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REHABILITATION OF BEDROCK STREAM CHANNELS: THE EFFECTS OF BOULDER WEIR PLACEMENT ON AQUATIC HABITAT AND BIOTA†

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ABSTRACT

The placement of boulder weirs is a popular method to improve fish habitat, though little is known about the effectiveness of these structures at increasing fish and biota abundance. We examined the effectiveness of boulder weir placement by comparing physical habitat, chemical and biotic metrics in 13 paired treatment (boulder weir placement) and control reaches in seven southwest Oregon watersheds in the summer of 2002 and 2003. Pool area, the number of boulders, total large woody debris (LWD) and LWD forming pools were all significantly higher in treatment than control reaches ($p < 0.05$). No differences in water chemistry (total N, total P, dissolved organic carbon) or macroinvertebrate metrics (richness, total abundance, benthic index of biotic integrity etc.) were detected. Abundance of juvenile coho salmon (*Oncorhynchuskisutch*) and trout (*O. mykiss* and O. *clarki*) were higher in treatment than control reaches ($p < 0.05$), while dace (*Rhinichthys* spp.; $p < 0.09$) were more abundant in control reaches and no significant difference was detected for young-of-year trout ($p > 0.20$). Both coho salmon and trout response to boulder weir placement were positively correlated with difference in pool area; $p < 0.10$), while dace and young-of-year trout response to boulder weir placement were negatively correlated with difference in LWD ($p < 0.05$). The placement of boulder weirs appears to be an effective technique for increasing local abundance of species that prefer pools (juvenile coho and trout >100 mm). Based on our results and previous studies on bedrock and incised channels, we suggest that the placement of boulder structures is a useful first step in attempting to restore these types of stream channels. Published in 2006 by John Wiley & Sons, Ltd.

KEY WORDS: restoration; rehabilitation; boulders; weirs; habitat structures; salmonids; macroinvertebrates; coho salmon

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INTRODUCTION

Many streams in the North America, Europe and elsewhere have been degraded and greatly simplified by log drives (floating of logs), stream cleaning (removal of logs), and other forestry activities (e.g. Sedell and Luchessa, 1984; House and Boehne, 1987; Muotka *et al*., 2002; Erskine and Webb, 2003). The simplification and incision of stream channels is a problem not only in forested areas but also in many areas with intensive land use such as grazing, agriculture, urbanization or in regulated rivers (Platts, 1991; Booth, 1990; Buijse *et al*., 2002). In forests of the Pacific Northwest United States splash damming and stream cleaning have resulted in stream channels devoid of wood and boulders (Sedell and Luchessa, 1984) and often produced narrow stream channels scoured to bedrock (Montgomery *et al*., 2003). Several instream habitat improvement techniques have been employed to try to improve or restore these stream channels. Adding large woody debris (LWD) and other log structures are common methods of improving stream channels (Reeves *et al*., 1991; Roni and Quinn, 2001a; Roni *et al*., 2002). In areas where LWD of adequate length and diameter are not readily

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available, boulder clusters, weirs, and other structures have been used. The placement of boulders and boulder weirs is a prevalent rehabilitation technique in streams dominated by sedimentary rock, such as in the southwest Oregon coast. However, few studies have examined the physical, chemical or biological effectiveness of this technique.

The need for rigorous evaluation of instream habitat enhancement and watershed restoration efforts has been noted for many years (Reeves *et al*., 1991; Kondolf and Micheli, 1995; Chapman, 1996; Kauffman *et al*., 1997; Roni *et al*., 2005a)). Early evaluations of instream structures focused durability, longevity, and whether they created pools (Ehlers, 1956; Gard, 1972; Frissell and Nawa, 1992; see Roni *et al.*, 2005b for a thorough review). Early failures often resulted from applying techniques developed for low gradient (<1% slope) streams, such as weirs and deflectors, to high gradient and higher energy streams (Platts and Rinne, 1985; Frissell and Nawa, 1992; White, 2002). More recent studies have demonstrated lower failure rates and improvements in physical habitat most likely due to improvements in project design and the placement of structures that mimic natural wood and boulder accumulations (Roper *et al.*, 1998; Roni et al., 2005a,b). The effectiveness of wood placement on fish abundance has been examined in several recent studies (e.g. Cederholm *et al*., 1997; Reeves *et al*., 1997; Solazzi *et al*. 2000; Roni and Quinn, 2001a; Roni, 2003). Most of these studies demonstrated increases in juvenile coho salmon (*Oncorhynchuskisutch*) abundance following wood placement and variable results for trout. In contrast, research on the effectiveness of boulder weir placement in North American streams has been limited to a handful of case studies with limited information on fish responses or inconsistent results (Moreau, 1984; Hamilton, 1989; House *et al*., 1989). For example, Moreau (1984) reported a 100% increase steelhead parr densities 2 years after boulder structure placement in a northern California stream, but a 50% decline in steelhead parr numbers in nearby control reaches. Fontaine (1987) and Hamilton (1989) found no effect of placement of boulder structures on juvenile steelhead in a California and an Oregon stream. In contrast, Van Zyll De Jong *et al*. (1997) found boulder structures more successful than log structures at increasing juvenile Atlantic salmon (*Salmo salar*) abundance in a Newfoundland stream. Several European studies have suggested increases in brown trout (*Salmo trutta*) and other species due to these boulder treatments (Näslund, 1989; Hvidsten and Johnsen, 1992, Linlokken, 1997; O'Grady *et al*., 2002). For example, Hvidsten and Johnsen (1992) found increased densities of brown trout following placement of weirs and boulder substrate in a canalized reach of a Norwegian river. These limited studies on boulder structures suggest potential benefits for steelhead, brown trout, and Atlantic salmon, but more rigorous evaluation is needed for these and other species.

The response of macroinvertebrates to placement of boulder structures has been less frequently examined but similar to fishes, has produced equivocal results. Again, most studies have focused on log structures rather than boulder structures (e.g. Tarzwell, 1938; Gard, 1961; Wallace *et al*., 1995; Hilderbrand *et al*., 1997). Gortz (1998) and Negishi and Richardson (2003) reported increases in macroinvertebrate species composition and abundance following placement of boulders. In contrast, Tikkanen *et al*. (1994), Laasonen *et al*. (1998), and Brooks *et al*. (2002) detected no change in macroinvertebrate species composition or abundance following boulder placement. Muotka *et al*. (2002) re-examined some of the streams sampled by Laasonen *et al*. (1998) several years later and found that macroinvertebrate density and diversity in restored streams were similar to those in natural stream reaches but higher than those in channelized stream reaches; indicating that the invertebrate response to restoration may take several years. The difference in results of previous macroinvertebrate studies underscores the need for additional research on macroinvertebrate response to habitat rehabilitation.

The effects of instream structures on water chemistry and organic matter retention has rarely been examined in North American studies though it has been examined in some European stream rehabilitation projects. Muotka and Laasonen (2002) and Negishi and Richardson (2003), found increased organic matter retention following boulder placement in small streams. We found, however, no published studies on effects of instream structures on water chemistry. Given the importance of wood and other channel obstructions in trapping gravel and organic matter (Bilby and Likens, 1980), it is possible that placement of boulder weirs and other structures might lead to changes in water chemistry.

