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tributaries (Fig. 23B). Chinook salmon were found in most of the same tributaries, but also occurred in portions of the mainstem Sixes River (Fig. 23C). Substantial numbers of chinook were seen in Dry Creek and in Sixes River from Dry Creek downstream to the estuary. Based on past studies (McGie 1969, Reimers 1971, Reimers 1978, Reimers et al. 1979) and surveys in recent years (G. Susac, unpublished data) by Oregon Department of Fish and Wildlife, large numbers of juvenile chinook spend the summer in Sixes River estuary.

Comparison with data for 1965-72 indicates that local contractions and expansions in distribution have occurred among both coho and chinook salmon (Table 13). The summer rearing range of each species has remained declined in the lower basin, has declined dramatically in the middle portion of the basin, and has expanded in the upper basin.

Range contractions have occured in tributaries and mainstem segments of the middle portion of Sixes basin. Coho and chinook salmon were absent from two middle-basin tributaries (Elephant Rock Creek, Little Dry Creek) and two lower-river tributaries (Beaver Creek, "Andrews Fork" of Crystal Creek) where they had been observed in 1965-72. Little Dry Creek was reported to support high densities of both coho and chinook juveniles prior to 1972, but from 1986 through 1991 it was entirely dry during summer months in its lower 1 km. A series of landslides followed road construction and clearcutting of portions of this drainage basin in the late 1960's and 1970's, and the channel today is highly aggraded with coarse sediment (Frissell et al. 1992b). Another tributary to upper Sixes River reported to contain summer populations of coho in 1965-72 was nearly dry when we visited it in 1990 and supported only a few trout fry. Finally, chinook are absent today from two warm sections of the mainstem Sixes River where they were apparently present during summer prior to 1972.

Figure 23. Maps of the number of salmonid age classes and species present in Sixes River during late summer in 1987-90 (A) and the distribution and abundance by species and age classes (B-G). Line pattern indicates approximate density (fish per square meter in pools, see key at bottom left). For chinook and coho salmon, dot-dash pattern adjacent to stream indicates distribution in 1969-71 as reported by Stein et al. (1972).

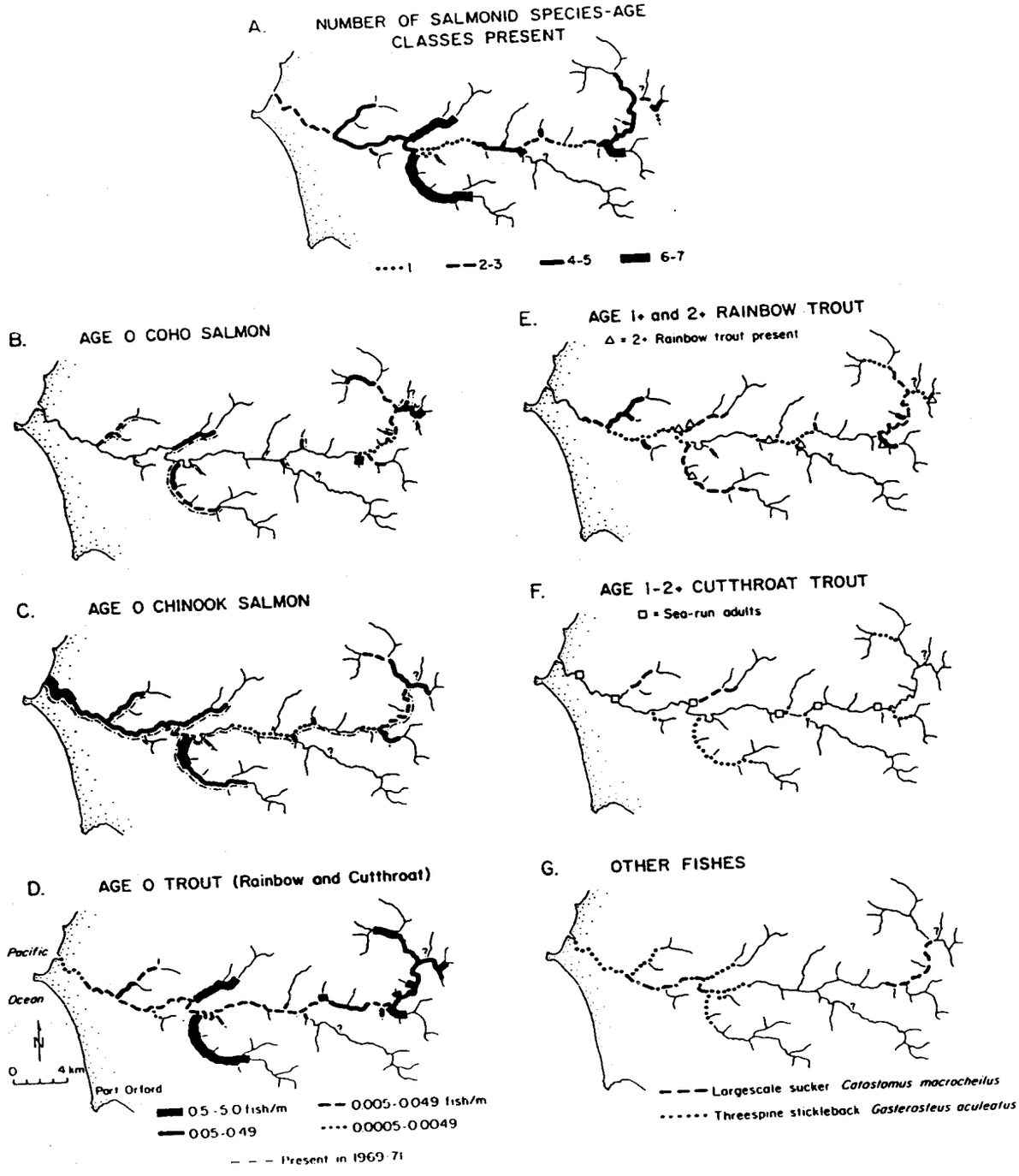
A. NUMBER OF SALMONID SPECIES-AGE CLASSES PRESENT
1
2-3
4-5
6-7
B. AGE 0 COHO SALMON
C. AGE 0 CHINOOK SALMON
D. AGE 0 TROUT (Rainbow and Cutthroat)
Pacific
Ocean
N
0
4 km
Port Orford
0.5-5.0 fish/m
0.05-0.49
0.005-0.049 fish/m
0.0005-0.0049
Present in 1969-71
E. AGE 1+ and 2+ RAINBOW TROUT
Δ = 2+ Rainbow trout present
F. AGE 1-2+ CUTTHROAT TROUT
□ = Sea-run adults
G. OTHER FISHES
Largescale sucker Catostomus macrocheilus
Threespine stickleback Gasterosteus aculeatus

