Determinates of Gibbons Creek Watershed Condition and Health: Results of the Gibbons Creek Watershed Analysis, 1997-1999

Prepared by:

Scott Barndt
Julia Taylor
Travis Coley
Brian Ensign
Jen Stone
Marv Yoshinaka

U.S. Fish and Wildlife Service
Columbia River Fisheries Program Office
9317 Highway 99, Suite I
Vancouver, Washington 98665

August 2003

Table of Contents

List of Figures

List of Tables

Introduction

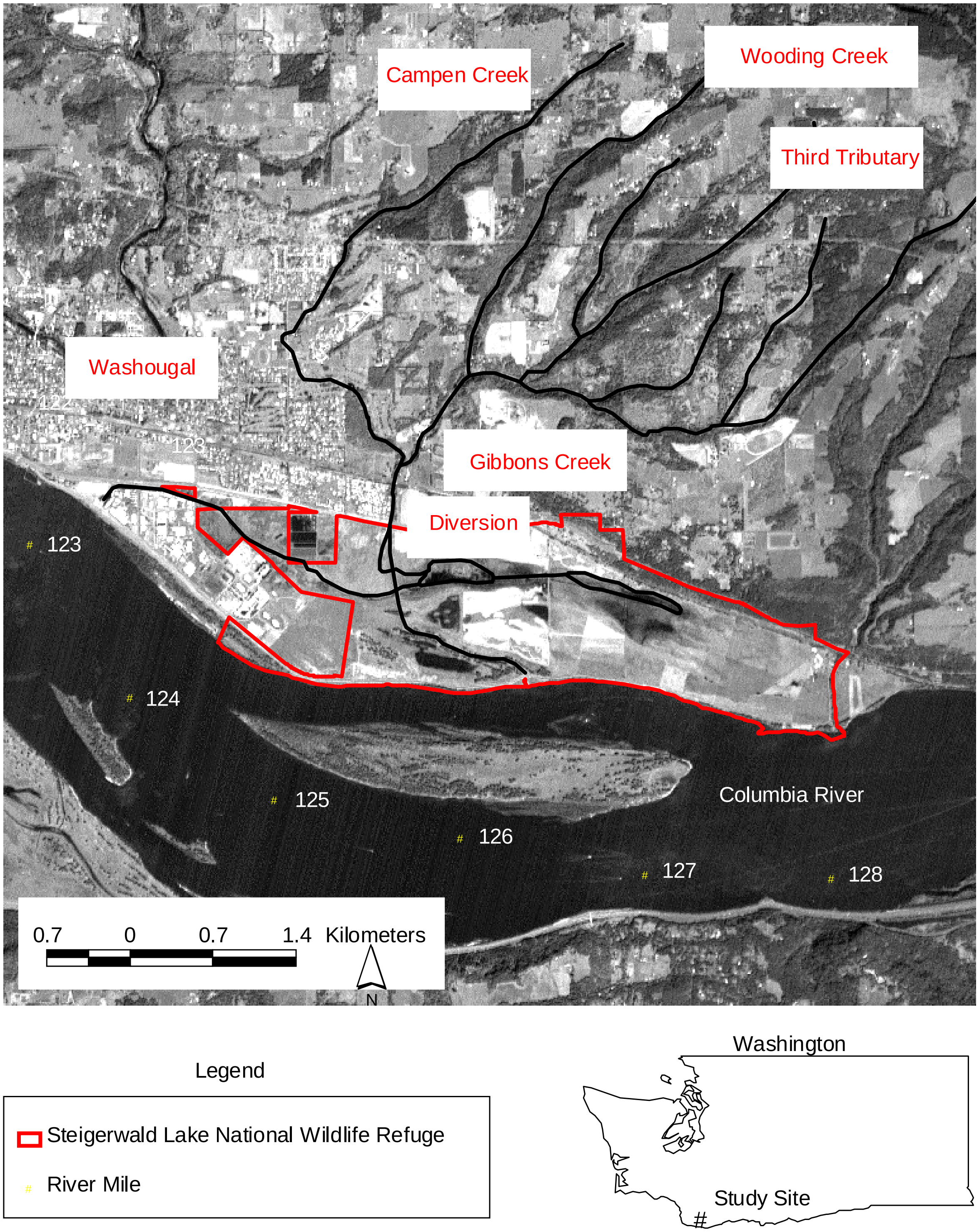
A watershed analysis, consisting of instream habitat and biotic surveys, was conducted in the Gibbons Creek, Washington watershed by the U.S. Fish and Wildlife Service, Columbia River Fisheries Program Office (CRFPO), between 1997 and 1999. This analysis was conducted to assess the overall condition of the watershed, its biota, and to determine preservation and restoration priorities.

Physical Description

The Gibbons Creek (GC), watershed is located at the west end of the Columbia River Gorge in the southeastern corner of Clark County, Washington (Figure 1). The drainage flows generally south and is composed of four primary tributaries: GC, Campen Creek (CC), Wooding Creek (WC) and one unnamed tributary, referred to in this report as Third Tributary (TT). Gibbons Creek joins the Columbia River at Columbia River km 202. About half of this 36 km² watershed is contained within the city limits of Washougal, Washington. Land ownership is mostly private, except for the lower 2.1 km where GC flows through Steigerwald Lake National Wildlife Refuge (SWR) and for two sections of Campen Creek that are City of Washougal property. Portions of the GC drainage east of Sunsetview Road are included within the Columbia River Gorge National Scenic Area. Gibbons Creek is classified as a Class A stream by the State of Washington (Washington Administrative Code 173-201A-030 section 2).

Background

Historically, GC flowed into Steigerwald Lake, which was contained in the Columbia River's floodplain, and then into the Columbia River. Gibbons Creek supported runs of coho salmon, steelhead, and coastal cutthroat trout. In 1962, Washington Department of Fisheries staff observed "many" juvenile coho in lower reaches of GC (Fiscus 1978). In the 1930s, attempts were made to drain the lake and wetlands for agricultural use (J. Clapp, Steigerwald Lake National Wildlife Refuge, personal communication). In 1966, a dike was constructed to isolate and drain this lake and its associated wetlands and the creek was diverted through a tidegate and pumping station at the Port of Camas-Washougal (Port). If the Columbia River was above an elevation of 11.5 feet, this tidegate would close and pumping would begin. At these times upstream migrating fish could not enter the creek and emigrating fish had to leave it through the pump (Bicknell 1988). Consequently, fish passage between Gibbons Creek and the Columbia River was dependent on the river's elevation during migration periods. Despite this limitation, anadromous coho salmon and steelhead were still found in the creek in 1985 (Bicknell 1988).



UTM NAD 27 Zone 10
Courtesy of USFWS and Ducks Unlimited

Figure 1. Gibbons Creek study area.

The entire drainage is within the Evolutionarily Significant Units (ESU) for a listed species and a candidate species under the Endangered Species Act (ESA): steelhead trout (*Oncorhynchus mykiss*), listed as threatened; and coho salmon (*Oncorhynchus kisutch*), candidate for listing. Steigerwald Lake National Wildlife Refuge, a 967 acre U.S. Fish and Wildlife Service (FWS) refuge encompassing the lower reaches of Gibbons Creek (Figure 1), was purchased by the U.S. Army Corps of Engineers as mitigation for the construction of the second powerhouse at Bonneville Dam (Bicknell 1988). Primary objectives of the purchase of SWR were to realign lower Gibbons Creek so that it would not flow through the Port area, thereby reducing pumping costs, and to provide uninhibited fish passage through the refuge to upstream spawning areas. The federal government, including the FWS, has spent over $7.5 million in acquiring land, constructing an elevated fish passage channel and a fish ladder at GC's confluence with the Columbia River, and re-routing GC through these structures to meet the objective of unrestricted fish passage through SWR (Figure 2). Resumption of unrestricted fish passage occurred in 1992.

A water diversion structure is present on GC just below the north boundary of SWR (Figure 2). This structure diverts GC from its old channel, which flowed into Steigerwald Lake, into the elevated channel that conveys water to the Columbia River. The structure is designed to pass 70 cubic feet per second (cfs) into the elevated channel and to force excess water through a screened opening into the old channel and the Steigerwald Lake bed. Bedload from storm events periodically reduced the transfer capacity of the structure in previous years and the channel immediately upstream of the diversion has required regular cleaning by excavator (USFWS, unpublished data).

In early 1996, a sediment plume from holding ponds of a gravel mining operation impacted about 1.1 km of coho and steelhead spawning habitat (USFWS, unpublished data). A monetary settlement was received from the responsible party as mitigation for these impacts. This settlement was to be used in the GC basin for fish habitat restoration and enhancement.

To identify habitat restoration and protection priorities and to evaluate the sources of the bedload which impede operation of the GC diversion structure, the FWS evaluated physical habitat and biological community structure and health in GC. Habitat surveys were conducted in 1997 and 1998 and biological monitoring was conducted from 1996 to1999.