Existing monitoring and evaluation of stream restoration projects has generally focused on changes in physical habitat with relatively few comprehensive biological evaluations. Boulder placement is a technique that is in need of more thorough biological evaluation. The goals of our research were to examine the effects of boulder weir placement on, not only physical habitat but also water chemistry and nutrients, fishes and macroinvertebrates.

METHODS

We used the extensive post-treatment design (Hicks *et al*., 1991) to compare the response of habitat, macroinvertebrates, nutrient levels and juvenile fishes to boulders and boulder weirs placed in southwest Oregon streams. This design involves comparison between treatment and control reaches at a large number of sites after restoration and has been used widely to assess habitat alterations on salmonids (e.g. Murphy and Hall, 1981; Grant *et al*., 1986; Reeves *et al*., 1993; Roni and Quinn, 2001a). Thirteen paired treatment and control reaches in seven different streams in the lower Umpqua and Coquille River basins were sampled once in the late summer of 2002 or 2003 (Figure 1 and 2). Treatment was defined as the artificial placement of boulders and boulder weirs within the active stream channel. The weirs were semi-natural structures composed of several boulders (>50 cm in diameter) spanning the channel, but they were not solid structures and allowed for upstream and downstream fish passage. We selected stream reaches 200 m long in each stream (>10 times the bankfull channel width) and at least 200 m apart to assure that fish movement between treatment and control reaches was minimal during our study period (Kahler *et al*., 2001; Roni and Quinn, 2001b). In streams with multiple treatment and control reaches (Middle, Paradise, and West Fork of the Smith River), treatment-control pairs were located two or more stream kilometers apart. Paired treatment-control reaches within a stream were of similar slope, width, riparian vegetation, discharge and length (Figure 2). The proximity of the reaches insured that discharges between reaches were essentially identical, though the distribution of point velocities might differ. All streams in the study region had a similar legacy of splash damming, stream cleaning and other forestry activities that have resulted in highly uniform incised bedrock dominated channels with few boulders and low levels of woody debris.

Approximately 30 boulder weir placement projects were examined, but only 13 had suitable treatment and control reaches with similar flow, channel width, gradient, confinement and riparian vegetation. Previous studies, such as Frissell and Nawa (1992) have indicated that boulder weirs and other structures may fail due to design factors (inadequate structure size) or stream gradient and power, sediment load, or other watershed scale factors. These design issues had been addressed during project construction and we focused our attention on examining biological response to functioning boulder weirs. The number of boulder weirs (spanning entire channel) and

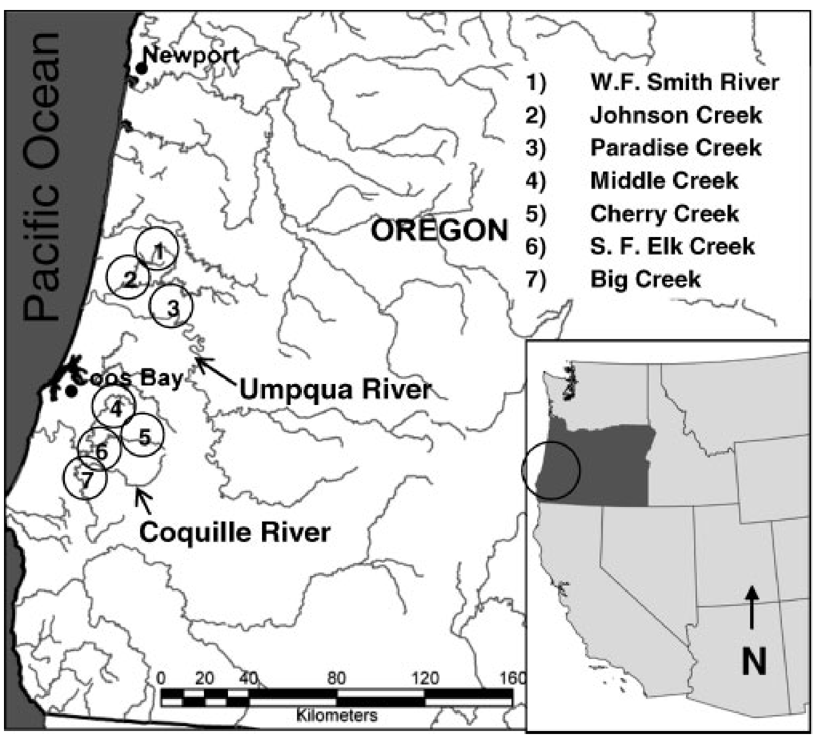

Figure 1. Map of streams sampled in southwest Oregon 2002 and 2003

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Figure 2. Example of typical control (top) and treatment (bottom) reaches from West Fork of Smith River. This figure is available in colour online at www.interscience.wiley.com/journal/rra

deflectors (spanning only a portion of channel) in treatment reaches ranged from 2 to 8 and project age at sampling ranged from to 1 to 20 years. Geology at most sites was sandstone and siltstone, except sites at Cherry and South Fork Elk creeks, which were predominantly mudstone and sandstone (Niem and Niem, 1990). Stream gradient ranged from 1 to 3% with treatment and control reaches being within 1% gradient of each other. Elevation of study sites ranged from approximately 75 to 150 m. Rainfall within watersheds ranges from 127 to 254 cm per year depending upon location and elevation. Riparian forests at study sites were dominated by deciduous trees including red alder (*Alnus rubra*), cottonwood (*Populus trichocarpa*), big leaf maple (*Acer macrophyllum*), as well as myrtle (*Umbellularia californica*) in sites in the Coquille basin. Conifers such as western red cedar (*Thuja plicata*), Douglas fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*) dominate upland areas in these basins and are also found in lower densities in riparian areas. Land use was predominantly commercial forest with most watershed composed of a large component of young (<25 years) to moderate age (25 to 80 years) forests.

We classified habitat units within each stream reach using a modification of the methods and habitat types described by Roni (2002) and Bisson *et al*. (1982) which included 7 slow water (pools and glides) and 3 fast water habitat types (riffles and cascades). Unique to these bedrock channels were bedrock pocket pools, which were glides consisting of several small (< 1 m in diameter) but deep (>30 cm) pools or depressions in the bedrock. Total surface area of each habitat was estimated by measuring the total habitat length and multiplying by the average of 3–5 width measurements. Discharge was estimated with a flow meter immediately following each survey. All boulders (rocks with an intermediate axis >0.5 m) and boulder weirs within the wetted channel were enumerated, the length and width measured, and whether they were natural or artificially placed noted. The diameter class (small: 10–20 cm, medium: 20–50 cm, and large: >50 cm) and length of all pieces of natural and artificially placed LWD within the wetted stream channel greater than 10 cm in diameter and 1.5 m long were recorded. The function of an individual piece of LWD or a boulder was classified into three categories based on its influence on pool formation: (1) dominant—primary factor contributing to pool formation, (2) secondary—influences zone of channel scour but not responsible for pool formation, or (3) negligible—may provide cover but not involved in scour (Montgomery *et al*., 1995). In addition, we visually estimated the percent of each piece of LWD that was in the low-flow wetted channel and within the bankfull channel.

