Rehabilitation of bedrock stream channels: the effects of boulder weir placement on aquatic habitat and biota

Project Completion Report for Interagency Agreement HAI013001

Phil Roni, Todd Bennett, Sarah Morley, George R. Pess, Karrie Hanson

Northwest Fisheries Science Center
National Marine Fisheries Service
2725 Montlake Boulevard East
Seattle, WA 98112
(206) 860-3307
phil.roni@noaa.gov

and

Dan Van Slyke and Pat Olmstead
Bureau of Land Management
Coos Bay District
1300 Airport Lane
North Bend, OR 97459
(541) 751-4452
Dan_vanslyke@or.blm.gov

Abstract

The placement of boulder weirs is a popular method to improve fish habitat though little is known about their effectiveness 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 7 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. 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 (*Oncorhynchus kisutch*) and trout (*O. mykiss* and O. *clarki*) were higher in treatment than control reaches ($p < 0.05$), while dace (*Rhinichthys* spp.; $p < 0.09$) numbers were higher in control reaches and no significant difference was detected for young-of-year trout ($p > 0.20$). Both coho salmon and trout response ($\log_{10}$(treatment density/reference density) to boulder weir placement was positively correlated with difference in pool area ($\log_{10}$(treatment/reference; $p < 0.10$), 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 > 100mm). 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.

Introduction

Many streams in the North America, Europe and elsewhere have been degraded and greatly simplified by log drives, splash damming, 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 in many areas with intensive land use such as grazing, agriculture, or 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 particularly common methods of improving stream channels (Reeves et al. 1991; Roni and Quinn 2001; Roni et al. 2002). In areas were LWD of adequate length and diameter are not readily available, boulder clusters, weirs, and other structures have been used. The placement of boulder is a particularly prevalent in streams dominated by sedimentary rock in southwest Oregon coast where boulders placed in the configuration of weirs are intended to function similar to key pieces of wood. Anecdotal information suggest that streams along the Oregon coast contained many larger boulders prior to twentieth century forestry activities (splash damming and stream cleaning). However, there is considerable discussion as to whether boulder and weir placement mimics natural conditions or is entirely artificial.

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 2001; Roni 2003). Most of these studies demonstrated increases in juvenile coho salmon (*Oncorhynchus kisutch*) abundance following wood placement. In contrast, research on the effectiveness of boulder weir placement has been limited to a handful of case studies which have often focused on physical variables with limited information on fish responses (Roni et al. 2004). House et al. (1989) reported higher levels of juvenile coho salmon, cutthroat (*Oncorhynchus clarki)* and steelhead (*O. mykiss*) in several north and central Oregon coast streams following a combination of LWD, boulder and gabion placement. They did not, however, distinguish fish response between boulder and LWD structures. Moreau (1984) reported a 100% increased steelhead parr densities two 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. 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. Boulder structures have also been commonly used in European streams and several studies have suggested increases in brown trout (*Salmo trutta*) and other species due to these treatments (Näslund 1989; Linlokken 1997; O'Grady et al. 2002). 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.

Stream morphology and biotic communities can differ by geology and channel type (Hicks and Hall 2003; Montgomery et al 1996; 1999). For example, basalt and sandstone stream channels in coastal Oregon have different morphological characteristics and fish community structure with sandstone channels having more pools, a lower gradient, and are typically dominated by coho salmon (Hicks and Hall 2003). Most evaluations of fish response to restoration have occurred in alluvial reaches or in stream channels in basalt or glacial geology. Moreover, these studies have generally occurred in relatively small streams (< 12 m bankfull width) and boulder structures are often placed in larger channels (Roni et al. 2002; Roni et al. 2004). The response of biota to placement of instream structures is likely to differ among geologic types but has not been examined in sandstone channels or in larger stream channels.

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), 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 restoration.

