Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams

Philip Roni and Thomas P. Quinn

Abstract: Thirty streams in western Oregon and Washington were sampled to determine the responses of juvenile salmonid populations to artificial large woody debris (LWD) placement. Total pool area, pool number, LWD loading, and LWD forming pools were higher in treatment (LWD placement) than paired reference reaches during summer or winter. Juvenile coho salmon (*Oncorhynchus kisutch*) densities were 1.8 and 3.2 times higher in treated reaches compared with reference reaches during summer and winter, respectively. The response (treatment minus reference) of coho density to LWD placement was correlated with the number of pieces of LWD forming pools during summer and total pool area during winter. Densities of age-1+ cutthroat trout (*Oncorhynchus clarki*) and steelhead trout (*Oncorhynchus 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 during summer was negatively correlated with increases in pool area. Trout fry densities did not differ between reaches, but the response of trout fry to treatment was negatively correlated with pool area during winter. Our research indicates that LWD placement can lead to higher densities of juvenile coho during summer and winter and cutthroat and steelhead during winter.

Résumé : Des échantillonnages dans trente cours d'eau de l'ouest de l'Oregon et du Washington ont permis d'étudier les réactions des populations de jeunes saumons à l'introduction artificielle de débris ligneux de grande taille (LWD) dans le lit du cours d'eau. La surface et le nombre de fosses, la charge de LWD, et l'incidence de fosses formées par la présence de LWD sont toutes plus élevés dans les zones expérimentales dans lesquelles on a ajouté des LWD, que dans les zones témoins appariées, tant en hiver qu'en été. Les densités des jeunes saumons coho (*Onchorhynchus kisutch*) sont 1,8 fois plus grandes que dans les zones témoins durant l'été et 3,2 fois plus élevées en hiver. La modification de la densité des saumons (densité expérimentale moins densité de la zone témoin) à la suite de l'introduction de LWD est reliée en été au nombre de pièces de LWD qui entraînent la formation de fosses et, en hiver, à la surface totale des fosses. Les densités de la truite fardée (*Oncorhynchus clarki*) et de la truite arc-en-ciel anadrome (*Oncorhynchus mykiss*) d'âge 1+ ne varient pas entre les zones expérimentales et les zones témoins en été, mais sont 1,7 fois plus élevées en hiver dans les zones expérimentales. La densité des truites arc-en-ciel d'âge 1+ en été est en corrélation négative avec l'augmentation de la surface des fosses. Les densités des alevins de la truite fardée ne varient pas d'une section à une autre, mais elles sont en relation négative avec la surface des fosses en hiver. Notre étude montre que l'addition de LWD peut conduire à des densités accrues de saumons coho juvéniles tant en été qu'en hiver, de même que des truites fardées et des truites arc-en-ciel en hiver.

[Traduit par la Rédaction]

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, at least for stream-rearing species (National Research Council 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; Chapman 1996). Unfortunately, the research and monitoring that has occurred has often been inadequate to determine the effectiveness of various stream restoration activities at increasing fish abundance (Reeves et al. 1991*b*; Chapman 1996).

Received April 5, 2000. Accepted November 8, 2000. Published on the NRC Research Press Web site on January 23, 2001.
J15706

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 DOI: 10.1139/cjfas-58-2-282

The restoration and enhancement of stream fish habitat has been occurring for more than 60 years in the United States (Reeves et al. 1991*a*). Stream restoration techniques were originally pioneered in the midwestern United States but have been modified for use in the steeper, high-energy streams common in western North America (Reeves et al. 1991*a*). 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 (Bisson et al. 1987). Prior to this time, "stream cleaning" (removal of wood from streams) was a common practice (Bisson et al. 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 (National Research Council 1996). In-stream 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, increasing overall stream productivity (Bisson et al. 1987). Consequently, in-stream LWD placement has become one of the most common techniques for improving 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) identified the need for better evaluation of habitat improvement projects. More recently, Reeves et al. (1991*a*), Chapman (1996), and Kauffman et al. (1997) reiterated the need for comprehensive evaluation of in-stream 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 studies have documented increases in pool frequency, pool depth, woody debris, and sediment retention following placement of in-stream structures (e.g., House et al. 1991; Riley and Fausch 1995; Cederholm et al. 1997). However, the extent to which the structures remain in place and functioning after several years is less clear (Frissell and Nawa 1992; Roper et al. 1998).