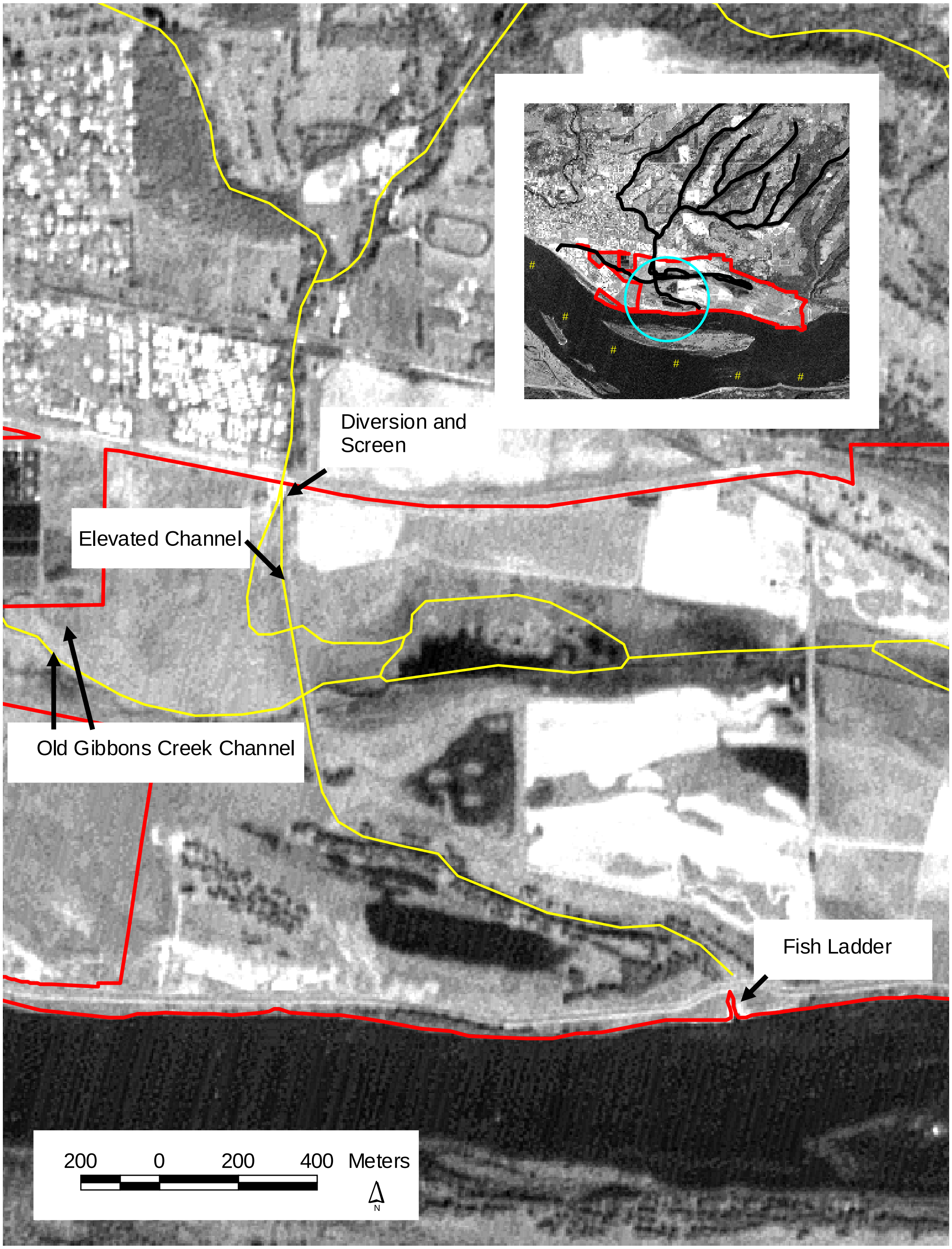


Figure 2. Lower Gibbons Creek.

Methods

Physical Habitat

Stream habitat surveys were conducted according to a modified U.S. Forest Service Level II Stream Inventory Protocol (Anonymous 1997, 1998). Our methods differed from the standard protocol in that we measured all standard habitat parameters including length, width, depth, and amount of large woody debris (LWD) at each habitat unit rather than estimating them. We standardized pool designations based on a bankfull width and residual pool depth relationship (Schuett-Hames et al. 1993). For GC streams, pools in channels with bankfull widths from 0 to 2.5 m were at least 0.1 m deep, whereas pools in channels between 2.5 and 5 m bankfull width were at least 0.2 m deep. Riparian habitat characteristics (e.g. canopy density, riparian species composition) and stream channel morphology (e.g. bankfull width, substrate composition) were measured as a systematic sample of at least 20 percent of surveyed habitat units. Channel morphology was classified according to Rosgen (1994). The LWD was classified into three size groups: small, one foot in diameter twenty five feet from the large end of the log; medium, two feet in diameter fifty feet from the large end; and large, three feet in diameter fifty feet from the large end (Anonymous 1997, 1998). Land uses were qualitatively described for each reach. GC, CC, and TT were broken down into three separate reaches and WC into two based on stream morphology and size (Table 1). All streams were surveyed to their headwaters.

Table 1. Start and end points of habitat survey reaches in Gibbons Creek drainage 1997 and 1998.

Reach	Start Point	End Point
GC1	Confluence with Columbia River	Evergreen Highway Bridge
GC2	Evergreen Highway bridge	Hans Nagel Road
GC3	Hans Nagel Road culvert	Spring
CC1	Confluence with GC	Acker Road (“Q” Street)
CC2	Acker Road (“Q” Street)	362nd Avenue
CC3	362nd Avenue	Spring (near 20th Street)
WC1	Confluence with GC	Power line right of way
WC2	Power line right of way	Spring (near SE 380th Avenue)
TT1	Confluence with GC	First tributary (0.7 km from confluence)
TT2	First tributary	Natural falls
TT3	Natural falls	Spring (near Moffet Road)

At sites where biological inventories were made, conductivity and temperature were measured with a calibrated Hach 44600 conductivity meter and pH was measured using a Hach pH test kit (Hach Company, Loveland, Colorado). Thermographs were deployed to continuously monitor temperatures in mainstem GC at river kilometers (Rkm) 0.1, 1.3, 2.3, and 6.4; in WC 30 meters above its confluence with GC; and in CC at Rkm 0.1 and 1.0 above its confluence with GC. Discharge was regularly measured in 1998 and 1999 above and below the diversion structure to evaluate both flow regimes and the diversion’s effects on flow. In spring 1998, a staff gage was installed at the diversion and a stage-discharge relationship calculated so that flow could be estimated from staff gage height. We reviewed the sporadic measurements taken at these sites prior to our study as part of our evaluation of the diversion structure (USFWS, unpublished data).

Surface substrate composition was measured by Wolman pebble counts (Wolman 1954) at representative units where all habitat parameters were measured. These units were selected in the first and second halves of reaches (1997) or at regular intervals throughout the surveyed reach (1998). The results of these counts were averaged to produce mean values for the sampled reaches. This method is designed to reveal broad patterns in substrate composition and stream energy, and not the substrate composition of salmonid spawning areas. However, general patterns of very small substrates (fines) and substrates suitable for salmonid spawning (6-128 mm) are discussed because of their importance to salmonid and aquatic community ecology (Kondolf and Wolman 1993, Kondolf 2000) (Table 2).

Table 2**.** Size range and definition of substrate classes used for substrate data collected using the Wolman pebble count method during habitat survey (Kondolf and Wolman 1993).

Substrate class (mm)	Definition
<6	fines
6-64	gravel (used for spawning by small bodied salmonids, <350 mm at maturity)
65-128	cobble (used for spawning by large bodied salmonids, >350 mm at maturity)
>129	boulder and other, large substrates, including bedrock

Although coho (*O. kisutch*) are often larger than 350 mm at maturity, they often spawn in substrates smaller than 65 mm (e.g., McMahon 1983, Reeves et al. 1989). Therefore, substrates suitable for smaller salmonids are considered as potentially suitable for coho as well. “Salmonids” are defined as Pacific salmon (chinook, *O. tshawytscha*; coho, *O. kisutch*; and chum, *O. keta*) and trout (cutthroat, *O. clarki*; and steelhead, *O. mykiss*) species for the purposes of this report.

Biological Inventory

<u>Outmigrant Salmonid Trapping</u>

In 1998 and 1999, emigrating salmonid smolts and other fish were trapped by winged fyke nets in GC 1.3 km upstream from its confluence with the Columbia River, on SWR. This location was chosen because it represented the downstream extent of salmonid habitat based on electrofishing and previous fyke net surveys (USFWS, unpublished data), because the narrow, clay-bottomed channel provided an excellent trap site, and because the trap site was easily accessible to agency personnel but was not visible to the public. In 1998, two 1-m-diameter fyke nets were fished as one trap: the second net was placed immediately downstream of the first and the catches of the two nets were combined. In 1999, a 1.3-m-diameter fyke net was operated alone. Except for diameter and the height of the wings (same as the diameter of the respective nets), all three fyke nets had the same characteristics: each net was double-throated, D-framed, and covered with 5 mm nylon mesh. The fyke net was placed in the thalweg. Two 5 mm nylon mesh wings, extending from the corners of the first fyke frame at about 45° upstream to each streambank, funneled fish to the trap. To provide debris and velocity refuge for trapped fish, a wood-framed live-box (92 cm long, 92 cm deep, and 61 cm wide), covered with 4 mm steel mesh, was attached to the fyke net's cod-end. The connection between the fyke net and live-box was a 15-cm-long, 15-cm-diameter steel pipe.