Figure 23

Table 13. Change in summer (July-August) distribution of juvenile coho and chinook salmon in upper, middle, and lower portions of Sixes River basin. Total habitat accessible to anadromous fish estimated from USGS 1:24,000-scale topographic maps, is about 170 lineal km.

Species	Section of Basin	Length of habitat Occupied (km) 1968-72	1987-89	% Change
0+ coho	Upper[a]	12.2	46.8	+284%
	Middle[b]	6.5	2.7	-58%
	Lower[c]	43.6	40.9	-6%
	Total, coho	62.3	90.4	+45%
0+ chinook	Upper	15.3	39.4	+158%
	Middle	42.2	13.7	-68%
	Lower	74.9	77.6	-3%
	Total, chinook	128.2	135.1	-5%

[a] Upper basin = from Rusty Butte Bridge upstream (river km 24-30; 50 lineal km of accessible habitat [29%]).
[b] Middle basin = above Dry Cr upstream to Rusty Butte Cr (river km 12-24; 42 lineal km of habitat [25%]).
[c] Lower basin = Estuary to Dry Cr (river km 0-12; 78 lineal km of habitat [46%]).

Coho salmon have expanded their summer rearing range in the upper basin; populations today are found throughout summer in Middle Fork Sixes River, Upper Sixes River, North Fork Sixes River, and Crafton Creek, none of which were reported to hold coho in summer during 1965-72. According to Stein et al. (1972), many of these streams supported coho in spring months, but these fish either moved from or perished in these habitats as water temperature increased in early summer. The range of summer-resident chinook salmon has likewise increased through headward expansion into upper basin tributaries. The summer-long presence of coho and chinook salmon in Crafton Creek, which formerly exceeded 25°C for much of the summer, suggests that upper-basin streams have cooled substantially in the past 20 y, probably due to vegetation growth in riparian areas that were logged in the 1960's.