In addition to biological objectives, the placement of boulders weirs and check dams is an increasingly common method of aggrading highly incised channels (Shields 1991; Cowx and Welcomme 1998). For example, Cowx and Welcomme (1998) described the use of check dams in the Danube River in Eastern Europe to raise the channel and water level to reconnect older river channels and increase water retention time (Cowx and Welcomme 1998). In mountain stream channels, log and boulder jams have been demonstrated to aggrade channels and to the formation of alluvial channels upstream of jams (Montgomery et al. 1996). In sand and gravel dominated alluvial channels, Shields et al. (1993;1995a,b) demonstrated that boulder weirs designed to aggrade highly incised stream channel led to increases in depth, width, and pool area following boulder weir placement. These studies suggest that the use of boulder weirs may benefit fish populations by changing bedrock channels to alluvial channels.

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 Micheli1995; Chapman 1996, Kauffman et al. 1997,Roni 2004)). Existing monitoring and evaluation of stream restoration projects has generally focused on changes in physical habitat with relatively few comprehensive biological evaluations. The goals of our research were to examine the effects of boulder weir placement on physical habitat, 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 reference 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 7 different streams in the lower Umpqua and Coquille River basins were sampled once in the late summer of 2002 or 2003 (Figure 1). Treatment was defined as the artificial placement of boulders and boulder weirs within the active stream channel. We selected stream reaches 200 m long in each stream (> 10 times the bankfull channel width) and at least 200m 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 2 or more stream kilometers apart. Paired treatment-reference reaches within a stream were of similar slope, width, riparian vegetation, discharge, and length. All streams in the study region had a similar legacy of splash damming, stream cleaning (removal of LWD) and other forestry activities that have resulted in highly uniform incised bedrock dominated channels with few boulders or woody debris. The proximity of the reaches insured that discharges between reaches were essentially identical, though the distribution of point velocities might differ.

Approximately 30 boulder weir placement projects were examined, but only 13 had suitable treatment and reference reaches with similar flow, channel width, gradient, confinement, and riparian vegetation. The number of boulder weirs (spanning entire channel) and deflectors (spanning only 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 and elevation of 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 of the watersheds composed 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) (Table 1). 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 and channel scour: (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. In stream less than 10 m wide, 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 $m^3 \cdot s^{-1}$ and 11 – 15°C during snorkel surveys.

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 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 a 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 two-minute 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 analyzed 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 (100m) and upstream end (200m) of each study reach Immediately after collection waters samples were frozen for later analysis of dissolved organic carbon, total nitrogen and phosphorous, and nutrient concentrations (e.g., NO3, NO4) using a spectrophotometer. The mean level for each water chemistry parameter was calculated by averaging the three samples for each reach.

Differences in habitat, LWD, and abundance and length of fish and salamanders between treatment and reference reaches were compared using paired t-tests. Fish densities 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, and total boulders. Pool area and LWD levels are known to be correlated with abundance and size of salmonid fishes (Roni and Quinn 2001) and sites with larger physical responses to restoration were predicted to have larger biological responses. All ratios of treatment to reference (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

Treatment and control reaches differed in those physical habitat features expected to respond to placement of instream structures though considerable variation existed in responses among sites. 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 2). 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_3, NO_2, or NH_4) between treatment and control reaches ($p > 0.10$; Table 3), though NO_2 was significantly higher in treatment reaches ($p = 0.08$).

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. Densities 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 4). Macroinvertebrate abundance, total taxa richness; relative abundance of EPT, shredders, and collectors, and BIBI did not differ between treatment and control reaches (Table 5.)

Pearson correlation analysis indicated that significant positive correlations existed between coho response $\log_{10}$(treatment density/reference density) and percent pool areas ($\log_{10}$(treatment/reference); Pearson correlation = 0.51, $p = 0.08$) and also for trout response and pool area response (Correlation = 0.54; $p = 0.06$; Table 6). Both YOY trout and dace response ($\log_{10}$treatment - $\log_{10}$ control) to boulder weir placement were negatively correlated with LWD (($\log_{10}$(treatment/reference); Correlation = -0.70 and 0.77 for YOY trout and dace, respectively; $p < 0.01$). The number of boulders, boulder weirs, and LWD forming pools ($\log_{10}$(treatment/reference) were not significantly correlated with any fish species response $\log_{10}$(treatment density/reference density.