Biological evaluations of in-stream restoration have also produced highly variable results. There have been a number of encouraging reports of increased densities of salmonids following restoration efforts. House et al. (1989) and House (1996) reported increased juvenile coho salmon (*Oncorhynchus kisutch*) densities in several Oregon coastal streams, with small increases in densities of juvenile cutthroat trout (*Oncorhynchus clarki*) and steelhead trout (*Oncorhynchus mykiss*). 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 in-stream restoration efforts and the need for a multispecies approach.

Our research was developed to provide a broad-scale physical and biological evaluation of in-stream LWD placement efforts (restoration) in western Oregon and Washington. The overall objectives were to determine whether the artificial placement of LWD produces a significant change in physical habitat and juvenile salmonid abundance. Specifically, we tested the null hypotheses that paired treatment and reference reaches would not differ in (*i*) densities of woody debris and pool area or (*ii*) densities of juvenile coho salmon and cutthroat and steelhead trout in summer and winter and that (*iii*) the magnitude of fish response to treatment would not depend on the magnitude of change in habitat and (*iv*) the sizes of the fish would not differ between treatment and reference reaches.

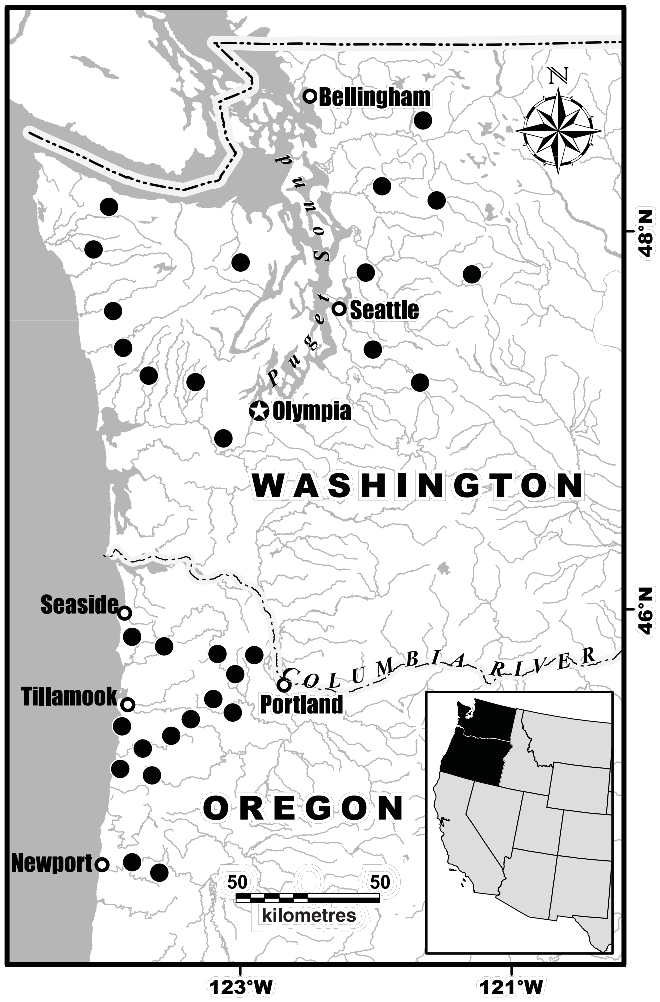
Materials and methods

We used the extensive posttreatment design (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 with pre- and post-treatment comparisons and comparisons between paired treatment and control streams. The extensive posttreatment design has frequently been used to assess the impacts of forestry and other land use practices on salmonids and their habitats and is particularly well suited to reach-scale habitat restoration projects (Hicks et al. 1991).

Thirty streams in western Oregon and Washington (Fig. 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. Study reaches were at least 10 times bankfull width in length, and most reaches were 100 m long. Each stream was surveyed once in summer (August–September) and winter (January–March) and reaches within a stream were sampled on the same day. Only those sites that had treatment and reference reaches of similar gradient, confinement, and channel width were selected.

