoring and evaluation of these efforts. Furthermore, these local citizen groups often lack the scientific expertise to design and evaluate the effectiveness of their restoration efforts. This stems in part from limited information on the effectiveness of stream restoration techniques to increase fish production and the need for large-scale comprehensive evaluation of habitat restoration efforts (Reeves et al. 1991a, Beschta et al. 1994; Chapman 1996, Kauffman et al. 1997).

Monitoring and evaluation of stream restoration projects in the Pacific Northwest generally focus on changes in physical habitat. Several authors have documented increases in pool frequency, pool depth, and other physical features in response to instream restoration (House et al. 1991, Crispin et al. 1993, Reeves et al. 1997, Thom 1997, and others). Ehlers (1956) and Frissell and Nawa (1992) documented high rates of physical failure of instream structures. However, more recently Roper et al. (1998) indicated that most U.S. Forest Service instream restoration projects had relatively low rates of physical failure. Biological evaluations have also produced variable results. House and Boehne (1986) and House (1996) reported significant increases in summer coho salmon (*Oncorhynchus kisutch*) densities following instream restoration. Cederholm et al. (1997) reported higher winter densities of coho salmon following instream LWD placement, but no significant differences in summer densities. Conversely, Reeves et al. (1997), Beschta et al. 1994, and Chapman et al. (1996) reported

no significant changes in coho salmon or other salmonid densities following instream restoration in several Columbia River tributaries. These and other restoration projects in the Pacific Northwest are aimed primarily at improving freshwater rearing habitat for one or two salmonid species with little consideration for other salmonids or resident stream fishes (Reeves et al. 1991b, Murphy 1995). Focusing exclusively on one or two anadromous species has produced inconsistent results, as a variety of factors during the marine and freshwater life history stages can influence abundance and survival. Bisson et al. (1992) reported that trends in abundance of individual species are often of limited use in assessing the impacts of habitat alteration within a watershed. They further suggested that changes in the composition of stream fish communities may provide more comprehensive evidence of the extent of environmental degradation and subsequent recovery. Platts and Rinne (1985) also emphasized the need for a better understanding of how enhancement projects affect fish communities.

The effects of habitat alteration and degradation on stream fish communities has been assessed successfully in other parts of North America, particularly the Midwest and southwestern United States (see Gorman and Karr 1978, Schlosser 1982, Meffe 1984, Fausch and Bramblett 1991, and others). Karr (1981) emphasized the importance of using fish communities as indicators of stream health. However, the fish fauna in the Midwest is much more diverse than in Pacific Northwest streams. Most coastal streams in Oregon and Washington are inhabited by 3-5 species of salmonids, 2-3 species of sculpin (*Cottus spp*.), 2-3 species of lamprey (*Lampetra* and *Entosphenus spp*.), 1-2 species of dace (*Rhinichthys* spp.) and possibly 1-3 species of fish of the families Catostomidae, Cyprinidae and Gasterosteidae (McPhail and Lindsey 1986). Little is

known about the habitat requirements and competitive interactions of most nonsalmonid fishes. The relatively depauperate fish fauna Pacific Northwest streams (McPhail and Lindsey 1986) and the limited knowledge of nonsalmonid fishes and their habitat requirements has inhibited application of a community level approach in determining the effects of habitat modification on fishes (Bisson et al. 1992). However, to completely evaluate the success of habitat restoration and enhancement efforts it is necessary to examine the response of different species, life stages, and interactions among species as well as species diversity and richness.

Response of juvenile salmonids and other fishes to habitat disturbance or enhancement have assumed that juvenile salmonids have small home ranges and do not move between reference and treatment streams or reaches (Gowan et al. 1994). However, this has recently been demonstrated to be an inappropriate assumption (Riley et al. 1992; Gowan et al. 1994). Moreover, large-scale movements of juvenile anadromous salmonids are common during spring and fall (Peterson 1982; Tchaplinski and Hartman 1983; Northcote 1992). Riley et al. (1992), Riley and Fausch (1995), and Gowan and Fausch (1996) reported that much of the increase in resident trout numbers in response to wood placement in Colorado streams was due to immigration. Thus quantifying movement of juvenile fishes among natural and artificial habitats is an important factor to consider when evaluating habitat improvement projects.

The following chapters outline the results of a large-scale research project designed to evaluate artificial LWD placement, one of the most common restoration techniques in Pacific Northwest streams. These studies were designed to answer four basic questions about LWD placement including: 1) the response of juvenile salmonids to

LWD placement; 2) the fish community response to artificial LWD placement; 3) the habitat-use patterns of fish and salamanders in small streams; and 4) the movements of juvenile salmonids among artificial and natural stream habitats.

CHAPTER 2: DENSITY AND SIZE OF JUVENILE SALMONIDS IN RESPONSE TO PLACEMENT OF LARGE WOODY DEBRIS IN WESTERN OREGON AND WASHINGTON STREAMS

Introduction

The factors controlling the populations of salmonid fishes are numerous and complex but it is widely believed that instream habitat plays a role in population density, as least for stream-rearing species (NRC 1996). In an effort to mitigate for degradation and loss of fish habitat from anthropogenic disturbance and stop or reverse the declines in salmonid populations in recent years, stream restoration projects have become common in the Pacific Northwest. The placement of boulders, logs and woody debris directly into the stream channel to create pools, provide cover and reduce gravel movement is an integral part of most of these stream restoration projects and recovery efforts for Pacific salmon (Rodgers et al. 1992). Unfortunately, the research and monitoring that has occurred has often been inadequate to determine the effectiveness of various stream restoration activities.

The restoration and enhancement of stream fish habitat has been occurring for more than 60 years in the United States (Ehlers 1956; Tarzwell 1934; Reeves et al. 1991a). Stream restoration techniques were originally pioneered in the Midwestern U.S. (Tarzwell 1934), but have been modified for use in the steeper, high-energy streams common in western North America (Reeves et al. 1991a). The interest in habitat restoration in the Pacific Northwest has increased dramatically since the early 1980s, when the importance of woody debris in maintaining and creating fish habitat became

widely accepted (Sedell et al. 1984, Bisson et al. 1987). Prior to this time, "stream cleaning" (removal of wood from streams) was a common practice (Bilby 1984, House and Boehne 1987). In the last 20 years, many studies have emphasized the critical role that large woody debris (LWD) plays in creating and maintaining fish habitat in streams. Instream LWD can create pools, increase habitat complexity, reduce sediment transport, trap gravel needed for spawning, stabilize stream channels, provide food for aquatic invertebrates, and provide stream nutrients and increasing overall stream productivity (Bilby and Likens 1980; Lisle 1986; Bisson et al. 1987; Robison and Beschta 1990; Fausch and Northcote 1992). Consequently, instream LWD placement has become one of the most common techniques to improve fish habitat in an effort to compensate for the reductions in LWD caused by stream cleaning and various land use practices (Kauffman et al. 1997).

More than a decade ago, Reeves and Roelofs (1982) and Platts and Rinne (1985) identified the need for better evaluation of habitat improvement projects. More recently, Chapman (1996), and Kauffman et al. (1997) reiterated the need for comprehensive evaluation of instream restoration efforts. Monitoring of stream restoration projects in the western United States and Canada has focused primarily on the physical habitat responses and whether LWD structures functioned as designed rather than on biotic responses. Many authors have documented increases in pool frequency, pool depth, woody debris and sediment retention following placement of instream structures (e.g., Anderson et al. 1984; Armantrout 1991; House et al. 1991; Crispin et al. 1993; Cederholm et al. 1997; Reeves et al. 1997). However, the extent to which the structures

remain in place and functioning after several years is less clear (Ehlers 1976; Armantrout 1991; Frissell and Nawa 1992; Roper et al. 1998).

Biological evaluations of instream restoration have also produced highly variable results. There have been a number of encouraging reports of increased densities of salmonids following restoration efforts. House and Boehne (1985), House et al. (1989), and House (1996) reported increased juvenile coho densities in several Oregon coastal streams, with small increases in densities of juvenile cutthroat (*O. clarki*) and steelhead (*O. mykiss*) trout. Cederholm et al. (1997) reported significantly more coho salmon smolts following restoration in a Washington stream. On the other hand, numerous authors have reported no significant biological response or even decreases in salmonid abundance following restoration. Reeves et al. (1997) reported no significant difference in coho parr or smolt numbers following restoration. Beschta et al. (1994) and Chapman (1996) reviewed several case studies on restoration efforts in the Columbia Basin and western United States and found little evidence of increased fish numbers. The inconsistent results of these studies further emphasize the need for continued biological evaluation of instream restoration efforts and the need for a multispecies approach.

This research was developed to provide a broad-scale physical and biological evaluation of instream LWD placement efforts (restoration) in western Washington and Oregon. The overall objectives were to determine whether the artificial placement of LWD produces a significant change in physical habitat and juvenile salmonid abundance. Specifically, I tested the null hypotheses that paired treatment and reference reaches would not differ in (1) densities of woody debris and pool area, (2) densities of juvenile coho salmon and cutthroat and steelhead trout in summer and winter, (3) the magnitude

of fish response to treatment would not depend on the magnitude of change in habitat, and (4) the sizes of the fish would not differ between treatment and reference reaches.

Materials and Methods

Experimental Design

I used the extensive post-treatment design (Hall et al. 1978, Hicks et al. 1991) to determine the response of juvenile salmonids to artificially placed LWD. This design involves comparison between treatment and reference reaches at a large number of sites after restoration efforts, in contrast to pre- and post-treatment comparisons and comparisons between paired treatment and control streams. The extensive post-treatment design has frequently been used to assess the impacts of forestry and other land-use practices on salmonids and their habitats (e.g., Murphy and Hall 1981, Grant et al. 1986, Reeves et al. 1993) and is particularly well suited to reach-scale habitat restoration projects.

Study Sites

Thirty streams in western Washington and northwest Oregon (Figure 2.1) with paired treatment and reference (control) reaches were sampled during both summer and winter between August 1996 and April 1999. Treatment was defined as the artificial placement of LWD within the active stream channel. Paired treatment and reference reaches 75-120 m long were selected in each stream. The length of study reaches where at least 10 times bankfull width, and most reaches were 100 m long. Treatment and

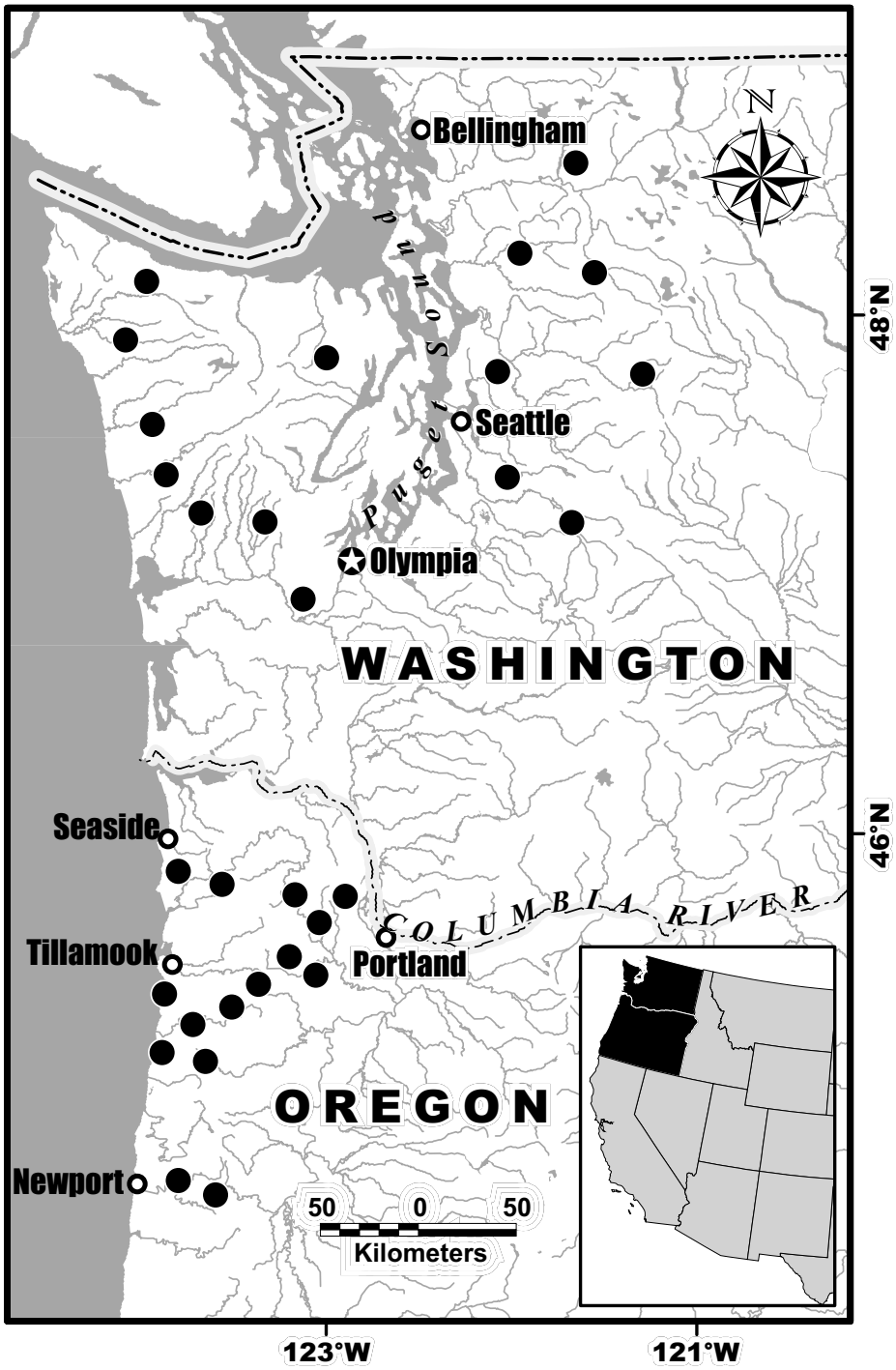


Figure 2.1. Map of 30 study streams in western Oregon and Washington, U.S.A.

reference reaches within a given stream were always the same length. Each stream was surveyed once in summer (August - September) and winter (January - March) and reaches within a stream were sampled on the same day. More than 100 LWD placement projects were examined in western Washington and Oregon, but only 30 had suitable treatment and reference reaches. The selection of study streams with paired treatment-reference reaches was based in part on physical and biological stream characteristics including stream size, bankfull width, channel type (Montgomery and Buffington 1997), gradient, and fish species composition. Only sites with treatment and reference reaches of similar gradient, confinement, and channel width were selected. Those sites that did not have suitable reference reaches generally had LWD placed throughout a uniform stream reach and the slope, bankfull width, channel type and riparian vegetation changed immediately upstream or downstream of the treatment reach. Reference reaches were generally located 200 m or more upstream from treatment reaches. Only projects in which the artificially placed large woody debris (LWD) remained in the channel after several high water events, usually after multiple winters were included.

The study streams ranged from 4 to 12 m in bankfull width and 0.5 to 4.0% slope (Table 2.1). The age of the restoration projects (date of last LWD placement to date of sampling) ranged from 1-10 years. Annual precipitation varied from and 107 to 315 cm. Dominant forest types were primarily Douglas fir (*Pseudtsuga menziesii*), Sitka spruce (*Picea sichensis*), and western hemlock (*Tsuga heterophylla*). Study sites were located in second and third growth forests on public and private forest lands. Land use within study watersheds was commercial forestry, with the exception of (Laughing Jacobs and

Table 2.1. Physical characteristics of study streams measured during summer. Channel type is based on Montgomery and Buffington (1997), pr= pool-riffle channel, fpr = forced pool-riffle, pb = planebed, sp = step-pool, and fsp = forced step-pool. Watershed area for a given stream was calculated as total area upstream of restoration site.

	Channel Type		No. of Pools		Percent Pool Area		Slope (percent)		Bankfull width (m)	
Stream	ref.	treat.	ref.	treat.	ref.	treat.	ref.	treat.	ref.	treat.
Oregon										
Bear	pr	fpr	3	5	0.29	0.79	1.2	1.5	10.0	10.2
Bergsvick	pr	fpr	4	8	0.81	0.76	1.0	0.9	10.0	8.5
Bewley	pr	fpr	5	7	0.55	0.62	0.5	1.1	6.7	7.0
Buster	pr	pr	5	4	0.89	0.79	0.8	0.5	7.6	8.0
Deer	fpr	fpr	9	8	0.86	0.90	0.4	1.2	4.6	4.5
Elliott	pr	fpr	5	7	0.55	0.55	1.4	2.4	10.4	11.6
Farmer	pr	pr	4	4	0.34	0.42	1.8	1.6	7.1	7.4
Kenusky	fpr	fpr	7	6	0.65	0.64	1.5	1.2	6.4	6.4
Killam	pr	pr	5	4	0.37	0.49	3.2	3.0	7.0	11.6
Klootchie	pr	fpr	5	8	0.36	0.64	2.2	1.9	9.3	8.4
Lobster (S.F.)	pr	fpr	4	8	0.44	0.65	1.8	1.7	9.3	10.5
Louisignot	pr	fpr	6	8	0.78	0.85	0.8	0.6	10.1	9.1
Ltl. Nestucca (S.F.)	pb	fpr	4	8	0.25	0.77	0.9	1.6	9.3	9.9
Rock (N.F.)	pb	fpr	3	6	0.25	0.52	1.3	0.7	9.8	10.0
Tobe	pr	fpr	4	6	0.38	0.51	2.5	2.8	5.9	5.8
Washington										
Beaver	fpr	fpr	8	7	0.72	0.74	1.8	2.3	5.8	5.1
Benson	pr	fpr	6	4	0.35	0.31	1.8	1.9	12.3	11.0
Burn	sp/pr	fpr/sp	7	6	0.55	0.69	2.2	2.0	6.3	6.4
French	pr	fpr	2	6	0.18	0.25	2.3	2.2	16.4	16.6
Harris	pr	fpr	5	6	0.29	0.66	1.1	1.0	7.3	7.0
Hoppers	pr	fpr	5	10	0.78	0.96	0.8	0.7	4.3	4.1
Hyas	pb	fpr	1	4	0.36	0.59	1.3	0.7	11.2	13.2
Laughing Jacobs	pr/sp	fpr	5	7	0.19	0.38	2.5	2.3	7.2	6.3
Midnight	sp	fsp	4	6	0.31	0.34	3.9	4.5	5.4	5.8
Newbury	fpr	fpr	6	6	0.45	0.46	1.8	1.9	5.6	6.0
Porter	pr	fsp	3	4	0.56	0.67	1.3	2.3	9.9	10.1
Punch	sp/pr	fsp	11	12	0.53	0.44	3.6	3.2	7.6	9.4
Shuwah	fpr	fpr	7	6	0.56	0.80	1.4	1.9	6.5	6.5
Soosette	pb	fpr	2	5	0.19	0.29	1.7	1.7	8.7	13.5
Townsend	sp	fsp	5	11	0.33	0.60	3.9	3.1	4.3	4.3

Soosette) which were in second growth forest downstream from suburban areas. The dominant watershed geology was volcanic, sedimentary or glacial-alluvial and varied by site, but was consistent for reaches within a stream (Table 2.2). The elevations of the study sites ranged from 40 to 789 m and watershed area upstream of my study reaches ranged from 124 to 2388 ha (Table 2.2).

Habitat within each stream reach was classified using a modification of the methods and habitat types described by Bisson et al. (1982). We used minimum residual depth and pool area to consistently identify pool habitats (Lisle 1987; Schuette-Hames et al. 1994). Total surface area of each habitat was estimated by measuring the total length and multiplying by the average of 3-5 width measurements. The gradient of each reach and individual habitat unit was measured using a hand level, survey (stadia) rod, and tape measure. Habitat-specific stream slope (gradient) was used to distinguish between riffles and cascades. Discharge was estimated with a flow meter prior to completion of each survey.

All natural and artificially placed LWD within the active channel greater 10 cm in diameter and 1.5 m long was inventoried. The diameter class (small 10-20 cm, medium 20-50 cm, and large > 50 cm) and approximate length were recorded. The function of an individual piece of LWD based on its influence on pool formation and channel scour was classified into one of three categories including: 1) dominant - primary factor contributing to pool formation, (2) secondary - influences zone of channel scour but not responsible for pool formation, (3) negligible - may provide cover but not involved in scour (Montgomery et al. 1995).

Table 2.2. Geology, precipitation and vegetation zone of watersheds sampled. Watershed area for a given stream was calculated as total area upstream of restoration site. Geology and elevation from United States Geologic Survey data, vegetation and precipitation from Daly et al. (1994). Age represents the number of years after large woody debris placement that habitat, fish and large woody debris surveys were conducted.

Stream Name	Age	Elev. (m)	Area (ha)	Precip. (cm/yr)	Dominant Geology	Vegetation zone (dominant)
Oregon sites						
Bear Cr.	4	244	1,580	320	Volcanics	Douglas fir-western hemlock
Bergsvik Cr.	3	122	540	308	Sedimentary	Sitka spruce-western hemlock
Bewley Cr.	3	12	639	235	Sedimentary	Sitka spruce-western hemlock
Buster Cr.	2	232	1,627	228	Sedimentary	Douglas fir-western hemlock
Deer Cr.	3	219	414	169	Sedimentary	Douglas fir-western hemlock
Elliott Cr.	1	427	720	236	Volcanics	Douglas fir-western hemlock
Farmer Cr.	3	73	727	260	Sedimentary	Sitka spruce-western hemlock
Kenusky Cr.	3	207	1,158	167	Volcanics	Douglas fir-western hemlock
Killam Cr.	3	110	863	298	Volcanics	Sitka spruce-western hemlock
Klootchie Cr.	3	61	1,011	299	Sedimentary	Sitka spruce-western hemlock
Lobster Cr.	11	207	1,254	233	Sedimentary	Douglas fir-western hemlock
Lousignont Cr.	2	244	1,715	201	Volcanics	Douglas fir-western hemlock
Ltl. Nestucca R. (SF)	3	122	981	250	Sedimentary	Sitka spruce-western hemlock
Rock Cr. (NF)	3	390	1,893	286	Volcanics	Douglas fir-western hemlock
Tobe Cr.	4	165	680	236	Volcanics	Douglas fir-western hemlock
Washington Sites						
Beaver Cr.	3	233	124	189	Glacial-Fluvial	Western Hemlock
Benson Cr.	7	320	459	217	Sedimentary	Western Hemlock
Burn Cr.	5	481	733	227	Sedimentary	Mountain Hemlock
French Cr.	6	172	1,783	213	Igneous	Western Hemlock
Harris Cr.	12	292	311	354	Volcanics	Silver Fir
Hoppers Cr.	1	73	467	269	Volcanics	Western Hemlock
Hyas Cr.	6	121	2,000	290	Sedimentary	Western Hemlock
Laughing Jacobs Cr.	2	23	335	119	Glacial-Fluvial	Douglas-Fir
Midnight Cr.	4	598	567	212	Igneous	Western Hemlock
Newbury Cr.	12	170	302	317	Volcanics	Western Hemlock
Porter Cr.	5	122	2,388	170	Volcanics	Western Hemlock
Punch Cr.	12	110	271	353	Sedimentary	Western Hemlock
Shuwah Cr.	1	197	305	297	Sedimentary	Western Hemlock
Soosette Cr.	3	45	1,225	108	Glacial-Fluvial	Douglas-Fir
Townsend Cr.	2	789	809	199	Volcanics	Douglas-Fir

Multiple-removal electrofishing was used in summer to estimate fish abundance within each individual habitat (Carle and Strub 1978). Each habitat was sampled separately by placing 3.2 mm mesh blocknets at the upstream and downstream boundaries of each habitat unit to prevent immigration or emigration during sampling. Three removals were made through each habitat and a fourth was made if a 50% or greater reduction in fish numbers was not seen between the second and third passes. Each electrofishing removal consisted of one upstream and one downstream pass using a pulsed DC electrofisher. All fish captured were anesthetized with tricaine methanesulfonate (MS-222), identified, measured to the nearest mm and then released. Based on length frequency distributions (Jearld 1983), all steelhead or cutthroat trout greater than 60 mm during summer or 80 mm during winter were considered age 1+. All trout less than these lengths were considered fry (age 0 or young-of-year). Such fry could not be reliably identified to species and so were simply called trout.

High flows during winter months often precluded the use of multiple-removal electrofishing so night snorkel surveys were used to estimate juvenile salmonid abundance during winter. Roni and Fayram (2000) demonstrated that winter night snorkeling was nearly as accurate as multiple-removal electrofishing and suitable for a wider range of conditions. Juvenile salmonids emerge from concealment 30-60 min. after sunset at temperatures below 8-9 °C (Campbell and Neuner 1985; Fraser et al. 1993; Contor and Griffith 1995). Therefore, snorkeling began at least 1 h after sunset, and only on nights with either complete cloud cover or no visible moonlight to assure that natural light levels were consistently low during night snorkel surveys. Surveys were conducted

in 1997 during February and March in Washington and January to mid-March of 1998 in Oregon to assure that sampling occurred prior to the outmigration of salmonid smolts.

One diver entered the habitat from the downstream end and slowly moved upstream, stopping occasionally to relay the number, sizes, and species of fish observed to a second individual on the bank. In streams greater than 10 m wide, two snorkelers worked side by side to cover the entire width of the stream. A halogen dive light was used to illuminate areas and identify fish. Fish length was visually estimated to the nearest 10 mm using a ruler attached to a dive glove. Water temperature and flow were measured downstream of each site before electrofishing and snorkeling. Discharge and temperature among streams ranged from 0.1 to 0.15 $m^3 \cdot s^{-1}$ and 8 – 16 °C during electrofishing surveys, and 0.14 and 2.1 $m^3 \cdot s^{-1}$ and 8 – 16 °C during snorkel surveys (Table 2.3).

Differences in habitat, LWD and fish abundance between treatment and reference reaches were compared using paired t-tests. A Bonferroni correction was applied to compensate for the increased likelihood of finding a significant difference when performing multiple t-tests (SPSS 1999). A family alpha level of 0.05 was used to determine significance and divided by the number of tests performed on each separate data set (fish, habitat, and LWD). This resulted in a significance level of 0.01 for each individual t-test. A log (x+1) transformation was used on fish data to meet basic assumptions of a t-test (additive data, normal distribution, equal variances) and account for any zero or low counts (Zar 1999). Multiple regression was used to examine the relationship(s) between fish response (ratio of treatment density/reference density) and

Table 2.3. Stream flow and temperature during summer electrofishing and winter snorkel surveys.

	Summer		Winter	
Stream	Flow (m^3/s)	Temp.°C	Flow (m^3/s)	Temp. °C
Oregon Sites				
Bear	0.08	12.9	1.41	6.8
Bergsvick	0.02	15.2	0.41	7.7
Bewley	0.03	14.4	0.97	7.9
Buster	0.01	15.0	0.85	5.1
Deer	<.01	13.4	0.14	6.8
Elliott	0.02	12.9	1.21	4.0
Farmer	0.10	12.8	0.96	7.8
Kenusky	0.02	13.3	0.36	6.5
Killam	0.08	13.6	0.99	6.5
Klootchie	0.07	16.7	0.46	7.2
Lobster	0.02	15.5	2.12	7.0
Louisignont	0.02	16.4	0.60	5.0
N.F. Rock	0.07	12.5	1.92	5.0
S.F. Ltl. Nestucca	0.06	13.2	1.05	7.3
Tobe	0.01	16.9	0.57	6.7
Washington Sites				
Beaver	0.01	7.5	0.22	4.3
Benson	0.06	15.5	0.61	7.7
Burn	0.01	10.0	0.08	2.5
French	0.18	13.5	1.27	4.1
Harris	0.01	13.5	0.24	6.4
Hoppers	0.04	13.0	0.27	7.5
Hyas	0.10	14.0	1.37	6.5
Laughing Jacobs	0.02	14.5	0.45	8.3
Midnight	0.03	9.5	0.13	4.4
Newbury	0.05	12.0	0.30	8.5
Porter	0.06	14.0	1.15	7.8
Punch	0.05	12.0	0.53	5.3
Shuwah	0.06	14.5	0.53	7.0
Soosette	0.03	15.0	0.22	6.8
Townsend	0.15	8.0	0.53	2.6

difference in physical variables including pool area, percent pool area, riffle area, pieces of LWD, pieces of LWD creating pools (pool-forming LWD), number of habitats, channel slope, geographic region (Washington or Oregon), structure type (engineered or naturally placed log). All ratios of treatment to reference (e.g., pool area, pieces of LWD, etc.) were also log transformed (log x) to meet assumptions of t-test (e.g., normal distribution, equality of variances, additive data). The average ratio of fish densities (treatment over reference) was calculated as a geometric mean. This was necessary to compensate for differences in fish densities among streams and to assure that all streams were given equal weight.