Discussion

Boulder weir placement produced the predicted changed in physical habitat including increased pools, LWD, and boulders as well an increase in fish abundance. This is consistent with many findings for other instream habitat rehabilitation methods which have reported large changes in physical habitat following treatment (see Roni et al. 2002 and Roni 2004 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. Had the placement of boulders been coupled with placement of large amounts of organic material (wood and leaves) or organic or inorganic nutrients (e.g., Ward and Slaney 1981; Slaney et al. 1994), we may have detected changes in water chemistry and primary productivity.

We detected significantly higher numbers of juvenile coho and trout (> 100 mm) in response to boulder weir placement, suggesting that boulder weirs are an effective method of creating summer habitat for juvenile coho salmon and age 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 2001; 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 trout was expected given their preference for pool habitat (Bisson et al. 1988; Roni and Quinn

2001, Roni 2003) and the fact that placement of boulder weirs led to an increase in pool area.

The lack of response of both young-of-year trout 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 (2001) 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). Roni and Quinn (2001) also 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. This may also explain the lack of response to boulder weir placement which is further supported by the 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), in a rare study on nonsalmonid fish response to boulder weir placement, 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 response and LWD suggests that increases in pool area and habitat complexity 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 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 their negative relationship between dace response and difference in LWD. However, our results on effects of boulder placement on dace should be viewed with caution 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; 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, it's possible that 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 smaller 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. As macroinvertebrates on bedrock and gravel substrates differ considerably in community structure (McCafferty 1991; 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.

Shields et al. (1993; 1995b) examined the effects of boulder weirs on physical habitat in incised stream channels in Mississippi, 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 suggest that weirs are effective at changing localized slope and aggrading the channel (Figure 4). 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. Previous studies have indicated that they also trap large amounts of gravel and aggrade the stream channel. They do not, however, increase habitat complexity (wood cover). We suggest that boulders weirs are merely the first step to restoring bedrock or incised stream channels and that weir placement should be coupled with measures to improve habitat complexity and protection of riparian areas to provide long-term inputs of LWD.

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, and determining the number of boulders needed to restore a stream channel. The latter could be achieved by examining historical data or data from undisturbed reference reaches. Spawner surveys and estimating egg-to-fry survival is another important step in determining the biological effectiveness of boulder weirs and other instream habitat enhancement techniques that continues to be overlooked. Finally, examining changes in fish survival at different life stages may be difficult to measure, but would provide a more accurate evaluation of project effectiveness.

Acknowledgements

We thank Sarah Baker, Holly Coe, Aimee Fullerton, Patricia Garcia, Morgan Heim, Kris Kloehn, Andrea Pratt, Brian Sodeman, and Danielle Werner, for field assistance and Martin “Lightning” Liermann for statistical advice as well as field assistance. Sarah Baker for assistance with preparing a site map. Tim Barnes of the Bureau of Land Management (BLM) for provided detailed information on basin geology. Scott Lightcap and Nikki Moore of BLM and various other staff at the Coquille Watershed Association assisted in site selection and land access issues. We thank Bob Alverts formerly of the BLM Oregon State office for providing partial funding for this project. We are grateful to Roseburg Resources, Plum Creek, Lone Rock Timber, and other private land owners for allowing us access to sites on their lands. Joe Ebersole and Jim Wigington from the U.S. Environmental Protection Agency provided assistance in coordinating our work on the West Fork of the Smith River. In addition, we thank Tim Beechie, Chris Jordan and anonymous reviewers for comments on earlier versions of this manuscript.

References

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Bisson, P. A., J. L. Nielsen, R. A. Palmason, and L. E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low stream flows. Pages 62-73 in N. B. Armantrout, editor. Acquisition and Utilization of Aquatic Habitat Inventory Information. American Fisheries Society, Bethesda, Maryland.

Bisson, P. A., K. Sullivan, and J. L. Nielsen. 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(3):262-273.

Booth, D. B. 1990. Stream channel incision following drainage-basin urbanization. Water Resources Bulletin 26:407-417.

Brooks, S. S., M. A. Palmer, B. J. Cardinale, C. M. Swan, and S. Ribblett. 2002. Assessing stream ecosystem rehabilitation: Limitations of community structure data. Restoration Ecology 10(1):156-168.