The selection of study streams with paired treatment and reference reaches was based in part on physical and biological stream characteristics including stream size, bankfull width, channel type, and fish species composition. The selection of similar treatment and reference reaches is critical for a study of this nature, and it is important to control "background" features not influenced by LWD placement. In particular, reaches within a stream need to be of similar slope, width, discharge, and length. These features can affect the fish community and also influence the action of LWD. The placement of LWD can alter the number, area, and size of pools and the substrate volume and size (Bisson et al. 1987; Reeves et al. 1991*b*; Cederholm et al. 1997). Therefore, within a given stream, we selected treatment and reference reaches of similar slope (paired t test, $p = 0.46$) and channel width (paired t test, $p = 0.13$) (Table 1). We fixed the lengths of the reaches to be identical, and the proximity of the reaches insured that the discharges were essentially identical, although the distribution of point velocities might differ. More than 100 LWD placement projects were examined in western Washington and Oregon, but only 30 had suitable treatment and reference reaches with similar flow, channel width, gradient, confinement, and riparian vegetation. At nine sites, the biologist responsible for constructing the project selected treatment and reference reaches prior to treatment, and we used these as our treatment and reference reaches when possible. Only projects in

Fig. 1. Map of the 30 study streams in western Oregon and Washington, U.S.A.



which the artificially placed LWD remained in the channel after several high-water events, usually over several winters, were included. Reference reaches were located 200 m or more upstream from treatment reaches.

The study streams ranged from 4 to 12 m in bankfull width and from 0.5 to 4.2% slope (Table 1). The age of the restoration projects (date of last LWD placement to date of sampling) ranged from 1 to 10 years. Annual precipitation varied from and 107 to 315 cm. Dominant forest types were primarily Douglas-fir (*Pseudotsuga menziesii*), Sitka spruce (*Picea sichensis*), and western hemlock (*Tsuga heterophylla*). The dominant drainage geology was volcanic, sedimentary, or glacial–alluvial and varied by site but was consistent for reaches within a stream. The elevations of the study sites ranged from 12 to 789 m and drainage area upstream of our study reaches ranged from 124 to 2388 ha (Table 1).

Habitat within each stream reach was classified using a modification of the methods and habitat types described by Bisson et al. (1982). Total surface area of each habitat was estimated by measuring the total habitat 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 to distinguish between riffles and cascades. Discharge was estimated with a flowmeter prior to completion of each survey.

All natural and artificially placed LWD within the active channel greater than 10 cm in diameter and 1.5 m long was inventoried. The diameter class (small, 10–20 cm; medium, 20–50 cm; 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: (*i*) dominant — primary factor contributing to pool formation, (*ii*) secondary — influences zone of channel scour but not responsible for pool formation, and (*iii*) negligible — may provide cover but not involved in scour (Montgomery et al. 1995). Preproject maps and the presence of anchoring material (cable, nonnative boulders, or rebar) or aluminum tags (placed on wood during construction) were used to assist in classifying LWD as artificially placed or natural.

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 millimetre, and then released. Based on length–frequency distributions, 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 generally emerge from concealment 30–60 min after sunset at temperatures below 8–9°C (Roni and Fayram 2000). 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 in 1998 from January to mid-March 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 the diver's glove. Water temperature and flow were measured downstream of each site before electrofishing and snorkeling. Discharge and temperature among streams ranged from 0.10 to 0.15 $m^3 \cdot s^{-1}$ and from 7.5 to 16.9°C during electrofishing surveys and from 0.14 to 2.1 $m^3 \cdot s^{-1}$ and from 2.5 to 8.5°C during snorkel surveys.

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. 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 $\log_{10}(x + 1)$ transformation was used on fish data to meet the basic assumptions of a *t* test (additive data, normal distribution, equal variances) and ac-

Table 1. Physical characteristics of study streams measured during summer.