Fishes in each habitat were enumerated using snorkel surveys. Endangered species concerns and the relatively large wetted stream width precluded the use of electrofishing in most of our study sites. 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 (Roni and Fayram, 2000). In streams greater than 10 m wide, two snorkelers worked side by side to cover the entire width of the stream. 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 snorkeling. Discharge and temperature among streams ranged from 0.01 to $0.12\,m^3\,s^{-1}$ and 11–15°C during snorkel surveys. The accuracy of snorkeling at enumerating juvenile fishes can be difficult in shallow water habitats such as riffles (Roni and Fayram, 2000). However, in our study the predominantly bedrock substrate in riffles coupled with low numbers of fish in shallow water habitats made estimation of fish abundance riffles relatively easy and accurate.

Common species observed during snorkel surveys included coho salmon, cutthroat and steelhead trout, three spine stickleback (*Gastreosteus aculeatus*) and dace (*Rhinichthys* spp.). Due to difficulty in distinguishing reliably between cutthroat and steelhead trout during snorkel surveys, they were referred to collectively as trout. Based on length frequency distributions, trout were separated into two age groups: all trout greater than or equal to 100 mm in length were considered age 1+ and referred to as trout, and all those <100 mm were considered young-of-year. Other species observed in small numbers included redside shiners (*Richardsonius balteatus*) and juvenile Chinook salmon (*Oncorhynchus tshawytscha*). Benthic species, such as larval lamprey (*Lampetra* spp.), Pacific giant salamanders (*Dicamptodon tenebrosus*) and sculpin (*Cottus* spp.) were present but rarely observed during snorkel surveys.

Benthic macroinvertebrates were collected in late summer and early fall, the typical index period for invertebrate sampling in the coastal Pacific Northwest streams as flows are relatively stable, taxa richness is high, and spawning anadromous fish have not yet begun to return in high numbers (Fore *et al*., 1996; Morley and Karr, 2002). At each control and treatment reach, a Surber sampler (500-μm mesh, $0.1\,m^2$ frame) was used to collect invertebrates from three separate riffles. These riffles were evenly spaced within a 200 m reach and chosen to be as similar as possible in regards to surface substrate, water depth, and canopy cover. Where present, riffles containing gravel (as opposed to bare bedrock) were targeted. In order to collect an adequate sample size, the Surber sampler was placed at three random locations within each riffle; these three samples were then combined for each of the three sample riffles. Substrate within the Surber frame was disturbed to a depth of 10 cm for a 2-min period. Mineral material was washed and removed from the sample, and all organic material retained on a 500-μm-mesh sieve preserved in 70% ethanol. Invertebrates were identified to genus (except where impractical; e.g. Chironomidae), and classified according to functional feeding group, voltinism, and disturbance tolerance (Merritt and Cummins, 1996; Barbour *et al*., 1999).

Invertebrate samples were analysed in four ways: (1) total abundance, (2) total taxa richness, (3) relative abundance (proportion of total abundance) of functional feeding groups (shredders and collectors) orders and EPT taxa (insects from the orders diptera and combined ephemeroptera, plecoptera and tricoptera), and (4) benthic index of biological integrity (B-IBI; Kerans and Karr, 1994; Fore *et al*., 1996). The B-IBI is a 10 metric regionally calibrated index that produces a reach-specific score of biological condition ranging from 10 to 50 (Dewberry *et al*., 1999; Karr and Chu, 1999; Morley and Karr, 2002). One B-IBI value per stream reach was calculated based on values from the three riffles: total abundance, taxa richness, and relative abundance of EPT and shredder and collector taxa were averaged across the three riffles. These response variables were selected based on previous studies that examined the effects of habitat enhancement on invertebrates (Wallace *et al*., 1995; Hilderbrand *et al*., 1997; Larson *et al*., 2001).

In conjunction with invertebrate sampling, three water samples were taken from the downstream (0 m), middle (100 m) and upstream end (200 m) of each study reach. Immediately after collection water samples were frozen for later analysis of dissolved organic carbon, total nitrogen and phosphorous, and nutrient concentrations (e.g. NO_3, NO_4) using a spectrophotometer. Because no longitudinal trends were detected among the water quality sample, the mean level for each water chemistry parameter was calculated by averaging the three samples for each reach.

Differences in habitat, LWD, and abundance of fish between treatment and control reaches were compared using paired *t*-tests. Fish numbers were $\log_{10}$ transformed to meet basic assumptions of a t-test (normal distribution, equal variances; Zar, 1999). Because detailed multiple linear regression was not believed to be appropriate given our small sample size ($n = 13$), simple correlation analysis (Pearson's correlation) was used to examine the relationship(s) between fish response ($\log_{10}$(treatment density/reference density)) and key physical variables including pool area, total LWD, LWD forming pools, boulder weirs, total boulders and project age. Pool area and LWD levels are known to be correlated with abundance and size of salmonid fishes (Bisson *et al.*, 1982; Nickelson *et al.*, 1992; Roni and Quinn, 2001a) and sites with larger physical responses to restoration were predicted to have larger biological responses. All ratios of treatment to control (e.g. pool area, pieces of LWD etc.) were also log transformed ($\log_{10}x$) to normalize residuals and meet statistical assumptions of linear regression. A $\log_{10}(x + 1)$ transformation was use on LWD, boulder, dace, and trout data to adjust for zeros in some fields (Zar, 1999). A 0.10 level of significance was used for all statistical tests.

RESULTS

Pool area, large woody debris, pool-forming LWD and boulder abundance were significantly higher in treatment than control reaches though considerable variation in response existed among sites ($p < 0.05$; Table I). In contrast, total number of habitat units was higher in control than treatment reaches ($p < 0.05$) and no difference was detected between total number of pools ($p = 0.90$). No difference existed in concentrations of DOC, total phosphorus, phosphate, SiO_4, total nitrogen or components of nitrogen (NO_3or NH_4) between treatment and control reaches ($p > 0.10$; Table II).

Juvenile coho salmon numbers were significantly higher in treatment than control reaches ($p < 0.01$), averaging 1.4 times the number found in control reaches. Number of trout larger than 100 mm were also higher in treatment than control reaches ($p = 0.05$) and lower for dace ($p < 0.09$) while differences for other species (young-of-year trout, dace, stickleback) were not significant (Table III). Macroinvertebrate abundance, total taxa richness; relative abundance of EPT, shredders, and collectors, and B-IBI did not differ between treatment and control reaches (Table IV).

Pearson correlation analysis indicated that positive correlations existed between coho response ($\log_{10}$(treatment/control)) and percent pool area ($\log_{10}$(treatment/control); Pearson correlation $= 0.51$, $p = 0.08$) and also for trout response and pool area response (Correlation $= 0.54$; $p = 0.06$; Figure 3; Table V). Both YOY trout and dace response ($\log_{10}$(treatment/control)) to boulder weir placement were negatively correlated with difference in LWD ($\log_{10}$(treatment/control); Correlation $= -0.70$ and 0.77 for YOY trout and dace, respectively; $p < 0.01$). Project age, boulders, the number of boulder weirs, and LWD forming pools ($\log_{10}$(treatment/control)) were not significantly correlated with any fish species response.