Buijse, A. D., and coauthors. 2002. Restoration strategies for river floodplains along large lowland rivers in Europe. Freshwater Biology 47(4):889-907.

Cederholm, C. J., R. E., Bilby, P. A. Bisson, T. W. Bumstead, B. R. Fransen, W. J. Scarlett, and J. W. Ward. 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.Cowx, I. G., and R. L.

Welcomme. 1998. Rehabilitation of rivers for fish. Fishing News Books, Oxford, England.

Chapman, D. W. 1996. Efficacy of structural manipulations of instream habitat in the Columbia River Basin. Northwest Science 5(4):279-293.

Dewberry, T.C., J.R. Karr, and J.W. Adams. 1999. Evaluating aquatic resources in the Clackamas River basin of Oregon. US EPA Contract #CR825041-01.

Erskine, W. D., and A. A. Webb. 2003. Desnagging to resnagging: New directions in river rehabilitation in southeastern Australia. River Research and Applications 19(3):233-249.

Fore, L.S., J.R. Karr, and R.W. Wisseman. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. Journal of the North American Benthological Society 15:212-231.

Gard, R. 1961. Creation of trout habitat by constructing small dams. Journal of Wildlife Management 52(4):384-390.

Gortz, P. 1998. Effects of stream restoration on the macroinvertebrate community in the River Esrom, Denmark. Aquatic Conservation: Marine and Freshwater Ecosystems 8(1):115-130.

Grant, J. W. A., J. Englert, and B. F. Bietz. 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.

Fontaine, B. L. 1987. An evaluation of the effectiveness of instream structures for steelhead rearing habitat in the Steamboat Creek basin. M.S. Thesis. Oregon State University, Corvallis.

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

Hicks, B. J., J. D. Hall, P. A. Bisson, and J. R. Sedell. 1991. Responses of salmonids to habitat changes. Pages 483-518 in W. R. Meehan, editor. Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. American Fisheries Society Special Publication 19.

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

Hicks, B. J., and J. D. Hall. 2003. Rock type and channel gradient structure salmonid populations in the Oregon Coast Range. Transactions of the American Fisheries Society 132:468-482.

Hicks, B. J., and coauthors. 1991. Responses of salmonids to habitat changes. Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. W. R. Meehan, American Fisheries Society Special Publication 19: 483-518.

Hilderbrand, R. H., A. D. Lemly, C. A. Dolloff, and K. L. Harpster. 1997. Effects of large woody debris placement on stream channels and benthic macroinvertebrates. Canadian Journal of Fisheries and Aquatic Sciences 54(4):931-939.

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

Kahler, T. H., P. Roni, and T. P. Quinn. 2001. Summer movement and growth of juvenile anadromous salmonids in small western Washington streams. Canadian Journal of Fisheries and Aquatic Sciences 58(10):1947-1956.

Kauffman, J. B., R. L. Beschta, N. Otting, and D. Lytjen. 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22(5):12-24.

Karr, J. R. and E.W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington D.C.

Kerans, B.L., and J.R. Karr. 1994. A benthic index of biological integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4:768-785.

Kondolf, G. M., and E. R. Micheli. 1995. Evaluating stream restoration projects. Environmental Management 19(1):1-15.

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

Larson, M.G., D.B. Booth, and S.A. Morley. 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.

McCafferty, W. P. 1981. Aquatic entomology: the fisherman's and ecologist's illustrated guide to insects and their relatives. Jones and Bartlett Publishers Inc., Boston.

Massong, T. M. and D. R. Montgomery. 2000. Influence of sediment supply, lithology, and wood debris on the distribution of bedrock and alluvial channels. Geological Society of America Bulletin 112(5): 591-599.

Merritt, R.W., and K.W. Cummins, editors. 1996. An introduction to the aquatic insects of North America. Kendall/Hunt Publishing Company, Dubuque, Iowa.

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

Montgomery, D. R., and coauthors. 1996. Distribution of bedrock and alluvial channels in forested mountain drainage basins. Nature 381:587-589.