Stream	Dominant geology	Elevation (m)	Drainage area (ha)	Precipitation (cm·year^{-1})	Bankfull width (m)	Slope (%)		Pool area (%)	
						R	T	R	T
Oregon									
Bear	Volcanics	244	1580	320	10.1	1.2	1.5	0.29	0.79
Bergsvik	Sedimentary	122	540	308	9.2	1.0	0.9	0.81	0.76
Bewley	Sedimentary	12	639	235	6.8	0.5	1.1	0.55	0.62
Buster	Sedimentary	232	1627	228	7.8	0.8	0.5	0.89	0.79
Deer	Sedimentary	219	414	169	4.6	0.4	1.2	0.86	0.90
Elliott	Volcanics	427	720	236	11.0	1.4	2.4	0.55	0.55
Farmer	Sedimentary	73	727	260	7.3	1.8	1.6	0.34	0.42
Kenusky	Volcanics	207	1158	167	6.4	1.5	1.2	0.65	0.64
Killam	Volcanics	110	863	298	9.3	3.2	3.0	0.37	0.49
Klootchie	Sedimentary	61	1011	299	8.9	2.2	1.9	0.36	0.64
Lobster	Sedimentary	207	1254	233	9.9	1.8	1.7	0.44	0.65
Louisignont	Volcanics	244	1715	201	9.6	0.8	0.6	0.78	0.85
Ltl. Nestucca (South Fork)	Sedimentary	122	981	250	9.6	0.9	1.6	0.25	0.77
Rock (North Fork)	Volcanics	390	1893	286	9.9	1.3	0.7	0.25	0.52
Tobe	Volcanics	165	680	236	5.9	2.5	2.8	0.38	0.51
Washington									
Beaver	Glacial–fluvial	233	124	189	5.5	1.8	2.3	0.72	0.74
Benson	Sedimentary	320	459	217	11.6	1.8	1.9	0.35	0.31
Burn	Sedimentary	481	733	227	6.4	2.2	2.0	0.55	0.69
French	Igneous	172	1783	213	16.5	2.3	2.2	0.18	0.25
Harris	Volcanics	292	311	354	7.2	1.1	1.0	0.29	0.66
Hoppers	Volcanics	73	467	269	4.2	0.8	0.7	0.78	0.96
Hyas	Sedimentary	121	2000	290	12.2	1.3	0.7	0.36	0.59
Laughing Jacobs	Glacial–fluvial	23	335	119	6.8	2.5	2.3	0.19	0.38
Midnight	Igneous	598	567	212	5.6	3.9	4.5	0.31	0.34
Newbury	Volcanics	170	302	317	5.8	1.8	1.9	0.45	0.46
Porter	Volcanics	122	2388	170	10.0	1.3	2.3	0.56	0.67
Punch	Sedimentary	110	271	353	8.5	3.6	3.2	0.53	0.44
Shuwah	Sedimentary	197	305	297	6.5	1.4	1.9	0.56	0.80
Soosette	Glacial–fluvial	45	1225	108	11.1	1.7	1.7	0.19	0.29
Townsend	Volcanics	789	809	199	4.3	3.9	3.1	0.33	0.60

Note: Drainage area (area) for a given stream was calculated as total drainage area upstream of the restoration site. Geology and elevation are from unpublished U.S. Geologic Survey data. R, reference; T, treatment.

count for any zero or low counts. Multiple regression was used to examine the relationship(s) between fish response (ratio of treatment density to reference density) and difference in physical variables including pool area, percent pool area, riffle area, pieces of LWD, pieces of LWD creating pools (functional LWD), number of habitats, channel slope, geographic region (Washington or Oregon), and structure type (engineered or naturally placed log). All ratios of treatment to reference (pool area, pieces of LWD, etc.) were also log transformed ($\log_{10}(x)$) to meet the assumptions of a t test (additive data, normal distribution, equality of variances). 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

Treatment and reference reaches had identical lengths and were similar in slope and bankfull width. However, they differed in physical habitat features expected to respond to LWD placement. The total number of pieces of LWD per 100 m was significantly higher in treatment than in reference reaches during both summer (20–80 versus 8–63, $p < 0.01$) and winter (16–78 versus 4–64, $p < 0.01$) (Table 2) and averaged 1.83 and 1.89 times greater in treatment reaches in summer and winter, respectively. The total number of pieces of functional LWD was also significantly higher in treatment than in reference reaches during both summer and winter ($p < 0.01$) and averaged 2.83 and 2.96 times greater in treatment than in reference reaches. The difference in functioning LWD (LWD creating pools) between treatment and reference reaches was correlated with the difference in pool area ($p = 0.02$, $r^2 = 0.18$).

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). 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). Total number of habitat units (an indicator of habitat complexity) was not significantly different between treatment and reference reaches during summer ($p = 0.05$) but

Table 2. Number of pieces (per 100 m) of artificially placed, functioning (creating pools, summer only), and total LWD in reference (R) and treatment (T) reaches for each stream.