Results

Physical Habitat

The total number of pieces of LWD was significantly higher in treatment than reference reaches during both summer (20 - 80 vs. 8 - 63) and winter (16 – 78 vs. 4 – 64; $P<.01$; Table 2.4) and averaged 1.8 and 1.9 times greater in treatment reaches. The total number of pieces of pool-forming LWD was also significantly higher in treatment than reference reaches during both summer and winter ($P < 0.001$) and averaged 2.8 and 3.0 times greater in treatment than reference reaches.

Treated stream reaches exceeded reference reaches in total wetted area, total number of habitat units, total pool area and total number of pools during both summer and winter ($p < 0.01$ in all cases; Table 2.1). However, total riffle area was not significantly different between treatment and reference reaches during either summer or winter ($p = 0.05$, and 0.09, respectively). Pool area in treatment reaches averaged 1.5

Table 2.4. Total number of pieces of artificially placed, forming pools (functioning) and total LWD both in reference (ref.) and treatment (treat.) reaches for each stream. Project age represents the number of years between LWD placement and my surveys.

Stream	Project age	No. pieces placed LWD (treat. only)	Functioning LWD		Total LWD summer		Total LWD winter	
			ref.	treat.	ref.	treat.	ref.	treat.
Oregon Streams								
Bear	4	11	1	5	9	40	21	54
Bergsvick	3	3	0	5	8	40	14	55
Bewley	3	6	0	2	11	35	10	16
Buster	2	14	2	3	41	27	25	24
Deer	3	6	5	4	34	31	29	24
Elliott	1	12	2	1	29	57	26	57
Farmer	3	13	0	5	41	46	26	37
Kenusky	3	6	0	3	26	66	29	73
Killam	3	8	1	1	17	39	14	40
Klootchie	3	8	0	3	27	50	14	48
Lobster	11	6	0	6	40	70	42	43
Louisignont	2	8	2	6	18	28	15	50
Ltl. Nestucca (South Fk.)	3	10	0	6	11	39	12	47
Rock (North Fk.)	3	7	1	4	27	48	43	55
Tobe	4	10	0	8	28	52	22	42
Washington Streams								
Beaver	3	11	7	8	25	54	31	73
Benson	7	18	2	6	18	40	42	52
Burn	5	15	2	8	35	80	25	66
French	6	42	2	8	23	55	20	66
Harris	12	12	0	9	22	24	22	27
Hoppers	1	10	5	10	33	35	43	46
Hyas	6	16	0	5	0	42	4	50
Laughing Jacobs	2	35	2	5	61	66	49	53
Midnight	4	24	1	9	27	28	31	30
Newbury	12	9	1	4	9	20	15	24
Porter	5	22	2	4	25	62	21	55
Punch	12	9	8	9	63	59	64	78
Shuwah	1	12	6	9	38	47	52	53
Soosette	3	28	1	2	9	48	16	72
Townsend	2	17	3	7	29	42	37	51

times that in reference reaches during both summer and winter, and riffle area was reduced (but not significantly) by a factor of 0.8 in both summer and winter. Treated reaches had 1.3 times more pools as reference reaches in summer and 1.5 times more in winter. The total number of habitat units (an indicator of habitat complexity) was 1.1 and 1.2 times higher in treatment than reference reaches during summer and winter, respectively.

Salmonid Densities

Summer

Juvenile coho salmon densities (fish·m^{-1}) were higher in treatment reaches than reference reaches in summer (1.81 times higher; $p < 0.01$), but densities of age 1+ cutthroat, age 1+ steelhead trout, and trout fry did not differ ($p = 0.08$ $p = 0.46$ $p = 0.24$ respectively; Tables 2.5, 2.6). There was a positive linear relationship between the response of summer coho salmon densities (ratio treatment/reference) and the number of pieces of pool-forming LWD ($p < 0.01$, $r^2 = 0.25$; Figure 2.2a). No significant relationship was detected between coho response and any other individual or combination of physical variables ($p > 0.10$). No significant relationship existed between any individual or combination of physical variables and cutthroat trout response to treatment ($p > 0.15$). Age 1+ steelhead trout densities were negatively correlated with a difference in pool area ($p = 0.01$, $r^2 = .32$), % pool area ($p < 0.01$, $r^2 = 0.45$; Figure 2.2b), and positively correlated with difference in riffle area ($p = 0.001$), but not with any other combination or individual physical variables. No response was detected between trout

Table 2.5. Densities (fish·m^{-1}) of salmonids in treatment (Treat.) and reference (Ref.) reaches of study streams during summer.

	Coho		Cutthroat		Steelhead		Trout Fry	
Stream	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.
Oregon								
Bear	1.01	3.09	0.32	0.25	2.97	0.96	0.73	0.96
Bergsvick	0.09	0.45	0.07	0.18	0.04	0.08	0.03	0.07
Bewley	0.19	0.13	0.15	0.10	0.06	0.06	0.00	0.00
Buster	0.47	0.77	0.08	0.09	0.01	0.00	0.00	0.00
Deer	0.44	0.85	0.07	0.13	0.00	0.00	0.00	0.01
Elliott	0.00	0.00	0.56	0.53	0.26	0.31	0.47	0.53
Farmer	0.02	0.08	0.56	0.51	0.36	0.54	0.18	0.53
Kenusky	0.35	0.66	0.35	0.17	0.00	0.01	0.00	0.04
Killam	0.13	0.27	0.35	0.39	0.32	0.57	0.03	0.01
Klootchie	0.17	0.48	0.25	0.29	0.12	0.11	0.03	0.02
Lobster	1.76	4.19	0.34	0.28	0.33	0.36	0.04	0.14
Lousignont	1.22	2.27	0.06	0.04	0.11	0.13	0.07	0.03
Ltl. Nestucca (South Fk.)	0.57	1.51	0.21	0.23	0.31	0.21	0.02	0.00
Rock (North Fk.)	1.98	3.23	0.28	0.48	0.76	0.87	0.39	0.61
Tobe	0.28	0.49	0.06	0.17	0.45	0.41	0.44	0.34
Washington								
Beaver Creek	2.73	2.88	0.01	0.44	0.00	0.00	0.08	0.53
Benson Creek	0.23	0.25	0.02	0.01	0.11	0.13	0.86	0.56
Burn Creek	1.40	5.35	0.03	0.02	0.28	0.20	0.06	0.00
French Creek	0.58	0.41	0.02	0.02	0.35	0.31	1.34	1.09
Harris Creek	0.00	0.00	0.21	0.22	0.00	0.00	0.01	0.01
Hoppers Creek	0.72	0.64	0.20	0.33	0.00	0.00	0.00	0.00
Hyas Creek	0.12	1.33	0.11	0.15	0.86	1.09	0.68	1.32
Laughing Jacobs Creek	0.46	0.34	0.80	0.87	0.00	0.00	1.14	0.61
Midnight Creek	0.13	1.96	0.30	0.48	0.03	0.07	0.46	1.05
Newbury Creek	0.23	0.38	0.44	0.33	0.00	0.00	0.10	0.02
Porter Creek	1.17	0.49	0.05	0.06	0.40	0.63	0.00	0.00
Punch Creek	0.34	0.60	0.26	0.21	0.00	0.00	0.11	0.16
Shuwah Creek	2.14	2.63	0.61	1.08	0.00	0.00	0.06	0.55
Soosette Creek	1.07	0.75	0.03	0.08	0.00	0.05	0.00	0.00
Townsend Creek	0.00	0.00	0.00	0.00	0.94	0.66	0.10	0.04

Table 2.6. Ratio (geometric mean) of salmonid densities for treatment to reference reaches for all thirty sites combined and separated by state.

Species	Oregon	Washington	All sites
	Summer		
Coho	2.08*	1.55	1.81*
Cutthroat (age 1+)	1.10	1.55	1.27
Steelhead (age 1+)	1.03	1.37	1.19
Trout Fry	1.31	1.05	1.21
	Winter		
Coho	4.25*	2.33*	3.23*
Cutthroat (age 1+)	1.90*	1.44	1.70*
Steelhead (age 1+)	1.82*	1.48	1.73*
Trout Fry	1.25	1.24	1.25

* significant difference ($p < 0.05$)

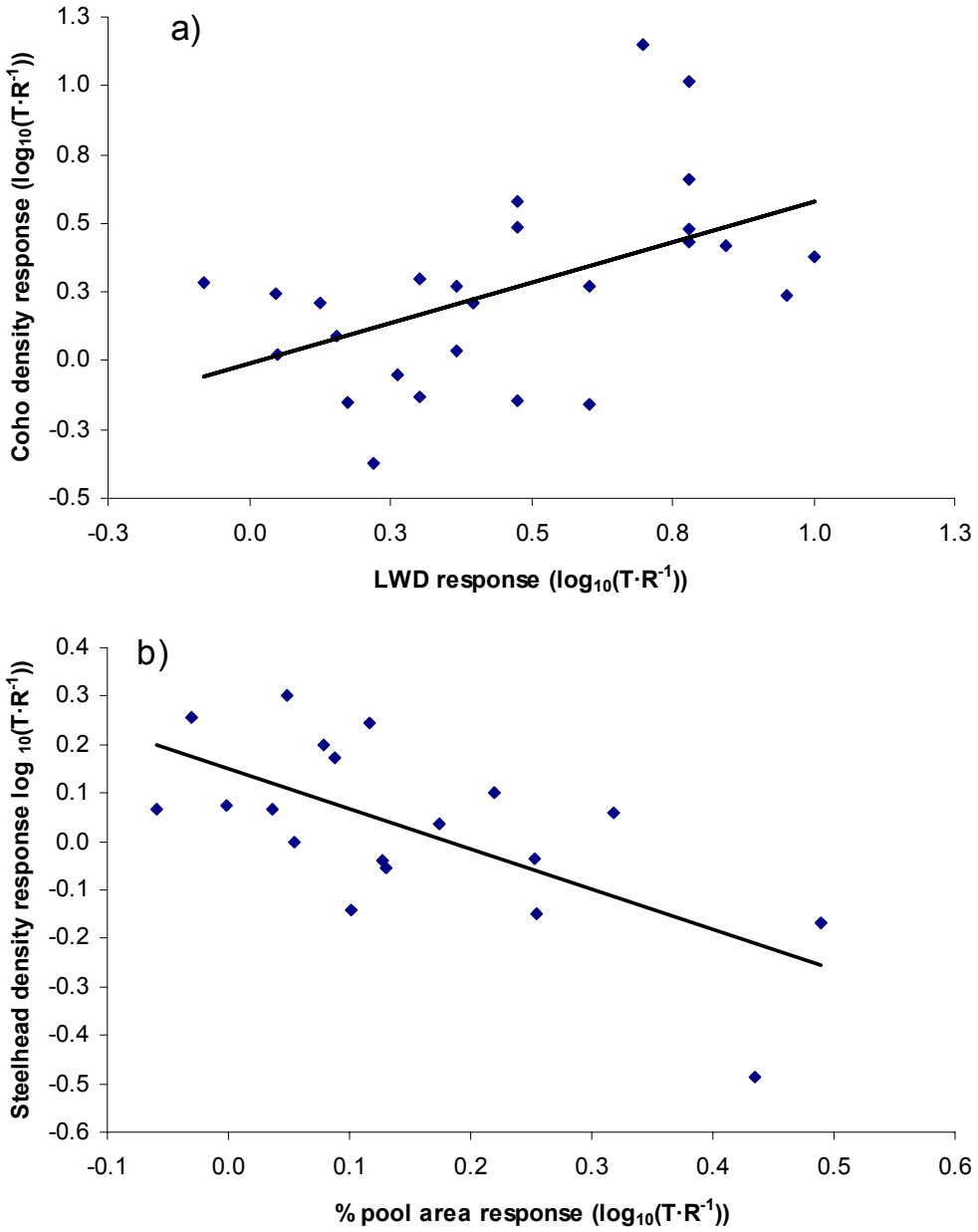


Figure 2.2. Relationship between (a) coho salmon response to restoration ($\log_{10}$ (treatment density/reference density)) and change in pool-forming LWD levels ($\log_{10}$(treatment/reference) for 27 sites inhabited by coho during summer ($y = 0.59x - 0.01$; $p < 0.01$, $r^2 = 0.25$); and relationship between (b) age 1+ steelhead response to restoration ($\log_{10}$(treatment density/reference density and change in percent pool area ($\log_{10}$(treatment/reference) for 20 sites containing 1+ steelhead during summer($y = -0.8323x + 0.1506$; $p < .01$, $r^2 = 0.45$). Treatment (T) consisted of artificial placement of logs and log structures and reference (R) represent unaltered stream reaches.

fry and any physical variables or combination of physical variables ($p > 0.50$ for all models) during summer.

Winter

Coho salmon densities were 3.23 times higher in treatment stream reaches ($p < 0.01$) and age 1+ cutthroat and steelhead trout densities were 1.70 and 1.83 times higher, respectively during winter ($p < 0.01$; Table 2.6). Trout fry densities did not differ between treatment and reference reaches ($p = 0.24$). Multiple regression analysis indicated that coho response during winter was significantly correlated with pool area and restoration type **(**engineered or natural**)** ($p < 0.01$, $r^2 = 0.38$), but not to any other physical variables either in combination or individually ($p > 0.10$). Pool area alone explained 27% of the variation in coho salmon response to restoration among sites ($p < 0.01$, Figure 2.3a). No relationships were detected between age 1+ cutthroat trout winter response to treatment and any combination or single physical variable ($p > 0.36$). Age 1+ steelhead response was not correlated with any combination or single physical variable ($p > .10$). Trout fry response to treatment were negatively correlated with difference in percent pool area ($p = 0.04$, $r^2 = 0.20$; Figure 2.3b).

Fish Length

No differences were observed in mean lengths of coho salmon, cutthroat trout, steelhead trout, or trout fry between treatment and reference reaches during summer ($p =$ 0.06, 0.11, 0.56, 0.5, respectively; Table 2.7) or winter ($p =$ 0.29, 0.22, 0.37, 0.16, respectively). However, mean coho length was negatively correlated with coho density

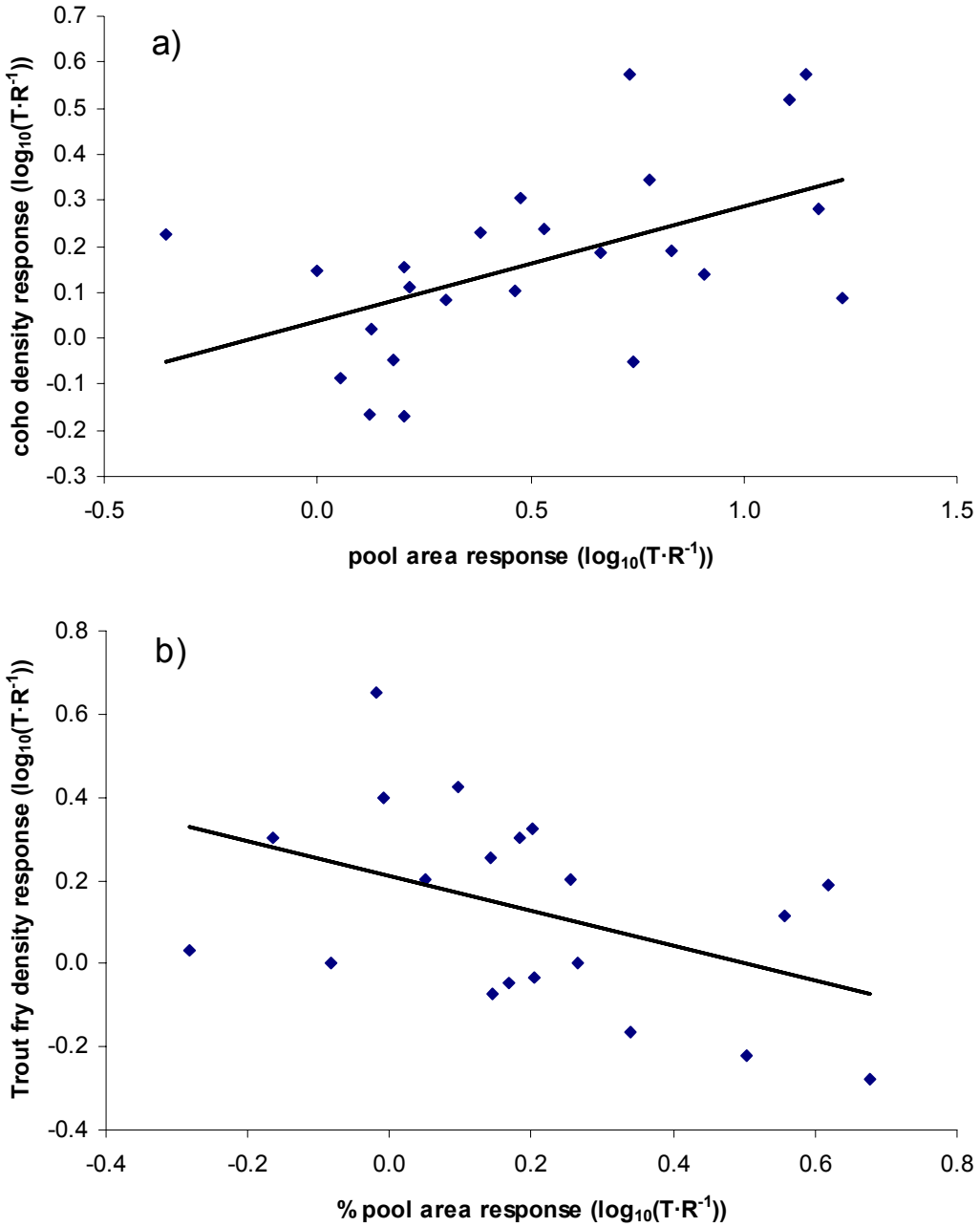


Figure 2.3. Relationship between (a) juvenile coho response to restoration ($\log_{10}$ (treatment/reference) and change in pool area ($\log_{10}$(treatment/reference)) during winter ($y = 0.25x + 0.04$; $p < 0.01$, $r^2 = 0.27$) for 24 sites inhabited by coho during winter; and (b) relationship between trout fry response ($\log_{10}$(treatment/reference)) to restoration and difference in percent pool area ($\log_{10}$ (treatment/reference)) for 20 sites inhabited by trout fry during winter ($y = -0.42x + 0.21$; $p = 0.04$, $r^2 = 0.20$). Treatment (T) consisted of artificial placement of logs and log structures and reference (R) represent unaltered stream reaches.

Table 2.7. Mean length (mm) of each species by stream and reach during summer. If n < 5 mean length was not calculated and designated na, and dashes (--) indicates species was not present.

	Coho		Cutthroat		Steelhead		Trout Fry	
Stream	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.
Oregon Sites								
Bear	76	67	90	134	83	90	56	54
Bergsvick	71	69	146	128	83	95	na	57
Bewley	75	79	128	116	118	--	--	--
Buster	62	61	149	127	na	--	--	--
Deer	65	61	114	106	--	--	--	NA
Elliott	na	na	108	107	87	85	53	53
Farmer	85	77	115	126	76	77	56	53
Kenusky	70	71	115	117	--	na	--	54
Killam	84	84	118	115	90	99	na	na
Klootchie	74	74	116	132	110	99	na	na
Lobster	62	60	115	120	85	93	57	56
Louisignont	63	61	171	187	105	129	52	48
Ltl Nestucca	73	73	106	118	96	87	na	--
Rock	67	66	127	119	104	98	54	50
Tobe	70	73	83	112	89	86	52	53
Washington Sites								
Beaver	50	52	na	82	--	--	30	36
Benson	63	64	na	na	113	104	41	42
Burn	57	55	93	159	104	103	24	
French	58	54	na	--	104	109	43	39
Harris	--	--	109	157	--	--	na	na
Hoppers	71	70	129	116	--	--	--	--
Hyas	86	75	128	122	77	84	55	53
Laughing Jacobs	69	69	74	81	--	--	53	54
Midnight	66	65	105	108	115	111	48	47
Newbury	72	75	92	99	--	--	58	na
Porter	79	82	172	195	109	95	--	--
Punch	75	70	106	125	--	--	47	53
Shuwah	66	66	91	89	--	--	48	50
Soosette	76	76	84	82	na	115	--	--
Townsend	--	--	--	--	133	142	50	52

($p < 0.01$, $r^2 = 0.29$; Figure 2.4b), but no difference existed between reaches (ANCOVA, $p = 0.57$). The difference in coho length (treatment - reference) during summer was negatively correlated with coho density response ($p < 0.01$, $r^2 = 0.29$; Figure 2.4a).

Discussion

Physical habitat

Increases in habitat complexity, pools and slow water habitats in response to restoration and specifically LWD placement have been well documented in western North America (e.g., Crispen et al. 1993; Riley and Fausch 1995; House 1996; Cederholm et al. 1997). My results support these findings; an overall increase in pool area, number of pools and LWD loading was detected in the 30 streams that I sampled. Large woody debris counts differed between summer and winter in some of my sites primarily due to transport of small logs (10 cm in diameter 1.5 to 3 m long) in or out of a reach between surveys. However, little difference was detected if only medium and large size classes (20-50 cm and > 50 cm) were examined. Thom (1997) examined physical responses of streams to LWD one year after treatment in six of the same streams I sampled and reported significant increases in number and volume of LWD, number of habitats and deep pools, but not in pool area. However, it may take several high flow events before the channel responds completely to LWD additions (Reeves et al. 1997) and I may have seen a larger physical response than reported by Thom because I sampled most sites 3-4 years after LWD placement.

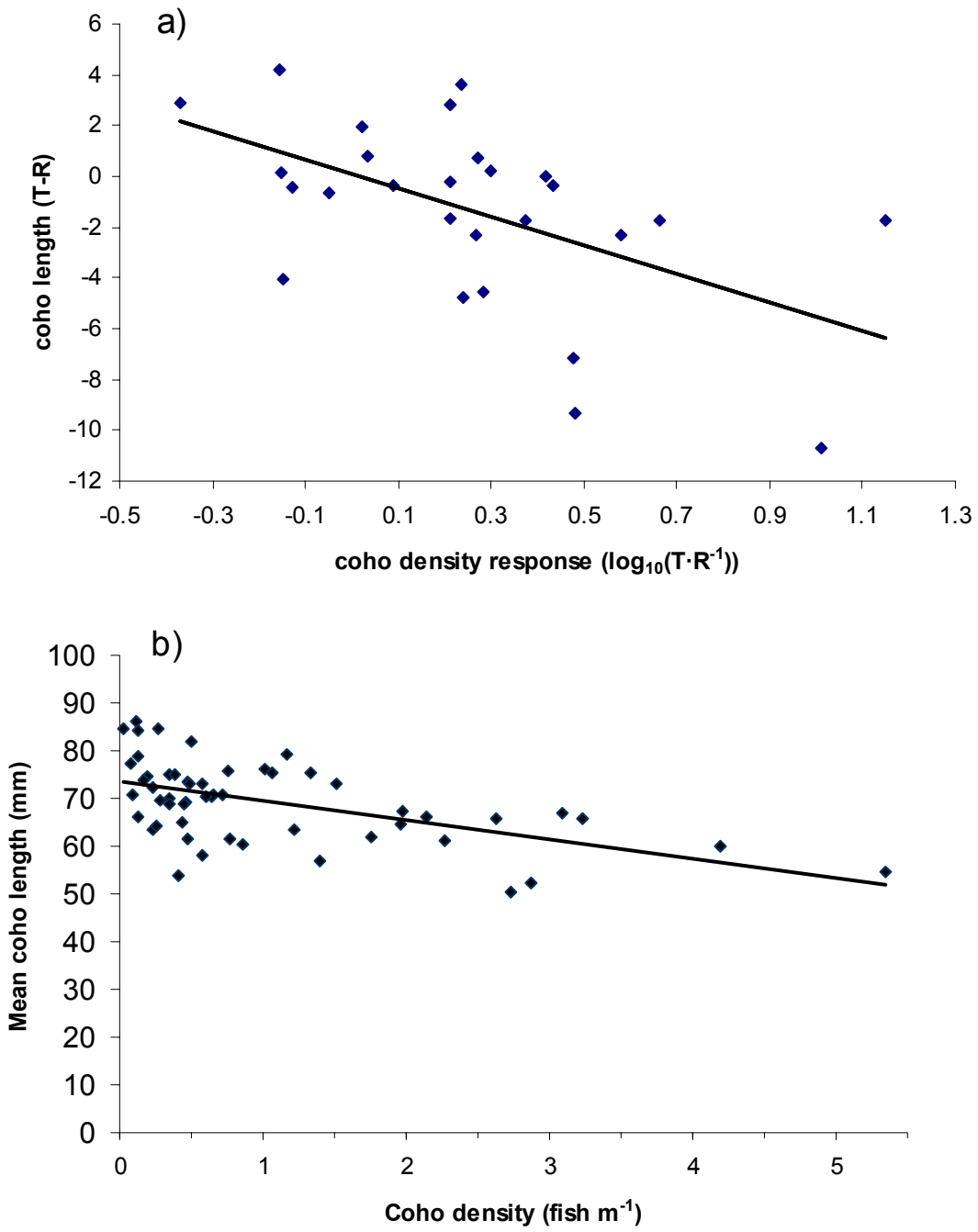


Figure 2.4. Relationship between (a) mean coho salmon length (treatment - reference) and coho salmon response to restoration ($\log_{10}$(treatment/reference) for 27 sites containing coho salmon during summer ($y = -5.61x + 0.09$; $p < 0.01$, $r^2 = 0.29$)); and (b) Relationship between coho salmon mean length (mm) and reach density ($y = 6.10x - 0.07$; $p < 0.01$, $r^2 = 0.29$).

Salmonid Densities

Coho salmon

Most studies on stream restoration or habitat improvement projects have reported significant physical responses to restoration, but evaluation of the responses of juvenile salmonids has been less extensive and the results less consistent. My sampling of 30 streams (27 utilized by juvenile coho) indicated a 1.8 fold increase in juvenile coho densities during summer in response to LWD placement and a 3.2 fold increase during winter months. Similarly, House and Boehne (1985, 1986), House (1989), House (1996) reported increased juvenile coho levels during summer in response to restoration in coastal Oregon streams. Nickelson et al. (1992a) found similar densities of juvenile coho in constructed and natural pools during both summer and winter in several coastal Oregon streams. However, Cederholm et al. (1997) found no significant difference in juvenile coho densities during summer in response to LWD placement in Porter Creek, Washington, but significantly higher levels of juveniles during winter and smolts during spring. In long-term monitoring of a restoration project in Fish Creek, Oregon, Reeves et al. (1997) found no significant increase in abundance of juvenile coho or coho smolts following restoration. With the exception of Nickelson et al. (1992a), these other evaluations occurred on individual streams and may not be broadly applicable elsewhere. For example, Reeves et al. (1997) indicated that many shallow low-gradient habitats such as glides were eliminated after restoration. I found that glides and shallow pools held the highest densities of juvenile coho during summer. Furthermore, during summer juvenile coho occupy pools, glides and other low gradient habitats but are almost exclusively

found in pools and slack water habitats during winter (Hartman 1965, Bustard and Narver 1975a, Bisson et al. 1982, Nickelson et al. 1992b).

I found that the response of coho to restoration was positively correlated with pool-forming LWD during summer and an increase in pool area during winter. It is unclear why, pool-forming LWD was not positively correlated with coho response during winter. However, determining the influence an individual piece of LWD has on creating a pool is difficult at higher flows common during my winter surveys. Coho are found almost exclusively in pools and backwater areas during winter and wood cover is an important element during both summer and winter (Bustard and Narver 1975a, Nickelson et al. 1992b). LWD loading is positively correlated with both pool area and frequency (Beechie and Sibley 1997). Therefore, my results are consistent with the general response of stream channels to LWD and the seasonal habitat preferences of juvenile coho salmon.

Cutthroat trout

I detected an increase in density of juvenile cutthroat trout (age 1+) related to LWD placement in winter and a weak ($P = 0.09$) increase in summer. During summer, cutthroat trout have more general habitat preferences than either steelhead or coho salmon and tend to be found in both pools and low gradient riffles (Bisson et al. 1982, 1988). Therefore, one might expect their response to restoration to be similar, but weaker than coho salmon. House and Boehne (1985, 1986), House (1996) and House et al. (1989) reported significant increases in juvenile cutthroat trout densities in response to placement of boulder structures and gabions in coastal Oregon streams during summer.

Juvenile coho salmon tend to competitively exclude both steelhead and cutthroat from pools during summer (Hartman 1965; Bisson et al. 1982; Glova 1987) and my results might have differed had I examined more sites inhabited only by cutthroat trout.