DISCUSSION

Boulder weir placement produced the predicted changes in physical habitat including increased pools, LWD, and boulders as well as an increase in fish abundance. This is consistent with findings for many other instream habitat rehabilitation methods, which have reported improvements in physical habitat following treatment (see Roni *et al.*, 2002, 2005a for a thorough review). The number of habitat units was actually lower in treated stream reaches, most likely because boulder weirs typically create large pools more than 20 m long. While boulder weirs modify physical habitat they appear to have little effect on water chemistry and nutrient levels (e.g. P, N, dissolved organic carbon).

We detected significantly higher numbers of juvenile coho and 1+ trout in response to boulder weir placement, suggesting that boulder weirs are an effective method of creating summer habitat for juvenile coho salmon and age

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Table I. Physical characteristics of study sites and results of paired t-tests (t-statistic, p-value) sampled in 2002 (Big, Cherry, Johnson, Middle, Paradise, S.F. Elk) and 2003 (W.F. Smith). Statistical comparisons of treatment and control reaches were performed on $\log_{10}$ transformed data

Stream/Reach (year constructed)	Number of habitats Units		Number of pools		Percent pool area		Function-ing LWD		Total LWD		Total boulders		Weirs & deflectors
	C	T	C	T	C	T	C	T	C	T	C	T	
Big Creek (1997)	17	7	8	4	0.74	0.84	0	0	18	15	0	109	3
Cherry Creek (2001)	15	17	6	9	0.18	0.76	0	0	9	5	3	954	7
Johnson Creek (1987)	29	20	14	10	0.70	0.72	4	7	48	32	733	503	4
Middle Creek I (1996)	19	15	7	10	0.51	0.81	0	4	12	24	3	335	4
Middle Creek II (1993)	26	29	12	21	0.52	0.72	0	0	39	31	214	178	2
Paradise Creek I (1986)	25	17	12	14	0.44	0.95	0	5	21	50	119	295	4
Paradise Creek II (1986)	37	30	22	16	0.58	0.59	0	5	21	24	132	128	7
S. Fork Elk Creek (1996[1])	30	29	18	13	0.59	0.68	2	11	18	73	6	199	4
WFS Beaver (1994[2])	15	14	7	9	0.42	0.93	6	1	22	88	21	635	8
WFS Crane (1983[2])	18	19	11	13	0.70	0.73	0	0	11	18	205	328	4
WFS Moore (1989)	21	14	8	6	0.61	0.63	0	5	9	34	26	463	4
WFS Skunk (1999)	16	15	9	9	0.70	0.93	0	0	20	153	48	133	3
WSF Upper (1989)	27	29	13	15	0.53	0.54	2	2	53	103	48	257	3
t-statistic	−2.240		0.057		2.909		2.212		2.600		3.803		—
p-value	0.045		0.955		0.013		0.051		0.023		0.003		—

[1]LWD added in 1999.
[2]Boulder clusters added in 1999.
WFS = West Fork of Smith River. C = control reaches, T = treatment reaches.

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Table II. Average nutrient levels in study reaches of 13 study sites and results of paired *t*-tests (*t*-statistic, *p*-value). Statistical comparisons of treatment and control reaches were performed on $\log_{10}$ transformed data

	DOC		Total phosphorus		Total nitrogen		PO_4		SiO_4	
	C	T	C	T	C	T	C	T	C	T
Big Creek	2.1	2.3	32.4	33.0	186.2	207.6	4.8	4.1	4967.9	4585.7
Cherry Creek	2.2	1.9	41.3	40.3	189.5	260.8	9.6	11.1	4392.3	5148.7
Johnson Creek	2.6	2.7	30.6	31.9	188.4	185.0	4.1	2.1	2985.5	2160.9
Middle Creek I	2.9	3.0	38.3	35.3	204.1	168.1	6.2	5.6	4892.0	2367.0
Middle Creek II	1.8	1.6	34.4	35.2	162.5	159.3	7.8	7.6	5128.8	4572.4
Paradise Creek I	2.8	2.8	51.9	61.8	140.5	220.0	13.0	14.9	5832.9	7005.9
Paradise Creek II	2.5	3.0	63.9	53.9	236.0	153.4	14.1	12.9	4830.9	4708.1
South Fork Elk	1.7	1.5	36.1	34.1	204.5	164.9	9.8	10.1	4891.8	5365.4
WFS Beaver Reach	1.2	1.3	47.0	44.6	256.7	258.0	5.5	5.4	1388.3	1191.0
WFS Crane Reach	1.8	1.7	38.3	37.0	208.4	211.1	3.6	4.0	4139.6	1600.1
WFS Moore Reach	1.9	1.7	40.7	40.3	224.2	239.4	4.9	4.8	4578.2	4624.1
WFS Skunk Reach	1.1	0.9	40.4	41.8	207.8	236.9	5.0	4.9	3829.9	3893.0
WFS Upper Reach	0.4	0.8	34.2	45.8	189.0	262.8	4.4	6.1	2952.3	3230.8
t-statistic	0.482		0.368		−0.625		−0.468		−1.480	
p-value	0.638		0.719		0.543		0.648		0.165	

WFS = West Fork of Smith River. C = control reaches, T = treatment reaches.

1 and older juvenile trout. These results are also consistent with previous studies on coho, cutthroat and steelhead trout for both boulder and LWD placement (e.g. Ward and Slaney, 1981; Moreau, 1984; Fontaine, 1987; House *et al*., 1989; Cederholm *et al*., 1997; Roni and Quinn, 2001a; Roni, 2003), as well as with studies on brown trout and Atlantic salmon (e.g. Näslund, 1989; Linlokken, 1997; Van Zyll De Jong *et al*., 1997; O'Grady *et al*., 2002). The correlation between percent pool area and fish response for both coho and age 1+ trout was expected given their

Table III. Total fish numbers (fish per 200 m) in treatment and control reaches of 13 study sites and results of paired *t*-tests (*t*-statistic, *p*-value). Statistical comparisons of treatment and control reaches were performed on $\log_{10}$ transformed data.

Stream	Coho		Dace		Stickleback		Trout < 100		Trout - > 100	
	C	T	C	T	C	T	C	T	C	T
Big Creek	298	402	362	297	131	369	5	1	3	6
Cherry Creek	366	716	101	183	493	194	2	13	2	7
Johnson Creek	294	323	0	0	0	0	15	20	3	6
Middle Creek I	82	134	17	5	14	33	2	0	0	2
Middle Creek II	413	648	9	14	0	0	4	40	0	4
Paradise Creek I	140	372	0	0	0	0	3	0	6	16
Paradise Creek II	181	140	0	0	0	0	25	14	4	1
S.F. Elk Creek	217	380	1	0	0	0	41	3	4	7
WFS Beaver Reach	265	285	5	4	0	0	98	61	8	19
WFS Crane Reach	568	494	183	88	0	0	43	23	9	4
WFS Moore Reach	329	501	38	22	0	0	102	39	2	1
WFS Skunk Cabbage Reach	560	791	32	6	0	0	135	27	3	10
WFS Upper Reach	479	719	0	0	0	0	119	149	2	2
t-statistic	3.659		−1.945		NA		−1.334		2.195	
p-value	0.003		0.088		NA		0.207		0.05	

WFS = West Fork of Smith River. C = control reaches, T = treatment reaches. NA = not applicable.