					Total LWD			
	Project	No. of pieces placed	Functioning LWD		Summer		Winter	
Stream	age	LWD (T only)	R	T	R	T	R	T
Oregon								
Bear	4	11	1	5	9	40	21	54
Bergsvik	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 Fork)	3	10	0	6	11	39	12	47
Rock (North Fork)	3	7	1	4	27	48	43	55
Tobe	4	10	0	8	28	52	22	42
Washington								
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

Note: LWD, large woody debris. Project age represents the number of years between LWD placement and our surveys.

was significantly higher in treatment reaches during winter ($p < 0.01$). Pool area in treatment reaches averaged 1.52 times that in reference reaches during summer and 1.51 during winter, and total wetted area increased by a factor of 1.11 in summer and 1.08 in winter. Treated reaches had 1.31 times more pools than reference reaches in summer and 1.48 times more in winter. The total number of habitat units was 1.11 and 1.22 times higher in treatment than in reference reaches during summer and winter, respectively.

Salmonid densities

Summer

Juvenile coho salmon densities (fish per metre) in summer were higher in treatment than in reference reaches (1.81 times higher, $p < 0.01$), but densities of age-1+ cutthroat, age-1+ steelhead trout, and trout fry did not differ ($p = 0.08$, 0.45, and 0.24, respectively) (Table 3). There was a positive linear relationship between the response of summer coho salmon densities (treatment to reference ratio) and the number of pieces of functioning LWD ($p < 0.01$, $r^2 = 0.25$) (Fig. 2). 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 ($p = 0.01$, $r^2 = 0.32$) and percent pool area ($p < 0.01$, $r^2 = 0.45$ (Fig. 2)) and positively correlated with a difference in riffle area ($p < 0.01$) but not with any other combination or individual physical variables. No response was detected between trout 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 1.73 times higher, respectively, during winter ($p < 0.01$) (Table 3). Trout fry den-

Table 3. Ratio (geometric mean) of salmonid densities for treatment to reference reaches for all 30 sites combined and separated by state.

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

*Significant difference ($p < 0.05$).

sities 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 with 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 ($p < 0.01$) (Fig. 3*a*). No relationships were detected between age-1+ cutthroat trout winter response to treatment and any combination of variables or single physical variable ($p > 0.36$). Age-1+ steelhead response was not correlated with any combination of variables or single physical variable ($p > 0.10$). Trout fry response to treatment was negatively correlated with difference in percent pool area ($p = 0.04$, $r^2 = 0.20$) (Fig. 3*b*).

Fish length

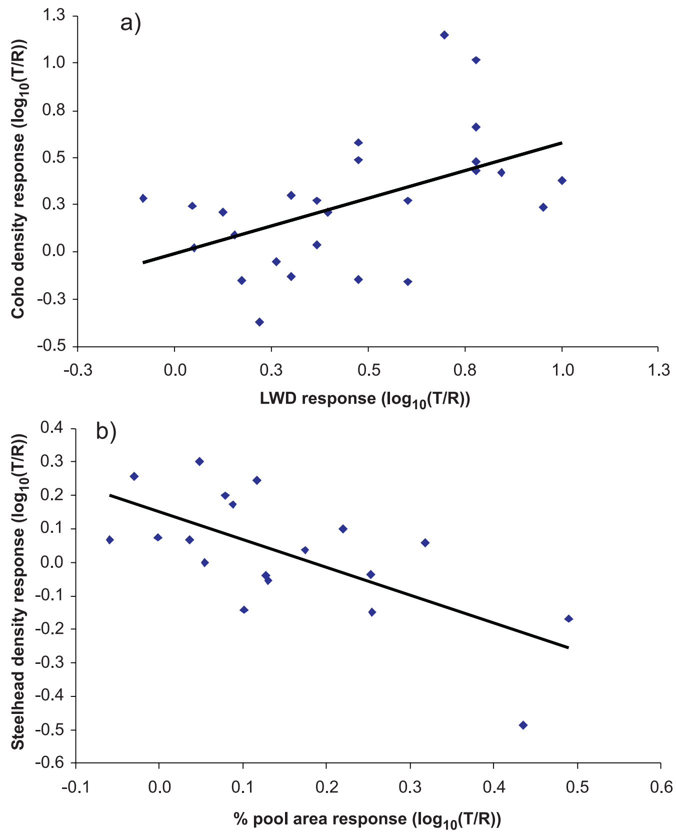
No differences were observed in mean lengths of coho salmon, cutthroat trout, steelhead trout, or trout fry between treatment and reference reaches during summer (paired *t* test, $p = 0.06$, 0.11, 0.56, and 0.50, respectively) or winter ($p = 0.29$, 0.22, 0.37, and 0.16, respectively). However, mean coho length was negatively correlated with coho density ($p < 0.01$, $r^2 = 0.29$) (Fig. 4*b*), but no difference existed between reaches (analysis of covariance, $p = 0.57$). The difference in coho length (treatment minus reference) during summer was negatively correlated with coho density response ($p < 0.01$, $r^2 = 0.29$) (Fig. 4*a*).