Winter densities of cutthroat trout were significantly higher (1.5 times) in treatment reaches. Unfortunately, few studies exist on the winter ecology of cutthroat trout and I found no studies on the response of coastal cutthroat trout to restoration in winter months. Bustard and Narver (1975b) found that during winter juvenile cutthroat occupied pools with cover. Glova (1986) found that agonistic behavior between cutthroat and coho was high during summer and lead to habitat segregation but aggression was low during winter and both species utilized pools, consistent with the observations of Bustard and Narver (1975a). The seasonal differences I observed in response of cutthroat trout to LWD placement are most likely due to a shift in habitat use and competition from summer to winter. I observed coho, steelhead and cutthroat almost exclusively in pools during winter so factors affecting pools might be expected to influence winter densities or distributions.

Steelhead trout

Steelhead trout densities did not differ between treatment and reference reaches during summer. Case studies examining steelhead response to LWD and boulder placement in individual streams have shown varying results. Cederholm et al. (1997) found no change in juvenile steelhead densities during summer or winter following LWD placement. Similarly, House and Boehne (1985), House (1996), Moreau (1984) and Chapman (1996) found no significant difference in steelhead parr or fry densities during

summer following placement of instream structures. However, steelhead response to LWD placement in my study was negatively correlated with changes in pool area and positively correlated with changes in riffle area. This is consistent with studies on habitat use which indicate that steelhead occupy riffles and fast water habitats during summer (Bisson et al. 1982, Bisson et al. 1988). Thus, the physical responses to LWD placement that tend to benefit coho salmon (increased pool area and decreased riffle area), may decrease juvenile steelhead densities at those sites in summer.

In contrast to summer, winter steelhead densities were significantly higher (1.7 times) in treatment than in reference reaches. Only Cederholm et al. (1997) and Reeves et al. (1997) evaluated winter or spring response of juvenile steelhead to LWD additions but neither found significant increases in juveniles or smolts. During winter I observed juvenile steelhead, cutthroat and coho primarily in pool habitats whereas in summer steelhead were generally found in low gradient riffles and glides. However, no significant relationship existed between change in pool area (treatment-reference) and steelhead densities (treatment-reference), even though my observations and data from other studies indicate that steelhead show a strong preference for pools and woody cover during winter (Hartman 1965; Bustard and Narver 1975a). The lack of a significant relationship between steelhead response to restoration and specific physical variables during winter may be due to the large natural variability among sites or my inability to quantify microhabitats (e.g., substrate, velocity preferences, etc) to which steelhead were responding. However, my results indicated that during winter age 1+ steelhead benefit from LWD placement and data on habitat preferences suggest that it is most likely due to an increasing preference for pools and LWD cover during winter.

Trout Fry

I found no significant increase in trout fry (age 0 cutthroat trout and steelhead) densities during either summer or winter in relation to restoration. House and Boehne (1986) and House et al. (1989) reported increases in trout fry densities following placement of instream structures but Hamilton (1989), House (1996) and Cederholm et al. (1997) reported no increase in trout fry following restoration, and Reeves et al. (1997) found a significant decline in steelhead fry following restoration. Bisson et al. (1988) indicated that age 0 steelhead trout showed no strong preference or avoidance of most habitat types during summer, except backwater pools where they were most abundant. Similarly, they found that age 0 cutthroat tended to avoid riffles and prefer pools and glides. Age 0 steelhead or rainbow trout tend to occupy stream margins during summer and winter (Hartman 1965; Campbell and Neuner 1985). Moore and Gregory (1988) found that cutthroat trout fry densities were positively correlated with lateral (edge) habitat and increased 2.2 fold when lateral habitat was experimentally increased. I did not quantify edge or lateral habitats, though my general observation was that little change occurred in lateral habitats as a result of LWD placement and the creation of pools by LWD may have eliminated some shallow water edge habitats. Trout fry response to LWD placement in my study was negatively correlated with pool area during winter, suggesting that increasing deep water habitats may eliminate some fry rearing areas.

Fish Length

Mean fish length did not differ between treatment and reference reaches. However, coho were generally smaller in treated stream reaches, whereas age 1+ cutthroat and steelhead tended to be slightly larger. The difference in fish length between treatment and reference reaches was positively correlated with density for coho salmon, indicating that LWD placement lead to more but smaller juvenile coho during summer months. Coho salmon growth in streams is inversely related to density (Fraser 1969), and overwinter survival and smolt to adult survival within a population are also size dependent (Quinn and Peterson 1996). To the extent that restoration leads to density-dependent reduction in growth, there may be smaller responses in adult abundance than might be projected from the increased juvenile densities.

Other factors influencing fish response

One purpose of stream restoration efforts is to increase abundance of fish. However, evaluations of restoration and habitat enhancement (including this one) tend to quantify fish response at small spatial scales. If the improved habitat simply concentrates fish that are moving among reaches and reduces their growth, the consequences for the population may be negligible. Consistent with this concern, Kahler (1999) reported that about half the juvenile coho salmon moved at least one habitat unit during the summer in three western Washington streams. Moreover, fish in poor quality habitat (smaller, shallower pools) were more likely to move than those in larger pools. A mark-recapture study conducted in one of my study streams found little or no movement of fish between the treatment and reference reaches during summer and winter (P. Roni, unpublished

data). Riley and Fausch (1995) and Gowan and Fausch (1996) reported increases in adult trout numbers following restoration in six Wyoming streams but recovery of marked and unmarked fish suggested that movement rather than survival was responsible for much of the increase. However, Riley and Fausch (1995) acknowledged that they did not monitor the sites long enough to determine whether there was a long-term increase in fish production due to LWD placement. Therefore, while movements may explain some differences in fish densities between treatment and reference reaches, there may also have been changes in survival, but I have no direct way to distinguish these factors.

Hamilton (1989) reported a two-fold increase in steelhead parr in treated stream reaches, while numbers in untreated reaches decreased by half over a two year period. Initially, placement of instream structures or LWD might lead to redistribution of juvenile and resident fish and it may take several years for fish populations to fully respond. Reeves et al. (1991) suggested that monitoring for two generations might be needed to detect population response for anadromous salmonids in an individual stream. I sampled most sites 3-7 years (mean = 4.4) after treatment so that the full biological response was likely to have occurred.

It is possible that differences in LWD placement technique or initial LWD abundance may account for some of the differences I observed. Most of the sites sampled in Washington consisted of "engineered LWD structures" such as log weirs and deflectors which were held in place with rebar or cable, whereas most of the Oregon sites were artificially created LWD jams with minimal use of cable. Sites in Oregon were generally selected for LWD placement only if stream surveys had indicated that there was little to no LWD present (Thom 1997). In contrast, some Washington sites had

moderate levels of natural LWD in reference and treatment reaches. These sites were selected for treatment due to reasons other than just LWD levels such as ease of access, available funds, etc. Differences in fish response between states (regions) were most pronounced for coho salmon in winter; abundance in treatment reaches was 4.3 times that in reference reaches in Oregon sites, compared to 2.3 in Washington sites. While I was not able to definitively attribute differences in fish response to region or structure type, sites that had low levels of LWD prior to treatment generally had the largest physical and biological responses. This is supported by the positive relationship between coho response and difference in LWD numbers between treatment and reference.

Everest et al. (1991) suggested that response to restoration should be monitored at a watershed level through smolt trapping rather than at a reach scale. However, most instream projects occur at reach scale (100-500 m) and monitoring at a watershed level is unlikely to detect changes in fish abundance or habitat use at the reach level. Smolt trapping treatment and reference reaches, such as that by Cederholm et al. (1997), would have provided estimates of smolt production, but would have been prohibitively expensive on the large number of streams I sampled. Because most coho and age 1+ trout are likely to have smolted shortly after my winter surveys, they provide reasonable estimates of smolt production from the study reaches. Ultimately, a study combining both reach scale and watershed scale juvenile and smolt production estimates is needed to estimate response at different scales and provide additional information on fish movements within restored and unrestored sections of a watershed.

My results provide strong evidence that artificially placed LWD leads to significantly higher densities of juvenile coho in summer and winter and higher densities

of cutthroat and steelhead during winter, especially in sites deficient in wood to begin with. However, density-dependent growth of coho salmon may reduce the net benefit to the population if subsequent survival is size-related. Summer densities of steelhead may be reduced by artificially placed LWD, but this may be more than compensated for by increases in winter rearing areas and densities. While the study design I employed was not designed to determine the effectiveness of individual projects, it provides insight into the factors that make projects successful. The relationship between coho abundance and pool area and pool-forming wood indicates that projects that dramatically increase pool area or LWD creating pools will provide the largest increases in fish abundance. However, I focused on small low-gradient streams on forested sites and the results of this study should be applied with caution in larger or steeper channels, in urban and agricultural areas, or in different geographic provinces. Finally, my results in no way negate the need to focus on restoring natural processes that create and maintain salmon habitat rather than relying on instream manipulations (e.g., placement of wood, boulders, nutrients) which can be costly and not address the ultimate factors limiting habitat quality and salmonid production.

CHAPTER 3: RESPONSES OF SALAMANDERS, SCULPIN AND LAMPREY TO ARTIFICIAL WOOD PLACEMENT IN SMALL PACIFIC NORTHWEST STREAMS

Introduction

Enhancement and restoration of aquatic habitats in western North America has focused primarily on improving habitat in small streams for one or two species of salmonid fishes (Reeves et al. 1991a). In the Pacific Northwest and many coastal regions with anadromous species, this emphasis on salmonids has produced inconsistent results, as a variety of factors during the marine and freshwater life history stages can influence abundance and survival (Bisson et al. 1992). Platts and Rinne (1985), Hunt et al. (1988), Reeves et al. (1991a), Beschta et al. (1994), Chapman (1996) and others have emphasized the need for a more comprehensive evaluation of restoration that includes examining the response of nonsalmonids to habitat enhancement efforts. However, the response of nonsalmonid fishes or other vertebrates to restoration or enhancement has seldom been examined. Bilby and Fransen (1992) reported increased speckled dace (*Rhinichthys osculus*) densities following instream large woody debris (LWD) placement but other factors (e.g., beaver activity and canopy removal) made a cause-and-effect relationship with LWD unclear. Even in regions with more diverse fish fauna than the Pacific Northwest, the response of the nonsalmonid fishes to habitat improvement efforts has rarely been examined. In a review of 41 stream habitat enhancement projects in Wisconsin, Hunt (1988) found only one that examined the response of nonsalmonid fishes. Angermeier and Karr (1984) examined responses of 10 warmwater fishes to wood

removal and placement in small Illinois stream. They found that most fish species and age classes > 75 mm were more dense in stream sections with woody debris. Lonzarich and Quinn (1995) found no effect of cover (woody debris) and depth on threespine stickleback (*Gasterosteus aculeatus*) or coastrange sculpin (*Cottus aleuticus*) habitat use, growth or survival in an artificial stream channel.

In contrast to the paucity of studies on nonsalmonids, numerous studies have examined the responses of juvenile salmonids to habitat enhancement and stream restoration, with inconsistent results. House and Boehne (1985), House et al. (1989) and House (1996) reported increased juvenile coho salmon (*Oncorhynchus kisutch*) densities in several Oregon coastal streams, and small increases in densities of juvenile cutthroat (*O. clarki*) and steelhead (*O. mykiss*) trout. Cederholm et al. (1997) reported that significantly more coho salmon smolts migrated from a western Washington stream following LWD placement (see also Chapter 2). Hunt's (1988) review of stream enhancement efforts in Wisconsin streams found higher densities of brown (*Salmo trutta*), brook (*Salvelinus fontinalis*) and rainbow trout (*O. mykiss*) following placement of instream structures. Conversely, others have reported no significant biological response or even decreased salmonid abundance following restoration (e.g., Reeves et al. 1997; see reviews by Beschta et al. 1994 and Chapman 1996). Karr (1981), Bisson et al. (1992) and others suggested that changes in the composition and organization of stream fish communities provide more comprehensive evidence of the extent of environmental degradation and subsequent recovery than trends in abundance of individual species.

Monitoring of stream restoration projects in the Pacific Northwest has often focused on changes in physical habitat and whether the structures functioned properly

from a hydrological perspective. Several authors documented increases in pool frequency, pool depth, woody debris abundance and sediment retention following placement of instream structures (e.g., Tripp 1986, Reeves et al. 1991a, Viola et al. 1991, Poulin and Associates 1991, Klassen 1991, House et al. 1991, Crispin et al. 1993, and others). Thus the physical response to habitat manipulations has been well documented and those species that prefer pools and abundant cover are likely to benefit from the placement of LWD.

The effect of restoration efforts on species richness or fish community diversity has not been examined, though diversity has frequently been correlated with habitat degradation (e.g., Karr 1981, Fausch et al. 1984, Angermeier and Karr 1986, Lyons et al. 1996, and others). Reeves et al. (1993) found that basins with high levels of timber harvest had lower diversity of salmonids and were more likely to be dominated by a singles species. They attributed these differences to the higher levels of LWD and habitat complexity in basins with low levels of timber harvest. However, Chen (1998) found only weak correlations between physical variables (pool depth, volume, and area) and salmonid diversity, and he suggested that habitat structure may not be important in determining community diversity at a reach or habitat unit level.

The effects of habitat alteration and degradation in streams has been assessed successfully using fish communities in other parts of North America, particularly the Midwest and southwestern United States (e.g., Gorman and Karr 1978, Schlosser 1982, Meffe 1984, Fausch and Bramblett 1991, Lyons et al. 1996 and others). However, most small coastal streams in the Pacific Northwest are inhabited by 3-5 species of salmonids, 2-3 species of sculpin (*Cottus* spp.), 2-3 species of lamprey (*Lampetra and Entosphenus*

spp.), 1-2 species of dace (*Rhinichthys* spp.) and possibly 1-3 species of fish of the families Catostomidae, Cyprinidae and Gasterosteidae (Wydowski and Whitney 1979, McPhail and Lindsey 1986). Few small streams in the region are inhabited by more than 5-7 species. Physical barriers to migration may further reduce the number of fish species present in reaches of small streams. Little is known about the effects of habitat restoration on these species, but existing information on habitat preferences suggests that those that prefer pools may respond positively to changes in pool area and complexity. For example, larval lamprey (*Entosphenus tridentatus* and *Lampetra spp*.) inhabit the benthos of pools and other slow water habitats (Wydowski and Whitney 1979; Scott and Crossman 1998) and are likely to be found in higher densities in stream reaches with high percentage of pool area and fine sediment. In contrast, other benthic fishes such as cottids are common in pools and riffles (Wydowski and Whitney 1979; Scott and Crossman 1998) and their response to restoration is unclear. In addition to fishes, giant salamanders (*Dicamptodon spp.*), a large (up to 300 mm in length) benthic predator, also inhabit small Pacific Northwest streams. Giant salamanders prey on both fish and aquatic invertebrates and are known to be sensitive to habitat alterations (Murphy et al. 1981, Hawkins et al. 1983, Corn and Bury 1989) and any study examining the response of fishes to habitat alterations would be incomplete without examining the response of giant salamanders.

The relatively depauperate fish fauna of the Pacific Northwest (McPhail and Lindsey 1986) and limited knowledge of the ecology of nonsalmonid fishes in this region has inhibited application of a community level approach in determining the effects of habitat modification on fishes (Bisson et al. 1992). Regardless of the diversity of the fish

community, it is still important to examine the response of different species to habitat manipulations. The objectives of this study were two fold: first, to quantify the response of nonsalmonids and giant salamanders to the artificial placement of LWD (restoration) in small streams; and second, to examine the effects of LWD placement on species richness and diversity.

Methods

Thirty streams in western Washington and northwest Oregon (Chapter 2; Figure 2.1) with paired treatment and reference (control) reaches were sampled during summer between August 1996 and September 1998. Treatment was defined as the artificial placement of LWD within the active stream channel. Paired treatment and reference reaches 75-120 m long were selected in each stream. Study reaches were at least 10 times longer than the bankfull width, and most reaches were 100 m long. Treatment and reference reaches within a given stream were always the same length.

The selection of study streams with paired treatment-reference reaches was based in part on physical and biological stream characteristics including stream size, bankfull width, channel type and fish species composition. Only sites with treatment and reference reaches of similar gradient, confinement, and channel width were selected. The study streams ranged from 4 m to 12 m in bankfull width and 0.5 % to 4.2% slope. Annual precipitation varied from 107 cm to 315 cm. Dominant forest types were primarily Douglas fir (*Pseudotsuga menziesii*), Sitka spruce (*Picea sichensis*), and western hemlock (*Tsuga heterophylla*). The dominant watershed geology was volcanic, sedimentary or glacial-alluvial and varied by site, but was consistent for reaches within a

stream (Chapter 2, Table 2.1). The elevations of the study sites ranged from 12 m to 789 m and drainage area upstream of our study reaches ranged from 124 ha to 2388 ha. Habitat units within each stream reach was classified using a modification of the methods and habitat types described by Bisson et al. (1982). I used minimum residual depth and pool area to consistently identify pool habitats (Lisle 1987; Schuette-Hames et al. 1994). Total surface area of each habitat was estimated by measuring the total length and multiplying by the average of 3-5 width measurements.

Discharge was estimated with a flow meter immediately following each survey. All pieces of natural and artificially placed large woody debris (LWD) within the wetted stream channel greater than 10 cm in diameter and 1.5 m long were inventoried. The diameter class (small: 10-20 cm, medium: 20-50 cm, and large: > 50 cm) and approximate length were recorded. The function of an individual piece of LWD based on its influence on pool formation and channel scour was classified into one of three categories including: 1) dominant - primary factor contributing to pool formation, (2) secondary - influences zone of channel scour but not responsible for pool formation, (3) negligible - may provide cover but not involved in scour (Montgomery et al. 1995). In addition, the percent of each piece of LWD that was in the low-flow wetted channel and within the bankfull channel was determined (Robison and Beschta 1990).

Multiple-removal elecrofishing was used to estimate fish abundance within each habitat unit and stream reach (Carle and Strub 1978). Each habitat was sampled separately by placing 3.2 mm mesh blocknets at the upstream and downstream boundaries to prevent immigration or emigration during sampling. Three removals were made through each habitat and fourth was made if a 50% or greater reduction in fish

numbers was not seen between the second and third passes. All fish captured were anesthetized with tricaine methanesulfonate (MS-222), identified, measured to the nearest mm and then released. Species commonly captured included juvenile coho salmon, cutthroat trout, steelhead, adult and juvenile torrent sculpin (*C. rhotheus*), reticulate sculpin (*C. perplexus*), Pacific and Cope's giant salamander (*Dicamptodon tenebrosus* and *D. copei*), and ammocoetes (larvae) of Pacific (*Entosphenus tridentatus*), river (*Lampetra ayresi*) and occasionally brook (*L. richarsoni*; Table 3.1) lamprey. Multiple-removal estimates were used to estimate fish and salamander abundance (Carle and Strub 1978), but were not applied to lamprey because of the low capture efficiency (capture probability < 0.25 in many habitats). Therefore, the total number captured in three removals was used as an estimate of total abundance for lamprey ammocoetes.

Based on length frequency distributions (Jearld 1983), all sculpin greater than 60 mm were considered age 1+ and all those less considered age 0. I could not reliably distinguish between Cope's giant salamander and Pacific giant salamander and there is some disagreement as to whether they are two separate species (Corkran and Thoms 1996). Therefore, I refer to them collectively as Pacific giant salamanders. Similarly, I could not reliably distinguish between larval (ammocoetes) Pacific, river, and brook lamprey and refer to them collectively as larval lamprey.

Table 3.1. Fish and salamander species present in study streams.

Stream	Salmonids			Sculpin		Giant	
	Coho	Cutthroat	Steelhead	Torrent	Reticulate	Salamander	Lamprey
Oregon							
Bear Cr.	X	X	X		X	X	X
Bergsvick Cr.	X	X	X	X	X		X
Bewley Cr.	X	X	X		X	X	X
Buster Cr.	X	X	X	X	X	X	X
Deer Cr.	X	X			X		X
Elliott Cr.		X	X		X	X	
Farmer Cr.	X	X	X		X	X	
Kenusky Cr.	X	X	X	X	X		X
Killam Cr.	X	X	X	X	X	X	
Klootchie Cr.	X	X	X	X	X	X	X
Lobster Cr. (South Fork)	X	X	X	X	X	X	X
Louisignot Cr.	X	X	X	X	X	X	X
Rock Cr. (North Fork)	X	X	X	X	X	X	X
Ltl. Nestucca (South Fork)	X	X	X		X	X	X
Tobe Cr.	X	X	X		X	X	X
Washington							
Beaver Cr.	X	X				X	
Benson Cr.	X	X	X	X		X	
Burn Cr.	X	X	X			X	
French Cr.[1]	X	X	X	X	X	X	
Harris Cr.		X		X	X		
Hoppers Cr.	X	X		X	X		X
Hyas Cr.	X	X	X	X	X		X
Laughing Jacobs Cr.[2]	X	X		X		X	X
Midnight Cr.	X	X	X		X	X	
Newbury Cr.	X	X			X		
Porter Cr.	X	X	X	X	X		X
Punch Cr.	X	X		X	X		X
Shuwah Cr.	X	X		X	X		X
Soosette Cr.	X	X	X	X			X
Townsend Cr.			X				

[1] longnose sucker (*Catostomus catostomus*) also present
[2] coastrange sculpin (*Cottus aleuticus*) also present

Two statistical approaches were taken to test the null hypothesis of no relationship between physical variables and the density of species or age groups of vertebrates. First, differences in species diversity and abundance between treatment and reference reaches were compared using paired t-tests. A Bonferroni correction was applied to compensate for the increased likelihood of finding a significant difference when performing multiple t-tests (SPSS 1999). A family alpha level of 0.05 was used to determine significance and divided by the number of t-tests performed on fish and salamander densities and diversity. However, this resulted in a significance level of less than 0.01 for each individual t-test, which was believed to be overly conservative. Thus, a minimum alpha level of 0.01 was used for each individual t-test. A $\log_{10}(x+1)$ transformation was used on fish and salamander data to meet basic assumptions of a t-test (additive data, normal distribution, equal variances) and account for any zero or low counts (Zar 1999).

In addition to the t-tests, multiple regression was used to examine the relationship(s) between fish response ($\log_{10}$(treatment density/reference density)) and difference in physical variables including pool area, percent pool area, total LWD, LWD in the low flow channel, LWD creating pools, channel slope, geographic region (Washington or Oregon), and structure type (engineered or naturally placed log). Sites with larger physical responses to restoration were predicted to have larger responses by the vertebrate populations. All ratios of treatment to reference (e.g., pool area, pieces of LWD, etc.) were also log transformed ($\log_{10}x$) to meet the assumptions of t-tests. Fish and salamander diversity for each stream reach were estimated using species richness (total number of species) and the Berger-Parker index (Magurran 1988), which

is a simple measure of dominance (fraction of most abundant species). Dominance was also estimated using the fraction of the two and three most abundant species.

Results

Response of Individual Species

The densities of several species were similar between treated and control reaches of streams: giant salamanders ($p = 0.34$), age 1+ reticulate sculpin ($p = 0 .41$), and both age class of torrent sculpin ($p > 0.49$; Table 3.2). Lamprey tended to occur at higher densities in treated reaches ($p = 0.03$) as did age 0 reticulate sculpin ($p = 0.02$), but at the alpha level of 0.01 (Bonferroni correction) I could not reject the null hypothesis of no difference. No relationship was found between any combination of physical variables and giant salamander or age 0 or age 1+ reticulate sculpin or torrent sculpin response to restoration ($\log_{10}$(treatment/reference)) and any physical variables ($p > 0.08$). However, the number of pieces of pool forming LWD was positively related to lamprey density among sites ($p < 0.01$; $r^2 = 0.44$; Figure 3.1).

Mean length did not differ between treatment and reference reaches for any species (paired t-test, $p > 0.30$; Table 3.3). Difference in mean length of age 1+ reticulate sculpin (treatment – reference) was positively correlated with difference in LWD in the low flow channel ($P < 0.01$, $r^2 = 0.28$; Figure 3.2). No single or combination of physical variables was correlated with length for any other species. Difference in length and difference in density between treatment and reference reaches were not correlated with each other for giant salamanders ($p = 0.09$), lamprey ($p = 0.77$), age 0 ($p = 0.80$) or age 1 reticulate sculpin ($p = 0.21$) or age 1+ torrent sculpin ($p = 0.55$). Difference in length

Table 3.2. Densities (number per m) of various species in treatment (Treat.) and reference (Ref.) reaches in study streams. Townsend Creek contained only salmonids and was omitted from the table.

	Giant Salamander		Lamprey		Reticulate Sculpin Age 0		Reticulate Sculpin Age 1+		Torrent Sculpin Age 0		Torrent Sculpin Age 1+	
Stream	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.
Bear	0.07	0.04	0.05	1.12	0.68	0.90	2.11	0.57	--	--	--	--
Bergsvick	--	--	0.02	0.50	0.39	0.87	0.51	0.53	0.11	0.84	0.30	0.36
Bewley	0.01	0.03	0.15	0.38	0.83	1.16	1.06	0.91	--	--	--	--
Buster	0.00	0.02	1.36	0.62	0.36	0.63	0.04	0.18	0.71	0.31	0.33	0.46
Deer	--	--	0.19	0.04	0.97	1.61	0.12	0.20	--	--	--	--
Elliott	0.07	0.02	--	--	0.78	1.65	0.80	1.31	--	--	--	--
Farmer	0.07	0.08	--	--	0.66	1.10	1.42	1.52	--	--	--	--
Kenusky	--	--	0.31	0.97	0.56	0.83	0.51	0.30	0.20	0.05	0.38	0.23
Killam	0.01	0.01	--	--	0.35	0.26	0.85	1.66	0.27	0.16	1.04	1.51
Klootchie	0.00	0.01	0.11	1.34	0.26	0.81	0.45	0.65	0.17	0.29	0.80	0.56
Lobster (S.F.)	0.11	0.21	0.46	0.43	2.67	2.69	0.66	0.89	--	--	--	--
Louisignont	0.10	0.04	0.20	0.91	0.77	1.30	0.31	0.39	0.76	0.60	0.36	0.42
Ltl. Nestucca	0.05	0.19	0.06	0.80	0.61	2.35	0.46	1.05	--	--	--	--
Rock (N.F.)	0.10	0.04	--	--	0.82	0.66	1.33	0.74	0.04	0.03	0.14	0.06
Tobe	0.23	0.37	0.00	0.11	0.68	1.42	0.94	1.65	--	--	--	--
Beaver	0.04	0.04	--	--	--	--	--	--	--	--	--	--
Benson	0.03	0.08	--	--	--	--	--	--	0.63	0.96	1.35	0.92
Burn	0.12	0.06	--	--	--	--	--	--	--	--	--	--
French	0.03	0.20	--	--	--	--	--	--	0.29	0.45	0.94	0.98
Harris	0.11	0.15	--	--	0.33	0.30	0.53	0.29	--	--	--	--
Hoppers	--	--	0.51	0.29	0.96	0.39	0.68	0.47	0.29	0.20	0.13	0.31
Hyas	--	--	0.01	0.15	0.17	0.45	0.17	0.52	0.22	0.60	0.92	1.19
Laughing Jacobs	0.01	--	0.06	0.06	--	--	--	--	0.03	0.03	0.09	0.16
Midnight	0.27	0.19	--	--	0.33	0.97	0.44	1.36	--	--	--	--
Newbury	--	--	--	--	0.48	0.59	0.74	0.63	--	--	--	--
Porter	0.00	0.11	1.19	0.37	0.85	0.97	0.41	0.45	2.33	2.63	1.77	2.11
Punch	0.01	--	0.07	0.17	0.09	0.57	0.08	0.34	0.22	0.58	0.32	0.22
Shuwah	--	--	0.35	0.31	0.34	0.05	0.73	0.09	0.24	0.21	0.48	0.16
Soosette	--	--	0.14	0.02	--	--	--	--	0.31	0.44	0.70	1.31

Table 3.3. Average lengths of fishes and salamanders in reference (Ref.) and treatment (Treat.) reaches of study streams. Mean length was only calculated if n > 2 in a given stream reach. Townsend Creek contained only salmonids and was omitted.