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Table IV. Selected macroinvertebrate metrics measured in treatment and control reaches of 13 study sites and results of paired *t*-tests (*t*-statistic, *p*-value). Statistical comparisons of treatment and control reaches were performed on $\log_{10}$ transformed data

Stream	Abundance											
	Total Abundance		Relative EPT Taxa		Relative Shredder		Relative Collector		Taxa Richness		B-IBI	
	C	T	C	T	C	T	C	T	C	T	C	T
Big Creek	1275	655	62	26	2	2	28	21	32	37	30	30
Cherry	1524	2315	52	63	1	5	12	19	36	40	32	36
Johnson	299	887	29	45	17	4	33	43	27	40	24	32
Mid I	1098	188	41	53	2	2	22	34	28	25	30	28
Mid II	1237	710	57	46	7	7	27	19	45	42	38	36
Paradise I	988	1230	67	36	12	6	29	32	51	40	40	34
Paradise II	710	1036	73	48	4	11	41	23	31	31	32	30
S. Fork Elk	3260	885	50	34	10	2	22	15	44	43	38	36
WFS Beaver	862	608	41	49	4	7	49	46	47	49	44	44
WFS Crane	1257	1625	48	39	3	7	46	34	33	37	30	30
WFS Moore	454	2366	53	44	4	9	29	36	48	41	42	36
WFS Skunk	1094	995	58	54	11	9	34	30	54	45	46	38
WFS Upper	1112	2207	53	56	6	14	40	45	47	46	44	40
t-statistic	−0.110		−1.329		0.414		−0.472		−0.057		−0.962	
p-value	0.991		0.208		0.686		0.646		0.956		0.355	

WFS = West Fork of Smith River. C = control reaches, T = treatment reaches.

preference for pool habitat (Bisson *et al*., 1988; Roni and Quinn, 2001a, Roni, 2003) and the fact that placement of boulder weirs led to an increase in pool area. It should also be noted that steelhead and cutthroat trout have different habitat preferences (Bisson *et al*., 1988, Roni, 2003) and had we been able to reliably distinguish between steelhead and cutthroat trout, we may have seen detected slightly different responses for both size groups of trout.

The lack of response of both young-of-year trout (steelhead and cutthroat combined) is also partially supported by previous studies, although the results of placement of instream structures on small cutthroat and steelhead trout have produced mixed results. For example, Hamilton (1989), House (1996), Cederholm *et al*. (1997) and Roni and Quinn (2001a) detected no significant response of young-of-year trout to placement of instream structures, while Reeves *et al*. (1997) found a significant decline. Trout fry (YOY) show no strong preferences for pools (Bisson *et al*., 1988; Roni, 2002) and prefer stream margins at least during summer (Hartman, 1965; Moore and Gregory, 1988). Moreover, Roni and Quinn (2001a) found a negative relationship between winter trout fry response to restoration and percent pool area and suggested that placement of pool-forming structures leads to a decrease in shallow edge habitat preferred by YOY. Moore and Gregory (1988) demonstrated that manipulation of edge habitat can lead to large changes in YOY trout abundance. Cederholm et al. (1997) suggested that the lack of YOY steelhead response they observed in there evaluation of LWD placement may have been due to loss of pool habitat and increased predation. Loss of preferred habitat for YOY trout and increased predation in pools may in part explain the lack of response to boulder weir placement and negative correlation we observed between YOY trout and woody debris.

Few studies have examined the response of non-salmonid fishes to the placement of instream structures and we found no studies that specifically examined the response of dace. Shields *et al*. (1995a,b) found a decrease in the proportion of cyprinds and an increase in centrarchids following placement of stone weirs. In our study dace showed little response to boulder structure placement though the negative correlation between dace and LWD suggests that increases in cover, habitat complexity, and pool area do not necessarily benefit dace. Similar to young-of-year trout, longnose and speckled dace prefer shallow habitats such as glides and riffles (Wydowski and Whitney, 2003). Dace in our study were most frequently observed in glides or in shallow water habitat and the large deep

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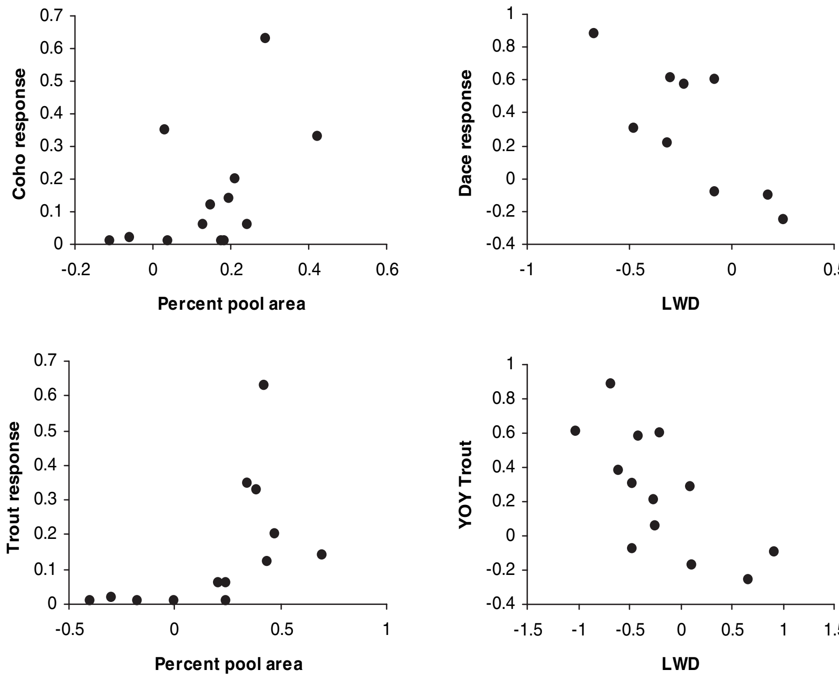

Figure 3. Partial correlation plots between fish (coho, trout, dace, YOY trout) response and percent pool or LWD response to boulder weir placement. All axes are a $\log_{10}$ scale ($\log_{10}$ (treatment/control)

pools typically created by boulder weirs and woody debris most likely eliminated preferred dace summer habitats. This may explain the negative response to boulder placement we detected for dace and the negative relationship between dace response and difference in LWD. However, our results on effects of boulder placement on dace should be viewed with caution as only eight of our study sites contained large numbers of dace.