Young-of-the-year trout (probably >90% rainbow, <10% cutthroat) were observed in all streams that had surface flow. The largest concentrations, however, were found in Dry Creek, Edson Creek, and upper basin tributaries (Fig. 23D). Age 1+ rainbow trout occurred in all major tributaries and certain portions of the mainstem; age 2+ rainbows were rare, but were distributed similarly (Figure 23E). Cutthroat trout (age 1 and older) were the rarest group, and appeared to be restricted to cooler tributaries (Fig. 23F). Juvenile cutthroat trout were not observed in the mainstem Sixes River, but we found mature sea-run cutthroat throughout summer holding in large pools, even where water temperature was quite high (Fig. 23F). Pre-1972 seining records did not distinguish between species or age classes of trout sampled in Sixes River, but as today, "trout" in some form were ubiquitous in the basin.

The two other fish species had distribution patterns markedly different from the salmonids (Fig. 23G). Largescale suckers were confined to the mainstem, with adults and fry observed in segments of the upper mainstem and lower mainstem. We never observed more than seven adult suckers in a pool, and observed few or no subadults

(5-20 cm). Only a few mainstem pools held any individuals of this species. Although we did not estimate a population size, the abundance of largescale suckers appeared to be markedly lower than it was in 1979, when Oregon Department of Fish and Wildlife researchers observed large schools, and captured and tagged 88 adults from a single pool (T. W. Downey, unpublished report, Elk River Research Laboratory). Threespine sticklebacks were abundant in the mainstem and tributaries of the lower river as both adults and fry. We commonly observed schools numbering in the hundreds or thousands in backwater pools. While snorkeling we observed juvenile salmonids preying on stickleback fry.

3.4 Basin-Wide Abundance

To assess relative abundance and large-scale patterns in distribution of species, we made crude estimates of total population size (Table 14). Although all three portions of Sixes basin contain roughly equal amounts of available habitat, the populations of all species and age classes are concentrated in upper and lower portions of the basin, with relatively few fish in the central basin. Most cutthroat trout and chinook salmon are found in the lower basin, with most chinook in the mainstem and cutthroat in the tributaries. Assuming that Oregon Department of Fish and Wildlife estimates of average mid-summer peak abundance of chinook salmon in the estuary since 1970 (Reimers and Downey 1980) reflect present-day estuary conditions, we estimate that about one-third of the mid-summer population of juvenile chinook in Sixes River resided in freshwater during 1987-89, with the remainder in the estuary. Based on our observations of tagged fish (unpublished data), we believe that many chinook in the mainstem of Sixes River are migrating downstream during late July and August. Peak migration into the estuary

Table 14. Available habitat and crude estimates of total abundance of juvenile salmonids by species/age class in upper, middle, and lower sections of Sixes River basin in midsummer of 1987-89. Upper, middle, and lower sections defined as in Table 3 except estuary (river km 0-5) broken out from lower basin.

	Estimated Abundance (% of Basin Total)				
	Upper Basin	Middle Basin	Lower Basin	Estuary	Total
0+ coho	8000 (66%)	40 (0.3%)	4100 (34%)	0	12,140
0+ chinook	5400 (6%)	460 (0.5%)	28,000 (30%)	60,000[a] (64%)	93,000
0+ trout	38,000 (29%)	4000 (3%)	88,000 (68%)	0	130,000
1+ rainbow	2,500 (50%)	100 (2%)	2,400 (48%)	0	5,000
1-2+ cutthroat	25 (6%)	5 (1%)	365 (93%)	?[b]	390?
Lineal km of accessible habitat[c]	50 (29%)	42 (25%)	73 (43%)	5 (3%)	170
Estimated total pool area ($X1000m^2$)[d]	125 (23%)	120 (22%)	195 (35%)	115 (20%)	555

[a] Based on ODFW estimates.
[b] ODFW sampling indicates estuary serves as nursery for juvenile cutthroat trout, but numbers are unknown.
[c] Estimated from US Geological Survey 1:24,000-scale topographic maps.
[d] Extrapolated from estimate of accessible stream length and field measurements of pool length and wetted width.

during the 1970's occurred in June or early July (Reimers et al. 1979, Reimers and Downey 1980).

3.5 Temperature, Abundance, and Assemblage Diversity

Among stream segments where we had both temperature and biological data for 1987-89, the diversity of the salmonid assemblage, measured by the number of species and age classes observed, declined as maximum water temperature increased (Fig. 24). This pattern reflected a progressive loss of thermally sensitive species with warming. The coolest streams held all four species, including three age classes of rainbow and cutthroat trout. Cutthroat, coho, and chinook dropped out in sequence as maximum temperature increased, with rainbow trout the only species present in waters exceeding 23°C.