The fyke net trap was fished for the entire spring emigration period both years (late March to early June) and was checked at 24 hour intervals. Captured fish of all species were anesthetized (40 mg/l MS-222), measured (fork length (FL), mm), weighed (g), and examined for anomalies or parasites. Prevalence, intensity, and abundance of parasites were summarized according to Margolis et al. (1982). Juvenile salmonids also were classified as parr or smolt (Hoar 1976) and were marked with a colored dye tag in the anal fin (New West Technologies, CA). Recaptured smolts were enumerated, the color of their mark recorded, and were released 100 m downstream of the trap. Newly marked salmonids were allowed to recover, then were released 1 km upstream of the trap. In 1999, we released fish 0.7 km upstream from the trap in a successful attempt to reduce the time of migration between the release site and the trap. In 1998, a subsample of smolts was held for 24 hours to determine short-term mortality and mark retention. In 1999, all marked smolts were held in a live-car until dusk daily, when mark retention and survival rates were noted. The fish were then released. In both years, the trapping season was divided into weekly strata; a new marking color was used for each 7 day period.

Smolt population abundance was estimated from the percentage of marked fish recaptured, corrected for short-term mortality and mark retention (Thedinga et al.1994; Murphy et al. 1996). Variance was calculated using a 1000 iteration bootstrapping method (Efron and Tibshirani 1986, Murphy et al. 1996). The weekly marking periods were statistically compared (Chi-square, alpha = 0.05) to detected differences in trap efficiencies between the periods. If no differences were detected, marking periods were pooled to increase sample sizes and maximize statistical confidence. When differences were detected, groups were not pooled, and estimates and variances were calculated separately. All estimates and variances were then summed to provide final estimates and variances for each year. Confidence bounds were calculated as 95% CL = 1.96*(V-2) where V is the variance found by bootstrapping.

During 1999, we examined scales from salmonids captured in the trap to determine their ages. Scales were collected from nearly all steelhead and coastal cutthroat handled and from a subsample of coho.

Spawning Ground Surveys

For spawning ground surveys, standardized stream reaches totaling 4.3 km were surveyed on each occasion and additional reaches were added as time permitted. The standard reaches were GC from the downstream end of the elevated channel to its confluence with TT, CC from its confluence with GC to Sunset View Road, and CC from its emergence onto Orchard Hills Golf Course to where it leaves the golf course.

Carcasses were measured (mm FL) and aged from acetate impressions made from scale samples. Spawning ground surveys for coho were conducted four times between October 22 and

December 1, 1997, and seven times between October 19 and December 16, 1998. Surveys for Pacific and western brook lamprey were conducted on May 6 and 9, 1998. During each survey, parts of both GC and CC were surveyed. Redd locations were flagged at the site and GPS coordinates were taken. Surveys were conducted at least once each week during spawning periods, discharge and visibility permitting.

Determination of Fish Distribution and Community Structure

Fish distributions and community structure in the GC drainage were determined by backpack electrofishing (Smith-Root Model 12-B, Smith-Root, Inc., Vancouver, Washington) October 6-8, 1998. Sampled stream reach lengths were 35 times their respective wetted widths (Lyons 1992). This technique resulted in stream lengths exceeding 100 m in all cases. Captured fish were handled as described earlier for migrating fish. Nine sites were selected: three in GC, three in CC, one in WC and two in TT. Sites were selected that represented the spectrum of habitat conditions in the drainage, both in terms of stream size and habitat quality. These sites included one upstream of each barrier to fish migration. As indices of abundance, catch-per effort (CPE = number of a species captured/length of stream sampled) and relative abundance (number of a species captured/total fish captured at a site) were calculated for some species captured (Simonson and Lyons 1995; Waite and Carpenter 2000). A subsample of captured fish was transported to the Lower Columbia River Fish Health Center (LCRFHC) for disease and parasite analysis. These fish were examined both internally (gills, liver, kidney, cranium) and externally (fins, skin, eyes) for a variety of pathogens and parasites. We examined otolith annuli from 34 sacrificed cutthroat and 10 coho to determine the ages of these fish.

Fish distribution information was augmented by four other methods: 1) pools in lower GC1 (below Rkm 0.8) were sampled four times by overnight fyke net sets (February 20 and June 6, 19, and 27, 1997) using a fyke net similar to that described previously; 2) a weir trap was operated intermittently during the fall and winter of 1997-1998 at GC Rkm 0.3 to sample migrating adults; 3) spawning ground surveys were conducted during spawning periods to evaluate the relative abundance and distribution of coho, Pacific lamprey, and western brook lamprey redds; and 4) recent electrofishing surveys (1994 to 1997) conducted by either FWS (6 surveys) or Washington Department of Fish and Wildlife (WDFW) (7 surveys) in the drainage were reviewed. These electrofishing surveys were not used to calculate total fish community information because in most cases only salmonids were netted or other species were netted only sporadically.

Biological condition of fish communities was assessed from electrofishing data by using a multimetric index because an Index of Biotic Integrity (Karr et al.1986) has not been developed for Washington streams. We used the metrics that Cuffney et al. (1997) used to describe Yakima River basin fish communities, except that we substituted the metric “number of sensitive species” for “percent individuals as tolerant species” (Table 3). We designated this as the Fish Community Index (FCI). We compared an additional metric, percentage of individuals as sensitive species, with the number of sensitive species metric to determine its ability to discriminate conditions in the smaller headwater streams in the drainage. We termed this comparison the Modified Fish Community Index (MFCI). Fish tolerance, trophic association, and origin were characterized following Zaroban et al. (1999). Scoring criteria followed Hughes and Gammon (1987) with higher scores indicating higher biological integrity (Table 3). Total scores were then used to assign sites to one of three qualitative classes of community integrity: good (16-20), fair (10-14), and poor (4-8).

Table 3. Fish community metrics and scoring criteria used to assess biological condition of sites in the Gibbons Creek watershed, 1998.

Metric	**Scoring criteria (%)**		
	1 (poor)	3 (fair)	5 (good)
Number of sensitive species	0	1-2	3+
Percentage of individuals as sensitive species	0-1	26-50	>50
Percentage of individuals as non-native species	>9	2-9	0-1
Percentage of individuals as omnivore/herbivores	>49	25-49	0-24
Percentage of individuals with external anomalies*	>5	2-5	0-1

*Includes external parasites

<u>Macroinvertebrate Collection, Distribution, and Benthic Index of Biotic Integrity (BIBI)</u>

Large macroinvertebrates, such as mussels and crayfish, were noted when they were encountered during habitat and fish sampling surveys. Invertebrate community health was determined from samples collected by kick-net from four of the locations at which fish community structure was evaluated: CC at the golf course and upstream of Q street, and GC upstream of Hans Nagel road and downstream of TT confluence with GC. Four kick-net samples were collected from riffles and pools at each site, and these samples were combined to create a composite sample (Plotnikoff 1994). A 300 organism subsample from each composite sample was identified to the lowest taxon possible, usually genus (Plotnikoff 1996). From these data, a Benthic Index of Biotic Integrity (BIBI) was calculated. This BIBI is composed of biological attributes or “metrics” chosen to reflect specific and predictable responses of stream macroinvertebrates to human activities across the landscape (Karr and Chu 1999). Nine metrics, calibrated for Puget Sound lowland ecoregion streams, were used to determine the BIBI score for GC (Plotnikoff 1999) (Table 4). Puget Sound lowland streams’ macroinvertebrate communities are very similar to those of streams in the Willamette Valley ecoregion, in which GC lies (Robert Plotnikoff, Washington Department of Ecology, personal communication).

Scores for each metric are 1, 3, or 5 and are summed to generate a total score between 9 and 45. These scores indicate condition of the stream habitat from which the invertebrates were collected. Scores 33-45 represent good conditions; 25-32, fair conditions; 9-24, poor conditions (modified from Karr and Chu 1999).

Table 4. Macroinvertebrate metrics and scoring criteria used to assess biological condition of sites in the Gibbons Creek watershed, 1998. Modified from Plotnikoff (1999).

Metric	**Scoring criteria**		
	1 (poor)	3 (fair)	5 (high)
Total number of taxa	<10	10-20	>20
Number of *Plecoptera* taxa	<3	3-5.5	>5.5
Number of *Ephemeroptera* taxa	<3	3-5.5	>5.5
Number of *Tricoptera* taxa	<2	2-4.5	>4.5
Number of long lived taxa	<0.5	0.5-2	>2
Number of intolerant taxa	<0.5	0.5-2	>2
Percent individuals in tolerant taxa	>50	20-50	<20
Percent of predator individuals	<5	5-10	>10
Percent dominance (3 taxa)	>75	50-75	<50

Results

Physical Habitat

In general, surveyed streams were characterized by high amounts (>25%) of fine sediments (Table 5). Exceptions were the upper two reaches of GC (GC2 and GC3) and TT3, where fine sediments comprised 19.4-21.8% of substrate composition (Table 5). Gravel (6-64 mm) comprised the second most common category of substrate (29.4-43.5%) (Table 5).