The lack of observed differences in invertebrate parameters between control and treatment reaches could be due to a number of factors: (1) the level of actual change produced by boulder additions in our study streams, (2) the types of habitats we sampled (e.g. riffles vs. pools), (3) the spatial scale at which we examined invertebrate response (stream reach vs. microhabitat), or (4) our sampling protocols. The first possibility is that boulder weirs did not sufficiently change habitat conditions within our study reaches to affect invertebrate assemblages. This conclusion agrees with a number of studies that have reported no change in macroinvertebrate abundance or diversity with placement of wood, boulders, or gravel (e.g. Tikkanen *et al*., 1994; Hilderbrand *et al*., 1997; Larson *et al*., 2001;

Table V. Pearsons correlation and p-values for relationships between physical variables ($\log_{10}$ (treatment/control) and fish response ($\log_{10}$ (treatment/control)

Fish response	Physical response				
	Project Age	Percent pool	LWD	Boulders	No. of boulder structures
Coho	−0.02	0.51*	0.11	0.37	−0.27
Trout (>100 mm)	−0.35	0.54*	−0.77**	0.21	0.18
YOY (trout <100 mm)	−0.01	0.32	−0.70**	−0.15	−0.07
Dace	−0.06	0.54	−0.06	0.24	−0.31

*$p < 0.10$.
**$p < 0.05$.

Laasonen *et al*., 1998; Brooks *et al*., 2002). Alternately, we may have sampled at an inappropriate spatial scale or habitat type to detect change. Results from our habitat surveys showed that treatment reaches contained a greater percentage of pool habitat, presumably forming as a result of boulder weir addition. Had we sampled pools rather than riffles, we may have observed differences in invertebrates between control and treatment reaches though the technique we employed for sampling invertebrates is not effective in pools. A third possibility is that by sampling over an entire stream reach, we missed a potentially finer scale response. Those studies that have reported changes in macroinvertebrates following placement of structures (Tarzwell, 1938; Gard, 1961; Wallace *et al*., 1995; Gortz, 1998), have generally found differences at the specific locations where the structures were placed, and associated changes in depth, velocity and substrate. Finally, we have to consider the effects of our sampling protocols. Because of the difficulty of collecting effective Surber samples on completely bare bedrock or in pools, we sampled riffles that contained patches of gravel when possible. As macroinvertebrates on bedrock and gravel substrates differ considerably in community structure (McCafferty, 1981; Merritt and Cummins, 1996), had we more randomly placed our benthic samples irrespective of the availability of gravel, we may have detected differences in invertebrates between control and treatment reaches. This would have been further supported by detailed quantification of gravel volumes in treatment and control reaches. Quantifying both gravel volume and surface area in future studies would allow quantification of the amount of increase in total surface area and invertebrate production due to boulder weir placement

We found no differences in water chemistry or nutrients with placement of boulder weirs. This may be because bedrock dominated treatment and control reaches and levels of organic matter and woody debris were relatively low, but also because boulder weirs were effective at trapping gravel but perhaps not finer organic matter. Both primary production and macroinvertebrates are known to respond to changes in water chemistry and nutrients (Rosemond *et al*., 1993; Kiffney and Richardson, 2001). This lack of change in nutrients and water chemistry may also explain why macroinvertebrate densities and diversity did not differ between treatment and control reaches. We suggest that future studies examine organic matter retention as well as primary production.

Shields *et al*. (1993; 1995b) examined use of boulder weirs to improve physical habitat in incised Mississippi stream channels and found large significant increases in both pool habitat and fish species abundance and diversity. This work and manuals on stream channel restoration recommend placement of weirs as a method of preventing channel incision or aggrading stream channels (Rosgen, 1996; Cowx and Welcomme, 1998). Further, Massong and Montgomery (2000) and Montgomery *et al*. (2003) indicated that logjams in conjunction with other roughness elements such as boulders, convert bedrock stream reaches to alluvial reaches by trapping gravel, aggrading stream channels and lowering stream gradient. We did not specifically examine the effects of boulder weirs on channel depth and incision though a simple reconstruction using our post-treatment long profile data suggested that weirs are effective at changing channel slope and aggrading the channel. Additional monitoring using pre and post long-profile surveys, is needed to accurately determine the level of channel aggradation due to boulder weir placement.

Based on our results, the placement of boulder weirs appears to be effective at improving habitat for trout and juvenile coho salmon by creating pools and low gradient habitats at a reach scale. Previous studies have indicated that they also trap large amounts of gravel and aggrade the stream channel. They do not, however, increase cover within a habitat unit and we suggest boulders weirs are merely the first step to restoring bedrock or incised stream channels. Boulder weir placement should be coupled with other measures to improve habitat complexity and protection of riparian areas to provide long-term inputs of LWD and fully restore bedrock or incised stream channels.

Our study also highlights some of the difficulties in examining chemical and biotic response to habitat restoration actions including the scale of analysis and the metrics examined. This was particularly evident for water chemistry and macroinvertebrates, but also holds for fishes as well. Macroinvertebrate community structure and water quality are often controlled by landscape scale factors such as elevation and geology, which may override subtle changes in habitat resulting from placement of structures within a stream reach (Malmqvist and Hoffsten, 2000; Weigel et al., 2000; Suren and McMurtrie, 2005). Thus a more complete evaluation would be to examine watershed-scale response to habitat rehabilitation. Finally, we suggest that future research should focus on the effects of boulder weirs on bed aggradation, spawner use of gravels trapped by boulder weirs, examining changes in fish survival, novel metrics for macroinvertebrates and water chemistry, and determining the number of boulders needed to restore a stream channel.

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REFERENCES

Barbour MT, Gerritsen J, Snyder BD, Stribling JB. 1999. *Rapid Bioassessment Protocols For Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish*, 2nd edn.. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water: Washington, D.C.

Bilby RE, Likens GE. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. *Ecology* **61**: 1107–1113.

Bisson PA, Nielsen JL, Palmason RA, 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*, Armantrout NB (ed.). *American Fisheries Society*: Bethesda; 62–73.

Bisson PA, Sullivan K, Nielsen JL. 1988. Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead, and cutthroat trout in streams. *Transactions of the American Fisheries Society* **117**: 262–273.

Booth DB. 1990. Stream channel incision following drainage-basin urbanization. *Water Resources Bulletin* **26**: 407–417.

Brooks SS, Palmer MA, Cardinale BJ, Swan CM, Ribblett S. 2002. Assessing stream ecosystem rehabilitation: limitations of community structure data. *Restoration Ecology* **10**: 156–168.

Buijse AD, Coops H, Staras M, Jans LH, Van Geest GJ, Grift RE, Ibelings BW, Oosterberg W, Roozen F. 2002. Restoration strategies for river floodplains along large lowland rivers in Europe. *Freshwater Biology* **47**: 889–907.

Cederholm CJ, Bilby RE, Bisson PA, Bumstead TW, Fransen BR, Scarlett WJ, Ward JW. 1997. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. *North American Journal of Fisheries Management* **17**: 947–963.

Chapman DW. 1996. Efficacy of structural manipulations of instream habitat in the Columbia River Basin. *Northwest Science* **5**: 279–293.

Cowx IG, Welcomme RL. 1998. *Rehabilitation of Rivers for Fish*. Fishing News Books: Oxford.