Discussion

Physical habitat

Increases in habitat complexity, pools, and slow-water habitats in response to habitat restoration, and specifically LWD placement, have been well documented in western North America (e.g., Riley and Fausch 1995; House 1996; Cederholm et al. 1997). Our results support these findings; an overall increase in pool area, number of pools, and LWD loading was detected in the 30 streams that we sampled. LWD counts differed between summer and winter in some of our sites primarily due to transport of small logs (10 cm in diameter, 1.5–3 m long) in or out of a reach between surveys. However, little difference was detected if only medium and large size-classes (20–50 and >50 cm) were examined.

Fig. 2. Relationship between (*a*) coho salmon response to restoration ($\log_{10}$(treatment (T) density/reference (R) density)) and change in LWD levels ($\log_{10}$(T/R)) for 27 sites inhabited by coho during summer ($y = 0.59x - 0.01$; $p < 0.01$, $r^2 = 0.25$) and (*b*) age 1+ steelhead trout response to restoration ($\log_{10}$(T density/R density)) and change in percent pool area ($\log_{10}$(T/R)) for 20 sites containing 1+ steelhead during summer ($y = -0.83x + 0.15$; $p < 0.01$, $r^2 = 0.45$). Treatment consisted of artificial placement of logs and log structures and reference represents unaltered stream reaches.



Thom (1997) examined physical responses of streams to LWD 1 year after treatment in six of the same streams that we sampled and reported significant increases in number and volume of LWD and 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 we may have seen a larger physical response than reported by Thom (1997) because we sampled most sites 3–4 years after LWD placement.

Salmonid densities

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. Our 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 et al. (1989) and House (1996) reported increased juvenile coho levels during summer in response to restoration in coastal

Fig. 3. Relationship between (*a*) juvenile coho salmon response to restoration ($\log_{10}$(treatment (T)/reference (R))) and change in pool area ($\log_{10}$(T/R)) during winter ($y = 0.25x + 0.04$; $p < 0.01$, $r^2 = 0.27$) for 24 sites inhabited by coho during winter and (*b*) trout fry response ($\log_{10}$(T/R)) to restoration and difference in percent pool area ($\log_{10}$(T/R)) for 20 sites inhabited by trout fry during winter ($y = -0.42x + 0.21$; $p = 0.04$, $r^2 = 0.20$). Treatment consisted of artificial placement of logs and log structures and reference represents unaltered stream reaches.