	Giant salamander		Larval lamprey		Reticulate sculpin Age 0		Reticulate sculpin Age 1+		Torrent sculpin Age 0		Torrent sculpin Age 1+	
Stream	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.	Ref.	Treat.
Bear	130.4	200.0	90.0	103.7	45.42	44.81	69.2	70.55	--	--	--	--
Bergsvick	--	--	--	119.3	50.5	49.3	74.9	78.9	53.4	44.3	85.7	92.1
Bewley	--	134.0	97.8	107.7	53.2	53.8	74.2	71.3	--	--	--	--
Buster	--	--	94.9	105.7	46.7	49.0	63.8	66.5	50.7	51.4	87.2	90.7
Deer	--	--	116.9	106.0	47.1	46.1	69.9	65.1	--	--	--	--
Elliott	163.3	--	--	--	53.6	52.6	68.4	67.6	--	--	--	--
Farmer	151.1	197.6	--	--	56.1	55.2	67.2	68.0	--	--	--	--
Kenusky	--	--	93.0	92.6	55.5	50.3	67.6	68.2	51.2	52.2	93.7	85.5
Killam	--	--	--	--	55.3	56.5	71.7	72.7	47.8	54.9	86.7	85.8
Klootchie	--	--	98.5	108.8	54.8	50.6	75.3	78.3	51.5	47.5	93.0	95.6
Lobster	149.1	111.0	71.2	86.7	51.3	50.7	67.0	66.9	--	--	--	--
Louisignont	89.7	209.5	128.0	118.8	49.6	48.4	66.7	66.3	46.5	50.8	88.7	82.0
Ltl Nestucca	152.0	143.7	70.0	111.9	50.1	52.1	68.6	74.6	--	--	--	--
Rock	123.3	126.5	--	--	54.8	51.2	67.5	68.7	55.3	41.0	73.8	72.8
Tobe	96.7	99.5	--	127.5	51.4	51.4	71.5	70.2	--	--	--	--
Beaver	155.0	155.7	--	--	--	--	--	--	--	--	--	--
Benson	141.7	144.9	--	--	--	--	--	--	57.5	56.4	75.2	75.3
Burn	131.5	185.7	--	--	--	--	--	--	--	--	--	--
French	151.7	150.9	--	--	--	--	--	--	54.8	55.9	85.0	89.2
Harris	103.9	114.9			49.5	47.4	73.8	71.3	--	--	--	--
Hoppers	--	--	101.7	112.5	43.8	47.6	83.7	77.8	54.5	55.6	84.5	90.3
Hyas	--	--	--	107.9	42.8	48.3	70.1	78.6	35.0	35.2	87.5	81.3
Jacobs	--	--	110.5	135.5	--	--	--	--	56.0	56.0	76.4	73.3
Midnight	127.9	143.9	--	--	47.8	52.2	76.5	78.2	--	--	--	--
Newbury	--	--	--	--	46.9	46.6	72.4	73.1	--	--	--	--
Porter	--	--	100.5	106.4	47.2	44.6	79.2	77.0	51.5	49.4	79.1	83.1
Punch	--	--	131.4	124.5	46.8	47.9	74.0	75.9	54.0	51.8	87.4	81.2
Shuwah	--	--	128.4	131.7	47.2	55.5	77.5	71.6	51.7	45.6	82.8	75.2
Soosette	--	--	107.1	--	--	--	--	--	32.3	29.1	88.8	86.0

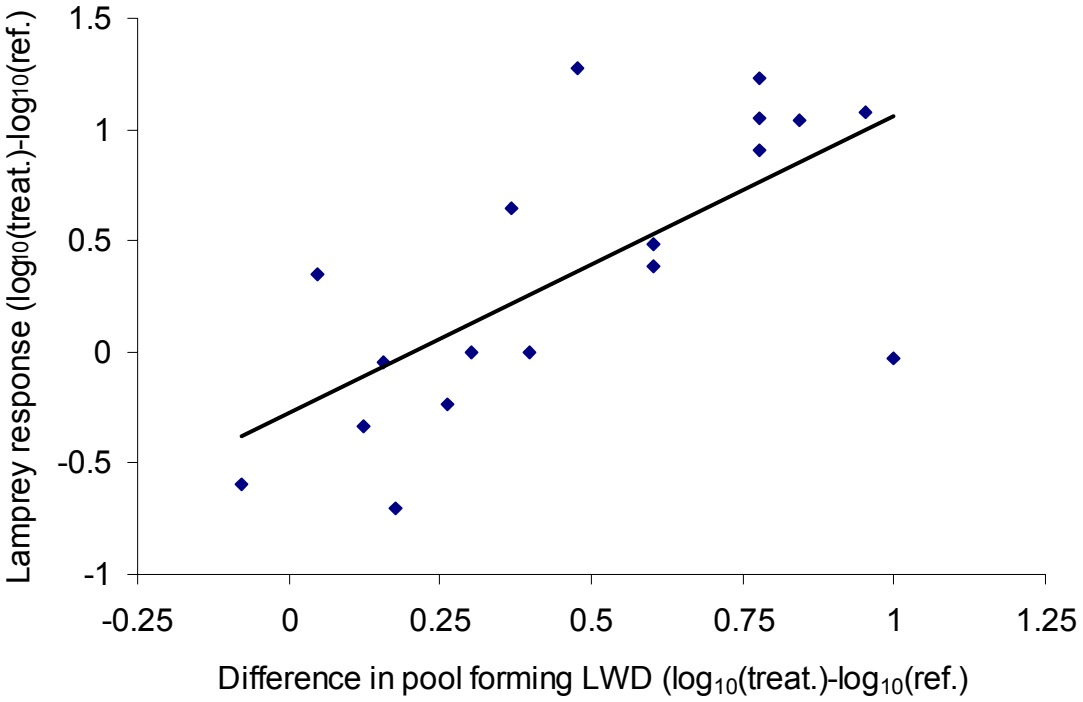


Figure 3.1. Relationship between larval lamprey density response ($\log_{10}$(treatment) - $\log_{10}$(reference)) to restoration and change in pool forming LWD (lamprey response = 1.320X-0.252; $p < 0.01$, $r^2 = 0.44$).

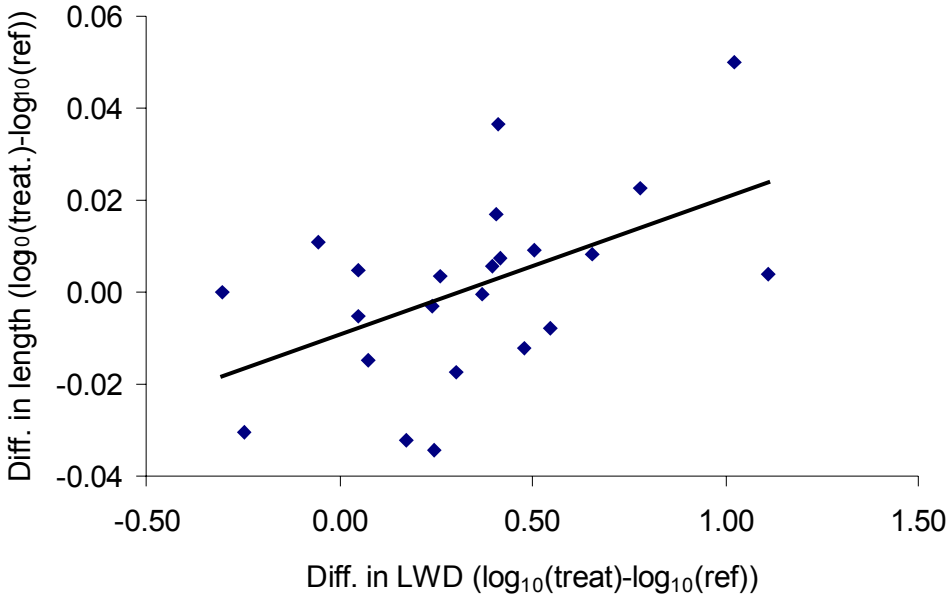


Figure 3.2. Relationship between difference in mean length of age 1+ reticulate sculpin and difference in large woody debris in the low flow channel. Difference in length = 0.03526x-.0122, $p < 0.01$, $r^2 = 0.28$.

was negatively correlated with difference in density for age 0 torrent sculpin ($p = 0.019$, $r^2 = 0.33$). However, one outlier existed and when this outlier was removed, no significant relationship was detected ($p = 0.17$).

Community Response

Community measures (species richness and diversity) did not differ between treatment and reference reaches (paired t-test; species richness: $p = 0.38$; species diversity (Berger-Parker dominance index): $p = 0.99$). Dominance calculated using the two and three most dominant species also indicated no significant differences between treatment and reference reaches ($p = 0.05$ and $p = 0.23$, respectively).

Discussion

Stream restoration efforts in the Pacific Northwest have focused on salmonid populations (Reeves et al. 1991a) and have either ignored nonsalmonid fishes or assumed that restoration activities had no effect on their abundance. After examining 30 LWD projects I found no significant difference in densities or length of larval lamprey, giant salamanders, and sculpin or diversity (richness and dominance) between treated and reference reaches. This supports the contention that LWD projects have little effect on nonsalmonid fishes. However, a number of factors may also explain the lack of response observed for these species, including habitat preferences of various species, sample size, low species diversity, or failure to measure physical habitat variables important to each species.

Pool dwelling species such as juvenile coho salmon and, to a lesser extent, cutthroat trout respond positively to LWD placement during summer (House and Boehne 1985, House et al. 1989, House 1996, also Chapter 2) and winter (Cederholm et al. 1997; Chapter 2 this study). Juvenile lamprey tend to occupy pools and other slow water habitats (Wydowski and Whitney 1979, Scott and Crossman 1998). Age 0 and age 1+ reticulate sculpin also tend to occupy pools, particularly in the presence of torrent sculpin (Bond 1963, Finger 1982, also Chapter 4). Thus it is somewhat surprising that a positive response to LWD placement was not observed for lamprey or reticulate sculpin given the significant increase in pool area between treated and control reaches. However, the relatively small sample sizes (18 sites contained lamprey and 23 contained reticulate sculpin) and conservative Bonferroni correction (alpha = 0.01 for individual t-tests) reduced the power to detect differences. For example, based on the observed variance in response ($\log_{10}$ treatment – $\log_{10}$ reference) among sites, a 0.01 level of significance and a power level of 0.80, a sample size of 42 paired stream reaches would be needed to detect a difference of 0.50 lamprey/m. Had a less conservative level of significance been used (i.e. greater than 0.025 for each t-test), a positive response would have been reported for both lamprey and reticulate sculpin.

Lamprey response among streams was, however, positively correlated with pool-forming LWD. Woody debris traps fine sediment, reduces velocity and forms pools (Harmon 1986, Bisson et al. 1987). Thus a positive relationship between wood and larval lamprey was predicted, given their affinity for pools and fine sediment. Surprisingly, no such relationship existed for reticulate sculpin, another species known to inhabit pools. However, difference in mean length of age 1+ reticulate sculpin (treatment – reference)

was positively correlated with difference in LWD in the low flow channel. Thus large increases in LWD levels within the active stream channel may increase growth of reticulate sculpin or provide better habitat for larger and older individuals. Presumably this would be due to improved feeding opportunities or shelter from predation.

Torrent sculpin and age 0 giant salamanders occupy riffles and fast water habitats (Wydowski and Whitney 1979; Kelsey 1995, also Chapter 4. Reduction in riffle area was negatively correlated with steelhead response to restoration (Chapter 2, but no relationship was detected for torrent sculpin or Pacific giant salamanders. Large woody debris placement did not decrease riffle area between treatment and reference reaches (Chapter 2, so the lack of response of torrent sculpin and Pacific giant salamanders was expected. Age 1+ giant salamanders occupy pools rather than riffles (Kelsey 1995). It was not possible to separate giant salamanders into age classes on the basis of length because so few were captured in each stream. Because young of the year and older salamanders may have very different responses to LWD placement, combining them may have masked numerical responses. In addition, giant salamanders inhabit streams with slopes ranging from 1% to more than 15% and my results may have differed had I sampled steeper streams.

Multiple-removal electrofishing is a reliable method for estimating juvenile salmonid abundance (Hankin and Reeves 1988), but is less reliable for benthic species, particularly lamprey (Pajos and Weise 1994). The number of lamprey was difficult to estimate using multiple-removal electrofishing because they burrow deep into fine sediment. Because of low capture efficiency for lamprey, the total catch of three passes was used as an estimate of lamprey abundance. However, if this method is not an

accurate index of lamprey abundance then the ability to detect responses was affected. Electrofishing equipment specifically designed to capture sea lamprey ammocoetes has been developed for the Great Lakes (Weisser and Klar 1990), but was not employed in this study. Lamprey densities are also affected by depth, subsurface substrate size, and stream size (Kelso 1993; Kelso and Todd 1993). However, I did not quantify subsurface sediment size and found no relationship between stream size and lamprey response to restoration. Additional work is needed to accurately estimate lamprey densities and examine habitat factors that may be important in influencing their response to LWD placement and habitat restoration.

Wood placement had no detectable effect on species richness or dominance. This is not altogether surprising given the relatively simple fish community in coastal Pacific Northwest streams (McPhail and Lindsey 1986), though Reeves et al. (1993) found differences in salmonid species dominance in streams with different levels of timber harvest. The small streams I sampled were generally inhabited by four or five species including two or three salmonids. Wood placement increased pool area and habitat complexity, but probably had little effect on overall water quality or stream productivity particularly because the LWD placement occurred in relatively short (< 1km) stream reaches. In addition, all streams were in second or third growth forests. Therefore, little response in species richness or dominance would be expected unless reference reaches were totally devoid of pools or differed markedly in canopy cover and water quality. Chen (1998) found no significant relationship between wood density and salmonid diversity and dominance at either a pool or reach level in southwest Oregon streams, though diversity and dominance were weakly correlated with pool depth and volume.

Chen further suggested that salmonid diversity was not sensitive to habitat structure at a reach scale. The results of the present study on fish and salamander diversity support the findings of Chen (1998) and suggest that species richness and dominance are not sensitive to changes in habitat structure at a reach scale.

In summary, LWD placement had little effect on density, species richness or diversity of nonsalmonid fishes and salamanders. Lamprey and age 1+ reticulate sculpin appeared to increase in density with the largest changes in LWD. Additional monitoring should be conducted on reticulate sculpin and lamprey to further elucidate their response to artificially placed LWD. Little is known about the competitive interactions of most of the species I examined and additional experiments on competition under different habitat manipulations is also needed. In general, however, the benefits of LWD projects for coho salmon (typically the species for which they are designed) do not seem to come at a cost to the rest of the vertebrate community.

CHAPTER 4: HABITAT USE BY FISHES AND PACIFIC GIANT SALAMANDERS IN SMALL WESTERN WASHINGTON AND OREGON STREAMS

Introduction

Differences in habitat use and partitioning among stream fishes are fundamental to understanding the effects of both natural and anthropogenic disturbance on fish distribution and abundance. In particular, the habitat preferences of many salmonids have been examined in North American streams in an effort to understand the effects of land-use (e.g., forestry activities, road building, hydropower development) and restoration activities (e.g., riparian, instream and off-channel habitat manipulations) on salmonid abundance and survival (Meehan 1991; Cunjak 1996). Forestry and other land-use activities often affect large woody debris abundance, fine sediment deposition, pool frequency, size and depth; habitat complexity and other changes in physical habitat (Bisson et al. 1987, Bilby and Ward 1991, Ralph et al. 1994). These physical changes have been linked to changes in vertebrate (fish and amphibian) abundance and community structure, particularly for salmonids such as juvenile coho salmon (*Oncorhynchus kisutch*) and cutthroat trout (*O. clarki*; Murphy and Hall 1981, Murphy et al. 1986, Reeves et al. 1993) and Atlantic salmon (*Salmo salar*; Grant et al. 1986). Moreover, changes in physical habitat alter the suitability of these streams for different members of the fish community and may increase the competition between endemic and exotic species (Moyle 1994).

Examination of fish and salamander densities at a microhabitat scale is useful in understanding species distributions within stream reaches. Numerous papers examine salmonid habitat use at a microhabitat scale (e.g., Bustard and Narver 1975a,1975b; Bisson et al. 1982; Heifetz et al. 1986; Nickelson et al. 1992b and others). Several of these studies have correlated woody debris, substrate size and other physical variables with densities of salmonids within individual habitats (e.g., Bustard and Narver 1975a; 1975b, Heifetz et al. 1986; Swales et al. 1989). Others have correlated differences in reach-scale fish or salamander densities with physical differences among streams or paired logged and unlogged reaches (e.g., Murphy et al. 1981, Hawkins et al. 1983, Corn and Bury 1989, Kelsey 1995, Rosenfeld et al. 2000). Rosenfeld et al. (2000) examined densities of cutthroat trout in 119 stream reaches and found that reach-scale physical variables were better predictors of density than those collected within individual habitats. Dunham and Vinyard (1997) demonstrated that examining fish habitat relationships at only one scale can lead to erroneous conclusions. These studies demonstrate that to examine and compare habitat associations of fish and other aquatic biota, collection and analysis of habitat use data are needed at both a microhabitat and reach scale.

Restoration activities seek to bring habitats from a degraded state back to some predisturbance condition through increased habitat complexity and creation of pools. The creation of pool habitat has been successful in increasing the abundance of certain species such as juvenile coho salmon that show strong preferences for pools and other slow water habitats (House and Boehne 1985, House 1996, Cederholm et al. 1997, Solazzi et al. 2000). Habitat preferences and densities of coho salmon in natural habitats are similar to those in artificially created habitats (Cederholm and Scarlett 1991; Nickelson et al.

1992a). Habitat modifications designed to benefit one species may affect other species (Minns et al. 1996), including non-salmonid fishes and other vertebrates in small streams, especially salamanders. Unfortunately, studies of habitat use and preference are often limited to salmonids or a few members of the fish community (e.g., Hartman 1965; Bisson et al. 1982; 1988, Nickelson et al. 1992b), a small geographic area (one stream, basin or province; e.g., Bustard and Narver 1975a, Murphy et al. 1984, Swales et al. 1986, Heifetz et al. 1986) and to summer low-flow periods (e.g., Bisson et al. 1982; 1988). While many studies have examined juvenile salmonid habitat use in western North American streams, little information exists on habitat use for other members of the vertebrate community in these streams such as sculpin (*Cottus spp.*), lamprey (*Lampetra and Entosphenus spp.*) or Pacific giant salamanders (*Dicamptodon tenebrosus*). Cottids are small resident benthic fishes that inhabit both streams and lakes and prey primarily upon invertebrates, but may also prey on other fishes as they grow larger (Wydowski and Whitney 1979; Scott and Crossman 1998). Pacific giant salamanders are also benthic predators that inhabits small streams and prey upon both vertebrates and invertebrates (Kelsey 1995). Similar to many salmonids (e.g., coho, cutthroat and steelhead), Pacific (*Entosphenus tridentatus*) and river lamprey (*Lampetra ayresi*) are anadromous, and juveniles (larvae or ammocoetes) may spend 5 or more years in freshwater before emigrating to sea. They are filter feeders that inhabit fine sediment in backwaters and slow areas in streams (Wydowski and Whitney 1979). Brook lamprey have similar habitat preferences to other lamprey, but are resident rather than anadromous. While information on the general habitat preferences for these species is available, it is unclear how their habitat use differs by size and age and what effect other habitat variables and

interspecific competition may have on their distribution within stream channels. Presumably, larger and more fit individuals will inhabit optimal habitats. For example, there is evidence that larger Pacific giant salamanders prefer pools (Kelsey 1995). Similarly, there is evidence that interspecific competition may influence habitat use between reticulate (*Cottus perplexus*) and torrent sculpin (*C. rhotheus*; Finger 1982). Furthermore, a thorough understanding of the seasonal habitat preferences of the entire fish community is needed to further elucidate the effects of natural disturbance, land-use and restoration activities on the vertebrate community.

The objectives of this study were to examine patterns of size and density of fishes and salamanders at both a microhabitat (i.e., pools and riffles) and reach scale. Specifically, I tested the following hypotheses for each species: 1) densities among habitat types were similar; 2) length and morphology (weight-length relationships) were similar between pools and riffles; 3) densities within microhabitats (pools) were not correlated with physical variables (e.g., LWD, substrate, size, etc.); and 4) densities and size of fish and salamanders among streams (reaches) were not correlated with physical variables measured at a reach (e.g., LWD, gradient) or watershed scale (e.g., elevation, drainage area, precipitation).

Methods

Thirty streams in western Washington and northwest Oregon were sampled during summer and winter between August 1996 and April 1999 to estimate fish abundance and habitat use at both reach and habitat unit scales (Chapter 2, Figure 2.1).

Streams and sites were selected as part of a larger study evaluating stream restoration projects in western Washington and Oregon. Two study reaches 75 to 120 m long and at least 10 times the local bankfull width were selected in each stream. Each stream was surveyed once in summer (August - September) and winter (January - March) and reaches within a stream were sampled on the same day.

The selection of study streams and reaches was based in part on physical and biological stream characteristics including stream size, bankfull width, channel type and fish species composition. These sites and reaches were selected as part of a larger study evaluating habitat enhancement (LWD placement) in western Washington and Oregon (see Chapter 2). The study streams ranged from 4 m to 12 m in bankfull width and 0.5 % to 4.2% slope. Annual precipitation varied from and 107 cm to 315 cm. Dominant forest types were primarily Douglas fir (*Pseudotsuga menziesii*), Sitka spruce (*Picea sichensis*), and western hemlock (*Tsuga heterophylla*). The dominant watershed geology was volcanic, sedimentary or glacial-alluvial and varied by site, but was consistent for reaches within a stream (Chap. 2, Table 2.1). The elevations of the study sites ranged from 12 m to 789 m and drainage area upstream of our study reaches ranged from 124 ha to 2388 ha.

Habitat units within each stream reach were classified using a modification of the methods and habitat types described by Bisson et al. (1982; Table 4.1). I used minimum residual depth and pool area to consistently identify pool habitats (Lisle 1987; Schuette-Hames et al. 1994). Total surface area of each habitat was estimated by measuring the total length and multiplying by the average of three to five width measurements. The gradient of each reach and individual habitat unit was measured using a hand level, survey (stadia) rod, and tape measure. Habitat-specific stream slope (gradient) was used

Table 4.1. Stream habitat types as modified from Bisson et al. (1982) and Nickelson et. al. (1992b).

Slow Water Habitats	
Dam Pool	A pool impounded upstream from a complete or nearly complete channel blockage often cause by a debris jam or beaver pond.
Backwater Pool	An eddy or slack water along the channel margin separated from the main channel margin by a gravel bar or small channel obstruction.
Scour Pool	A scoured basin or depression either (1) near the channel margin caused by flow being directed to one side of the stream by a partial channel obstruction, (2) near the center of the channel usually caused by a channel constriction or high gradient riffle or cascade, or (3) a slot or trench in a stable substrate such as bedrock or clay.
Plunge Pool	A basin or depression scoured by a vertical drop over a channel obstruction.
Glide	A moderately shallow reach with an even, laminar flow and no pronounced turbulence or obstructions.
Fast Water Habitats	
Riffle	a) Low Gradient - A shallow reach with a moderate current velocity, moderate turbulence, and a gradient of # 2%.
	b) High Gradient - A shallow reach with a moderate current velocity, moderate turbulence, and a gradient between 2% and 4%.
Cascade	A shallow reach with high current velocity and considerable turbulence with a gradient of > 4% or a series of small steps of alternating small waterfalls and small pools with a gradient > 4%

to distinguish between riffles and cascades. Discharge was estimated with a flow meter immediately following each survey. All pieces of natural and artificially placed large woody debris (LWD) within the wetted stream channel greater than 10 cm in diameter and 1.5 m long were inventoried. The diameter class (small: 10-20 cm, medium: 20-50 cm, and large: > 50 cm) and approximate length were recorded. In addition, the percent of each piece of LWD that was in the low-flow wetted channel and within the bankfull channel was determined (Robison and Beschta 1990).

Multiple-removal elecrofishing was used in summer to estimate fish abundance within each habitat unit (Carle and Strub 1978). Each habitat was sampled separately by placing 3.2 mm mesh blocknets at the upstream and downstream boundaries to prevent immigration or emigration during sampling. Three removals were made through each habitat and fourth was made if a 50% or greater reduction in fish numbers was not seen between the second and third passes. Each electrofishing removal consisted of one upstream and one downstream pass using a pulsed DC electrofisher. All fish captured were anesthetized with tricaine methanesulfonate (MS-222), identified, measured to the nearest mm and then released. Fish species commonly captured included juvenile coho salmon, cutthroat trout, steelhead, adult and juvenile torrent sculpin, reticulate sculpin, Pacific and Cope's giant salamander (*Dicamptodon copei*), juvenile Pacific, river and occasionally brook lamprey (*Lampetra richarsoni;* Chapter 3; Table 3.2). In addition, coast range sculpin (*Cottus aleuticus*) and juvenile longnose sucker (*Catostomus catostomus*) were each present in one stream but were not used in the analysis. Based on length frequency distributions (Jearld 1983), all steelhead or cutthroat trout greater than 60 mm during summer or 80 mm during winter were considered age 1+. All trout less

than these lengths were considered fry (age 0 or young-of-year). Such fry could not be reliably identified to species and so were simply called trout. Similarly, length frequency distributions for torrent and reticulate sculpins indicated that age 0 sculpins were less than 60mm and larger sculpins were age 1+. I could not reliably distinguish between Cope's giant salamander and Pacific giant salamander and there is some disagreement as to whether they are two separate species (Corkran and Thomas 1996). Therefore, I refer to them collectively as Pacific giant salamanders. Similarly, I could not reliably distinguish between larval (ammocoetes) Pacific, river, and brook lamprey and refer to them collectively as larval lamprey.

High flows during winter months often precluded the use of multiple-removal electrofishing so night snorkel surveys were used to estimate juvenile salmonid abundance during winter. Winter night snorkeling was nearly as accurate as multiple-removal electrofishing and suitable for a wider range of conditions (Roni and Fayram 2000). Juvenile salmonids emerge from concealment 30-60 min. after sunset at temperatures below 8-9°C (Campbell and Neuner 1985; Contor and Griffith 1995). Therefore, snorkeling began at least 1 h after sunset, and only on nights with either complete cloud cover or no visible moonlight to assure that natural light levels were consistently low during night snorkel surveys. Sculpins, salamanders and lamprey were rarely observed during snorkel surveys and accurate counts could only be obtained for salmonids during winter. Surveys were conducted in 1997 during February and March in Washington and January to mid-March of 1998 in Oregon to assure that sampling occurred prior to the outmigration of salmonid smolts.

One diver entered the habitat from the downstream end and slowly moved upstream, stopping occasionally to relay the number, sizes, and species of fish observed to a second individual on the bank. In streams greater than 10 m wide, two snorkelers worked side by side to cover the entire width of the stream. A halogen dive light was used to illuminate areas and identify fish. Fish length was visually estimated to the nearest 10 mm using a ruler attached to a dive glove. Water temperature and flow were measured downstream of each site before electrofishing and snorkeling. Discharge and temperature among streams ranged from 0.1 to 0.15 $m^3 \cdot s^{-1}$ and 8 – 16°C during electrofishing surveys, and 0.14 and 2.1 $m^3 \cdot s^{-1}$ and 8 – 16°C during snorkel surveys.

Box plots with median and the interquartile range were used to examine differences in species specific densities among the eight habitat types. To assure that low numbers of spawning adult salmonids or rare occurrences did not bias examination of habitat preferences, only those sites where 10 or more of a given species were captured were used to examine habitat preferences. Because certain habitat types were rare (e.g., backwater pools, dam pools, glides) and classification of different pool types subjective, statistical comparisons of fish and amphibian densities and length were made only between pools and riffles (riffles and cascades combined). Glides were excluded from statistical analysis because they were rare and represent an intermediate habitat type between riffles and pools. Paired t-tests were used to compare densities and mean lengths of fish in pools and riffles (paired by stream). A Bonferroni correction was applied to compensate for the increased likelihood of finding a significant difference when performing multiple t-tests (SPSS 1999). A family alpha level of 0.05 was used to

determine significance and divided by the number of tests performed on each separate data set (summer and winter). However, in some instances this resulted in a significance level of less than 0.01 for each individual t-test, which may be overly conservative. In these instances, a minimum alpha level of 0.01 was used for each individual t-test. Step-wise multiple regression was used to examine the relationship between physical habitat variables and individual pool densities and reach-scale densities. Fish densities within individual pools were $\log_{10}(x + 0.01)$ transformed for regression analysis to normalize residuals. Physical variables in individual pools included slope (pool water surface), LWD counts, residual depth, percent wood cover, and substrate type (categorized as silt, sand, gravel, cobble, boulder, and bedrock). Reach-scale variables included drainage area (upstream of the site), average annual precipitation, stream gradient, LWD counts, site elevation, and percent pool area.

Results

The frequency of the eight habitat types differed among streams; backwater pools, dam pools and glides were relatively scarce. The mean gradients of pools and riffles differed, though riffles were categorized by slope (Table 4.2). Box plots of fish and salamander densities from the 30 streams indicated that habitat use differed by season (salmonids only; Figures 4.1 and 4.3) and among species (Figures 4.1 and 4.2).

Salmonids - summer

Summer coho salmon densities were highest in backwater, dam, and scour pools and, to a lesser extent, glides. Cutthroat trout were most dense in scour and plunge pools

Table 4.2. Average gradient (slope) for each habitat type during summer and winter all streams combined.