Dewberry TC, Karr JR, Adams JW. 1999. *Evaluating Aquatic Resources in the Clackamas River Basin of Oregon*. Contract #CR825041-01. U.S. Environmental Protection Agency: Portland.

Ehlers R. 1956. An evaluation of stream improvement project devices constructed eighteen years ago. *California Fish and Game* **42**: 203–217.

Erskine WD, Webb AA. 2003. Desnagging to resnagging: New directions in river rehabilitation in southeastern Australia. *River Research and Applications* **19**: 233–249.

Fontaine BL. 1987. *An Evaluation of the Effectiveness of Instream Structures for Steelhead Rearing Habitat in the Steamboat Creek Basin*. MS thesis. Oregon State University: Corvallis.

Fore LS, Karr JR, Wisseman RW. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. *Journal of the North American Benthological Society* **15**: 212–231.

Frissell CA, Nawa RK. 1992. Incidence and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. *North American Journal of Fisheries Management* **12**: 182–187.

Gard R. 1961. Creation of trout habitat by constructing small dams. *Journal of Wildlife Management* **52**: 384–390.

Gard R. 1972. Persistence of headwater check dams in a trout stream. *Journal of Wildlife Management* **36**: 1363–1367.

Gortz P. 1998. Effects of stream restoration on the macroinvertebrate community in the River Esrom, Denmark. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**: 115–130.

Grant JWA, Englert J, Bietz BF. 1986. Application of a method for assessing the impact of watershed practices: effects of logging on salmond standing crops. *North American Journal of Fisheries Management* **6**: 24–31.

Hamilton JB. 1989. Response of juvenile steelhead to instream deflectors in a high gradient stream. In *Practical Approaches to Riparian Resource Management*, Gresswell RE, Barton BA, Kershner JL (eds). U.S. Bureau of Land Management: Billings; 149–158.

Hartman GF. 1965. The role of behavior in the ecology and interaction of underyearling coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). *Journal of the Fisheries Research Board of Canada* **22**: 1035–1081.

Hicks BJ, Hall JD, Bisson PA, Sedell JR. 1991. Responses of salmonids to habitat changes. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, Meehan WR (ed.). Special Publication 19. American Fisheries Society: Bethesda; 483–518.

Hilderbrand RH, Lemly AD, Dolloff CA, Harpster KL. 1997. Effects of large woody debris placement on stream channels and benthic macroinvertebrates. Canadian *Journal of Fisheries and Aquatic Sciences* **54**: 931–939.

House R. 1996. An evaluation of stream restoration structures in a coastal Oregon stream 1981–1993. *North American Journal of Fisheries Management* **16**: 272–281.

House RA, Boehne PL. 1987. The effect of stream cleaning on salmonid habitat and populations in a coastal Oregon drainage. *Western Journal of Applied Forestry* **2**: 84–87.

House R, Crispin V, Monthey R. 1989. *Evaluation of Stream Rehabilitation Projects—Salem District (1981-1988)*. Technical Note, T/N OR-6. U.S. Dept. Interior Bureau of Land Management, Oregon State Office: Portland.

Hvidsten NA, Johnsen BO. 1992. River bed construction: impact and habitat restoration for juvenile Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L. *Aquaculture and Fisheries Management* **23**: 489–498.

Kahler TH, Roni P, Quinn TP. 2001. Summer movement and growth of juvenile anadromous salmonids in small western Washington streams. *Canadian Journal of Fisheries and Aquatic Sciences* **58**: 1947–1956.

Karr JR, Chu EW. 1999. *Restoring Life in Running Waters: Better Biological Monitoring*. Island Press: Washington DC.

Kauffman JB, Beschta RL, Otting N, Lytjen D. 1997. An ecological perspective of riparian and stream restoration in the western United States. *Fisheries* **22**: 12–24.

Kerans BL, Karr JR. 1994. A benthic index of biological integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications* **4**: 768–785.

Kiffney PM, Richardson JS. 2001. Effects of nutrient enrichment on periphyton and vertebrate (Ascaphus truei) and invertebrate grazers in experimental channels. *Copeia* **2001**: 422–429.

Kondolf GM, Micheli ER. 1995. Evaluating stream restoration projects. *Environmental Management* **19**: 1–15.

Laasonen P, Muotka T, Kivijarvi I. 1998. Recovery of macroinvertebrate communities from stream habitat restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**: 101–113.

Larson MG, Booth DB, Morley SA. 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. *Ecological Engineering* **18**: 211–226.

Linlokken A. 1997. Effects of instream habitat enhancement on fish population of a small Norwegian stream. *Nordic Journal of Freshwater Research* **73**: 50–59.

Malmqvist B, Hoffsten PO. 2000. Macroinvertebrate taxonomic richness, community structure and nestedness in Swedish streams. *Archives für Hydrobiologie* **150**: 29–54.

Massong TM, Montgomery DR. 2000. Influence of sediment supply, lithology, and wood debris on the distribution of bedrock and alluvial channels. *Geological Society of America Bulletin* **112**: 591–599.

McCafferty WP. 1981. *Aquatic Entomology: The Fisherman's and Ecologist's Illustrated Guide to Insects and Their Relatives*. Jones and Bartlett Publishers Inc.: Boston.

Merritt RW, Cummins KW (eds). 1996. *An Introduction to the Aquatic Insects of North America*. Kendall/Hunt Publishing Company: Dubuque.

Montgomery DR, Buffington JM, Smith RD, Schmidt KM, Pess G. 1995. Pool spacing in forest channels. *Water Resources Research* **31**: 1097–1105.

Montgomery DR, Massong TM, Hawley SCS. 2003. Influence of debris flows and log jams on the location of pools and alluvial channel reaches, Oregon Coast Range. *Geological Society of America Bulletin* **115**: 78–88.

Moore KM, Gregory SV. 1988. Summer habitat utilization and ecology of cutthroat trout fry (Salmo clarki) in Cascade mountain streams. *Canadian Journal of Fisheries and Aquatic Sciences* **45**: 1921–1930.

Moreau JK. 1984. Anadromous salmonid habitat enhancement by boulder placement in Hurdygurdy Creek, California. In *Pacific Northwest Stream Habitat Management Workshop*, Hassler TJ (ed.). American Fisheries Society, Humboldt Chapter, Humboldt State University: Arcata; 97–116.

Morley SA, Karr JR. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conservation Biology* **16**: 1498–1509.

Muotka T, Laasonen P. 2002. Ecosystem recovery in restored headwater streams: the role of enhanced leaf retention. *Journal of Applied Ecology* **39**: 145–156.

Muotka T, Paavola R, Haapala A, Novikmec M, Laasonen P. 2002. Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. *Biological Conservation* **105**: 243–253.

Murphy ML, Hall JD. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences* **38**: 137–145.

Näslund I. 1989. Effects of habitat improvement on the brown trout, Salmo trutta L., population of a northern Swedish stream. *Aquaculture and Fisheries Management* **20**: 463–474.

Negishi JN, Richardson JS. 2003. Responses of organic matter and macroinvertebrates to placements of boulder clusters in a small stream of southwestern British Columbia, Canada. *Canadian Journal of Fisheries and Aquatic Sciences* **60**: 247–258.