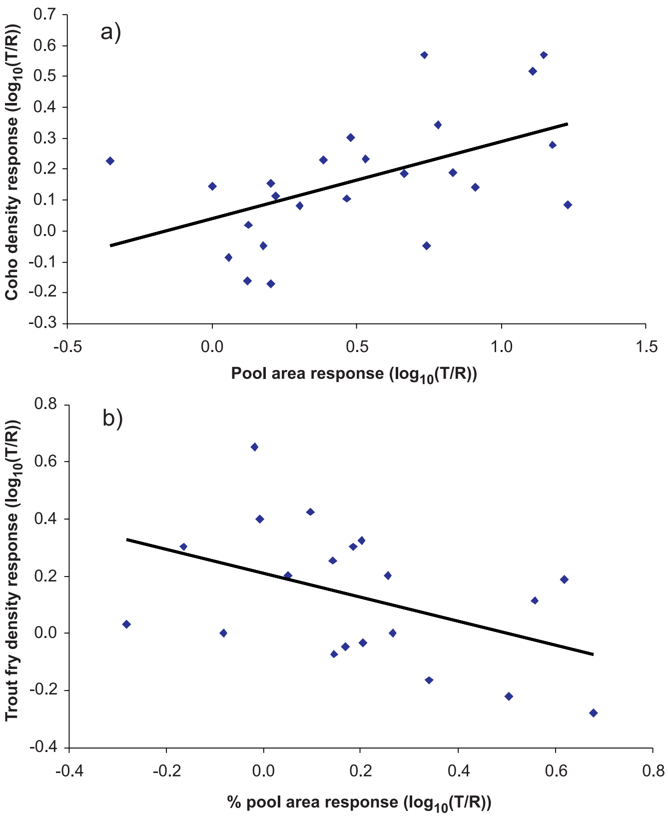
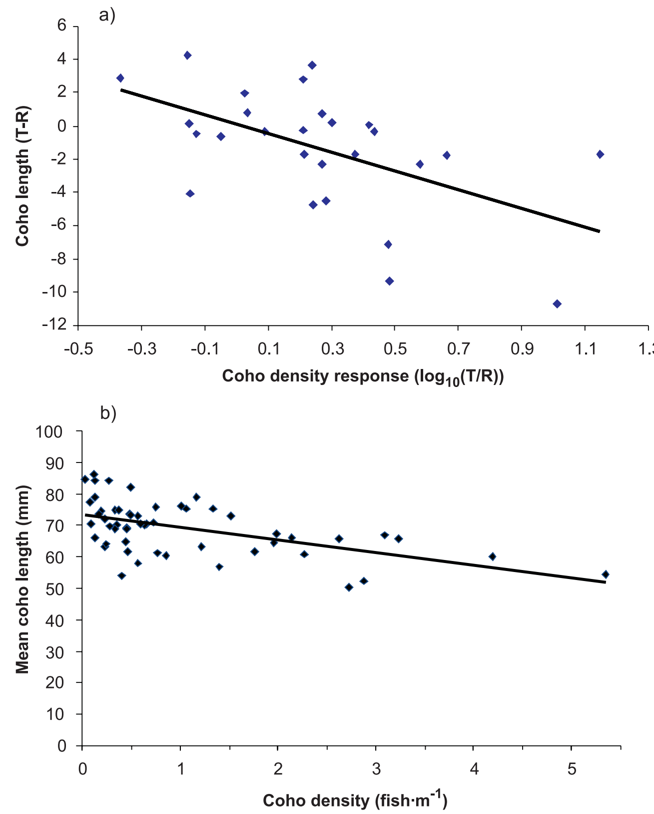


Fig. 4. Relationship between (*a*) mean coho salmon length (treatment (T) – reference (R)) and coho response to restoration ($\log_{10}$(T/R)) for 27 sites containing coho during summer ($y = -5.61x + 0.09$; $p < 0.01$, $r^2 = 0.29$) and (*b*) coho mean length and reach density ($y = 6.10x - 0.07$; $p < 0.01$, $r^2 = 0.29$).



Oregon streams. Nickelson et al. (1992*b*) 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. (1992*b*), these other evaluations occurred on individual streams and may not be broadly applicable elsewhere. Reeves et al. (1997) indicated that many shallow, low-gradient habitats such as glides were eliminated after restoration. We found that glides and shallow pools held the highest densities of juvenile coho during summer. Furthermore, juvenile coho occupy pools, glides, and other low-gradient habitats during summer but are almost exclusively found in pools and slack-water habitats during winter (Hartman 1965; Bustard and Narver 1975*a*; Bisson et al. 1982). Nickelson et al. (1992*a*) indicated that winter habitat was limiting coho salmon smolt production in many Oregon streams, and the larger response that we found for coho salmon during the winter compared to summer supports their findings.

We found that the response of coho to restoration was positively correlated with functioning (pool-forming) LWD during summer and an increase in pool area during winter. It is unclear why functioning LWD was not positively correlated with coho response during winter. However, determining the influence that an individual piece of LWD has on creating a pool is difficult at the higher flows common during our 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 1975*a*, 1975*b*; Nickelson et al. 1992*a*). LWD loading is positively correlated with both pool area and pool frequency (Montgomery et al. 1995). Therefore, our results are consistent with the general response of stream channels to LWD and the seasonal habitat preferences of juvenile coho salmon.