Historic timber harvest (ca. >30 years) was evident along all reaches of stream. As a result, although riparian canopy density was high (>85%) in most surveyed reaches, trees in the riparian zone were mostly early seral (young and small). In most reaches of stream, alder dominated the overstory. The nature of the riparian forest (young and alder dominated) is reflected in the size of the LWD in stream channels; the small class of LWD predominated all stream reaches except GC3 and TT3 (Table 5). The lower canopy densities in GC1, CC1, and TT1 reflect the elevated channel reach of GC1, the Orchard Hills Golf Course reach of CC1, and the road along TT1, all areas where large portions of riparian vegetation are removed or suppressed. Except for WC1, WC2, and GC2 (23.4-36.7 pieces/km), the LWD counts for most stream reaches were low (Table 6). Channelization was evident in GC1, CC1 in the golf course, and TT1.

Riffle habitat dominated most stream reaches, generally comprising >80% of habitat area (Table 6). Exceptions were GC1, GC3, and CC1, where riffles were 33.8-60.4% of habitat area (Table 6). The relatively high amount of pool habitat in GC1 is somewhat misleading, reflecting a long series of shallow, wide pools in the lower 1.3 km of this reach; the upper 2.28 km of this reach is largely channelized riffle habitat. Residual pool depths generally decreased with decreasing stream size (Table 6). However, GC3 average residual depth (0.4 m) was the second deepest in the watershed, reflecting the high quality of pools in that reach.

Table 5. Habitat survey results in Gibbons Creek drainage, 1997-98.

	Reach										
Parameter	GC1	GC2	GC3	CC1	CC2	CC3	WC1	WC2	TT1	TT2	TT3
Distance surveyed (km)	3.58	2.59	2.93	2.75	2.59	1.14	1.25	2.56	0.70	1.07	2.02
Percent Habitat Area											
Side Channels (%)	0.7	1.4	4.0	0	0.1	2.7	4.3	9.7	0.8	0.0	0
Riffles (%)	33.8	85.2	59.2	60.4	92.2	97.3	87.2	79.9	86.3	83.6	94.9
Pools (%)	65.5	13.4	35.0	39.4	6.5	0.0	8.5	4.5	12.5	4.0	2.0
Pool Quality											
Pools/km	11.4	17.8	23.9	19.7	17.7	0.0	23.1	19.6	25.8	14.0	11.4
Total Pool Area (m^2)	14265	1202	3266	2319	387	0.0	269	264	208	114	94
Average Residual Pool Depth (m)	0.68	0.38	0.40	0.40	0.17	--	0.29	0.20	0.30	0.23	0.21
Large Wood Per Km											
Large & Medium	2.3	5.8	7.9	1.1	5.0	0.0	8.8	4.3	1.4	1.9	8.9
Small	11.4	21.7	6.5	5.8	10.8	6.2	27.9	19.1	7.1	12.1	7.9
All classes	13.7	27.5	14.4	6.9	15.8	6.2	36.7	23.4	8.5	14.0	16.8
Riparian Canopy Density											
% Canopy Density (stan. dev.)	64.9 (31.9)	84.6 (17.7)	96.5 (12.4)	63.5 (43.0)	92.5 (15.1)	88 (20.8)	84.9 (18.0)	93.6 (10.3)	78.9 (25.2)	87.3 (11.0)	96.4 (5.0)
% of Total in Substrate Class											
Sample Size Substrate Size	2	3	3	3	10	3	2	10	2	7	9
<6 mm	49.6	19.4	21.8	53.5	34.4	63.3	32.5	37.2	39.6	44.7	21.3
6-64 mm	30.7	38.7	43.5	38.5	35.9	30.5	33.3	36.4	29.4	30.6	38.7
65-128 mm	15.5	24.3	27.7	5.5	16.1	2.7	18.0	12.5	15.4	12.8	20.4
>129 mm - bedrock	4.4	17.6	7.0	0.6	16.5	3.5	16.3	13.7	15.7	12	19.7
Sinuousity	1.02	1.15	1.26	1.34	1.22	1.00	1.03	1.16	1.00	1.19	1.0
Rosgen Type	C4	C3b	C4b	C4	C4b	B6a	B4a	C4b	B4a	B4a	B6a

Off-channel habitat (side-channels, off-channel ponds) was uncommon in most surveyed reaches (Table 6). However, side-channels were important habitat features in WC1, WC2, and

GC3, providing 4-9.7% of habitat area (Table 6).

Water temperatures were about 4 degrees cooler on average during summer months at GC Rkm 6.4 and WC Rkm 0.1 than other locations in the watershed (Table 6). During all months, stations in CC tended to be warmer than other stations and warmed quickest in spring (Table 6). Temperatures exceeded 18.0°C only once during 204 recorded days between April and October 1998 at GC Rkm 6.4 (just above Hans Nagel Road) whereas temperatures exceeded 18.0°C 37 times in 214 recorded days during the same months at GC Rkm 2.34, downstream of the CC confluence (Table 7). Temperatures in CC exceeded 18.0°C on 85 of 214 recorded days April to October 1998. Conversely, WC exceeded 18.0°C only twice on 204 recorded days during those months (Table 7). Point measurements of dissolved oxygen ranged from 9.81-10.31 mg/l, and pH ranged from 7.5-8.0.

Table 6**.** Mean monthly temperatures and range (^{o}C) at bimonthly intervals, May 1998-March 1999, at five thermograph stations in the Gibbons Creek watershed.

	Month					
Location	May	July	September	November	January	March
GC 0.8	11.5 (8.7-17.6)	16.8 (13.4-22.1)	15.3 (13.4-18.6)	12.0 (10.9-13.4)	10.7 (8.4-12.7)	11.56 (8.7-14.8)
CC 0.1	12.4 (9.5-12.1)	18.6 (14.0-18.0)	16.5 (15.0-18.0)	13.4 (7.7-12.4)	9.7 (10.9-12.9)	13.2 (6.0-12.7)
CC 1.0	14.4 (9.4-19.3)	17.8 (14.4-24.9)	15.8 (13.3-20.8)	13.4 (7.7-12.4)	12.9 (5.1-9.3)	13.2 (6.0-12.7)
WC 0.1	10.8 (8.6-14.1)	14.5 (12.0-18.6)	13.4 (11.0-16.6)	9.1 (7.8-10.7)	7.5 (5.5-9.0)	8.1 (5.6-11.6)
GC 6.43	11.8 (8.6-14.1)	14.6 (11.7-18.2)	11.2 (10.8-16.7)	8.7 (7.5-10.2)	7.3 (6.1-8.6)	7.8 (5.0-10.5)

Table 7**.** Number of days that temperatures exceeded state water quality standards (18^{o}C) at six thermograph stations in the Gibbons Creek watershed, 1998.

	Month						
Location	May	June	July	August	September	October	Total
GC 1.3	0	1	17	15	4	0	36
GC 2.3	0	1	17	16	4	0	37
CC 0.1	1	5	26	22	28	1	86
CC 1.0	2	7	26	31	15	1	85
WC 0.1	0	0	2	0	0	0	2
GC 6.43	0	0	1	0	0	0	1

A review of 13 flow measurements taken during August and September 1997-1998 at the diversion structure revealed a mean base flow of 3.8 cfs (range, 1.5-9.7 cfs). When mainstem GC habitat measurements were surveyed in 1997, flow was 3.0 cfs. The maximum flow we measured in the elevated channel was 71 cfs in May 1998, but the amount of flow in the elevated channel depended on the amount of sediment build-up in the stream channel above the diversion. Sediment build-up forced more water over the concrete sill and out of the stream channel. For example, in December 1996, at high levels of sediment build-up, 44 cfs was flowing in the elevated channel and 58 cfs was spilling over the concrete sill. By contrast, the 71 cfs flow in the elevated channel came after the channel upstream of the diversion had been excavated. At high flows, the screened overflow clogged, diverting almost all excess flow over the unscreened sill.

Biological Inventory

Outmigrant Salmonid Trapping

Three species of smolt salmonids were captured: steelhead, coastal cutthroat trout, and coho salmon. Coho were most abundant, both by numbers captured and by abundance estimates (Table 8). Not enough trout smolts were captured in 1998 to generate abundance estimates by species, so steelhead and coastal cutthroat catches were combined to produce an estimate (Table 8). Steelhead smolts were more abundant both in catch and abundance estimates in 1999 (Table 8). Mean lengths were longer for all species in 1999 (Table 8). Trap efficiencies were higher for all species in 1999 as well (Table 8). Most coastal cutthroat and steelhead smolts were age 2, and all coho smolts were age 1, based on the sample of fish that was aged (Table 9).

Table 8. Number captured (n), mean fork length (MFL), abundance estimate (AE), and mean trap efficiency (MTE) for salmonid smolts trapped in Gibbons Creek, WA, 1998-99. An * denotes a significant difference between trap efficiencies in one or more trapping periods.

		1998		
Species	n	MFL (range)	AE (95% CI)	MTE (range)
Coho	558	108 (60-146)	1935 (1398-2472)	0.25 (0.12-0.31)*
Steelhead	42	147 (81-199)	760 (86-1434)	0.10
Coastal Cutthroat	61	164 (72-267)		
		1999		
Coho	485	119 (87-170)	1253 (1126-1380)	0.55
Steelhead	179	185 (93-249)	366 (297-435)	0.50
Coastal Cutthroat	133	170 (136-182)	402 (292-511)	0.35

Table 9. Age distributions of salmonid smolts in Gibbons Creek, WA, 1998-99. Age distributions are represented as the proportion of the number of fish aged (n) in each age group.