Habitat Type	n	Slope (%) Mean	Variance
	Summer		
Dam pool	10	0.08	0.02
Backwater pool	31	0.13	0.07
Plunge pool	38	0.17	0.07
Scour pool	253	0.21	0.19
Glide	33	0.31	0.17
Low gradient riffle	98	1.09	0.29
High gradient riffle	99	2.85	0.27
Cascade	90	7.08	19.83
	Winter		
Dam pool	16	0.37	0.08
Backwater pool	37	0.16	0.07
Plunge pool	52	0.23	0.18
Scour pool	16	0.27	0.06
Glide	279	0.44	0.20
Low gradient riffle	125	1.15	0.25
High gradient riffle	97	2.86	0.37
Cascade	90	6.95	12.28

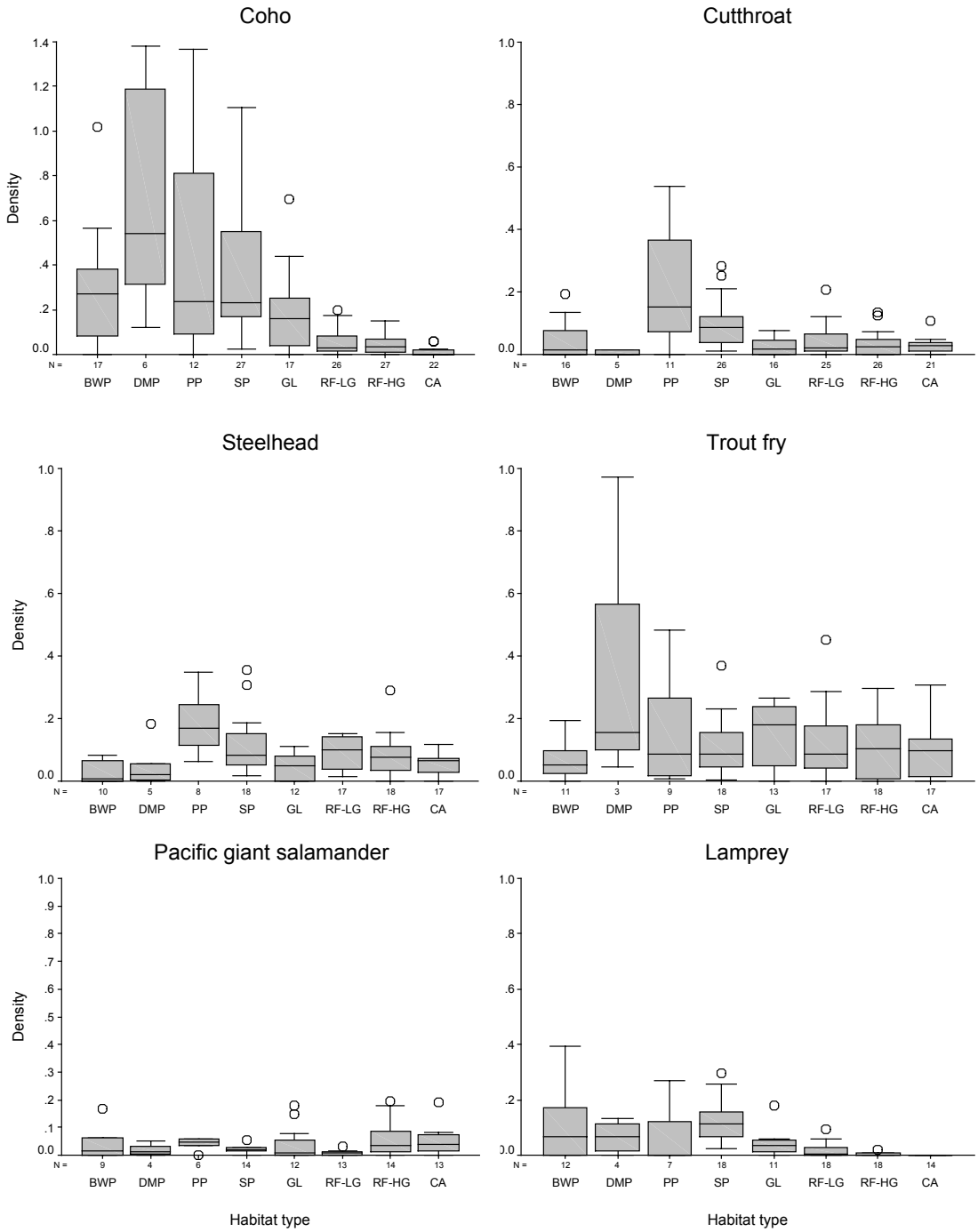


Figure 4.1. Summer densities (fish·m^{-2}) of salmonids, lamprey and Pacific giant salamanders for each of eight habitat types. Densities are displayed in box plots with median and interquartile range. Open circles represent outliers. N represents the number of streams with that habitat type present. DMP = dam pool, BWP = backwater pool, PP = plunge pool, SP = scour pool, RF - LG = low gradient riffle, RF-HG = high gradient riffle, CA = cascade.

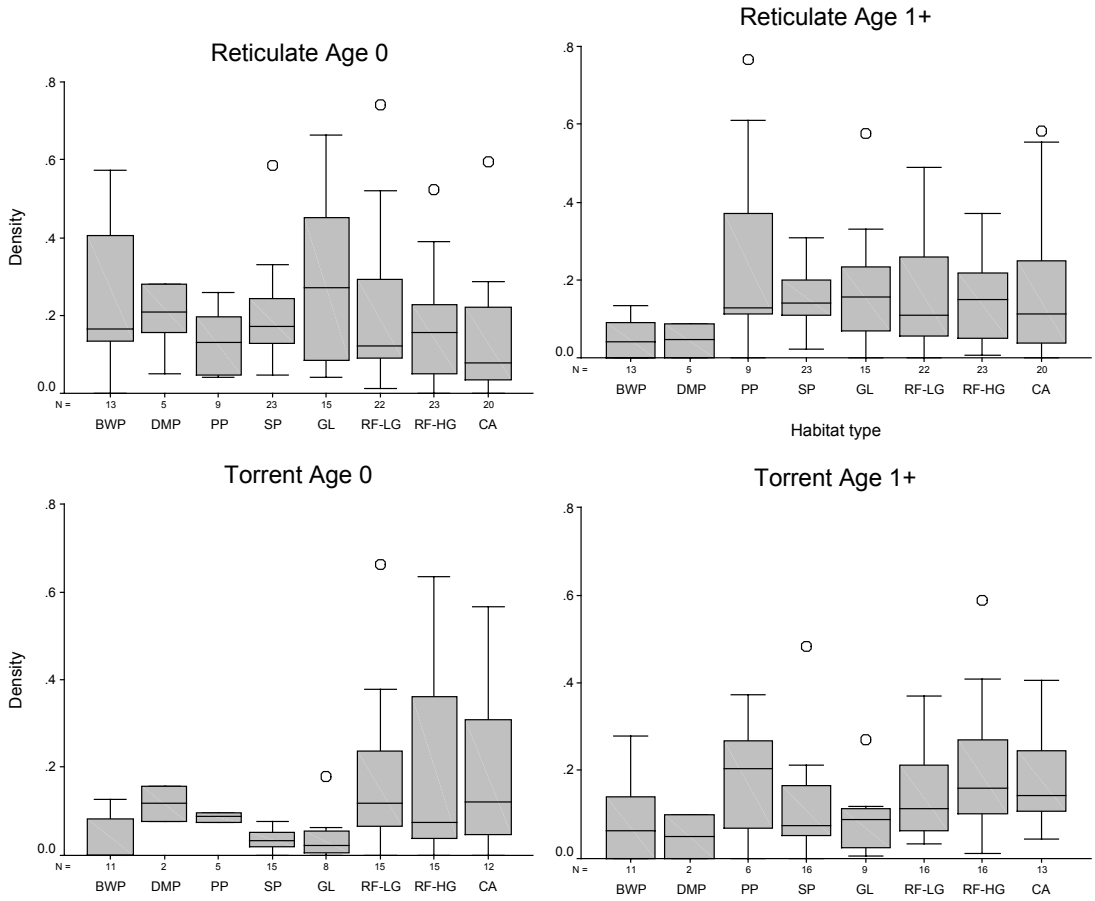


Figure 4.2. Summer densities (fish·m^{-2}) of age 0 and age 1+ torrent and reticulate sculpin. Densities are displayed in box plots with median and interquartile range. Only those streams were at least 10 fish of a given species were captured were included in analysis. Open circles represent outliers. N represents the number of streams with that habitat type present. DMP = dam pool, BWP = backwater pool, PP = plunge pool, SP = scour pool, RF - LG = low gradient riffle, RF-HG = high gradient riffle, CA = cascade.

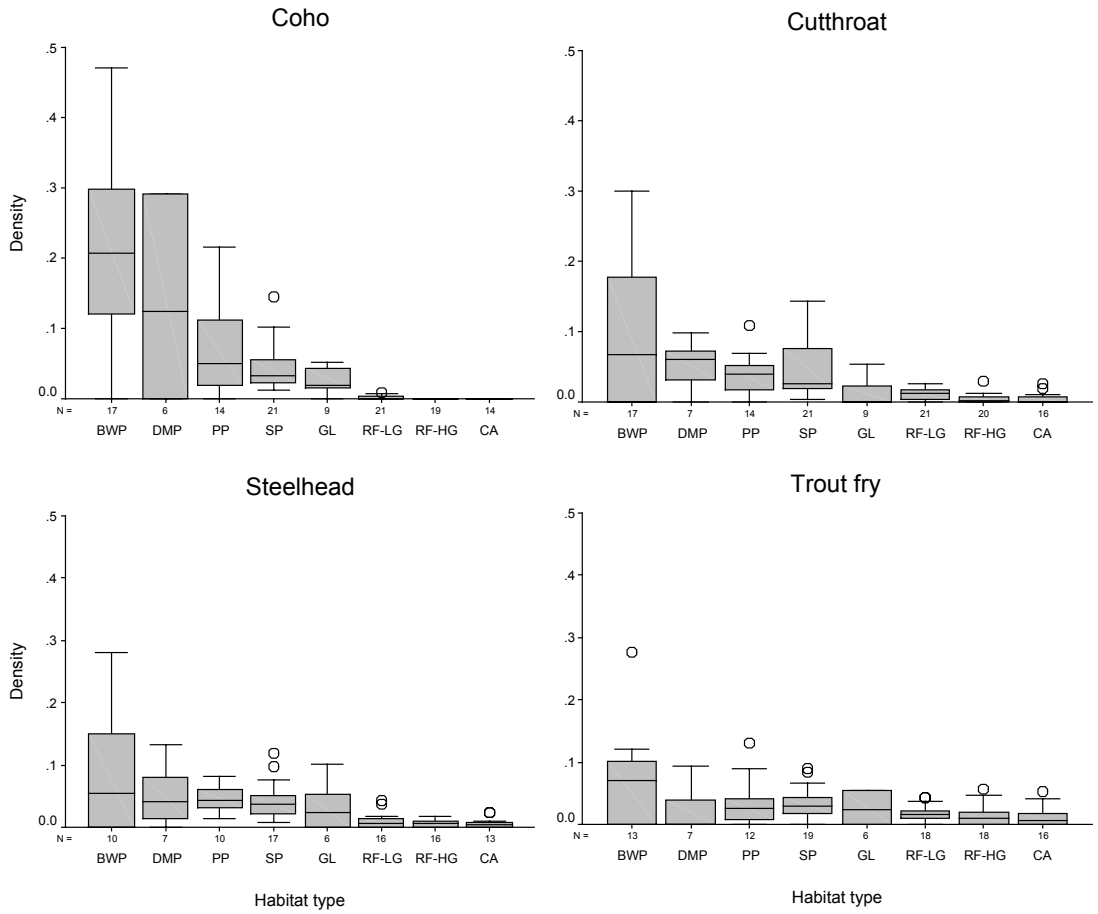


Figure 4.3. Winter densities (fish·m^{-2}) of salmonids for each of eight habitat types. Densities are displayed box plots with median and interquartile range. Only those streams were at least 10 fish of a given species were captured were included in analysis. Open circles represent outliers. N represents the number of streams with that habitat type present. DMP = dam pool, BWP = backwater pool, PP = plunge pool, SP = scour pool, RF - LG = low gradient riffle, RF-HG = high gradient riffle, CA = cascade.

(Figure 4.1). Steelhead did not demonstrate strong habitat preferences, though the highest densities were found in plunge pools, scour pools and low gradient riffles. Trout fry were most dense in dam pools and backwater pools, two relatively rare habitat types, but no other differential habitat use patterns were apparent. When the four pool types and three riffle types were classified as simply pools and riffles, summer densities of coho and cutthroat trout were significantly higher in pools ($p < 0.01$; Table 4.3). However, no difference were detected in densities of steelhead ($p = 0.29$) or trout fry ($p = .83$).

LWD was positively correlated with summer coho densities in pools ($p < 0.01$, $r^2 = 0.03$), but no other physical variables. Dominant substrate type and wood cover were positively correlated with cutthroat summer densities ($p < 0.01$; $r^2 = 0.05$). When examined at a stream reach level, coho summer densities were positively correlated with both site elevation and pieces of LWD in the low flow channel (multiple regression; $p < 0.01$; $r^2 = 0.37$; Table 4.4). Summer cutthroat trout densities were negatively correlated with drainage area ($p = 0.01$; $r^2 = 0.22$). Site elevation, drainage area, and annual precipitation explained 32% of the variation among streams for steelhead during summer ($p = 0.03$; $r^2 = 0.32$). Trout fry densities were negatively correlated with average annual precipitation and percent pool area ($p < 0.01$; $r^2 = 0.41$) during summer. Both salmonid density and total fish densities (all species combined) were positively correlated with site elevation ($p < 0.01$; $r^2 = 0.38$ and 0.37, for salmonids and all species combined, respectively).

Table 4.3. Average densities (fish·m^{-2}) and standard error (SE) of fish and salamanders from pools and riffles for those streams where species were present. Sample size (n) represents the number streams used in analysis and inhabited by a species.

Species		Pool	SE	Riffle	SE	n	p-value
Summer							
Coho		0.404	0.072	0.076	0.026	27	0.000
Cutthroat		0.113	0.022	0.041	0.009	26	0.000
Steelhead		0.104	0.019	0.089	0.023	18	0.294
Trout Fry		0.112	0.025	0.114	0.023	18	0.833
Pacific giant salamander		0.029	0.007	0.033	0.009	14	0.507
Larval lamprey		0.127	0.019	0.010	0.003	18	0.000
Reticulate sculpin	Age 0	0.194	0.024	0.186	0.038	23	0.858
	Age 1+	0.165	0.024	0.152	0.025	23	0.478
Torrent sculpin	Age 0	0.071	0.021	0.254	0.064	14	0.007
	Age 1+	0.147	0.038	0.195	0.038	15	0.267
Winter							
Coho		0.054	0.009	0.002	0.001	24	0.000
Cutthroat		0.038	0.008	0.007	0.001	25	0.000
Steelhead		0.041	0.007	0.007	0.002	20	0.000
Trout Fry		0.035	0.006	0.014	0.002	21	0.000

Table 4.4. Multiple regression equations for species density (fish·m^{-1}) and physical variables. Torrent sculpin density was not significantly correlated with any physical variables. Elev. = site elevation in meters, LWD = total pieces of large woody debris interacting with low flow channel, area = drainage area in hectares, %pool = percent pool area, and precip. = annual precipitation in mm, r^2 is adjusted r^2.

Species	Equation (density = ax+ by +...c)	p	r^2
	Summer		
Coho	= (0.00108*elev.) +(0.006589*LWD) - 0.230	0.00	0.37
Cutthroat	= (-0.0000566*area) + 0.129	0.01	0.22
Steelhead	= (0.0002195*elev.) + (0.0000625*area) + (0.001083*precip.) - 0.305	0.03	0.32
Trout fry	= (-0.000693*precip) - (0.256*%pool) + 0.412	0.01	0.41
Giant salamander	= (-0.00434*BFW) + (0.01872*slope) + 0.03605	0.00	0.63
Lamprey	= (0.0000346*area + 0.175*% pool - 0.0543	0.02	0.33
Reticulate sculpin			
Age 0	= (-0.0000871*areas) - (0.00180* precip.) + 0.75	0.00	0.43
Age 1+	= (-0.0000616*area) - (0.000628*precip.) - (0.401*%pool) + 0.620	0.01	0.36
All species	= (0.02417*elev.) - 3.168	0.00	0.38
All salmonids	= (0.6147*elev.) - 9.927	0.00	0.37
	Winter		
Coho	= (0.00007969*elev.) + 0.00000874*area - 0.00469	0.00	0.36
Cutthroat	= (-0.0001197*precip) - (0.00497*BFW) + 0.03009	0.01	0.29
Steelhead	= (-0.0372*%pool) + (0.00001093*area) + 0.0274	0.03	0.29

Cutthroat trout, steelhead, Pacific giant salamanders, and torrent sculpin found in pools were all significantly longer (paired t-test, $p < 0.009$) than those found in riffles (Table 4.5). In addition, weight-length regressions differed between pools and riffles for steelhead ($p < 0.01$) and trout fry ($p < 0.01$; Table 4.6). Steelhead were heavier for a given length in pools compared to riffles, while trout fry were lighter. Mean length and density were positively correlated for coho salmon ($p < 0.01$; $r^2 = 0.43$) and cutthroat trout ($p < 0.01$; $r^2 = 0.30$), but not for steelhead ($p = 0.42$) or trout fry ($p = 0.07$). Mean coho length was negatively correlated with elevation ($p < 0.01$; $r^2 = 0.30$; Y = -0.0358*elevation + 75.908). Drainage area explained 43% of the variation in cutthroat trout length among sites ($p < 0.01$; $r^2 = 43$; Y=-0.0267*area+91.718) and precipitation explained 25% of the variation in steelhead length ($p = 0.02$; $r^2=0.25$; Y= -0.190*precipitation + 146.403). Density was not incorporated into length multiple regression model because of autocorrelation with significant physical variables.

Salmonids -Winter

During winter months coho salmon, cutthroat and steelhead, were found in highest densities in backwater and dam pools, and trout fry in backwater pools. Densities tended to decrease with increasing gradient of habitats (Figure 4.3). Coho salmon, cutthroat and steelhead and trout fry densities were significantly higher in pools than riffles ($P < 0.01$; Table 4.3) and cutthroat were significantly longer in pools than in riffles. The number of pieces of LWD, pool gradient (slope), and dominant substrate type explained 10 % of the variation in winter coho densities among pools (multiple regression, $p < 0.01$, $r^2 = 0.10$). Percent wood cover and residual depth explained 4% of

Table 4.5. Mean length (mm) of fish and salamanders from pools and riffles for those streams where species were present and length measurements of more than 3 fish in riffles and pools were recorded. An asterisk indicates significant difference ($p < 0.01$) and n represents the number of streams used in analysis. Summer lengths were from fish measured during electrofishing and winter data were from lengths estimated during snorkel surveys.

Species	Pool	Riffle	n	p-value
Summer				
Coho	68.1*	67.6	21	0.451
Cutthroat	114.2*	94.5	20	0.000
Steelhead	104.1*	90.5	16	0.000
Trout Fry	49.7	50.6	16	0.109
Pacific giant salamander	166.2*	110.7	12	0.000
Reticulate sculpin	60.6	59.8	23	0.583
Torrent sculpin	73.3*	64.3	16	0.009
Winter				
Coho	84.0*	NA[1]	25	NA[1]
Cutthroat	141.2*	117.8	14	0.000
Steelhead	124.9	116.7	12	0.076
Trout Fry	67.6	67.2	19	0.689

[1] coho were found in few riffles during winter

the variation in trout fry densities among pools during winter ($p < 0.01$; $r^2 = 0.04$), and percent wood cover and substrate size were positively correlated with winter cutthroat trout densities ($p < 0.01$; $r^2 = 0.06$) and steelhead densities ($p < 0.01$; $r^2 = 0.07$) in pools. Reach-level coho density were positively correlated with elevation and drainage area ($p < 0.01$; $r^2 = 0.36$) and winter cutthroat densities were negatively correlated with precipitation and bankfull width ($p = 0.01$; $r^2 = 0.29$). During winter, drainage area was positively correlated and percent pool area negatively correlated with steelhead densities ($p = 0.03$; $r^2 = 0.29$). Reach-scale trout fry densities were not significantly correlated to any variables during winter ($p > 0.10$).

Sculpin

Habitat specific densities did not differ substantially between age 0 and age 1+ reticulate or riffle sculpins (Figure 4.2). However, age 1+ reticulate sculpin densities were lowest in backwater and dam pools and both age classes were found at highest densities in glides. Densities for age 0 and 1+ torrent sculpins were highest in riffles and cascades. However, no significant differences in density existed between riffles and pools for Age 0 and age 1+ reticulate sculpin ($p = 0.478$ and $p = 0.108$) and age 1+ torrent sculpin ($p = 0.267$; Table 4.3). In contrast, age 0 torrent sculpin ($p = 0.007$) densities were higher in riffles than pools. Drainage area and precipitation were negatively correlated with age 0 reticulate sculpin densities ($p < 0.01$; $r^2 = 0.43$, while drainage area, precipitation and percent pool area were negatively correlated with age 1+ reticulate sculpin densities ($p < 0.01$; $r^2 = 0.36$; Table 4.4). Torrent sculpin densities were

not correlated with any single or combination of reach-scale physical variables ($p > 0.060$).

Torrent sculpin were significantly longer in pools than riffles ($p < 0.01$). Weight-length regressions differed between riffles and pools for both reticulate ($p < 0.01$) and torrent sculpin ($p < 0.01$; Table 4.6). Torrent and reticulate sculpin were heavier for a given length in riffles. No relationship existed between length and density or length and any single or combination of reach-scale physical variables for either age class of torrent or reticulate sculpin ($p > 0.09$).

Pacific giant salamanders

Giant salamanders showed a highly variable distribution among habitat types and appeared to prefer plunge pools and high gradient riffles and cascades (Figure 4.1), though no difference existed between densities in pools and riffles ($p = 0.51$). Giant salamander densities were negatively correlated with both stream gradient (slope) and bankfull width ($p < 0.01$; $r^2 = 0.63$; Table 4.4). Mean length was significantly higher in pools than riffles ($p < 0.01$; Table 4.5). No relationship existed between length and density ($p = 0.05$) or length and any physical variables ($p > 0.38$).

Lamprey

Lamprey densities were highest in four pool types and nearly zero in high gradient riffles and cascades (Figure 4.1). Densities were significantly higher in pools than riffles (paired t-test, $p < 0.01$; Table 4.3). Substrate size and LWD explained about

Table 4.6. Weight-length regression equations for different fish and salamanders. Equations reported separately in riffles and pools for those species where pool and riffle equations were significantly different. Basic equation is $\log_{10}(\text{weight}) = \log_{10}(\text{length})$ *slope – intercept; $p < 0.001$ for all weight length equations. Weight at 75 mm was predicted from equations.

				Equation		Weight at
Species	Habitat	n	r^2	Intercept	Slope	75 mm
Coho	combined	4122	0.97	-4.920	3.002	5.12
Cutthroat	combined	1355	0.99	-4.753	2.883	4.50
Steelhead[a]	pools	817	0.99	-4.811	2.962	5.53
	riffles	677	0.99	-4.867	2.959	4.80
Trout fry[b]	pools	515	0.92	-5.350	3.237	5.24
	riffles	694	0.94	-5.647	3.401	5.37
Giant salamander	combined	314	0.97	-5.057	2.947	2.94
Lamprey	combined	1238	0.96	-5.291	2.790	0.87
Reticulate sculpin[b]	pools	3302	0.97	-5.260	3.186	5.18
	riffles	2273	0.96	-5.148	3.129	5.24
Torrent sculpin[b]	pools	869	0.98	-5.184	3.152	5.32
	riffles	1267	0.98	-4.910	3.007	5.35

[a] intercepts of equations signficantly different

[b] slopes of regression equations signficantly different

6% of the variation lamprey densities among pools ($p < 0.01$, $r^2 = 0.06$). Lamprey densities among streams were positively correlated with a drainage area and percent pool habitat ($p = 0.19$; $r^2 = 0.33$; Table 4.4). Differences in lamprey size could not be compared between pools and riffles, because only two streams had more than three lamprey captured in riffles. Lamprey length among streams was negatively correlated with density ($p = 0.03$; $r^2 = 0.22$; Y= -116.445*density+119.845), but not any single or combination of physical variables ($p > 0.05$).

Discussion

Salmonids, sculpins, lamprey and Pacific giant salamanders demonstrated differences in both habitat use patterns among species as well as differences in size and condition factor among habitat types. The results of this study also indicate that they are in part a result of differences in physical variables within individual habitats and stream reaches. However, these differences may be the result of a number of factors that including inter- and intraspecific competition, differences in food availability and trophic ecology, and differences in growth and survival.

Salmonids

The results for juvenile salmonids supported previous findings that juvenile coho salmon tend to occupy a variety of pool types during summer and especially backwater pools during winter (Bustard and Narver 1975a, 1975b; Bisson et al. 1982, Heifetz et al. 1986; Nickelson et al. 1992b). Nickelson et al. (1992b) found the highest densities (~ 1.6 fish·m^{-2}) of juvenile coho salmon in alcoves, but this habitat type was not present in the

streams I sampled. Similarly, the results supported previous findings that age 1+ cutthroat trout occur at higher densities in pools than riffles during summer (Bisson et al. 1982; 1988; Rosenfeld et al. 2000) and show an increased preference for pools (backwater in particular) during winter (Bustard and Narver 1975b). Age 1+ steelhead showed a strong preference for pools during winter, but were found in similar densities in pools and riffles during summer. Bisson et al. (1982;1988) indicated that steelhead prefer riffles during summer months and Heiftez et al. (1986), Bustard and Narver (1975a) and Hartman (1965) found that steelhead prefer pools with abundant cover during winter. During winter, coho salmon, cutthroat and steelhead densities were highest in backwater pools, consistent with previous findings (e.g., Bustard and Narver 1975a;1975b, Heifetz et al. 1986, Nickelson et al. 1992b). Trout fry (age 0 cutthroat and steelhead) are generally found along stream margins during summer and winter (Hartman 1965, Moore and Gregory 1988; Bisson et al. 1988). I did not quantify lateral or edge habitats, but found no difference in trout fry densities between pools and riffle. However, during winter months trout fry densities were highest in backwater pools, which were often shallow, stillwater pools with a high proportion of edge habitat.

Densities of salmonids and other fishes in pools were only weakly correlated with physical variables (residual depth, substrate size, pool gradient, LWD, wood cover). Similarly, Nickelson et al. (1992b) found that maximum pool depth explained only 4% of the variation in juvenile coho density among pools in Oregon coastal streams. However, when salmonid densities were compared among streams, physical variables explained from 22 to 41% of the variation among streams. High elevation sites (streams) with high levels of LWD had the highest densities of coho salmon. LWD levels have been

positively correlated with coho densities (e.g., Murphy et al. 1985,1985; Bisson et al.1987; Murphy 1995). Wood was not positively correlated with reach-scale coho density in my study, but most sites had moderate to high levels of LWD, and LWD was positively correlated with density in individual pools. The positive relationship between coho density and elevation is surprising. However, high elevation streams were usually smaller, colder and often contained lower numbers of sculpin or other species. In addition, mean length of coho among streams was negatively correlated with elevation. Thus a combination of smaller size and reduced interspecific competition likely explain the higher densities of coho and other salmonids at higher elevations. Drainage area, precipitation, and elevation were positively correlated with steelhead summer densities, suggesting that steelhead were found in higher densities in larger streams and high elevation streams. Steelhead winter densities were positively correlated with drainage area and negatively correlated with percent pool area (Table 4.4). The negative correlation with percent pool area for steelhead during winter was unexpected, given this species' preference for pools during winter. In contrast, drainage area was negatively correlated with summer cutthroat densities and precipitation and bankfull width were negatively correlated with winter cutthroat densities. This suggests that smaller streams had higher densities of cutthroat during both summer and winter. Rosenfeld et al. (2000) found that cutthroat and coho densities were highest in small streams and that bankfull width explained more than 50% of variation in cutthroat densities in small coastal British Columbia streams. However, Rosenfeld et al. (2000) sampled a larger number of streams over a smaller geographic area than I did and this may explain the slight difference in predictive variables between their study and this one.