We detected an increase in density of juvenile cutthroat trout (age 1+) related to LWD placement in winter and a weak insignificant 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). Therefore, one might expect their response to restoration to be similar to but weaker than that of coho salmon. House et al. (1989) and

House (1996) 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 1986), and our results might have differed had we 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 we found no studies on the response of coastal cutthroat trout to restoration in winter months. Bustard and Narver (1975*b*) 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 led to habitat segregation, but aggression was low during winter and both species utilized pools, consistent with the observations of Bustard and Narver (1975*b*). The seasonal differences that we observed in the response of cutthroat trout to LWD placement are most likely due to a shift in habitat use and competition from summer to winter. We 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 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 (1996), Moreau (1984), and Chapman (1996) found no significant difference in steelhead parr or fry densities during summer following placement of in-stream structures. However, steelhead response to LWD placement in our study was negatively correlated with changes (increase) in pool area and positively correlated with changes (decrease) in riffle area. This is consistent with studies on habitat use that indicate that steelhead occupy riffles and fast-water habitats during summer (Bisson et al. 1982). 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 with 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, we 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 minus reference) and steelhead densities (treatment minus reference), even though our 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 1975*a*). 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 our inability to quantify microhabitats (substrate, velocity preferences, etc.) to which steelhead were responding. However, our 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.

We found no significant increase in trout fry (age-0 cutthroat and steelhead) densities during either summer or winter in relation to restoration. House et al. (1989) reported increases in trout fry densities following placement of in-stream 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). 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. We did not quantify edge or lateral habitats, although our 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 our study was negatively correlated with pool area during winter, suggesting that increasing deepwater 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 led 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

The purpose of most stream restoration efforts is to increase the abundance of fish. However, evaluations of restoration (including ours) 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 our 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) 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 we have no direct way to distinguish these factors.

Hamilton (1989) reported a twofold increase in steelhead parr in treated stream reaches, while numbers in untreated reaches decreased by half over a 2-year period. Initially, placement of in-stream 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*b*) suggested that monitoring for two generations might be needed to detect population response for anadromous salmonids in an individual stream. We 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 that we observed. Most of the sites taht we sampled in Washington consisted of "engineered LWD structures" such as log weirs and deflectors that 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 with 2.3 in Washington sites. While we were 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 sites.

The placement of LWD has been shown to trap gravels (House et al. 1989), and the placement of LWD likely led to differences in substrate size and possibly number of spawning salmonids in treatment and reference reaches. We did not quantify spawning salmonids or their habitat, but spawner abundance of coho was reportedly low in some Oregon streams that we sampled. Low numbers of adults may result in compensatory high survival rates of fry, and large-scale movements of juvenile coho during spring and fall (Kahler 1999) would lead to redistribution among treatment and reference reaches. The juvenile densities that we observed suggest that these streams were at moderate seeding levels, and we feel that differences between reaches are more likely related to differences in physical habitat rather than differences in spawner abundance.

Reeves et al. (1991*b*) suggested that response to restoration should be monitored at a watershed level through smolt trapping rather than at a reach scale. However, most in-stream projects occur at the 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 because of the large number of streams that we sampled. Because most coho are likely to have smolted shortly after our 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.

In summary, our 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 at 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 that we 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 functioning wood indicates that projects that dramatically increase pool area or LWD creating pools will provide the largest increases in fish abundance. However, we focused on forested sites, and the response in highly urbanized and agricultural areas may be different. Finally, our results in no way negate the need to focus on restoring natural processes that create and maintain salmon habitat rather than relying on in-stream manipulations that can be costly and not address the ultimate factors limiting habitat quality and salmonid production.

Acknowledgements

This study would not have been possible without the assistance of numerous individuals in selecting sites, including Barry Thom and Kelly Moore with the Oregon Department of Fish and Wildlife, Barry Olson, Karen Chang, Roger Nichols, and Phil Decillis with the U.S. Forest Service, Bob Alverts, Bob Rudiger, and others with the Bureau of Land Management, Kerry Bauman with King County, as well as others. Funding for this study was provided by the National Marine Fisheries Service with additional funds for Oregon sites provided by the Bureau of Land Management. Helpful input and comments were provided by Drs. Robert Bilby, Loveday Conquest, James Karr, Dave Montgomery, Robert Iwamoto, and Mike Murphy. We would also like to thank

Andy Fayram and others for their assistance with collecting field data and two anonymous reviewers for their helpful comments.

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