		1998		
Species	n	Age 1	Age 2	Age 3
Coho	0	--	--	--
Steelhead	20	0.20	0.75	0.05
Coastal Cutthroat	27	0.26	0.70	0.04
		1999		
Coho	16	1.00	0	0
Steelhead	130	0.10	0.75	0.15
Coastal Cutthroat	81	0.27	0.68	0.05

In 1998, the smolt emigration period (23 April – 14 May) was up to a month shorter in duration than 1999 (1 April – 1 June), with the vast majority of fish emigrating within a two week period (Table 10). Temperatures were warmer during the migration period in 1999 than 1998 (Table 11). In both years, the vast majority of migration for all species took place when mean daily temperatures at the trap were between 12 and 16°C (Table 11). Steelhead and coho emigration showed a significant difference with respect to water temperature. However, cutthroat showed no significant difference in migration.

Table 10. Emigration quartiles for coho, cutthroat and steelhead smolts emigrating from Gibbons Creek, 1998-1999.

	Emigration quartile			
Species	25%	50%	75%	100%
1998				
Coho	2 May	4 May	7 May	23 May
Cutthroat	25 April	30 April	1 May	5 May
Steelhead	25 April	1 May	2 May	12 May
1999				
Coho	21 April	30 April	18 May	16 June
Cutthroat	20 April	2 May	20 May	13 June
Steelhead	17 April	25 April	5 May	13 June

Table 11. Percentage of total coho, coastal cutthroat, and steelhead smolts emigrating within five temperature categories.

	Temperature category				
Species	8 to <10	10 to <12	12 to <14	14 to <16	16 to <18
1998					
Coho	3 (21)	12 (43)	42 (21)	43 (15)	0 (0)
Cutthroat	6 (8)	25 (31)	20 (23)	49 (38)	0 (0)
Steelhead	15 (8)	62 (42)	24 (29)	36 (21)	0 (0)
1999					
Coho	0 (0)	5 (15)	53 (58)	36 (22)	6 (5)
Cutthroat	0 (0)	7 (15)	62 (58)	27 (22)	4 (5)
Steelhead	0 (0)	1 (15)	57 (58)	33 (22)	9 (5)

Mean discharge during migration period was 12 cfs (range, 7-20 cfs) in 1998, and 15 cfs (range, 10-35 cfs) in 1999. Emigration was not correlated with periods of increasing discharge in 1998; discharge was decreasing for the initial 25 days of coho emigration, during which over 75% of coho emigrated. All cutthroat and steelhead emigrated within the period of decreasing discharge in 1998. However, in 1999 25% of coho, 33% of steelhead and 33% of cutthroat emigrated during the 12 days (16% of the migration period) that discharge was increasing. In both years, the entire emigration period occurred during increasing photoperiod (daylength).

Spawning Ground Surveys

Coho salmon

The 1997 peak counts were 28 coho adults (6.5 fish/km) on November 6, and 72 redds (10.0/km) on November 12. Of redds counted, 65% were found in our standard survey reaches. Nineteen coho carcasses were sampled; average length of females was 606 mm (n=10) and of males was 578 mm (n=9). All were age-3. Rainy weather and turbid water made viewing conditions difficult during 1998. Only one carcass was sampled, and the highest peak count in 1998 was 2 coho (0.5/km) on November 12. Eighteen total coho redds were counted in 1998 (4.0/km), 17 in GC and 1 in CC. Coho spawning was documented in the same reaches as juvenile coho were found: CC1, CC2, GC1, and GC2.

Pacific Lamprey

In 1998, we counted 30 Pacific lamprey redds. We measured 23 redds that averaged 0.43 m long, 0.45 m wide and 0.14 m deep. Of the 23 measured redds, 12 were on pool tailouts and 11 were in riffles. Redds were found in GC1, GC2 and CC1. Pacific lamprey spawning occurred in early May.

Western Brook Lamprey

We observed and measured 5 individual western brook lamprey redds. These redds averaged 0.23 m long, 0.34 m wide, and 0.08 m deep. Most western brook lamprey redds were found in clusters on pool tailouts (n=15, clusters counted once each). In three other instances, western brook lamprey adults were observed spawning inside of Pacific lamprey redds. Western brook lamprey redds were observed in riffles only in instances in which they were inside of a Pacific lamprey redd. We captured and measured 11 female and 6 male actively spawning western brook lamprey. Both sexes had the same average length (120mm). Redds were observed in GC1, GC2, GC3, TT1, and CC1.

Determination of Fish Distribution and Community Structure

We documented seventeen fish species in the GC watershed (Table 12). Three of these species, pumpkinseed sunfish (*Lepomis gibbosus)* and eastern banded killifish (*Fundulus diaphanus*), and brown trout (*Salmo trutta*) are not native to the Pacific Northwest. The only surveyed reach in which no fish were found was TT3 above a natural barrier falls (Table 5). Cutthroat were present in all areas surveyed except TT3 and were the only fish present upstream of the natural barrier falls in WC (Table 12). Where present they were either the dominant species (WC, TT2) or subdominant (GC3, Golf Course in CC1) except where coho were present (GC2, CC1 Rkm 0.8). They were most abundant in TT2 (CPE, 15.3) and WC upstream of the natural barrier chute (CPE, 17.6) (Table 13). Reticulate sculpin (*Cottus perplexus)* dominated catch in all reaches except GC3 (Table 13). The only place they were not present was upstream of the natural barriers in WC and TT (Table 5). Both western brook lamprey (*Lampetra richardsoni*) and reticulate sculpin were present in all reaches surveyed except TT3 and upper WC2 (Table 12).

Three species appeared incidental in GC: a single male chinook salmon (*Oncorhynchus tshawytscha)* was trapped in the weir at Rkm 0.3, and two chinook fry were captured in a fyke net set near Rkm 0.8; a single adult eulachon smelt or candlefish (*Thaleichthys pacificus)* was found dead during a spawning survey at Rkm 2.0, and two brown trout were captured: one, a gravid female, was trapped at the weir and another was electrofished in Campen Creek, adjacent to Washougal High School.

Coho juveniles were present in the lower reaches of GC and CC (Table 12). Steelhead juveniles were only captured in the GC mainstem up to Hans Nagel Road (Table 12), but were captured here consistently by WDFW in the shocking surveys we reviewed (WDFW, unpublished data). Pacific lamprey ammocoetes were only captured on SWR, but adult Pacific lamprey have been observed spawning elsewhere in the drainage (see below).

Peamouth, three-spined stickleback, pumpkinseed sunfish, and eastern banded killifish were captured only in fyke net sets in the lower 1.3 km of GC on SWR, a reach dominated by shallow, wide, and warm pools. Adult largescale suckers were captured in the smolt emigration trap beginning in late April each year and ending in mid-May as they emigrated after spawning. Spawning suckers were observed just below the diversion structure. Sexually mature peamouth were captured in early June, 1997 in a fyke net set at Rkm 1.2. Juvenile largescale sucker, speckled dace, longnose dace, three-spined stickleback, reticulate sculpin, prickly sculpin, red legged frogs (*Rana aurora*), bullfrogs (*Rana catesbeiana*), and western painted turtles (*Chrysemys picta*) were also captured in the emigrant trap.

The crayfish, *Pacifisticus lenisculus*, was found in every surveyed reach. The western pearlshell mussel, *Margaritifera falcata*, was found only in CC1.

Table 12**.** Selected aquatic species distributions in the Gibbons Creek watershed. Reaches are listed by stream and reach number. Stream abbreviations are: GC = Gibbons Creek; CC = Campen Creek; WC = Wooding Creek; and TT = Third Tributary. Species abbreviations are: COS = coho salmon; CHS = chinook salmon; CUT = cutthroat trout; STD = steelhead trout; BRN = brown trout; RES = reticulate sculpin; PRS = prickly sculpin; PCL = Pacific lamprey; WBL = western brook lamprey; LRS = largescale sucker; CAN = candlefish; PMO = peamouth; LND = longnosed dace; SPD = speckled dace; PUS = pumpkinseed sunfish; KIL = eastern banded killifish; TTS = three-spined stickleback; WEP = western pearlshell mussel; LIM = limpet (genus *Fisherola*); CRF = crayfish.

	Reach									
Species	GC1	GC2	GC3	CC1	CC2	CC3	WC1	WC2	TT1	TT2
COS	X	X		X			X		X	
CHS	X									
CUT	X	X	X	X	X	X	X	X	X	X
STD	X	X								
BRN	X			X						
RES	X	X	X	X	X	X	X	X	X	X
PRS	X									
PCL	X	X								
WBL	X	X	X	X	X	X	X	X	X	X
LRS	X			X						
CAN	X									
PMO	X									
LND	X	X		X						
SPD	X			X						
PUS	X									
KIL	X									
TTS	X									
WEP				X						
LIM		X		X	X					
CRF	X	X	X	X	X	X	X	X	X	X

Table 13. Catch per effort, multiplied 100X and relative abundance at each site electrofished for species presence in October 1998. Catch per effort is calculated as effort per length sampled, and relative abundance is proportion of total catch of all species at a site.