Interspecific competition is often thought to be one of the major reason salmonids such as coho salmon, cutthroat and steelhead trout segregate among habitats (e.g., Hartman 1965, Chapman 1966, Glova 1986, Sabo and Pauley 1997). Coho salmon may competitively exclude steelhead trout (Hartman 1965) or cutthroat trout from pools (Glova 1986; Sabo 1997). Bisson et al. (1988) suggested that morphological differences explain the variation in habitat preferences among juvenile steelhead, cutthroat and coho. Further, they suggested that steelhead, which are thinner than coho, are adapted for life in riffles rather than pools. It is unclear whether the habitat specific morphology of steelhead and trout fry resulted from occupancy of pools by deeper bodied fish or simply larger individual size (steelhead and cutthroat were also larger in pools than riffles) providing some competitive advantage that resulted in heavier body.

Sculpin

The habitat preferences of sculpins and other Pacific Northwest resident stream fishes have rarely been examined in detail (Lonzarich 1994). Torrent sculpin were found in higher density in riffles and cascades than pools but no difference in densities was detected between riffles and pools for reticulate sculpin. Wydowski and Whitney (1979) also reported that torrent sculpin generally prefer riffles and other fast water habitats whereas reticulate sculpin inhabit both pools and riffles. Bond (1963) and Finger (1982) indicated that torrent sculpin may exclude reticulate sculpin to pools when the two are sympatric. Lonzarich (1994) also found torrent sculpin primarily in riffles and glides and reticulate sculpin in glides and pools as well as size (age) based difference in habitat use of torrent sculpin. In contrast, Pasch and Lyford (1972) found no evidence for habitat

partitioning or competition between these two species. I examined differences in densities of allopatric populations of reticulated sculpin and populations which were sympatric with torrent sculpin. Densities of age 0 and age 1+ reticulate sculpin were slightly higher in riffles than pools in the absence of torrent sculpin and higher in pools in the presence of torrent sculpin (Figure 4.4). Furthermore, densities age 0 and age 1+ reticulate sculpin in riffles were higher in allopatric populations than sympatric populations ($p = 0.04$, and $p < 0.01$ for age 0 and 1+, respectively). Pool densities were higher in allopatric than sympatric populations of reticulate sculpin for age 0 ($p = 0.04$), but not age 1+ ($p < 0.01$). This suggests habitat partitioning and supports the findings of Bond (1963) and Finger (1982). I also found that age 1+ torrent sculpin were longer in pools than riffles, suggesting that larger or possibly older (age 2+) individuals may prefer pools or competitively exclude smaller individuals. Differences in length-weight relationships between fish in pools and riffles suggested differences in feeding opportunity or competition. Torrent and reticulate sculpin in riffles were heavier for a given length than those in pools, whereas steelhead and trout fry in pools were heavier for a given length than those in riffles. Macroinvertebrate production is higher in riffles than pools (Hawkins et al. 1983; Rosenfeld and Hudson 1997) and higher food availability may explain in part why benthic species, such as sculpin, were heavier in riffles. However, one might expect them to grow longer rather than become heavier. There may be some morphological advantage for sculpin inhabiting riffles to be deeper bodied similar to that suggested by Bisson et al. (1988) for juvenile salmonids.

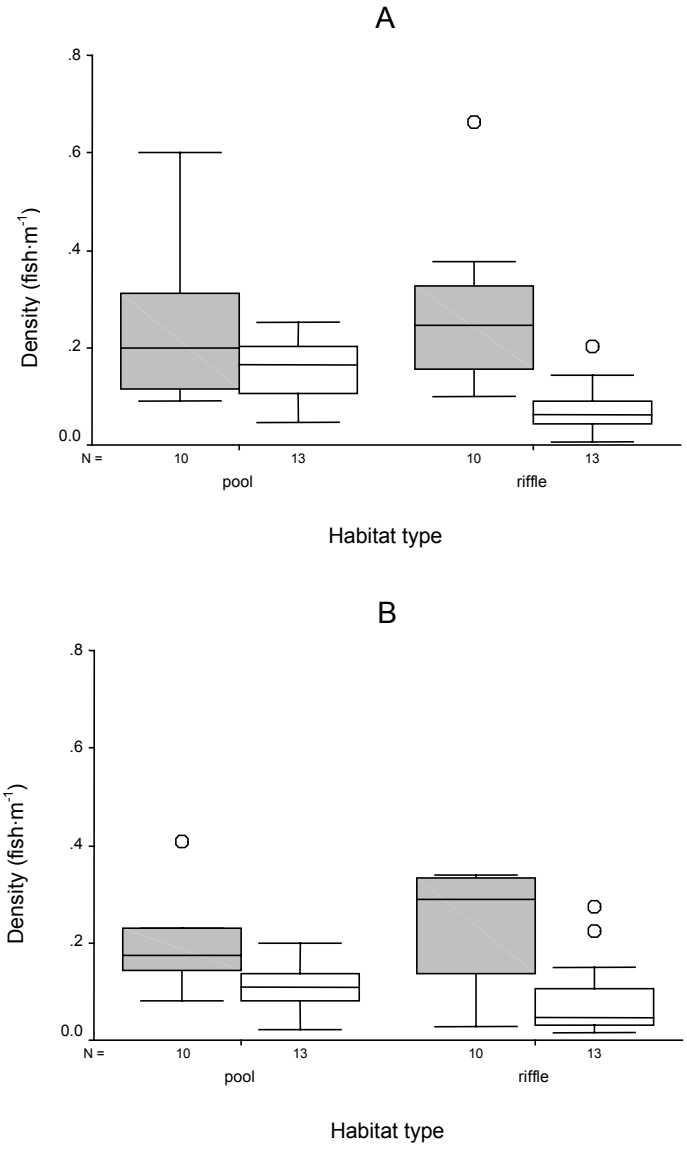


Figure 4.4. Densities in pools and riffles of allopatric populations (shaded box plots) of age 0 (A) and age 1+ (B) reticulated sculpin and when they occurred sympatrically (white or open box plots) with torrent sculpin. N represents the number of streams with sympatric or allopatric populations of reticulate sculpin.

Physical variables measured in individual habitats had little influence on sculpin densities. However, reticulate sculpin densities were negatively correlated with drainage area and precipitation, suggesting that small streams held the highest densities of reticulate sculpin. However, unlike salmonids, density had little effect on sculpin size, which suggests that food may not be limiting growth of reticulate sculpin. It is unclear why torrent sculpin densities were not correlated with any physical variables at the reach scale. Other factors such as substrate size or embeddedness (which I did not measure at a reach scale) may explain the variation in torrent sculpin densities among streams and within individual habitats.

Pacific giant salamanders

Pacific giant salamanders prefer small streams with small to large boulders and abundant woody debris (Corkran and Thoms 1996). I found no difference in salamander densities among habitat types, but the average size was larger in pools than riffles, suggesting that the largest salamanders preferred pools. This is consistent with Kelsey (1995), who indicate that riffles are important areas for first-year Pacific giant salamander larvae and that larger second and third-year larvae prefer pools. Unfortunately, salamander densities were not high enough (< 20 at most sites) to accurately separate them into size or age classes. Giant salamander densities at a reach scale were positively correlated with slope and negatively correlated with bankfull width, suggesting that highest densities were in small high gradient streams, generally consistent with the findings of Corn and Bury (1989) and Kelsey (1995). Streams sampled by

Corn and Bury (1989) and Kelsey (1995) were mostly less than 2 m wide and greater than 8% slope, whereas the sites I sampled were less than 4.5 % slope and more than 5 m in bankfull width. Thus both stream size and gradient may be better predictors of salamander density in lower gradient, higher order (larger) streams but gradient may be of primary importance among very small, steep streams.

Lamprey

Lamprey showed a strong preference for pool habitats and densities were only weakly correlated with LWD and dominant substrate size in pools. However, I found most lamprey larvae in deposits of fine sediment and organic material within backwaters and margins of pools, which is consistent with that reported for larval lamprey by Wydoski and Whitney (1979) and Scott and Crossman (1998). Drainage area and percent pool habitat explained one third of the variation in lamprey densities among sites, indicating that larval lamprey prefer pools and that they are found in higher densities in larger streams. Adult lamprey spawn in gravel in riffles and had I sampled earlier in the summer, I may have found a larger number of lamprey in riffles or in habitats with course sediment. Lamprey growth appears to be density dependent as length was negatively correlated with density. Backwater pools and areas with high levels of silt and fine organic matter were rare in streams I sampled. Given that larval lamprey are filter feeders and inhabit slow water habitats, suitable habitat may have resulted in high competition and reduced growth in those streams where they were abundant.

The information on habitat preferences provided in this study indicates how habitat alterations might affect various species found in small Pacific Northwest streams.

Anthropogenic activities such as logging and agriculture typically alter physical habitat (e.g., pool frequency and area, LWD loading, water quality) and biotic factors (e.g., Murphy 1981; Hawkins et al. 1983; Bisson et al. 1987; Corn and Bury 1989; Hicks et al. 1991). The loss of woody debris, habitat complexity and pools following logging has been linked to reductions in pool-preferring species such as coho salmon (Murphy et al. 1984; Heifetz et al 1986; Hicks et al. 1991). Restoration activities that create pool habitat and increase habitat complexity have increased densities of juvenile coho salmon and, to a lesser extent, cutthroat and steelhead trout (e.g., House et al. 1989; House 1996, see Chapter 2 this document). It is less clear how changes in pools and pool quality might impact other species that prefer pools such as lamprey. Their strong preference for pools and fine sediment, usually associated with LWD accumulations, suggested that they would respond negatively to decreases in LWD, pools and other slow water habitats.

Pools and riffles explained much of the difference in distribution of fish within stream reaches. However, physical variables such as LWD and substrate size were poor predictors of density among individual habitats explaining 10% or less of the variation in individual species distribution among pools. In contrast, reach-scale analysis indicated that physical variables explained from 22 to 63% of the variation in densities among streams. Rosenfeld et al. (2000) also found reach or watershed scale factors to be better predictors of cutthroat abundance than microhabitat features. Clearly a large portion of the variation both within and among streams is unaccounted for and it is likely the result of sampling error, natural variation and important physical variables I did not quantify.

Finally, this study was not designed to examine competition within and among species. With the exception of studies on salmonids species (e.g., Hartman 1965, Glova

1986), few studies have examined competition among different species and species groups. Controlled experiments are needed to examine interspecific competition and interactive relationships among various species and how the presence or absence of various species may affect habitat use.

CHAPTER 5: EFFECTS OF ARTIFICIAL WOOD PLACEMENT ON MOVEMENTS OF TROUT AND JUVENILE COHO SALMON IN NATURAL AND ARTIFICIAL STREAM CHANNELS

Introduction

Populations of many fish species have been reduced by a combination of factors including over-fishing and degradation of habitat. Habitat modification has frequently been undertaken in an effort to restore fish populations or facilitate fishing in both freshwater and marine environments. In some cases artificial habitats are explicitly designed to concentrate fish to increase fishing success, such as "fish aggregating devices" (FADs) used to attract tuna (Holland et al. 1990; Higashi 1994). In other cases such as artificial reefs in freshwater and coastal marine areas, the structures clearly function in more than one way (Bassett 1994; Lindberge 1997; Bohnsack et al. 1997; Kelch 1999). Artificial reefs are quickly colonized by adult fish, suggesting movement among habitats, even though many of these species are assumed to spend all or almost all of their time in very restricted home ranges (e.g., rockfishes: Matthews 1990a, 1990b). Artificial habitats may also provide refuge and increase survival of recruits, and so benefit the population as a whole (Lindberge 1997). The question of whether structures increase survival and recruitment or merely relocate fish is a difficult one to answer, yet it is important to the evaluation of such structures.

The enhancement and restoration of stream habitat for salmonid fishes through the instream placement of structures and large woody debris (LWD) has increased dramatically in western North America in recent years. Numerous studies have

documented increased density of resident and anadromous salmonids after instream habitat enhancement or restoration techniques (e.g., Hunt 1976, House and Boehne 1985, Binns 1994, House 1996, Cederholm et al. 1997). This is consistent with observations that the density of juvenile salmonids varies with habitat, and specifically that coho salmon (*Oncorhynchus kisutch*) tend to occupy pools that are associated with woody debris (Bisson et al. 1982).

Studies relating salmonid density to variation in habitat quality (either natural or post-restoration) generally assume that the fish have small home ranges and seldom move between treated (restored or enhanced) and unaltered stream reaches, or between natural reaches that vary in quality. This assumption allows one to infer that differences in density result from increased survival. Immigration to restored sites would create the appearance of benefit where it did not exist, and emigration from restored sites would minimize the apparent survival benefit. The assumption of restricted movement was primarily based on studies prior to 1990, which indicated that juvenile and adult salmonids had limited home ranges and rarely moved far except for spawning migrations or emigration to sea (reviewed by Gowan et al. 1994).

Gowan et al. (1994) demonstrated that most studies on movement of stream fishes were unlikely to detect movement and that movements of several hundred meters over the course of a week or even day are common among juvenile and adult resident salmonids. Riley et al. (1992), Riley and Fausch (1995) and Gowan and Fausch (1996) reported that most of the increase in trout abundance in high elevation Colorado streams enhanced with log structures was due to immigration. Other studies indirectly suggest that movement may play a role in increased fish abundance following habitat

manipulations. Angermeier and Karr (1984) reported increased abundance of warm-water fishes in response to short-term habitat manipulations, which was clearly due to immigration. Hamilton (1989) found a two fold increase in juvenile steelhead (*O. mykiss*) abundance in a stream reach treated with boulders, while steelhead densities in a nearby reference reach declined by half.

Large-scale movements (reach or watershed scale) of juvenile anadromous salmonids in streams often occur in fall as the fish relocate to off-channel habitat (e.g., Bustard and Narver 1975a, 1975b; Peterson 1982, Tchaplinski and Hartman 1983, Cederholm and Scarlett 1984) and during seaward migration in spring (reviewed by Northcote 1992, McCormick et al. 1998). However, it unclear how common movement is during the summer. Heggenes et al. (1991) and Harvey et al. (1999) found that most age 1+ cutthroat trout (*O. clarki*) in small coastal streams moved less than 50 m throughout the year. However, Kahler (1999) demonstrated that over the course of a summer approximately 50% of the juvenile coho salmon, cutthroat, and steelhead trout in three streams moved from their original tagging location. Most movements were about five habitat units or 30-50 m but a few fish moved more than 100 m. It is important to evaluate the extent of movement and the ways in which it might affect assessment of population-level benefits from habitat restoration.

It has been generally assumed that not only do few fish move but that the movers are less fit than those that remain. Large size and territorial possession provide great advantages in competition for space (Rhodes and Quinn 1998, and references therein). Dominant juvenile salmonids may force smaller conspecfics to move to other habitats or suboptimal feeding areas (Chapman 1962, Nielsen 1992) and it seems reasonable to

assume that movers would grow slower than residents. However, Kahler (1999) found that fish that moved had higher subsequent growth rates than those that did not move. Fish in poor habitat (small, shallow pools) tended to move to higher quality habitat, whereas fish in higher quality habitat stayed, despite high densities that reduced growth. Harvey et al. (1999) found that adult coastal cutthroat trout in habitats lacking LWD moved more frequently than those in habitats with LWD but I know of no study that explicitly examined the movements of juvenile anadromous fish in response to the artificial placement of LWD or other instream structures.

In this study I examined the movements of juvenile coho salmon and trout during fall and winter in a natural stream channel, testing the null hypothesis that few fish would move between unrestored (reference) and restored (treatment) reaches against the predicted alternative that fish would tend to move from unrestored to restored habitat. I also examined the movement patterns and growth rates of juvenile coho salmon in artificial stream channels with and without woody debris, testing the null hypothesis that the frequency of movement would be independent of habitat quality, and that movement would be independent of size and growth, against the predicted alternatives that smaller fish would tend to move, and would grow slower than residents.

Methods

Movements in Shuwah Creek

Juvenile coho salmon, cutthroat and steelhead trout were given group-specific photonic tags (Kahler 1999) in two reaches of Shuwah Creek, Washington to determine the level of movements among treatment (artificial LWD placement) and reference (no

LWD placement) stream reaches. The study section of Shuwah Creek was a small (6.5 m bankfull width), low gradient (slope = 1.5 %) forced-pool riffle channel (Montgomery and Buffington 1997) near Forks, Washington. A restoration project, including the placement of log structures within the bankfull channel, was conducted on approximately 500 m of Shuwah Creek in the summer of 1996. We selected two 90 m reaches within the creek, one at the upper end of the restored stream reach and one approximately 100 m farther upstream in an unrestored or reference reach. There were 35 and 47 pieces of large woody debris (>10 cm in diameter and 2 m long) in the reference and treatment reaches, respectively, in September 1998. The treatment (restored) reach we sampled included a total of 7 log habitat structures (i.e. deflectors, log weirs and cover structures) and had more pool area than the reference reach (80% versus 56% pool area, September 1998). Besides coho salmon, cutthroat and steelhead trout, the reaches contained torrent sculpin (*Cottus rhotheus*), reticulate sculpin (*C. perplexus*) and larval Pacific lamprey (*Entosphenus tridentatus*).

On September 6-8, 1998, salmonids were caught with three-removal electrofishing, anesthetized, and the dorsal fin of each fish was given a red or blue mark depending upon whether it was captured in the treatment (red) or reference (blue) stream reach. The fish was then returned to the habitat unit (e.g., pool or riffle) where it was captured. A total of 114 coho salmon, cutthroat, steelhead and trout fry (not distinguishable to species) were marked in the restored reach and 71 in the reference reach. Complications with the tagging equipment prevented tagging of 31 fish captured during electrofishing in the restored reach and 7 in the reference reach. The total numbers of trout and coho captured were identical to multiple removal-abundance

estimates using Carle and Strub (1978) and capture probability during three-pass electrofishing exceeded 0.73.

Monthly night snorkel surveys were conducted October 1998 through April 1999 to count tagged and untagged fish in the two reaches. Juvenile salmonids become nocturnal when temperatures drop below approximately 9°C and night snorkeling provides similar abundance estimates to multiple-removal electrofishing during winter (Roni and Fayram 2000). One diver with a halogen dive light entered the habitat from the downstream end and slowly moved upstream, stopping occasionally to relay the number, approximate size, mark (tag color), and species of fish observed to a second individual on the bank. Water temperature, underwater visibility and flow were measured downstream of each site before snorkeling (Table 5.1). Underwater visibility was measured using a horizontal secchi disk. Snorkel surveys were initiated one hour after sunset and only conducted on nights with cloud cover or no visible moonlight to assure similarity in light levels.

The numbers of marked and unmarked fish were compared between the two reaches over the 8-month sampling period. Most trout observed were cutthroat but we could not reliably distinguish juvenile cutthroat from steelhead during snorkel surveys. Therefore, we refer to these species collectively as trout throughout our analysis. Regression and analysis of covariance (ANCOVA) were used to examine the relationship between survey date and abundance within and between stream reaches. T-tests were used to examine the difference in length between fish in the treatment and reference reaches captured during electrofishing (initial marking).

Table 5.1. Water temperature, flow, and underwater visibility of Shuwah Creek during initial tagging (September) and monthly snorkel surveys.

Month	Temp. (C)	Visibility (m)	Discharge ($m^{-3} \cdot s^{-1}$)
Sept	13.0	na[1]	0.01
Oct	8.5	3.0	0.07
Nov	8.4	3.0	0.38
Dec	7.1	2.0	0.40
Jan	6.5	2.5	0.53
Feb	5.8	3.0	0.66
Mar	6.7	3.0	0.16
April	7.1	2.5	0.05

Movements in artificial channels

Experiments were conducted in a semi-natural outdoor stream system (45 by 6 m, 3% grade) at the National Marine Fisheries Service Manchester Field Station near Manchester, Washington (Berejikian et al. 2000). The stream system was divided evenly down the middle into two separate, identical channels (A and B) by a 1 m high wooden wall and each channel was further divided into 8 individual habitats units 5 m long by 3 m wide (A1-8 and B1-8; Figure 5.1). Well water was supplied at 80 L min^{-1} and was recirculated at a flow of 0.05 $m^3 \cdot s^{-1}$ (0.025 $m^3 \cdot s^{-1}$ in each channel). A 15 cm high wooden weir created a hydraulic drop between units within each channel. Each habitat unit contained equal amounts of 3 – 5 cm gravel which was graded to form shallow plunge pools. Depths were standardized between habitats to 30 cm at the upstream end, 10 cm at midpoint and decreased gradually to 2 cm at the downstream end. The exception was the lower-most habitats in each channel (A8 and B8) where water backed up against the screens and depths were 40 cm at the top and 30 cm at the bottom. Screens could be placed between each unit to allow for sampling of individual habitats.

Wood cover in the form of denuded 1.8 m long Douglas-fir *(Pseudotsuga menziesi*) trees (commercially farmed Christmas trees) and red alder *(Alnus ruba)* logs (1.2 m long by 15 cm diameter) was placed in habitat units 1, 2, 4 and 6 in channel A (Figure 5.1). Additionally, a single Douglas fir tree was placed in habitat units A8 and B8 because of the high concentration of fish in these units and concerns over heavy avian predation in these habitats in the absence of cover. Nevertheless, some predation likely

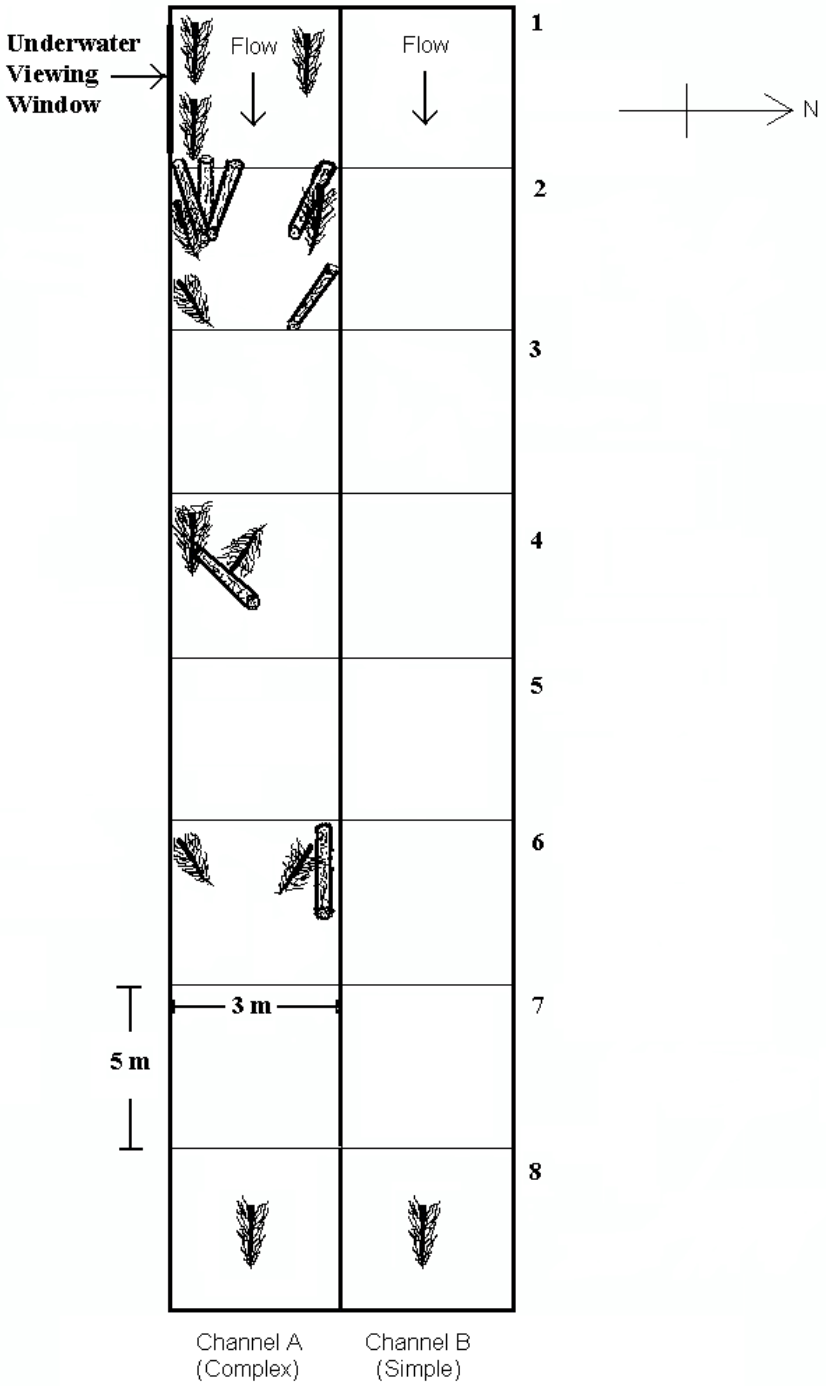


Figure 5.1. Diagram of National Marine Fisheries Service Manchester artificial stream used for experiment on difference in size, growth rate and movement of coho in complex (with wood) and simple (without woody) channels.

occurred during the study as avian predators such as kingfishers (*Ceryle alcyon*) and blue herons (*Ardea herodias*) had access to the channel. Wood was placed in an orientation similar to that found in many stream habitat enhancement projects (Figure 5.1). Water temperatures ranged from 8.1 to 15°C over the course of the study. Abundant production of aquatic insects (>16,500 chironomids·m^{-2}) within the stream channel made it unnecessary to feed the fish over the course of the study.

Five hundred and six 1998 brood juvenile coho salmon were obtained from the Washington Department of Fish and Wildlife Minter Creek Hatchery on April 21, 1999 and placed into the experimental channel. Thirty coho salmon were placed in each unit, and the remaining 26 fish were split equally between units A1 (complex) and B1 (simple). The fish were allowed to acclimate and move freely within channels for two weeks. On May 5 the fish were recaptured anesthetized, implanted with passive integrated transponder (PIT) tags, measured, weighed and released into the habitat in which they were captured. The fish were then sampled three times (May 19, June 2, June 16) to determine their location and size. Screens were placed between habitats to prevent movement and then fish were captured by making five passes through each habitat with a 6-mm mesh stick seine. All wood was temporary removed from a habitat prior to sampling and replaced shortly after sampling. Fish were anesthetized with tricaine methanesulfonate (MS222) and their length, weight and PIT tag number recorded. To minimize handling, fish were not weighed on May 19 and June 2.

Only those fish that were recovered on the last sampling date (June 16) were used in our analysis of length, weight and growth and only fish for which we had records for at

least 3 of the 4 weeks were included in the movement analysis. A "holder" was defined as a fish that was captured in the same habitat on all sampling dates. A "mover" was defined as fish that was recovered at least once in a habitat other than the habitat it was found in on May 5 (original tagging location). Length-weight relationships between holders and movers were compared using ANCOVA and analysis of variance (ANOVA) was used to examine differences in length, weight, growth rate (length on May 5 minus length on June 15) between movers and holders and between channels. A chi-square test was used to compare proportions of movers and holders between the two channels.

Results

Movements in Shuwah Creek

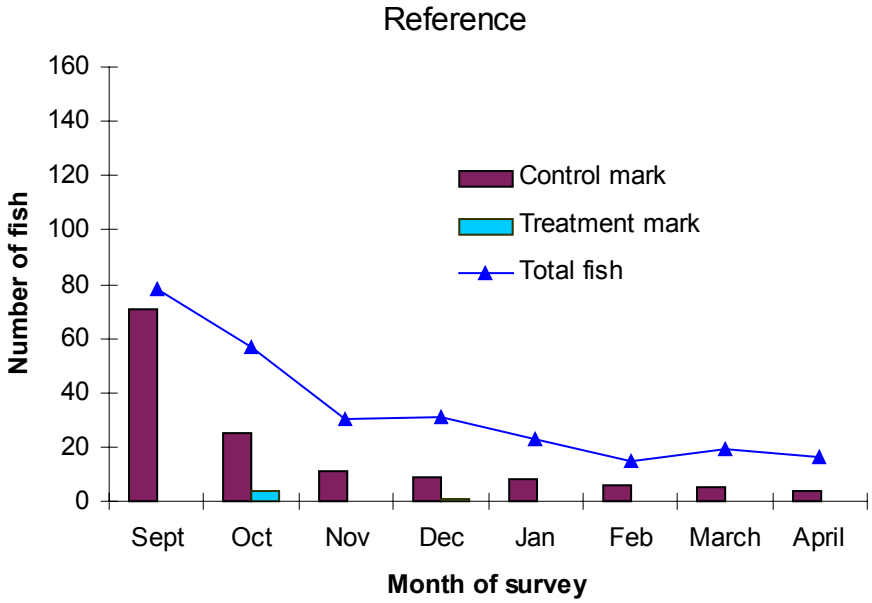
Little movement was observed between reference and treatment reaches in Shuwah Creek from September through April (Figure 5.2). The total number of fish marked in the two reaches differed (78 and 114 in reference and treatment, respectively) but similar numbers of marked fish were observed in the reference (5 to 29 fish) and treatment (5 to 24) reaches from November to April. From 0 to 4 treatment marked fish (0 to 14% of marked fish present) were observed in the reference reach during monthly snorkel surveys (October to April) and 1 to 3 reference marked fish (4 to 33% of marked fish present) were observed in the treatment reach. The percentage of fish observed during monthly snorkel surveys that had marks ranged from 26 to 51% in the reference reach and 11 to 55% in the treatment reach but generally declined with time in both reaches (Table 5.2). The total number of fish observed in the treatment reach ranged from 36 to 145 and was consistently higher than the total number in the reference reach

(15 to 78 fish; Fig. 5.2). Total monthly fish counts were nearly significantly different between reaches (paired t-test, $p = .06$), but not when coho and trout were examined separately ($p = 0.17$ and 0.26 for coho and trout, respectively).