Nickelson TE, Rodgers JD, Johnson SL, Solazzi MF. 1992. Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Sciences* **49**(4): 783–789.

Niem AR, Niem WA. 1990. *Geology and Oil, Gas, and Coal Resources, Southern Tyee Basin, Southern Coast Range, Oregon*. Open-File Report O-89-3. State of Oregon Department of Geology and Mineral Industries: Portland.

O'Grady M, Gargan P, Delanty K, Igoe F, Byrne C. 2002. *Observations in relation to changes in some physical and biological features of the Glenglosh River following bank stabilization*. *Proceedings of the 13th International Salmonid Habitat Enhancement Workshop*. Central Fisheries Board, Dublin.

Platts WS. 1991. Livestock grazing. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, Meehan WR (ed.). Special Publication 19. American Fisheries Society: Bethesda; 389–423.

Platts WS, Rinne JN. 1985. Riparian and stream enhancement management and research in the Rocky Mountains. *North American Journal of Fisheries Management* **5**: 115–225.

Reeves GH, Everest FH, Sedell JR. 1993. Diversity of juvenile anadromous salmonid assemblages in coastal Oregon Basins with different levels of timber harvest. *Transactions of the American Fisheries Society* **122**: 309–317.

Reeves GH, Hall JD, Roelofs TD, Hickman TL, Baker CO. 1991. Rehabilitating and modifying stream habitats. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, Meehan WR (ed.). Special Publication 19. American Fisheries Society: Bethesda; 519–557.

Reeves GH, Hohler DB, Hansen BE, Everest FH, Sedell JR, Hickman TL, Shively D. 1997. Fish habitat restoration in the Pacific Northwest: fish Creek of Oregon. In *Watershed Restoration: Principles and Practices*, Williams JE, Wood CA, Dombeck MP (eds). American Fisheries Society: Bethesda; 335–359.

Roni P. 2002. Habitat use by fishes and pacific giant salamanders in small western Oregon and Washington streams. *Transactions of the American Fisheries Society* **131**: 743–761.

Roni P. 2003. Response of benthic fishes and giant salamanders to placement of large woody debris in small Pacific Northwest streams. *North American Journal of Fisheries Management* **23**: 1087–1097.

Roni P, Beechie TJ, Bilby RE, Leonetti FE, Pollock MM, Pess GR. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management* **22**: 1–20.

Roni P, Fayram A. 2000. Estimating winter salmonid abundance in small western Washington streams: a comparison of three techniques. *North American Journal of Fisheries Management* **20**: 683–692.

Roni P, Fayram A, Miller M. 2005a. Monitoring and evaluating instream habitat enhancement. In *Monitoring Stream and Watershed Restoration*, Roni P (ed.). American Fisheries Society: Bethesda; 209–236.

Roni P, Hanson K, Beechie T, Pess G, Pollock M. 2005b. *Habitat Rehabilitation for Inland Fisheries: A Global Review of Effectiveness and Guidance for Rehabilitation of Freshwater Ecosystems*. FAO Fisheries Technical Paper 484, Food and Agriculture Organization of the United Nations, Rome, Italy.

Roni P, Quinn TP. 2001a. Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams. *Canadian Journal of Fisheries and Aquatic Sciences* **58**: 282–292.

Roni P, Quinn TP. 2001b. Effects of wood placement on movements of trout and juvenile coho salmon in natural and artificial stream channels. *Transactions of the American Fisheries Society* **130**: 675–685.

Roper B, Konnoff D, Heller D, Wieman K. 1998. Durability of Pacific Northwest instream structures following floods. *North American Journal of Fisheries Management* **18**: 686–693.

Rosemond AD, Mulholland PJ, Elwood JW. 1993. Top-down and bottom-up control of stream periphyton: effects of nutrients and herbivores. *Ecology* **74**: 1264–1280.

Rosgen D. 1996. *Applied River Morphology.* Wildland Hydrology: Pagosa Springs.

Sedell JR, Luchessa KJ. 1984. Using the historical record as an aid to salmonid habitat enhancement. In *Acquisition and Utilization of Aquatic Habitat Inventory Information*, Armantrout NB (ed.). American Fisheries Society: Bethesda; 210–222.

Shields FD, Cooper CM, Knight SS. 1993. Initial habitat response to incised channel rehabilitation. *Aquatic Conservation: Marine and Freshwater Ecosystems* **3**: 93–103.

Shields FD, Cooper CM, Knight SS. 1995a. Experiment in stream restoration. *Journal of Hydraulic Engineering* **121**: 494–503.

Shields FD, Knight SS, Cooper CM. 1995b. Incised stream physical habitat restoration with stone weirs. *Regulated Rivers: Research and Management* **10**: 181–198.

Solazzi MF, Nickelson TE, Johnson SL, Rodgers JD. 2000. Effects of increasing winter rearing habitat on abundance of salmonids in two coastal Oregon streams. *Canadian Journal of Fisheries and Aquatic Sciences* **57**: 906–914.

Suren AM, McMurtrie S. 2005. Assessing the effectiveness of enhancement activities in urban streams: II. Response of invertebrate communities. *River Research and Applications* **21**: 439–453.

Tarzwell CM. 1938. An evaluation of the methods and results of stream improvement in the Southwest. *Transactions of the 3rd North America Wildlife Conference* **1938**: 339–364.

Tikkanen P, Laasonen P, Muotka T, Kuusela K. 1994. Short-term recovery of benthos following disturbance from stream habitat rehabilitation. *Hydrobiologia* **270**: 121–130.

Van Zyll De Jong MC, Cowx IG, Scruton DA. 1997. An evaluation of instream habitat restoration techniques on salmonid populations in a Newfoundland stream. *Regulated Rivers: Research and Management* **13**: 603–614.

Wallace JB, Webster JR, Meyer JL. 1995. Influence of log additions on physical and biotic characteristics of a mountain stream. *Canadian Journal of Fisheries and Aquatic Sciences* **52**: 2120–2137.

Ward BR, Slaney PA. 1981. Further evaluations of structures for the improvement of salmonid rearing habitat in coastal streams of British Columbia. In *Proceedings: Propagation, Enhancement, and Rehabilitation of Anadromous Salmonid Populations and Habitat Symposium*, Hassler TJ (ed.). American Fisheries Society, Humboldt Chapter, Humboldt State University: Arcata; 99–108.

Weigel BM, Lyons J, Paine LK, Dodson SL, Undersander DJ. 2000. Using stream macroinvertebrates to compare riparian land use practices on cattle farms in southwestern Wisconsin. *Journal of Freshwater Ecology* **15**: 93–106.

White RJ. 2002. Restoring streams for salmonids: where have we been? Where are we going? In *Proceedings of the 13th International Salmonid Habitat Enhancement Workshop Westport, County Mayo, Ireland*, O'Grady M (ed.). Central Fisheries Board: Dublin, Ireland, 1–33.

Wydowski RS, Whitney RR. 2003. *Inland Fishes of Washington*. University of Washington Press: Seattle.

Zar JH. 1999. *Biostatistical Analysis*. Prentice Hall: Upper Saddle River.

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