Montgomery, D. R., E. M. Beamer, G. Pess, and T. P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. Canadian Journal of Fisheries and Aquatic Sciences 56:377-387.

Montgomery, D. R., T. M. Massong, and S. C. S. Hawley. 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, K. M., and S. V. Gregory. 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, J. K. 1984. Anadromous salmonid habitat enhancement by boulder placement in Hurdygurdy Creek, California. Pages 97-116 in T. J. Hassler, editor Pacific

Northwest Stream Habitat Management Workshop. American Fisheries Society, Humboldt Chapter, Humboldt State University, Arcata, CA.

Morley, S.A., and J.R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. Conservation Biology 16:1498-1509.

Muotka, T., R. Paavola, A. Haapala, M. Novikmec, and P. Laasonen. 2002. Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. Biological Conservation 105(2):243-253.

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

Naslund, 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, J. N., and J. S. Richardson. 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, T. E., J. D. Rodgers, S. L. Johnson, and M. F. Solazzi. 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, A.R. and Niem, W.A, 1990, Geology and Oil, Gas, and Coal Resources, Southern Tyee Basin, Southern Coast Range, Oregon: State of Oregon Department of Geology and Mineral Industries Open-File Report O-89-3, 44 p.

O'Grady, M. and coauthors. 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, Ireland.

Platts, W. S. 1991. Livestock grazing. Influences of forest and rangeland management on salmonid fishes and their habitats. W. R. Meehan, American Fisheries Society Special Publication 19: 389-423.

Reeves, G. H., F. H. Everest, and J. R. Sedell. 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, G. H., J. D. Hall, T. D. Roelofs, T. L. Hickman, and C. O. Baker. 1991. Rehabilitating and modifying stream habitats. Pages 519-557 in W. R. Meehan, editor. Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats, volume American Fisheries Society Special Publication 19. American Fisheries Society, Bethesda, MD.

Reeves, G. H., and coauthors. 1997. Fish habitat restoration in the Pacific Northwest: Fish Creek of Oregon. Pages 335-359 in J. E. Williams, C. A. Wood, and M. P. Dombeck, editors. Watershed Restoration: Principles and Practices. American Fisheries Society, Bethesda, Maryland.

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., editor. 2004. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, MD.

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(4):743-761.

Roni, P., and coauthors. 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):1-20.

Roni, P., and A. Fayram. 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., A. Fayram, and M. Miller. 2004. Monitoring and evaluating instream habitat enhancement. Pages xx to xx in P. Roni, editor. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Roni, P., and T. P. Quinn. 2001. 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(2):282-292.

Rosgen, D. 1996. Applied river morphology. Wildland Hydrology, Pagosa Springs, Colorado.

Sedell, J. R., and K. J. Luchessa. 1984. Using the historical record as an aid to salmonid habitat enhancement. Pages 210-222 in N. B. Armantrout, editor. Acquisition and Utilization of Aquatic Habitat Inventory Information. American Fisheries Society, Western Division, Bethesda, Maryland.

Shields, F. D., C. M. Cooper, and S. S. Knight. 1993. Initial habitat response to incised channel rehabilitation. Aquatic Conservation: Marine and Freshwater Ecosystems 3:93-103.

Shields, F. D. J., C. M. Cooper, and S. S. Knight. 1995a. Experiment in stream restoration. Journal of Hydraulic Engineering 121(6):494-503.

Shields, F. D., S. S. Knight, and C. M. Cooper. 1995b. Incised stream physical habitat restoration with stone weirs. Regulated Rivers: Research and Management 10:181-198.

Slaney, P. A., B. O. Rublee, C. J. Perrin, and H. Goldberg. 1994. Debris structure placements and whole-river fertilization for salmonids in a large regulated stream in British Columbia. Bulletin of Marine Science 55:1160-1180.

Solazzi, M. F., T. E. Nickelson, S. L. Johnson, and J. D. Rodgers. 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.

Tarzwell, C. M. 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., and coauthors. 1994. Short-term recovery of benthos following disturbance from stream habitat rehabilitation. Hydrobiologia 270(2): 121-130.