Despite the small number of tagged fish that moved between reaches, the appearance of untagged fish indicated considerable movement. The proportion of marked fish in the reference reach decreased from 91% (71 fish) in September to 31% (5 fish) in April. Similarly, the proportion of marked fish in the treatment reach decreased from 79 to 16% (114 to 5 fish) over the same period (Table 2.2). Juvenile salmonid abundance decreased dramatically in November in the treatment reach. The total number of fish remained relatively constant throughout the winter (December to April), particularly in the treatment reach; numbers decreased gradually over the winter in the reference reach. Regression analysis indicated that the relationship between survey date (month) and fish abundance was significant for the reference reach ($p < 0.01$, $r^2 = 0.78$), but not for the treatment reach ($p = 0.11$, $r^2 = 0.37$). However, no significant difference existed between the slopes and elevations of survey date-abundance equations between the reaches (ANCOVA, $p > 0.15$). When examined by species, a significant decline over time was observed in the reference reach for coho salmon ($P < 0.04$, $r^2 = 0.91$) and trout ($r^2 = 0.55$) and in the treatment reach for coho salmon ($r^2 = 0.54$, $p = 0.04$) but not for trout ($p = 0.46$; Figure 5.3). No significant difference existed in the survey date-abundance regression equations between reaches for coho salmon (ANCOVA $p > 0.20$) or trout ($p > 0.15$).

Table 5.2. Percent of marked fish observed in reference and treatment reaches of Shuwah Creek during monthly surveys from September 1998 to April 1999.

Month	Reference	Treatment
Sept	91%	79%
Oct	51%	55%
Nov	37%	36%
Dec	26%	34%
Jan	35%	33%
Feb	40%	31%
March	26%	20%
April	31%	16%



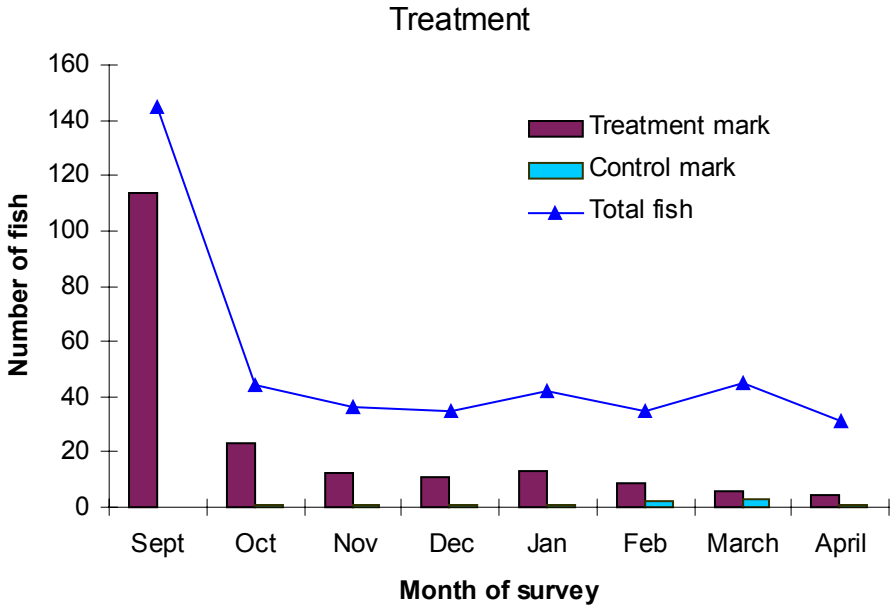
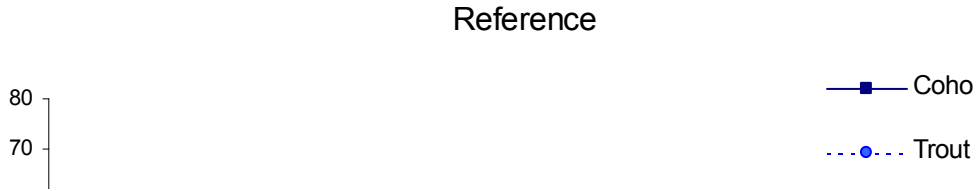


Figure 5.2. Number of marked fish (coho and trout combined) observed in reference and treatment reaches of Shuwah Creek from October 1998 through April 1999 and total number of fish observed in each reach. Presence of control marked fish in treatment or vice versa indicates level of movement among reaches. September counts represent total fish captured during tagging (electrofishing) all other dates are snorkel surveys.



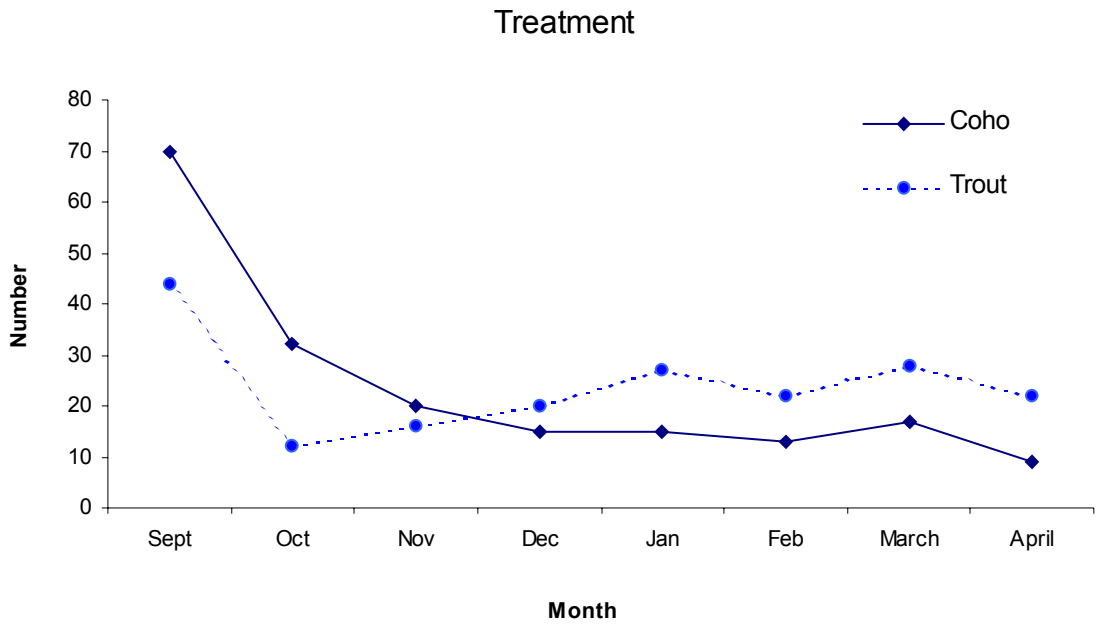


Figure 5.3. Total number of trout and coho observed in monthly surveys of reference and treatment reaches of Shuwah Creek from September 1998 through April 1999. September counts represent total fish captured during tagging (electrofishing) and all other dates represent snorkel counts.

When tagged in September, the coho salmon ranged from 60 to 130 mm long and the trout from 47 to 187 mm. The fish were significantly smaller in the reference reach than the treatment reach (coho: 73.5 mm vs. 77.6 mm; $p = 0.02$; trout: 90.8 mm vs. 105.3 mm; $p = 0.04$). Fish lengths from October through April were based on visual snorkel estimates and no analysis was performed on these data.

Movements in artificial channels

Of the 506 juvenile coho salmon released in the channel, 466 (93%) were recaptured approximately 2 months later. Recovery rates were not significantly different between the simple and complex channels (98 and 86 %, respectively; $p = 0.98$). A larger fraction of the fish moved in the complex channel than the simple channel (37 vs. 29% movers; Chi-square, $p < 0.01$; Table 5.3). However, of the fish that moved, a greater fraction moved only once (rather than several times) in the complex channel ($p = 0.01$; Table 5.3). The net direction of movement (upstream vs. downstream) was similar between the two channels; 34% and 43% of the movement was upstream in simple and complex channels, respectively ($p = 0.98$; Table 5.3).

No difference in length, weight or growth was detected between movers and holders in the complex channel (ANOVA, $p > 0.30$). However in the simple channel movers were longer ($p = 0.04$), heavier ($p < 0.01$), and grew more rapidly ($p < 0.01$) than holders. Length-weight relationships between holders and movers were similar in the complex channel (ANCOVA, $p = 0.91$), but differed in the simple channel ($p < 0.01$; Table 5.4) with movers being heavier for given length. The fish moved farther in the

Table 5.3. Percentage of fish holding, moving, and direction of their movement in experimental channels with (simple) and without woody debris (complex). Equal direction movement indicates that the fish moved an equal number of habitats up and downstream over the course of the study.

		Channel A (Complex)	Channel B (Simple)
Movers		37%	22%
Number of moves			
	one	75%	51%
	two	18%	38%
	three	7%	11%
Direction	up	43%	34%
	down	48%	40%
	equal	9%	26%

Table 5.4. Mean length, weight, and growth rate of juvenile coho salmon in simple (A) and complex (B) artificial channels. Length and weight were measured at the end of the experiment.

	Channel	
	Complex (A)	Simple (B)
Movers		
mean length (mm)	87.5	89.9
mean weight (g)	7.9	8.6
growth (mm)	23.2	25.3
Holders		
mean length (mm)	88.6	87.6
mean weight (g)	8.1	7.7
growth (mm)	24.1	22.5

simple than complex channel (6.7 vs. 4.4 habitat units or 33.5 vs. 22 m; t-test, $p < 0.01$), but there was no relationship between length or weight and distance moved (ANOVA, $p = 0.94$) or frequency of movement ($p > 0.35$) in either channel. The distribution of fish differed between the simple and complex channels; more fish were found in upstream habitats of the complex channel (where wood was present) than the simple channel. However, most fish in both channels were found in the downstream-most habitats (Figure 5.4).

Discussion

Gowan et al. (1994) and others have suggested that movements of salmonids may in part account for the increased density observed at instream restoration projects. In contrast, Reeves and Roelefs (1982) indicated that it was unlikely that LWD placement or other habitat structures would concentrate fish in treated stream reaches, as juvenile salmonids would disperse to those habitats vacated by others. The results of our tagging study on Shuwah Creek were consistent with this latter conclusion, as few tagged fish moved between the restored and reference reaches. However, a number of other factors may explain the limited exchange I observed between reaches including loss of marks, emigration of marked fish or immigration of unmarked fish, mortality or the spatial scale examined.

The numbers of marked fish decreased rapidly in both study sections from October through December and I interpret this as emigration. Juvenile coho, cutthroat and other anadromous salmonids make large-scale movements in the fall and winter following heavy rains (Peterson 1982, Cederholm and Scarlett 1984, Swales and Levings

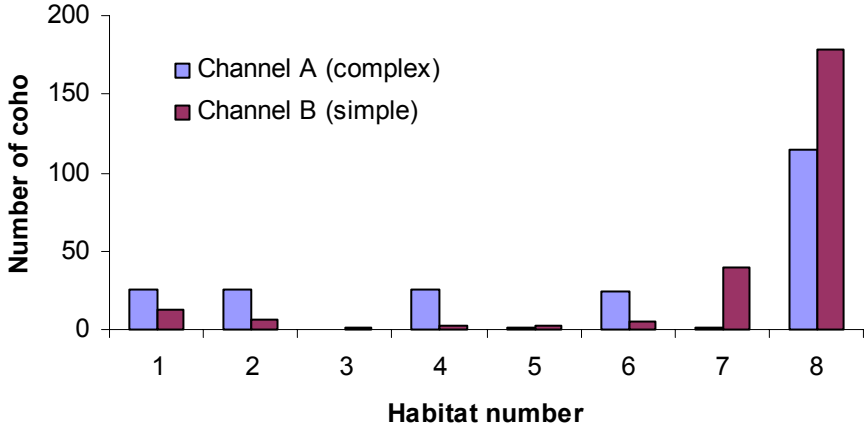


Figure 5.4. Distribution of juvenile coho in artificial channels with (complex) and without woody debris (simple) on June 15, 1999. Habitat numbers (x axis) are in order from upstream to downstream. Habitat units 1, 2, 4 and 6 in channel A all contained woody debris and habitat units 8 in both channels were deeper than other units because screens at the downstream end of the channel caused water to pond.

1989, Shirvell 1994) and the rapid decrease in densities observed in Shuwah Creek during fall was consistent with these studies. Some over-winter mortality doubtless occurs (on the order of 50 – 75% - Quinn and Peterson 1996), and this probably contributed to the density decreases I observed. The decrease from September to October may be due in part to electrofishing being used to initially capture fish and estimate abundance in September, while night snorkel counts were used during all other months. However, this would not explain declines from October to November. Moreover, Roni and Fayram (2000) reported that night snorkel estimates were similar to multiple-removal electrofishing estimates when water temperature and fish densities are less than approximately 9ºC. The percentage of unmarked fish also increased over the course of the winter. This might result from mark degradation, but our experience indicates that the blue and red photonic marks on the dorsal fin I used are readily visible for nine months or more. Furthermore, I noticed no deterioration of the mark over the course of this study. Therefore, I conclude that the increased numbers of unmarked fish resulted from both immigration of unmarked fish and emigration of marked fish, and that the declines in abundance resulted from movement as well as mortality.

I did not monitor the100 m reach between the treatment and reference sections, nor did I monitor areas upstream and downstream of these reaches. However, Kahler (1999) examined summer movements of individually marked trout and coho salmon from approximately 178 m downstream of our treatment reach to 120 m upstream of our study reaches and found few fish that moved from treatment and reference reaches into areas above, between or below the two reaches during summer (T. Kahler, 1410 Market Street, Kirkland, WA 98033 unpublished data). Moreover, Kahler (1999) found that on average

fish moved 5 habitat units (approximately 35 m) and few fish moved more than 100 m. However, my study in Shuwah Creek was not designed to detect movement among microhabitats, but rather among stream reaches. My fall and winter results and Kahler's summer results both suggested limited movement (exchange) between treatment (restored) and reference reaches but for different reasons. Summer movements seem to be frequent but of limited scale, hence most fish move within rather than between reaches. Fall movements, inferred in our study from the immigration of many unmarked fish into my study sections and disappearance of most marked fish, seem to be of such large scale that they exchange few fish between nearby reaches.

The results in the artificial channels are consistent with those of Heggenes et al. (1991) and Kahler (1999); although a substantial fraction of the fish moved, most moved only once, and moved a few habitat units. The percentage of movers was higher in the complex than the simple channel, though fish moved less frequently and shorter distances in the complex channel (Table 5.3). This suggests that the placement of wood (the only difference between channels) resulted in frequent short distance movements and in the absence of structure fish moved more frequently in search of suitable habitat. Harvey et al. (1999) found that adult cutthroat marked in habitats without woody debris or cover were more likely to move and moved farther than those tagged in habitat with woody debris. Similarly, Kahler (1999) indicated that during summer juvenile trout and coho salmon were more likely to move from shallow habitats lacking cover to deeper, more complex habitats. My results from Shuwah Creek also suggest that fish are less likely to move from more complex habitat, as the decreases in trout and coho abundance during fall and winter were more rapid in the reference reach than the treatment reach. My

results from both natural and artificial channels coupled with those of other recent studies, suggest that trout and juvenile coho salmon move shorter distances in channels with abundant cover and woody debris.

I found no difference in growth or size between movers and holders in the artificial channel with woody debris. Similarly, Spalding et al. (1995) found that varying levels of woody debris had little effect on growth of juvenile coho salmon but that density strongly affected growth in a semi-natural channel system. However, movers were initially larger and grew faster than holders in the channel without woody debris. This suggests that in poor habitat conditions some of the larger juvenile coho salmon may move to new habitats. This result is consistent with Kahler's (1999) work but not with previous studies on juvenile salmonids (e.g., Chapman 1962, Nielsen 1992), which indicated or implied that movers are smaller and less fit than fish that maintain territories. For example, Neilsen (1992) examined dominance hierarchies in juvenile coho salmon within individual habitats and found that "floaters" or fish that moved around an individual habitat grew less than dominant fish. However, she examined dominance and movement within individual habitat units, whereas I examined movement among habitats. Other studies have reported that mobile salmonids are larger (Riley et al. 1992; Young 1994), but may be in poorer condition than those not moving (Naslund et al. 1993; Gowan and Fausch 1996). The differences in size between mobile and static juvenile coho salmon needs further investigation, especially if fish that move enjoy higher overwinter survival rates (Quinn and Peterson 1996).

Coho salmon and trout were significantly larger in the treatment than the reference reach of Shuwah Creek at initial tagging, despite higher densities. This is

unusual as juvenile coho length is known to be inversely related to density in natural streams (Fraser 1969; see also Chapter 2). This may indicate that differences in habitat quality between the two reaches (i.e. differences in LWD and pool areas) can influence size in some way. High quality areas might be occupied by larger fish if smaller ones are forced out, or high quality areas might permit higher growth rates. Unfortunately, I did not have accurate length measurements for the remaining seven sampling dates to further examine differences in growth rates between reaches or between fish that moved or remained static in Shuwah Creek.

The majority of coho salmon in our artificial channels congregated in the lower most habitats and some fish may have been attempting to emigrate. Wilzbach (1985) examined emigration of cutthroat trout from artificial channels and found that emigration was higher at low food abundance and that cover influenced emigration only at high food levels. In contrast, Mesick (1988) found that reduced food levels resulted in little emigration of Apache (*O. apache*) and brown trout (*Salmo trutta*) from artificial channels regardless of the level of cover. In our study, benthic macroinvertebrates were abundant (>16,500 $\cdot m^{-2}$) and growth rates of coho salmon in the artificial channels exceeded 20 mm over a 6 week period, suggesting that both food levels and growth were high and thus emigration due to lack of food was unlikely. Furthermore, the density of coho in our artificial channels were near 2.0 fish$\cdot m^{-2}$ throughout my study, which is high but not unusual for natural streams (Nickelson et al 1992a, 1992b; P. Roni unpublished data). Spalding et al. (1995) found little emigration of juvenile coho salmon from artificial channels at higher densities than I examined. Lonzarich and Quinn (1995) found that depth was more important than structure in determining coho salmon distribution in

artificial channels. Thus I believe the higher numbers of fish in the lower habitat units in our study is due to these habitat being deeper in these units rather than fish attempting to emigrate from the channel.

The two movement studies I conducted suggest that movements among individual habitats (small scale) and movement among reaches (large scale) are common for juvenile coho salmon and juvenile and adult trout. While little exchange of marked fish occurred between treatment (restored) and reference reaches, large-scale movements make it difficult to determine the level of emigration from or immigration to a restored stream reach. Our results supports the conclusions of Gowan et al. (1994) and Riley and Fausch (1995), suggesting that both large and small scale movements patterns should be considered when evaluating stream restoration and habitat utilization. Additional research is needed to determine whether the placement of instream structures leads to emigration of juvenile salmonids from less suitable habitats and concentration in treated stream reaches.

CHAPTER 6: OVERALL CONCLUSIONS

Salmonid response to LWD placement

The results of this study support the findings of House et al. 1989; 1996, Cederholm et al. 1997 and provide strong evidence that artificially placed LWD leads to significantly higher densities of juvenile coho in summer and higher densities of coho, cutthroat and steelhead during winter, especially in sites deficient in wood to begin with. However, density-dependent growth of coho salmon may reduce the net benefit to the population if subsequent survival is size-related. Summer densities of steelhead may be reduced by artificially placed LWD, but this may be more than compensated for by increases in winter rearing areas and densities. While the study design I employed was not designed to determine the effectiveness of individual projects, it provides insight into the factors that make projects successful. The relationship between coho abundance and pool area and pool-forming wood indicates that projects that dramatically increase pool area or LWD creating pools will provide the largest increases in fish abundance. However, I examined fish response at a relatively small scale. Other studies (e.g., Bradford et al. 1997, Sharma 1998) looked at coho production at a watershed scale and found positive relationships between smolt production per kilometer of stream length and pool area and other physical variables. It would be inappropriate to couple my reach scale results for coho during winter and suggest that smolt production throughout a watershed could be increase threefold by placing LWD throughout the active stream channel. Additional research is needed to determine the effect instream LWD placement might have on overall smolt production within a watershed. Caution should also be used

in applying the results of my study to streams of different size, land use, or streams in different geographic areas. Finally, my results do not negate the need to restore natural processes that create and maintain salmon habitat rather than relying on instream manipulations which may not address the underlying factors limiting fish production.

Response of lamprey, sculpin and salamanders

While the response of salmonids to habitat manipulations has been frequently examined, little information exists on the effects of these habitat manipulations on nonsalmonids. It has generally been assumed that habitat restoration efforts either benefit nonsalmonids or have no adverse impacts. Until now, it has been unclear what effect habitat manipulations might have on other members of the fish community. The results of my research suggest that LWD placement has little effect on nonsalmonids and salamanders densities or species richness or diversity. The lack of response of the fish community to habitat disturbance (or improvement in this study) as reported in other studies is likely due to the relatively depauperate fish fauna found in these and other small Pacific Northwest streams. However, lamprey and age 1+ reticulate sculpin, two species that prefer pools appear to respond in those streams with the largest changes in LWD. These results indicate that habitat restoration efforts designed to increase juvenile salmonids do not appear to negatively impact other stream fishes and may benefit those that prefer pools. However, additional research and monitoring is needed to further elucidate the response of reticulate sculpin and lamprey to artificially placed LWD and to examine competitive interactions among these and other members of the fish community.

Habitat Preferences of Fishes and Amphibians

Difference in pool and riffle area explained much of the difference in distribution and density of fish and salamanders within and among streams. The results of my analysis of habitat preferences support previous studies which indicate that juvenile coho and cutthroat prefer pools during summer, and that coho, cutthroat and steelhead show strong preference for pools during winter. Larval lamprey were found almost exclusively in pools, whereas torrent sculpin were found at highest densities in riffles. Giant salamanders and reticulate sculpin densities did not differ between pools and riffles, but reticulate sculpin habitat use appeared to be influenced by the presence of torrent sculpin.

Physical variables such as LWD and substrate size were poor predictors of density among individual habitats explaining 10% or less of the variation in individual species distribution among pools. In contrast, reach-scale analysis indicated that physical variables explained from 22 to 63% of the variation in densities among streams. Rosenfeld et al. (2000) also found reach or watershed scale factors to be better predictors of cutthroat abundance than microhabitat features. Clearly a large portion of the variation both within and among streams is unaccounted for and it is likely the result of sampling error, natural variation and important physical variables I did not quantify.

This study was not designed to examine competition within and among species. With the exception of studies on salmonids species (e.g., Hartman 1965, Glova 1986), few studies have examined competition among different species and species groups. Controlled experiments are needed to examine interspecific competition and interactive

relationships among various species and how the presence or absence of various species may affect habitat use.

Movement of salmonids

The two movement studies I conducted suggest that movements among individual habitats (small scale) and movement among reaches (large scale) are common for juvenile coho salmon and juvenile and adult trout. While little exchange of marked fish occurred between treatment (restored) and reference reaches, large-scale movements make it difficult to determine the level of emigration from or immigration to a restored stream reach. My results supports the conclusions of Gowan et al. (1994) and Riley and Fausch (1995), suggesting that both large and small scale movements patterns should be considered when evaluating stream restoration and habitat utilization. Additional research is needed to determine whether the placement of instream structures leads to emigration of juvenile salmonids from less suitable habitats and concentration in treated stream reaches.

ACKNOWLEDGEMENTS

This project was funded by a cooperative agreement between the National Marine Fisheries Service and the Bureau of Land Management (BLM). Bob Alverts of the BLM, Kelly Moore and Barry Thom of Oregon Department of Fish and Wildlife were instrumental in expanding the project to Oregon. I would also like to thank Drs. Tom Quinn, Loveday Conquest, Jim Karr, Dave Montgomery , and Michael Murphy whose ideas improved the study design, data analysis and final product. I thank Andy Fayram, Jose Carrasquerro, Melissa Boles, Kristina Ramstad for their long hours in the field and their assistance collecting and entering field data. Numerous individuals donated their time to provide project maps, field tours and assist in site selection including: Barry Olson, Karen Chang, Rich McConnell, Phil DeCillis, Mike Donald, Roger Nichols, Brian Riordon, and Jim Doyle of the U.S. Forest Service; Mike Kelly of the U.S. Fish and Wildlife Service; Jeff Cederholm of the Washington Department of Natural Resources; Bob Rudiger, Dave Roberts, Effie Fraser, and Matt Walker with the BLM; Kerry Bauman and Laird O'Rollins with King County, Brian Erickson with Columbia Pacific, and Kurt Nelson with the Tulalip Tribes. I would also like to thank Boise Cascade, Olympic Resources Management, Stimpson Timber, Willamette Timber, and Simpson Timber for allowing me access to sites on their lands.

LIST OF REFERENCES

Anderson, J.W., Ruediger, R.A. and Hudson, W.F. 1984. Design, placement and fish use of instream structures in southwestern Oregon. *In* Proceedings: Pacific Northwest Stream Habitat Management Workshop. *Edited by* T. J. Hassler. American Fisheries Society, Humboldt Chapter, Humboldt State University, Arcata, CA. pp. 165-180

Angermeier, P.L., and Karr, J.R. 1984. Relationship between woody debris and fish habitat in a small warmwater stream. Trans. Am. Fish. Soc. 113: 716-726.

Armantrout, N.B. 1991. Restructuring streams for anadromous salmonids. Am. Fish. Soc. Symp. 10: 136-149.

Bassett, C.E. 1994. Use and evaluation of fish habitat structures in lakes of the eastern United States by USDA Forest Service. Bul. Mar. Sci. 55: 1137-1148.

Beechie, T.J., and Sibley, T. 1997. Relationship between channel characteristics, woody debris, and fish habitat in northwestern Washington Streams. Trans. Am. Fish. Soc. 126: 217-229.

Berejikian, B.A., Tezak, E.P., Flagg, T.A., LaRae, A.L., Kummerow, E., and Mahnken, C.V.W. 2000. Social dominance, growth, and habitat use of age-0 steelhead (*Oncorhynchus mykiss*) grown in enriched and conventional hatchery rearing environments. Can. J. Fish. Aquat. Sci. 57:628-636.

Beschta, R. L., Platts, W.S., Kauffman, J B., and Hill, M.T. 1994. Artificial stream restoration - money well spent or expensive failure? *In* Proceedings of Environmental Restoration, UCOWR 1994 Annual Meeting, Big Sky, Montana. pp. 76-104.

Bilby, R.E. 1984. Removal of woody debris may affect stream channel stability. J. For. 82: 609-613.

Bilby, R.E. and Fransen, B.R. 1996. Effect of habitat enhancement and canopy removal on the fish community of a headwater stream. Northwest Sci. 66: 137-137.

Bilby, R.E. and Likens, G.E. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61: 1107-1113.

Bilby, R.E. and Ward, J.W. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut and second-growth forests in southwestern Washington. Can. J. Fish. Aquat. Sci. 48: 2499-2508.

Binns, N.A. 1994. Long-term responses of trout and macrohabitats to habitat management in a Wyoming headwater stream. N. Am. J. Fish. Manage.14: 87-98.

Bisson, P.A., Nielsen, J.L., Palmason, R.A., and Grove, LE.. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low stream flows. *In* Acquisition and utilization of aquatic habitat inventory information. *Edited by* N. B. Armantrout. American Fisheries Society, Bethesda, Maryland. pp. 62-73.

Bisson, P.A., Sullivan, K, and Nielsen, J.L. 1988. Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead, and cutthroat trout in streams. Trans. Am. Fish. Soc. 117: 262-273.

Bisson, P.A., Bilby, R.E., Bryant, M.D., Dolloff, C.A., Grette, G.B., House, R.A., Murphy, M.L., Koski, K.V., and Sedell, J.R. 1987. Large woody debris in forested streams in the Pacific Northwest: Past, present, future. *In* Streamside management: Forestry and fisheries interactions. *Edited by* E. O. Salo and T. D. Cundy University of Washington College of Forest Resources, Seattle, WA. pp. 143-190.