	Electrofishing Sites							
Species	GC1	GC2	GC3	CC1 0.8	CC1 1.0	CC2	WC2	TT2
COS	0.4 (0.02)	3.8 (0.25)	--	1.6 (0.18)	--	--	--	--
CUT	--	2.3 (0.15)	6.0 (0.31)	0.7 (0.07)	3.4 (0.21)	3.7 (0.24)	7.7 (1.0)	5.9 (0.51)
STD	--	1.0 (0.07)	--	--	--	--	--	--
RES	17.1 (0.86)	4.8 (0.32)	13.0 (0.70)	6.2 (0.71)	12.6 (0.76)	11.2 (0.72)	--	5.6 (0.49)
SPD	0.09 (0.01)	--	--	0.1 (0.02)	--	--	--	--
LND	1.0 (0.05)	2.8 (0.19)	--	--	--	--	--	--
TTS	0.4 (0.02)	--	--	--	--	--	--	--
PCL*	0.2 (0.01)	--	--	--	--	--	--	--
WBL*	0.7 (0.04)	0.3 (0.02)	0.3 (0.02)	0.1 (0.02)	0.5 (0.03)	0.6 (0.04)	--	--

*Note: electrofishing equipment was on settings for maximum efficiency in capturing salmonids. Lamprey CPE is included for general comparison only.

Benthic Index of Biotic Integrity (BIBI) and Fish Community Characteristics

The BIBI scores collected in riffle habitat were good in the upper reaches of both GC and CC and were poor in the golf course reach of CC (Table 14). The FCI characterized the three CC sites as fair and the remaining five sites as good. When the metric "percent of individuals as sensitive" was substituted for "number of sensitive species" (MFCI), two sites (GC1 on SNWR and GC2) dropped from good to fair, one site (CC1 at golf course) dropped further in the fair category, and two sites already characterized as good (WC and TT) increased in score. The metric most responsible for lowered metric scores was "percent anomalies". Anomalies were very common among sensitive species (salmonids) at five locations: CC Rkm 2.6 (67%); CC Rkm 1.0 (golf course - 77%); CC Rkm 0.1 (59%), GC Rkm 4 (21%) and GC Rkm 0.8 (60%). Conversely, anomalies were much less common at the three upper watershed sites: WC (5%), GC Rkm 6.3 (6%) and TT (0%). The most common anomalies were clubbed or worn fins, "black spot" from *Neascus* sp., and *Salmincola californiensis* infection. Among emigrant salmonids, 19% of coho, 45% of cutthroat, and 45% of steelhead had anomalies. The vast majority of anomalies among the trout were *S. californiensis* , with 60% and 80% of cutthroat and steelhead anomalies attributable to this copepod, respectively. Among other anomalies recorded were tapeworms, protruding anus, fungus, cysts on body and fins, and eroded or clubbed fins.

The age distribution for 34 cutthroat sacrificed for fish health (range, 64 mm to 223 mm FL) was age-0 18%, age-1 55%, age-2 24%, and age-4 3%. Ten coho (range, 76-98 mm FL) were all age-0.

Table 14. Results and scores (in parentheses) for a riffle benthic index of biotic integrity (BIBI), a multimetric fish community index (FCI), and a modified FCI at eight sites within the Gibbons Creek drainage.

Stream/Rkm	Site characteristics	BIBI	FCI*	MFCI**
GC 1.3	artificial stream channel, SNWR	---	good (16)	fair (14)
GC 4.4	second growth forest	good (37)	good (16)	fair (14)
GC 6.3	horse farm, intact riparian forest	good (41)	good (16)	good (16)
CC 0.1	suburban landscaping	---	fair (14)	fair (14)
CC 1.0	golf course	poor (21)	fair (14)	fair (12)
CC 2.6	suburban w/riparian buffer	good (42)	fair (14)	fair (14)
WC 3.6	suburban w/riparian buffer	---	good (16)	good (18)
TT 1.2	second growth forest	---	good (18)	good (20)

*Includes # sensitive species metric
** Includes % sensitive species metric

Fish Passage

Seven fish passage barriers were found in the watershed, at least on in each stream surveyed (Table 15). Two barriers were natural, and 5 (all culverts) were man-made. These culverts blocked 6.9 km of stream to fish migrations: 3.0 km upstream of the Hans Nagel culvert on Gibbons Creek and 3.9 km upstream of the Q street culvert on Campen Creek. (Table 15).

Table 15. Fish barrier types and locations in Gibbons Creek drainage.

Reach	Barrier	Distance from confluence (Rkm**)**	Distance from headwaters (Rkm**)**
GC3	culvert at Hans Nagel Rd	6.2	3.0
GC3	culvert on private land	6.4	2.8
CC1	culvert at Q street	2.5	3.9
CC3	culvert at unnamed site	5.4	3.8
CC3	culvert at unnamed site	6.1	3.1
WC2	natural chute	1.5	2.3
TT2	natural falls	1.8	2.0

Fish Health

Four microparasite genera and one virus species were identified from coho and coastal cutthroat examined by LCRFH staff (Table 16). Another parasite, the copepod *Salmincola californiensis*, was present in juveniles sampled by both electrofishing and outmigrant trapping (Table 17). Prevalence, intensity, and abundance were significantly different between coho and the other two species but not between cutthroat and steelhead.

Table 16. Microparasite genera and pathogens, by number and (proportion), identified from fish in the Gibbons Creek drainage, 1998. Microparasites were identified to genus; BKD is *Renibacterium salmonarium*, pathogen for bacterial kidney disease.

Species	n	*Gyrodactylus*	*Scyphidia*	*Epistylus*	*Nanophyetus*	BKD
Coho	19	1 (0.05)	2 (0.11)	3 (0.16)	12 (0.63)	2 (0.11)
Cutthroat	35	1 (0.03)	4 (0.11)	0	22 (0.63)	2 (0.06)

Table 17. Prevalence, abundance, and intensity of *Salmincola californiensis* on stream resident and migrating salmonids in the Gibbons Creek drainage, 1998-99.

Species	n	Prevalence (%)	Intensity	Abundance
Coho (non-migratory)	46	0	0	0
Coho (migratory)	615	0.007	---	0.16
Cutthroat (non-migratory)	97	9	2.1	0.21
Cutthroat (migratory)	138	27	2.98	0.97
Steelhead (non-migratory)	4	25	3.0	0.75
Steelhead (migratory)	183	36	3.68	0.92

Discussion

Overall, our study identifies four primary negative impacts to the watershed's aquatic ecosystem: 1) habitat fragmentation, especially by road culverts; 2) riparian vegetation removal; 3) instream habitat simplification by LWD input reduction and LWD removal; and 4) spawning habitat degradation by heavy inputs of fine sediment. The result of these impacts includes far lower salmonid production than would be expected from a watershed of this size. For example, Fiscus (1978) estimated that the drainage should produce about 6,700 coho smolts; our estimates were 28.5% (1998) and 18.7% (1999) of this estimate. Clearly, removing barriers to the 6.9 km of habitat currently unavailable to anadromous fish, especially the very high quality habitat in GC3, will improve anadromous salmonid productivity.

After habitat fragmentation, the dearth of LWD in the watershed may reduce productivity of the drainage more than any other factor. In streams with similar characteristics to those we measured in the GC watershed, LWD is critical for sediment storage, maintaining stream channel morphology and stability, forming pools, and retaining nutrients (summarized in Lassetre and Harris 2001). These factors combine to control productivity of biotic communities, including salmonids and their food organisms. For example, studies of several salmonid species, including those species found in GC, determined that LWD is positively correlated with abundance, biomass, and spawning habitat availability and quality (summarized in Lassetre and Harris 2001). Studies of reference streams indicate that GC should have at least 50 pieces of LWD/km, twice as much as found in any reach except WC1, which is still about 10 pieces per km low (USDA et al. 1995). As a consequence, the number of pools and amount of pool area in all reaches of the drainage, which should range from 34-58/km and 0.5-0.75 habitat area depending on channel width, are lower than expected for a stream of this size (USDA et al. 1995, Hogan and Ward 1997). As with other studies (Bisson et al. 1988, Bugert et al. 1991), coho distribution appears associated with the quality of pool habitat available in the drainage, as we found them in the reaches with greatest residual pool depth (GC1, GC2, CC1) and in large pools in the downstream ends of WC1 and TT1. Remedies for these conditions include LWD placement and off-channel habitat development (Hogan and Ward 1997). Studies estimate that mainstem habitat development can increase coho, cutthroat and steelhead production by 1.8, 1.7 and 2.3 fold, respectively (Koning and Keeley 1997). Coho production can be dramatically enhanced by sidechannel or off-channel pond development; for example, a 1 ha pond can produce about 3,000 coho smolts (Koning and Keeley 1997). Suitable water temperatures throughout the year would be essential in enhanced sidechannel or off-channel pond habitat if they are to benefit rearing coho salmon. The optimum summer temperature range for coho rearing is between 10°C and 15°C and the upper lethal limit is 25.8°C (Laufle et al.1986). LWD or other overhead vegetation would be needed to provide protective cover and shade for rearing juvenile coho.