Van Zyll De Jong, M. C., I. G. Cowx, and D. A. Scruton. 1997. An evaluation of instream habitat restoration techniques on salmonid populations in a Newfoundland stream. Regulated Rivers: Research and Management 13:603-614.

Wallace, J. B., J. R. Webster, and J. L. Meyer. 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, B. R., and P. A. Slaney. 1981. Further evaluations of structures for the improvement of salmonid rearing habitat in coastal streams of British Columbia. Pages 99-108 in T. J. Hassler, editor Proceedings: Propagation, Enhancement, and Rehabilitation of Anadromous Salmonid Populations and Habitat Symposium. American Fisheries Society, Humboldt Chapter, Humboldt State University, Arcata, CA.

Wydowski, R. S., and R. R. Whitney. 2003. Inland fishes of Washington. University of Washington Press, Seattle.

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

Table 1. Stream habitat types as modified from Bisson et al. (1982), Nickelson et. al. (1992b), and Roni (2003).

Slow Water Habitats	
Dam Pool	A pool impounded upstream from a complete or nearly complete channel blockage often cause by a debris jam or beaver pond.
Backwater Pool	An eddy or slack water along the channel margin separated from the main channel margin by a gravel bar or small channel obstruction.
Scour Pool	A scoured basin or depression either (1) near the channel margin caused by flow being directed to one side of the stream by a partial channel obstruction, or (2) near the center of the channel usually caused by a channel constriction or high gradient riffle or cascade,
Trench Pool	Similar to a scour pool, but a slot or trench in a stable substrate such as bedrock or clay.
Plunge Pool	A basin or depression scoured by a vertical drop over a channel obstruction.
Bedrock Pocket Pool	A backwater feature that has exposed bedrock at the upstream and downstream end. The bedrock is perpendicular to the flow and acts as a elevation control on either end. The flow within the unit is slower than the main flow and often consists of many bedrock pockets or small pools.
Glide	A moderately shallow reach with an even, laminar flow and no pronounced turbulence or obstructions.
Fast Water Habitats	
Riffle	a) Low Gradient - A shallow reach with a moderate current velocity, moderate turbulence, and a gradient of # 2%.
	b) High Gradient - A shallow reach with a moderate current velocity, moderate turbulence, and a gradient between 2% and 4%.
Cascade	A shallow reach with high current velocity and considerable turbulence with a gradient of > 4% or a series of small steps of alternating small waterfalls and small pools with a gradient > 4%

Table 2. 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. WFS = West Fork of Smith River, C = control reaches, T = treatment reaches.

Stream/Reach (year constructed)	Number of habitats Units		Number of pools		Percent pool area		Functioning 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	2909	2.212	2.600	3.803	-
p – value	0.045	0.955	0.013	0.051	0.023	.003	-

[1] LWD added in 1999

[2] Boulder clusters added in 1999

Table 3. 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. WFS = West Fork of Smith River, C = control reaches, T = treatment reaches.

	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	

Table 4. Fish numbers 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. WFS = West Fork of Smith River, C = control reaches, T = treatment reaches

	Coho		Dace		Stickleback		Trout < 100		Trout > 100	
Stream	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					15	20	3	6
Middle Creek I	82	134	17	5	14	33	2			2
Middle Creek II	413	648	9	14			4	40		4
Paradise Creek I	140	372					3		6	16
Paradise Creek II	181	140					25	14	4	1
S.F. Elk Creek	217	380	1				41	3	4	7
WFS Beaver Reach	265	285	5	4			98	61	8	19
WFS Crane Reach	568	494	183	88			43	23	9	4
WFS Moore Reach	329	501	38	22			102	39	2	1
WFS Skunk Cabbage Reach	560	791	32	6			135	27	3	10
WFS Upper Reach	479	719					119	149	2	2
t statistic	3.659		-1.945		-		-1.334		2.195	
p- value	0.003		.088		-		.207		0.05	

Table 5. Selected macroinvertebrate metrics measured in treatment and control reaches of 13 study sites and results of paired t-tests (t statistic, p value). WFS = West Fork of Smith River. Statistical comparisons of treatment and control reaches were performed on $\log_{10}$ transformed data. WFS = West Fork of Smith River, C = control reaches, T = treatment reaches.