Bisson, P.A., Quinn, T.P., Reeves, G.H., and Gregory, S.V. 1992. Best management practices, cumulative effects, and long-term trends in fish abundance in Pacific Northwest river systems. *In* Watershed Management. *Edited by* R. J. Naiman Springer-Verlag, New York. pp189-232.

Bohnsack, J.A., Eklund, A.M., and Szmant, A.M. 1997. Artificial reef research: Is there more than the attraction-production issue? Fisheries 22(4): 14-23.

Bond, C.E. 1963. Distribution and ecology of freshwater sculpins, genus *Cottus*, in Oregon. Doctoral Dissertation. University of Michigan, Ann Arbor.

Bradford, M.J., Taylor, G.C., and Allan, J.A. 1997. Empirical review of coho smolt abundance and the prediction of smolt production at the regional level. Trans. Am. Fish. Soc. 126:49-64.

Bustard, D.R., and Narver, D.W. 1975a. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). J. Fish. Res. Board. Can. 32: 667-680.

Bustard, D.R., and Narver, D.W. 1975b. Preferences of juvenile coho salmon (*Oncorhynchus kisutch*) and cutthroat trout (*Salmo clarki*) relative to simulated alteration of winter habitat. J. Fish. Res. Board. Can. 32: 681-687.

Campbell, R.F., and Neuner, J.H. 1985. Seasonal and diurnal shifts in habitat utilized by resident rainbow trout (*Salmo gairdneri*) observed in western Washington

Cascade mountain streams. *In* Symposium on small hydropower and fisheries. *Edited by* F.W. Olson, R.G. White, and R.H. Hamre. American Fisheries Society, Western Division and Engineering Section, Bethesda, Maryland. pp. 39-48.

Carle, F.L., and Strub, M.R.1978. A new method for estimating populations size from removal data. Biometrics 34: 621-630.

Cederholm, C. J., and Scarlett W.J. 1984. Seasonal immigrations of juvenile salmonids into four small tributaries of the Clearwater River, Washington, 1977-1981. *In* Proceedings of the Salmon and Trout Migratory Behavior Symposium. *Edited* by E.L. Brannon and E.O. Salo. School of Fisheries, University of Washington, Seattle. pp. 98-110.

Cederholm, C.J., and Scarlett, W.J. 1991. The beaded channel: A low-cost technique for enhancing winter habitat of coho salmon. *In* Fisheries Bioengineering Symposium. *Edited by* J. Colt and R.J. White. American Fisheries Society Symposium, no. 10. American Fisheries Society, Bethesda, Maryland. pp. 104-108.

Cederholm, C. J., Bilby, R.E., Bisson, P.A., Bumstead, T.W., Fransen, T.W., Scarlett, W.J., and Ward, J.W. 1997. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. N. Am. J. Fish. Manage. 17: 947-963.

Chapman, D.W. 1962. Aggressive behavior in juvenile coho salmon as a cause of emigration. J. Fish. Res. Bd. Can. 19: 1047-1079.

Chapman, D.W. 1966. Food and space as regulators of salmonid populations in streams. Am. Nat. 100: 345-357.

Chapman, D.W. 1996. Efficacy of structural manipulations of instream habitat in the Columbia River Basin. Northwest Sci. 5: 279-293.

Chen, G.K. 1998. The relationship between stream habitat complexity and anadromous salmonid diversity and habitat selection. Ph.D. Dissertation. Oregon State University, Corvallis.

Contor, C.R., and Griffith, J.S. 1995. Nocturnal emergence of juvenile rainbow trout from winter concealment relative to light intensity. Hydrobiol. 299: 179-183.

Corkran, C.C., and Thoms, C. 1996. Amphibians of Oregon, Washington and British Columbia. Lone Pine Publishing, Edmonton, Alberta.

Corn, S.P., and Bury, R.B. 1989. Logging in western Oregon: response of headwater habitats and stream amphibians. For. Ecol. Manage. 29: 39-57.

Crispin, V., House, R. and Roberts, D. 1993. Changes in instream habitat, large woody debris, and salmon habitat after the restructuring of a coastal Oregon stream. N. Am. J. Fish. Manage. 43: 96-102.

Cunjak, R.A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. Can. J. Fish. Aquat. Sci. 53: 267-282.

Daly, C., Neilson, R.P., and Phillips, D.L.. 1994. A statistical-topographic mold for mapping climatological precipitation over mountainous terrain. J. Appl. Meterol. 33: 140-158.

Dunham, J.B., and Vinyard, G.L. 1997. Incorporating stream level variability into analyses of site level fish habitat relationships: Some cautionary examples. Trans. Am. Fish. Soc. 126: 323-329.

Ehlers, R. 1956. An evaluation of stream improvement project devices constructed eighteen years ago. Cal. Fish. Game 42: 203-217.

Everest, F.H., Sedell, J.R. and Reeves, G.H. 1991. Planning and evaluating habitat projects for anadromous salmonids. *In* Fisheries and Bioengineering Symposium. *Edited by* J. Colt and R. J. White. American Fisheries Society, Bethesda, Maryland. pp. 68-77

Fausch, K.D., and Northcote, J.G. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Can. J. Fish. Aquat. Sci. 49: 682-693.

Fausch, K.D., and Bramblett, RG.. 1991. Disturbance and fish communities in intermittent tributaries of a western Great Plains river. Copeia 1991: 659-674.

Fausch, K.D., Karr, J.R. and Yant, P.R. 1984. Regional application of an index of biotic integrity based on stream fish communities. Trans. Am. Fish. Soc. 113: 39-55.

Finger, T.R. 1982. Interactive segregation among three species of sculpins *(Cottus*). Copeia 1982: 680-694.

Fraser, F.J. 1969. Population density effects on survival and growth of juvenile coho salmon and steelhead trout in experimental stream channels. *In* Symposium on salmon and trout in streams. *Edited by* T.G. Northcote. H.R. McMillian Lectures in Fisheries. University of British Columbia, Vancouver, B.C. pp. 253-266.

Fraser, N. H. C., Metcalfe, N.B., and Thorpe, J.E. 1993. Temperature-dependent switch between diurnal and nocturnal foraging in salmon. Proc. R. Soc. Lond. B. 252: 135-139.

Frissell, C. A., and R. K. Nawa. 1992. Incidence and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. N. Am. J. Fish. Manage. 12: 182-187.

Glova, G.J. 1986. Interaction for food and space between experimental populations of juvenile coho salmon (*Oncorhynchus kisutch*) and coastal cutthroat trout (*Salmo clarki*) in a laboratory stream. Hydrobiol. 155-168.

Glova, G.J. 1987. Comparison of allopatric cutthroat trout stocks with those sympatric with coho salmon and sculpins in small streams. Environ. Biol. Fish. 20: 275-284.

Gore, J.A.1985. The restoration of rivers and streams, theories and experience. Butterworth Publishers, Stoneham, MA.

Gorman, O.T., and Karr, J.R.1978. Habitat structure and stream fish communities. Ecology 59: 507-515.

Gowan, C. and K. D. Fausch. 1996. Mobile brook trout in two high-elevation Colorado streams: re-evaluating the concept of restricted movement. Can. J. Fish. Aquat. Sci. 53: 1370-1381.

Gowan, C., Young, M.K., Fausch, K.D., and Riley, S.C. 1994. Restricted movement in resident stream salmonids: A paradigm lost? Can. J. Fish. Aquat. Sci. 51: 2626-2637.

Grant, J.W.A., Englert, J., and Bietz, B.F. 1986. Application of a method for assessing the impact of watershed practices: Effects of logging on salmonid standing crops. N. Am. J. Fish. Manage. 6: 24-31.

Hall, J. D., Murphy, M.L., and Aho, R.S. 1978. An improved design for assessing impacts of watershed practices on small streams. Verh. Int. Ver. Limnol. 20: 1359-1365.

Hamilton, J.B. 1989. Response of juvenile steelhead to instream deflectors in a high gradient stream. *In* Practical approaches to riparian resource management. *Edited by* R.E. Gresswell, B.A. Barton, and J.L. Kershner. U.S. Bureau of Land Management, Billings, MT. pp. 149-158.

Hankin, D.G., and Reeves, G.H. 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Can. J. Fish. Aquat. Sci. 45: 834-844.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W., Cromack, K., and Cummins, K.W. 1986. Ecology of coarse woody debris in temperate ecosystems. Advan. Ecolog. Res. 15: 133-302.

Hartman, G. F. 1965. The role of behavior in the ecology and interaction of underyearling coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). J. Fish. Res. Bd. Canada. 22: 1035-1081.

Harvey, B.C., Nakamoto, R.J., and White, J.L.. 1999. Influence of large woody debris and a bankfull flood on movement of adult resident coastal cutthroat trout (*Oncorhynchus clarki*). Can. J. Fish.Aquat. Sci. 56: 2161-2166.

Hawkins, C.P., Murphy, M.L., Anderson, N.H., and Wilzbach, M.A..1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Can. J. Fish. Aquat. Sci**.** 40**:**1173-1185.

Heggenes, J., T. G. Northcote, and A. Peter. 1991. Spatial stability of cutthroat trout (*Oncorhynchus clarki*) in a small, coastal stream. Can. J. Fish. Aquat. Sci. 48:757-762.

Heifetz, J., Murphy, M.L., and Koski, K.V. 1986. Effects of logging on winter habitat of juvenile salmonids in Alaskan streams. N. Am. J. Fish. Manage. 6: 52-58.

Hicks, B. J., J. D. Hall, P. A. Bisson, J. R. Sedell. 1991. Responses of salmonids to habitat changes. Am. Fish. Soc. Spec. Pub. 19: 483-518 .

Higashi, G.R. 1994. Ten years of fish aggregation device (FAD) design development in Hawaii. Bull. Mar. Sci. 55: 651-666.

Holland, K., Brill, R., and Chang, R.K.C.1990. Horizontal and vertical movements of yellowfin and bigeye tuna associated with fish aggregating devices. Fishery Bul. 88: 493-507.

House, R. 1996. An evaluation of stream restoration structures in a coastal Oregon stream 1981 to1993. N. Am. J. Fish. Manage. 16: 272-281.

House, R.A., and Boehne, P.L. 1985. Evaluation of instream enhancement structures for salmonid spawning and rearing in a coastal Oregon stream. N. Am. J. Fish. Manage. 5: 283-295.

House, R.A., and Boehne, P.L. 1986. Effects of instream structures on salmonid habitat and populations in Tobe Creek, Oregon. N. Amer. J. Fish. Manage. 6: 38-46.

House, R. A. and Boehne, P.L. 1987. The effect of stream cleaning on salmonid habitat and populations in a coastal Oregon drainage. West. J. Appl. For. 2: 84-87.

House, R., Crispin, V., and Monthey, R. 1989. Evaluation of stream rehabilitation projects - Salem District (1981-1988). Technical Note, T/N OR-6, U.S. Department of Interior, Bureau of Land Management, Oregon State Office, Portland, Oregon.

House, R., Crispin, V., and Suther, J.M. 1991. Habitat and channel changes after rehabilitation of two coastal streams in Oregon. *In* Fisheries Bioengineering Symposium 10. *Edited by* J. Colt and. R.J. White. American Fisheries Society, Bethesda, Maryland. pp. 150-159.

Hunt, R.L. 1976. A long-term evaluation of trout habitat development and its relation to improving management-related research. Trans. Am. Fish. Soc. 105: 361-364.

Hunt, R.L. 1988. A compendium of 45 trout stream habitat development evaluations in Wisconsin during 1953-1985. Technical Bulletin 162, Wisconsin Department of Natural Resources, Madison.

Jearld, A. 1983. Age determination. *In* Fisheries Techniques. *Edited by* L. A. Nielson and D.L. Johnson. American Fisheries Society, Bethesda, MD. pp. 301-324.

Kahler, T.H. 1999. Summer movement and growth of individually marked juvenile salmonids in western Washington streams. M.S. Thesis, University of Washington, Seattle.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6): 21-27.

Kauffman, J.B., Beschta, R.L., Otting, N., and Lytjen, D.1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22: 12-24.

Kelch, D.O., Snyder, F.L.,and Reutter, J.M. 1999. Artificial reefs in Lake Erie: biological impacts of habitat alteration. Am. Fish. Soc. Symp. 22: 335-347.

Kelsey, K. 1995. Response of headwater stream amphibians to forest practices in western Washington. Doctoral Dissertation, University of Washington, Seattle.

Kelso, J.R.M. 1993. Substrate selection by *Geotria australis* ammocoetes in the laboratory. Ecol. Freshwat. Fish. 2(3):116-120.

Kelso, J.R.M. and Todd, P.R. Instream size segregation and density of *Geotria australis* ammocoetes in two New Zealand streams. Ecol. Freshwat. Fish. 2(3):108-115.

Klassen, H.D. 1991. Operational stream rehabilitation trial at Clint Creek, Sewell Inlet. Land Management Report 68, B.C. Ministry of Forests, Victoria, B.C.

Kondolf, G.M., and Micheli, E.R. 1995. Evaluating stream restoration projects. Enviro. Manage. 19:1-15.

Koski, K.V. 1992. Restoring stream habitats affected by logging activities. *In* Restoring the nation's marine environment. *Edited by* G. W. Thayer, Maryland Sea Grant College, College Park, MD. pp. 343-404.

Lindberg, W.J. 1997. Can science resolve the attraction-production issue? Fisheries 22(4):10-13.

Lisle, T.E. 1986. Effects of woody debris on anadromous salmonid habitat, Prince of Wales Island, southeast Alaska. N. Am. J. Fish. Manage. 6: 538-550.

Lisle, T.E. 1987. Using "residual depths" to monitor pool depths independently of discharge. Res. Note PSW-394, Pacific Southwest Forest and Range Experimental Station, Forest Service, U.S. Department of Agriculture, Berkeley, CA.

Lonzarich, DG. 1994. Stream fish communities in Washington: Patterns and processes. Doctoral Dissertation, University of Washington, Seattle, WA.

Lonzarich, D.G., and Quinn, T.P. 1995. Experimental evidence for the effect of depth and structure on the distribution, growth, and survival of stream fishes. Can. J. Zool. 73: 2223-2230.

Lyons, J., Wang, L. and Simonson, T.D. 1995. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. N. Am. J. Fish. Manage. 16**:**241-256.

Magurran, A.E. 1988. Ecological diversity and its measurement. Princeton University Press, Princeton, NJ.

Matthews, K.R. 1990a. An experimental study of the habitat preferences and movement patterns of copper, quillback, and brown rockfishes (*Sebastes* spp.). Environ. Biol. Fish. 29:161-178.

Matthews, K.R. 1990b. A telemetric study of the home ranges and homing routes of copper and quillback rockfishes on shallow rocky reefs. Can. J. Zool. 68: 2243-2250.

McCormick, S.D., Hansen, L.P., Quinn, T.P., and Saunders, R.L. 1998. Movement, migration, and smolting of Atlantic salmon. Can. J. Fish. Aquat. Sci. 55 (Supplement 1): 77-92.

McPhail, J.D. and Lindsey, C.C. 1986. Zoogeography of the freshwater fishes of Cascadia (the Columbia system and rivers north to the Stikine. *In* Zoogeography of North American freshwater fishes. *Edited by* C.H. Hocutt. and E.O. Wiley. John Wiley and Sons, New York. pp. 615-637.

Meehan, W.R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. Am. Fish. Soc. Spec. Pub. 19. Bethesda, MD.

Meffe, G.K. 1984. Effects of abiotic disturbance on coexistence of predator-prey fish species. Ecology 65: 1525-1534.

Mesick, C.F. 1988. Effects of food and cover on the numbers of Apache and brown trout establishing residency in artificial stream channels. Trans Am. Fish. Soc. 117: 421-431.

Minns, C.K., Kelso, J.R.M., and Randall, RG. 1996. Detecting the response of fish to habitat alterations in freshwater ecosystems. Can. J. Fish. Aquat. Sci. 53: 403-414.

Montgomery, D.R., and Buffington, J.M. 1997. Channel-reach morphology in mountain drainage basins. Geol. Soc. Am. Bull. 109: 596-611.

Montgomery, D.R., Buffington, J.M., Smith, R.D., Schmidt, K.M., and Pess, G.. 1995. Pool spacing in forest channels. Water Resourc. Res. 31: 1097-1105.

Moore, K.M.S., and Gregory, S.V. 1988. Response of young-of-the-year cutthroat trout to manipulation of habitat structure in a small stream. Trans. Am. Fish. Soc. 117: 162-170.

Moreau, J.K. 1984. Anadromous salmonid habitat enhancement by boulder placement in Hurdygurdy Creek, California. In Pacific Northwest Stream Habitat Management Workshop. Edited by T.J. Hassler. Humboldt State University, Arcata, CA. pp. 97-116.

Moyle, P.B. 1994. Biodiversity, biomonitoring, and the structure of stream fish communities. *In* Biological monitoring of aquatic systems. *Edited by* L. L. and. A. S. Stanford. CRC Press, Boca Raton, Florida. pp. 171-186

Murphy, M.L. 1995. Forestry impacts on freshwater habitat of anadromous salmonids in the Pacific Northwest and Alaska--requirements for protection and restoration. 7, NOAA Coastal Ocean Office, Silver Springs, MD.

Murphy, M.L., and Hall, J.D. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Can. J. Fish. Aquat. Sci. 38: 137-145.

Murphy, M.L., Thedinga, J.F., Koski, K.V., and Grette, G.B. 1984. A stream ecosystem in an old-growth forest in southeast Alaska: Part V: Seasonal changes in habitat utilization by juvenile salmonids. *In* Fish and Wildlife Relationships in Old-Growth Forests, Proceedings of Symposium, *Edited by* W.R. Meehan, T.R. Merrell, Jr., and T.A. Hanley. American Institute of Fishery Research Biologists. Juneau, Alaska. pp. 89-98.

Murphy, M.L., Heifetz, J., Johnson, S.W., Koski, K.V., and Thedinga, J.F. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Can. J. Fish. Aquat. Sci. 43: 1521-1533.

Murphy, M.L., Koski, K.V., Heifetz, J., Johnson, S.W., Kirchoffer, D. and Thedinga, J.F. 1985. Role of large organic debris as winter habitat for juvenile salmonids in Alaska streams. Preceedings of the Annual Conference of the western Association of Fish and Wildlife Agencies 64: 251-262.

Naslund, I., Milbrink, G., Eriksson, L.O., and Holmgren, P. 1993. Importance of habitat productivity differences, competition and predation for the migratory behavior of arctic charr. Oikos 66: 538-546.

National Research Council (NRC). 1996. Upstream: salmon and society in the Pacific Northwest. National Academy Press, Washington, D.C.

National Marine Fisheries Service (NMFS). 1995. Proposed recovery plan for Snake River salmon. U.S Department of Commerce National Oceanic and Atmospheric Administration, Silver Springs, MD.

Nickelson, T.E., Rodgers, J.D., Johnson, S.L., and Solazzi, M.F. 1992b. Seasonal changes in habitat use by juvenile coho salmon *(Oncorhyncus kisutch*) in Oregon coastal streams. Can. J. Fish. Aquat. Sci. 49: 783-789.

Nickelson, T.E., Solazzi, M.F., Johnson, S.L., and Rodgers, J.D. 1992a. Effectiveness of selected stream improvement techniques to create suitable summer and winter rearing habitat for juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. Can. J. Fish. Aquat. Sci. 49:790-794.

Nielsen, J.L. 1992. Microhabitat-specific foraging behavior, diet, and growth of juvenile coho salmon. Trans. Am. Fish. Soc. 121: 617-634.

Northcote, T.G. 1992. Migration and residency in stream salmonids - some ecological considerations and evolutionary consequences. Nordic J. Fresh. Res. 67 :5-17.

Pajos, T.A., and Weise, J.G. 1994. Estimating populations of sea lamprey with electrofishing sampling methods. N. Am. J. Fish. Manage. 14: 580-587.

Pasch, R.W. and Lyford, J.H. Jr. 1972. The food habits of two species of *Cottus asper* and Cottus *rhotheus*, in British Columbia. Copeia 954: 25-28.

Peterson, N.P. 1982. Immigration of juvenile coho salmon (*Oncorhynchus kisutch*) into riverine ponds. Can. J. Fish. Aquat. Sci. 39:1308-1310.

Platts, W.S., and Rinne, J.N. 1985. Riparian and stream enhancement management and research in the Rocky Mountains. N. Am. J. Fish. Manage. 5: 115-125.

Poulin, V.A., and Associates Ltd. 1991. Stream rehabilitation using LOD placements and off-channel pool development. Land Management Report 61, B.C. Ministry of Forests, Victoria.

Quinn, T.P., and Peterson, N.P. 1996. The influence of habitat complexity and fish size on over-winter survival and growth of individually marked juvenile coho salmon (*Oncorhynchus kistuch*) in Big Beef Creek, Washington. Can. J. Fish. Aquat. Sci. 53: 1555-1564.

Ralph, S.C., Poole, G.C., Conquest, L.L., and Naiman, R.J. 1994. Stream channel morphology and woody debris in logged and unlogged basins in western Washington. Can. J. Fish. Aquat. Sci. 51: 37-51.

Reeves, G.H., and Roelofs, T.D. 1982. Rehabilitating and enhancing stream habitat: 2 field applications. General Technical Report PNW-140, U.S. Forest Service., Portland, OR.

Reeves, G.H., Everest, F.H.,and Sedell, J.R. 1991b. Responses of anadromous salmonids to habitat modification: How do we measure them? Am. Fish. Soc. Sym. 10: 62-67.

Reeves, G.H., Hall, J.D., Roelofs, T.D., Hickman, T.L., and Baker, C.O. 1991a. Rehabilitating and modifying stream habitats. Am. Fish. Soc. Spec. Pub. 19: 519-557.

Reeves, G.H., Everest, F.H., and Sedell, J.R. 1993. Diversity of juvenile anadromous salmonid assemblages in coastal Oregon Basins with different levels of timber harvest. Trans. Amer. Fish. Soc. 122: 309-317.

Reeves, G.H., Hohler, D.B., Hansen, B.E., Everest, F.H., Sedell, J.R., Hickman, T.L., and Shively, D. 1997. Fish habitat restoration in the Pacific Northwest: Fish Creek of Oregon. *In* Watershed restoration: principles and practices. *Edited by* J.E. Williams, C.A. Wood, and M.P. Dombeck. American Fisheries Society, Bethesda, Maryland. pp. 335-359

Rhodes, J.S., and Quinn, T.P. 1998. Factors affecting the outcome of territorial contests between hatchery and naturally reared coho salmon parr in the laboratory. J. Fish Biol. 53:1220-1230.

Riley, S.C., and Fausch, K.D. 1995. Trout population response to habitat enhancement in six northern Colorado streams. Can J. Fish. Aquat. Sci. 52: 34-53.

Riley, S.C., Fausch, K.D., and Gowan,C. 1992. Movement of brook trout (*Salvelinus fontinalis*) in four small subalpine streams in northern Colorado. Ecol. Fresh. Fish 1:112-122.

Robison, E.G., and Beschta, R.L. 1990. Coarse woody debris and channel morphology interactions for undisturbed streams in southeast Alaska, U.S.A. Earth Surf. Proc. Land. 15: 149-156.

Rodgers, J.D., Johnson, S.L., Nickelson, T.E., and Solazzi, M.F. 1992. The seasonal use of natural and constructed habitat by juvenile coho salmon (*Oncorhynchus kisutch*) and preliminary results from two habitat improvement projects on smolt production in Oregon coastal streams. *In* Proceedings of the Coho Workshop, Nanaimo, B.C. May 26-28, 1992. *Edited by* L. Berg and P. W. Delaney. Nanaimo, B.C. pp. 334-351.

Roni, P. and A. Fayram. 2000. Estimating winter salmonid abundance in small western Washington streams: A comparison of three techniques. N. Am. J. Fish. Manage. 20:682-691.

Roper, B., Konnoff, D. Heller, D., and Wieman, K. 1998. Durability of Pacific Northwest instream structures following floods. N. Am. J. Fish. Manage. 18: 686-693.

Rosenfeld, J., Porter, M., and Parkinson, E. 2000. Habitat factors affecting the abundance and distribution of juvenile cutthroat trout (*Oncorhynchus clarki*) and coho salmon (*Oncorhynchus kistuch*). Can. J. Fish. Aquat. Sci. 57: 766-774.

Rosenfeld, J.S., and Hudson, J.J. 1997. Primary production, bacterial production, and invertebrate biomass in pools and riffles in southern Ontario streams. Arch. Hydrobiol. 139:301-316.

Sabo, J.L., and Pauley, G.B. 1997. Competition between stream-dwelling cutthroat trout (*Oncorhynchus clarki*) and coho salmon (*Oncorhynchus kisutch*): effects of relative size and population of origin. Can. J. Fish. Aquat. Sci. 54:2609-2617.

Schlosser, I.J. 1982. Trophic structure, reproductive success, and growth rate of fishes in a natural and modified headwater stream. Can. J. Fish. Aquat. Sci. 39: 968-978.

Schuett-Hames, D., Pleus, A., Bullchild, L., and Hall, S. 1994. Timber Fish and Wildlife Ambient monitoring program manual. TFW-AM9-94-001, Northwest Indian Fisheries Commission, Olympia, WA.

Scott, W.B. and Crossman, E.J. 1998. Freshwater fishes of Canada. Galt House Publishing Oakville, Ontario.

Sharma, R.1998. Influence of habitat on smolt production in coho salmon (Oncorhynchus kisutch) in fourteen western Washington streams. M.S. Thesis. University of Washington, Seattle.

Sedell, J.R., Swanson, F.J., and Gregory, S.V. 1984. Evaluating fish response to woody debris. *In* Pacific Northwest Stream Habitat Management Workshop. *Edited by* T.

J. Hassler. American Fisheries Society, Humboldt Chapter, Humboldt State University, Arcata, CA. pp. 222-245.

Shirvell, C.S. 1994. Effect of changes in streamflow on the microhabitat use and movements of sympatric juvenile coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*O. tshawytscha*) in a natural stream. Can. J. Fish. Aquat. Sci. 51: 1644-1652.

Solazzi, M.F., Nickelson, T.E., Johnson, S.L., and Rodgers, J.D. 2000. Effects of increasing winter rearing habitat on abundance of salmonids in two coastal Oregon streams. Can. J. Fish. Aquat. Sci. 57: 906-914.

Spalding, S., Peterson, N.P., and Quinn, T.P. 1995. Summer distribution, survival, and growth of juvenile coho salmon under varying experimental conditions of brushy instream cover. Trans. Am. Fish. Soc. 124:124-130.

SPSS. 1999. SPSS 9.0 Applications guide. SPSS Inc. Chicago, IL.

Swales, S., and Levings, C.D. 1989. Role of off-channel ponds in the life cycle of coho salmon (*Oncorhynchus kisutch*) and other juvenile salmonids in the Coldwater River, British Columbia. Can. J. Fish. Aquat. Sci. 46: 232-242.

Tarzwell, C.M. 1934. Stream Improvement Methods. Stream Improvement Bulletin R-4, Division of Scientific Inquiry, Bureau of Fisheries, Ogden, UT.

Thom, B. 1997. The effects of woody debris additions on the physical habitat of salmonids: A case study on the Northern Oregon Coast. M.S. Thesis, University of Washington, Seattle, WA.

Tripp, D.B. 1986. Using large organic debris to restore fish habitat in debris-torrented stream. B.C. Ministry of Forests, Land Management Report 47, Victoria.

Tschaplinski, P.J., and Hartman, G.F. 1983. Winter distribution of juvenile coho salmon (*Oncorhynchus kistuch*) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Can. J. Fish. Aquat. Sci. 40: 452-461.

United States Forest Service (USFS) and Bureau of Land Management (BLM). 1994. Record of Decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. Regional Ecosystems Office, Portland, Oregon.

Viola, A.E., Schuck, M.L., and Nostrant, S.A. 1991. An evaluation of instream habitat alterations in southeast Washington. FM 91-11, Washington Department of Wildlife, Fisheries Management Division. Olympia.

Weisser, J.W., and Klar, G.T. 1990. Electric fishing for sea lamprey (*Petromyzon marinus*) in the Great Lakes region of North America, p.59-64. *In* Developments in electric fishing. *Edited by* I.G. Cowx Blackwell Scientific Publications, Ltd. London.

Wilzbach, M.A. 1985. Relative roles of food abundance and cover in determining habitat distribution of stream-dwelling cutthroat trout. Can. J. Fish. Aquat. Sci. 42: 1668-1672.

Wydoski, R.S., and Whitney, R.R. 1979. Inland fishes of Washington. University of Washington Press, Seattle.

Young, M.K. 1994. Mobility of brown trout in south-central Wyoming streams. Can. J. Zool. 72: 2078-2083.

Zar, J.H. 1999. Biostatistical analysis. Prentice Hall, Upper Saddle River, New Jersey, USA.