Consideration should be given to developing a spring-fed sidechannel for starting a chum salmon run in the drainage as part of recovery effort for that species. Chum salmon spawning has been most successful at restored habitat or constructed spawning channels that have clean gravel of suitable size and spring flow or upwelling groundwater (Greg Johnson, WDFW, personal communication; Matthew Foy, Fisheries and Oceans Canada, personal communication). Reports indicate that Campen Creek, and perhaps Gibbons Creek downstream of SR14, may have historically supported chum salmon (Hugh Fiscus, WDFW, personal communication).

The lack of LWD is excaberated in some reaches (e.g. GC1, GC2 along Sunset View Road, CC1 in the golf course, and TT1) by channelization due to encroachment by roads and other types of development. The recent acquisition of land adjacent to GC1 provides the opportunity to explore opportunities to use this property to establish more natural stream morphology in this reach, thereby improving both spawning and rearing habitat for salmonids. An alternative to this strategy is to develop rock-riffles in these reaches following the methods of Newbury et al. (1997). If this strategy is followed we recommend pursuing development of sidechannels and off-channel ponds as described above (Koning and Keeley 1997).

Amounts of surface fine sediments were high for most reaches, but particularly so for CC, WC and TT. In TT1, the road adjacent to the stream almost certainly contributes to the high levels of fines in that reach. Although there are no activities in some of these reaches (WC and TT2) that appear to be contributing to these sediment levels, persistence of the effects of past timber harvest along these reaches may contribute to these conditions (Salo and Cundy 1987). High amounts of fine sediment (>30%) can reduce salmonid spawning and overwintering success as well as degrade macroinvertebrate communities (Hicks et al. 1991, Kondolf 1999).

Both biotic indices we used indicate that elevated temperatures resulting from riparian vegetation removal in the CC drainage, coupled with high levels of fine sediment influx, has significantly degraded this stream. The high levels of anomalies among salmonids, especially in the lower reaches of this stream, indicate that it is at best marginal salmonid habitat. We did not measure for pesticides or herbicides, but these chemicals may also be contributing to the poor biological condition of this stream. Additional degradation likely will render this stream unsuitable to salmonids. Conversely, increasing stream shading will serve to lower stream temperatures, reducing the number of days the stream exceeds water quality standards (18°C - WAC -173-201A) and improving both salmonid habitat quality and overall biological integrity. Protecting riparian vegetation in all stream corridors is important to maintaining and improving habitat conditions.

Taken together, the high number of anomalies among both non-migratory and migratory salmonids is troubling. Examination of the literature indicates that the infection intensities we report for *S. californiensis* are not likely deleterious (Fasten 1921, Black 1982); however, the literature does not discuss the relationship between habitat conditions and infection characteristics. Even if *S. californiensis* infections are subtracted, the number of anomalies found in fish is still higher that would be expected (Hughes and Gammon 1987). In the Tualatin River basin, Friesen and Ward (1996) found 3.6% of coastal cutthroat trout with anomalies; by comparison, 18% of cutthroat smolts and 22% of non-migratory cutthroat we sampled had anomalies, excluding those with *S. californiensis*.

The BIBI was more sensitive than the suite of fish metrics we used (FCI or MFCI) at discriminating both good and poor sites. This may reflect the need for examining and perhaps modifying the metric used. For example, if as discussed above, the parasite taxa are not deleterious or representative of poor habitat quality, they should be excluded from the anomalies category. The MFCI appears more useful than FCI because it was able to better discriminate small, good quality headwater sites that only support one sensitive species (cutthroat). As such it provides useful information regarding habitat suitability for a coldwater fish assemblage, demonstrating differences in the relative biological health of aquatic habitats in the GC watershed.

Our emigrant trapping results indicate that emigrations are cued by both temperature and discharge. Temperature appears to be the primary cue, with localized bursts associated with discharge. As such, activities modifying either the temperature or discharge regimes of the watershed (Meehan 1991) will likely impact emigration timing (Holtby et al. 1989), alter life history characteristics (Hicks et al. 1991, Lichatowich et al. 1995), and perhaps affect the ultimate survival of the smolts (Bilton et al. 1982, Hicks et al. 1991). As a practical matter, if the diversion structure is not modified, or at the very least cleaned regularly, smolts emigrating to discharge cues may be entrained by spills over the concrete sill and lost to the population. The design of the diversion structure does not account for bedload or debris movement at even modest flows and thus will require frequent cleaning for both debris and bedload. Possible solutions are widening the elevated channel to accommodate at least bankfull flows; building a sediment trap of some kind upstream of SR 14; or removing the structure and allowing GC to reaccess Steigerwald Lake. Although it may be expensive and complex to accomplish, we recommend the latter solution. This solution will provide important spawning and rearing habitat for salmonids: Fiscus (1978) reported that prior to diking in the 1960s the reach of GC between SR 14 and Steigerwald Lake was heavily used by spawning coho. This solution will also allow riparian vegetation to be established along the stream corridor, will allow natural stream dynamics to operate, reducing the need for human intervention and maintenance, and will provide habitat for native coolwater species (peamouth, largescale sucker, etc.). One caution and consideration to this solution: implementation of this solution must consider the potential to create additional habitat for non-native fishes (sunfishes and especially predators such as bass *Micropterus* sp.) and native predatory fish (northern pikeminnow, *Ptychocheilus oregonensis*).

Smolt ages for the Gibbons Creek salmonid populations are consistent with other populations of similar latitudes: coho emigrate as age-1 (Weitkamp et al. 1995), and cutthroat and steelhead primarily as age-2 (Busby et al. 1996, Trotter 1997). The predominance of cutthroats is typical for a drainage the size of GC (Hartman and Gill 1968) and level of historic disturbance (Reeves et al. 1997). The majority of species sympatric with salmonids in the watershed are native species (Zaroban et al. 1999), and the documentation of both spawning peamouth and largescale sucker, and rearing juveniles, indicate that Gibbons Creek is important habitat for these native fish.

Our study provides a baseline of useful information for directing restoration and monitoring activities. Methods used in this study can be replicated to determine trends in biological integrity, whether or not restoration actions are undertaken. If restoration actions are undertaken, we recommend that they be monitored carefully. Of the methods we used to evaluate biotic integrity, both macroinvertebrate (BIBI) and fish community (MFCI) methods are cost-effective in terms of both time and total cost. Macroinvertebrates are reasonably easily collected following Plotnikoff (1994), but fish collection will require trained electrofishing crews. Both methods require accurate species identification. Smolt monitoring is fairly expensive in terms of both time and cost. However, it is probably the most effective way to evaluate the effects of habitat change (including restoration actions) on anadromous salmonids (Reeves et al. 1991).

Our study demonstrates that the GC watershed has been heavily impacted by past and present land management practices. However, in spite of these impacts, the watershed retains at least remnants of important components of its aquatic community, including threatened steelhead. In fact, some portions of the watershed maintain high biotic integrity as measured by both fish and macroinvertebrate communities. As such, the watershed should be protected by both regulation and community education to protect existing watershed integrity, and a program of restoration initiated by community and agency personnel to restore integrity to degraded areas.

Conclusions and Recommendations

By order of importance:

- Protect good habitats where they exist (WC, TT2, GC2, GC3).

- Remove barriers to fish migration, especially culverts at Hans Nagel Road and upstream on private land. GC3 contains the best combination of fish habitat and biotic integrity in the drainage.

- Re-establish riparian vegetation where it is currently suppressed or removed (Orchard Hills Golf Course and upstream to Q Street on CC).

- Pursue opportunities to create pools by LWD placement and other methods, including rock-riffle reaches where necessary (e.g. GC2 along Sunset View Road.

- Remove the diversion structure and modify the dike to allow lower Gibbons Creek to re-establish a natural stream channel to the Columbia River via Steigerwald Lake.

- Construct off-channel ponds and side-channel habitats for coho salmon where possible and appropriate. The newly acquired property adjacent to GC north of SR 14 provides opportunity for this type of activity, although temperatures may be prohibitive because little shading vegetation is currently established at this site.

- Study opportunities to restore natural channel morphology and habitat structure. The newly acquired property adjacent to GC north of SR 14 may provide opportunity for this type of activity as well. If the channel is re-routed, study the possibility of developing the abandoned channel as an off-channel pond or side-channel. Some suitable riparian vegetation and substrate for rearing already exist at this location. The reach of CC at Orchard Hills Golf Course is another channelized reach that would benefit greatly from restoration, especially in conjunction with riparian vegetation renewal.

- Study the potential to start a chum salmon population in the Gibbons Creek watershed by restoring or constructing spawning habitat at sites where spring water or groundwater is available.

- Monitor both condition of the watershed and results of restoration actions.

Literature Cited

Anderson, R.O. and R.M. Neumann. 1996. Length, weight, and associated structural indices. Pages 447-481 in B.R. Murphy and D.W. Willis, editors. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, MD.

Anonymous. 1998. Stream inventory handbook, level I and II. Version 9.8. U.S. Forest Service, Pacific Northwest Region. 84 p.

Anonymous. 1997. Stream inventory handbook, level I and II. Version 9.7. U.S. Forest Service, Pacific Northwest Region. 84 p.

Bailey, N.T.J. 1951. On estimating the size of mobile populations from recapture data. Biometrika 38:293-306.