	Abundance											
	Total Abundance		Relative EPT Taxa		Relative Shredder		Relative Collector		Taxa Richness		B-IBI	
Stream	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	

Table 7. Pearsons correlation and p-values for relationships between physical variables ($\log_{10}$ (treatment/control) and fish response ($\log_{10}$ (treatment/control).

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

* $p < 0.05$, **$p< 0.10$

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

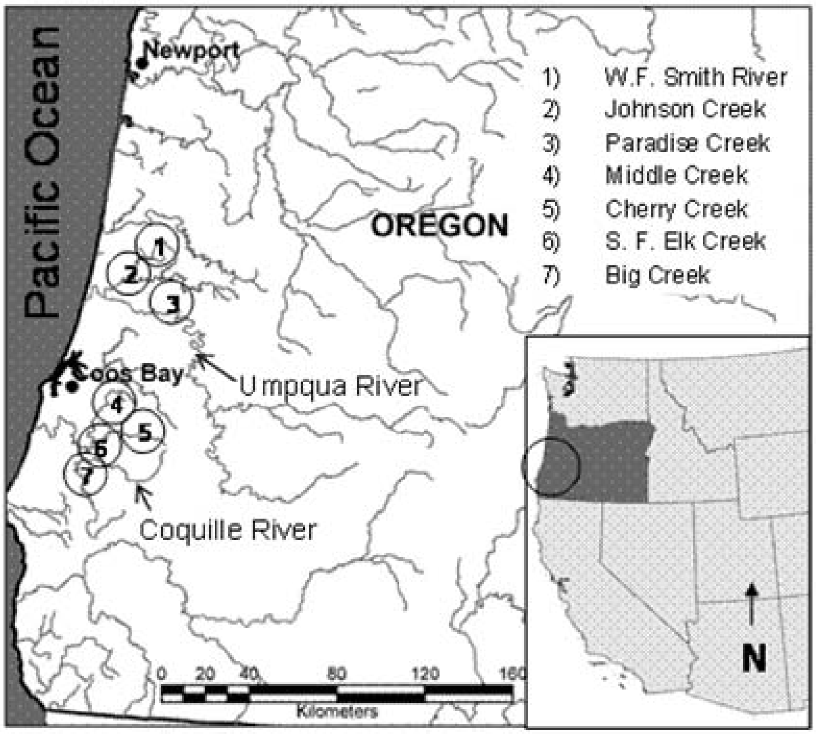

Figure 2. Example of typical control (top) and treatment (bottom) reaches from West Fork of Smith River.