Bicknell, R.J. 1988. Steigerwald Lake National Wildlife Refuge Annual Report. Washington Department of Fish and Wildlife, Vancouver, WA.

Bilton, H. T., D. F. Alderdice, and C. B. Murray. Influence of time and size at release of juvenile coho salmon (*Oncorhynchus kisutch*) in returns at maturity. Canadian Journal of Fisheries and Aquatic Sciences 39:426-447.

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. Transactions of the American Fisheries Society 117:262-273.

Black, G. A. 1982. Gills as an attachment site for *Salmincola edwardsii*. Journal of Parasitology 68:1172-1173.

Bugert, R. M., T.C. Bjornn, and W. R. Meehan. 1991. Summer habitat use by young salmonids and their responses to cover and predators in a small southeast Alaska stream. Transactions of the American Fisheries Society 120:474-485.

Busby, P.J. and 6 co-authors. 1996. Status review of west coast steelhead from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC-27.

Cuffney, T. F., M. R. Meador, S. D. Porter, and M. E. Gurtz. 1997. Distribution of fish, benthic invertebrate, and algal communities in relation to physical and conditions, Yakima River basin, Washington, 1990. U.S. Geological Survey, Water Resources Investigations Report 96-4280, Raleigh, NC.

Efron, B. and R. J. Tibshirani. 1993. An introduction to the bootstrap. Chapman and Hall, New York.

Fasten, N. 1921. Studies on the parasitic copepods of the genus *Salmincola*. American Naturalist 55:449-456.

Federal Register. 2000. Designated Critical Habitat: Critical Habitat for 19 Evolutionarily Significant Units of Salmon and Steelhead in Washington, Oregon, Idaho, and California. Vol. 65, (32): 7764.

Fiscus, G. 1978. A review of Gibbons Creek resources. Washington Department of Fisheries. Unpublished report. 17 p.

Friesen, T. A. and D. L. Ward. 1996. Status and condition of fish assemblages in streams of the Tualatin River basin, Oregon. Northwest Science 70(2):120-131.

Hartmann, G. F. and C. A. Gill. 1968. Distributions of juvenile steelhead and cutthroat trout (*Salmo gairdneri* and *S. clarki clarki*) within streams in southwestern British Columbia. Journal of the Fisheries Research Board of Canada. 25:33-48.

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, ed. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19.

Hoar, W. S. 1976. Smolt transformation: evolution, behavior, and physiology. Journal of the Fisheries Research Board of Canada 33:1234-1252.

Hogan, D. L. and B. R. Ward. 1997. Watershed geomorphology and fish habitat. Chapter 2 *in*, P. A. Slaney and D. Zaldokas, eds. Fish habitat rehabilitation procedures. Watershed Restoration Circular 9, British Columbia Watershed Restoration Program.

Holtby, L.B., T. E. McMahon, and J.C. Scrivener. 1989. Stream temperatures and interannual variability of coho salmon (*Oncorhynchus kisutch*) smolts and fry and chum salmon (*O. keta*) fry from Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 1396-1405.

Hughes, R. M. and J. R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. Transactions of the American Fisheries Society 116(2):196-209.

Karr, J.R. and E.W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, D.C. 206p.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. Environmental Management, 5:55-68.

Kondolf, G.M. and M.G. Wolman. 1993. The sizes of salmonid spawning gravels. Water Resources Research 29(7):2275-2285.

Kondolf, G. M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129:262-281.

Koning, C. W. and E. R. Keeley. 1997. Salmonid biostandards for estimating production benefits of fish habitat rehabilitation techniques. Chapter 3 *in*, P. A. Slaney and D. Zaldokas, eds. Fish habitat rehabilitation procedures. Watershed Restoration Circular 9, British Columbia Watershed Restoration Program.

Lassetre, N. S. and R. R. Harris. 2001. The geomorphic and ecological influence of large woody debris in streams and rivers. University of California - Berkeley. 69 p.

Laufle, J.C., G. B. Pauley, and M.F. Shepard. 1986. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest)--coho salmon. U.S. Fish and Wildlife Service Biol. Rep. 82(11.48). U.S. Army Corps of Engineers, TR EL 82-4. 18p.

Lichatowich, J., L. Morbrand, L. Lestelle, and T. Vogel. 1995. An approach to the diagnosis and treatment of depleted Pacific salmon populations in Pacific Northwest watersheds. Fisheries 20:10-18.

Lyons, J. 1992. The length of stream to sample with a towed electrofisher when fish species richness in estimated. North American Journal of Fisheries Management 12:198-203.

Margolis, L., G. W. Esch, J. C. Holmes, A. M. Kuris, and G. A. Schad.1982. The use of ecological terms in parasitology (report of an ad hoc committee of the American Society of Parasitologists). Journal of Parasitology 68:131-133.

McMahon, T.E. 1983. Habitat suitability index models: coho salmon. U.S. Department of the Int., U.S. Fish and Wildlife Service. FWS/OBS-82/10.49. 29 p.

Meehan, W. R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19.

Murphy, M. L., J. F. Thedinga, and J. J. Pella. 1996. Bootstrap confidence intervals for trap efficiency estimates of migrating fish. National Marine Fisheries Service, Juneau, AK.

Newbury, R., M. Gaboury, and D. Bates. 1997. Restoring habitats in channelized or uniform streams using riffle and pool sequences. Chapter 12 *in*, P. A. Slaney and D. Zaldokas, eds. Fish habitat rehabilitation procedures. Watershed Restoration Circular 9, British Columbia Watershed Restoration Program.

Platts, W. S. 1987. Methods for evaluating riparian habitats with applications to management. U.S. Forest Service. General Technical Report INT-221. Pages 58-60.

Plotnikoff, R.W. 1994. Instream biological assessment monitoring protocols: benthic macroinvertebrates. Washington Department of Ecology. Publication 94-11.

Plotnikoff, R.W. 1996. Taxonomic laboratory protocol for stream macroinvertebrates collected by the Washington State Department of Ecology. Washington State Department of Ecology, Olympia, WA.

Plotnikoff, R.W. 1999. The relationship between stream macroinvertebrates and salmon in the Quilceda/Allen drainage. Washington Department of Ecology. Publication 99-311.

Reeves, G.H., F.H. Everest, and T.E. Nickelson. 1989. Identification of physical habitats limiting the production of coho salmon in western Oregon and Washington. Gen. Tech. Rep. PNW-GTR-245. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 18 p.

Reeves, G. H., F. H. Everest, and J. R. Sedell. 1991. Responses of anadromous salmonids to habitat modifications: how do we measure them? American Fisheries Society Symposium 10:62-67.

Reeves, G. H., J. D. Hall, and S. V. Gregory. 1997. The impacts of land-management activities on coastal cutthroat trout and their freshwater habitats. Pages 138-144 *in* J.D. Hall, P.A. Bisson, and R. E. Gresswell, editors. Sea-run cutthroat trout: biology, management, and future conservation. Oregon Chapter, American Fisheries Society, Corvallis, OR.

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

Schuett-Hames, D., A. Pleus, and L. Bullchild. 1993. Habitat unit survey module. Timber-Fish-Wildlife Ambient Monitoring Program, Northwest Indian Fisheries Commission. 32 p.

Salo, E. O. and T. W. Cundy. 1987. Streamside management: forestry and fishery interactions. Contributions no. 57. University of Washington, Institute of Forest Resources. 471 p.

Simonson, T. D. and J. Lyons. 1995. Comparison of catch per effort and removal procedures for sampling stream fish assemblages. North American Journal of Fisheries Management 15:419-427.

Thedinga, J. F., S. W. Johnson, K. V. Koski, J. M. Lorenz, and M. L. Murphy. 1994. Determination of salmonid smolt yield with rotary screw traps in the Situk River, Alaska, to predict effects of glacial flooding. North American Journal of Fisheries Management 14:837-851.

Trotter, P. C. Sea-run cutthroat trout: life history profile. 1997. Page 7-15 *in* J.D. Hall, P.A. Bisson, and R. E. Gresswell, editors. Sea-run cutthroat trout: biology, management, and future conservation. Oregon Chapter, American Fisheries Society, Corvallis, OR.

USDA (U.S. Department of Agriculture), Forest Service, and U.S. Department of the Interior, Bureau of Land Management. 1995. Environmental assessment: interim strategies for managing andromous fish-producing watersheds in eastern Oregon and Washington, Idaho, and portions of California. U.S. Forest Service and U.S. Bureau of Land Management, Washington, D.C.

Waite, I. R. and K. D. Carpenter. 2000. Associations among fish assemblage structure and environmental variables in Willamette basin streams, Oregon. Transactions of the American Fisheries Society 129:754-770.

Washington Administrative Code. 1997. Water quality standards for surface waters of the State of Washington. WAC 173-201A-010.

Weitkamp, L. A. and 6 co-authors. 1995. Status review of coho salmon from Washington, Oregon, and California. NOAA Technical Memorandum NMFSNWFSC-24.

Wolman, M.G 1954. A method of sampling coarse bed material. Trans. Amer. Geophysical Union 35:951-956.

Zaroban, D. W., M. P. Mulvey, T. R. Maret, and R. M. Hughes. 1999. Classification of species attributes for Pacific Northwest freshwater fishes. Northwest Science 73(2): 81-93.