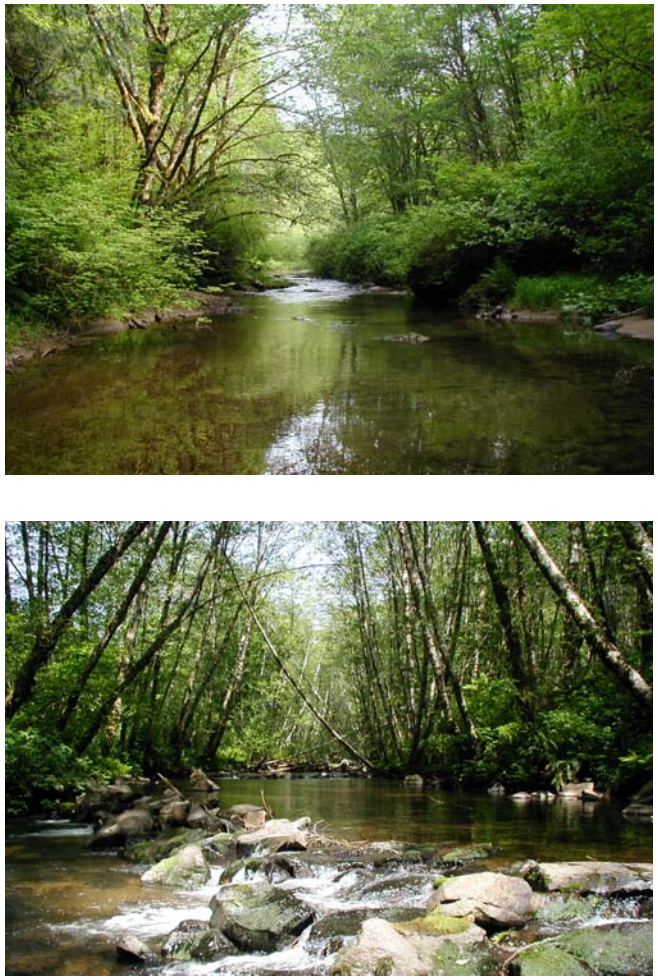

Figure 3. Partial correlation plots between fish (coho, trout, dace, YOY trout) response and percent pool or LWD response to boulder weir placement. All axis are a $\log_{10}$ scale ($\log_{10}$ (treatment/control).

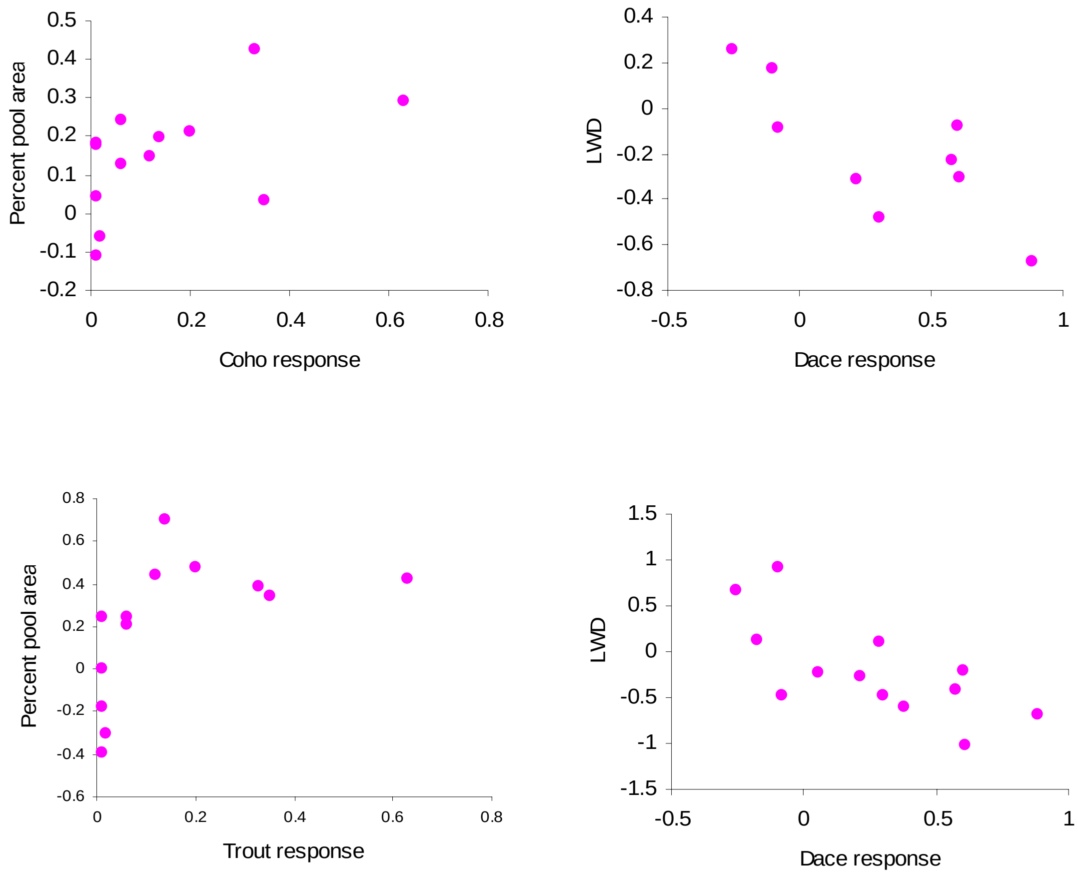

Figure 4. Long profile from Paradise Creek Study Site II showing channel with estimated channel profile before boulder weir placement (estimated) and after boulder weir placement (field measurement). Arrows indicate location of boulder weirs.