Overall abundance of salmonids declined as maximum temperature increased (Fig. 25). However, the response to maximum temperature varied among species and age classes. Several species showed evidence of a theshold response (due to variation in density among cool reaches, simple linear regressions were not significant). Coho salmon were absent or rare in segments exceeding 21°C (Fig. 25B), chinook salmon were absent from two of three segments exceeding 23°C (Fig. 25C), and cutthroat trout were absent where temperature exceeded 21°C (Fig. 25D). The small numbers of chinook and coho salmon in certain warmer segments were in every case found in association with anomalous, small cool pockets in otherwise warm reaches.

Age 0 rainbow trout showed a linear response to temperature, with small numbers present in even the warmest streams (Fig. 25E). The pattern for age 1+ rainbow trout was less clear, with some indication of a threshold response above 22°C

Figure 24. Salmonid diversity (SD) as measured by the number of species and age classes present, in relation to annual maximum water temperature (MWT). Numbers indicate the number of overlapping observations with identical coordinates.

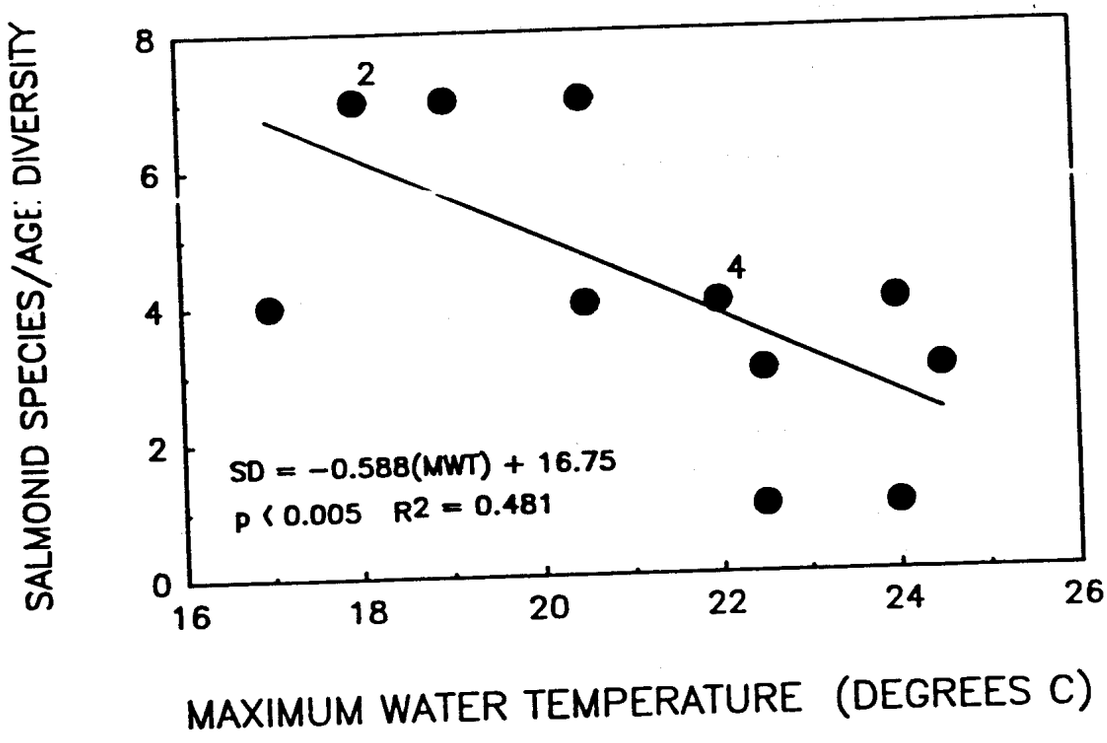
SALMONID SPECIES/AGE DIVERSITY
8
6
4
2
0
16
18
20
22
24
26
2
4
SD = −0.588(MWT) + 16.75
p < 0.005 R2 = 0.481
MAXIMUM WATER TEMPERATURE (DEGREES C)

Figure 24

Figure 25. Abundance of salmonids in relation to maximum water temperature in segments of Sixes River during 1987-90. Data are presented as total combined salmonids (A) and separate species and age classes (B-G). Note log scale on y-axis of total salmonid plot (A). Correlation coefficients, p-values and fitted curves are indicated for statistically significant ($p<0.10$) linear regressions of log-transformed data (A,B,D,E) (transformation necessary to stabilize variance) or non-transformed data (F).

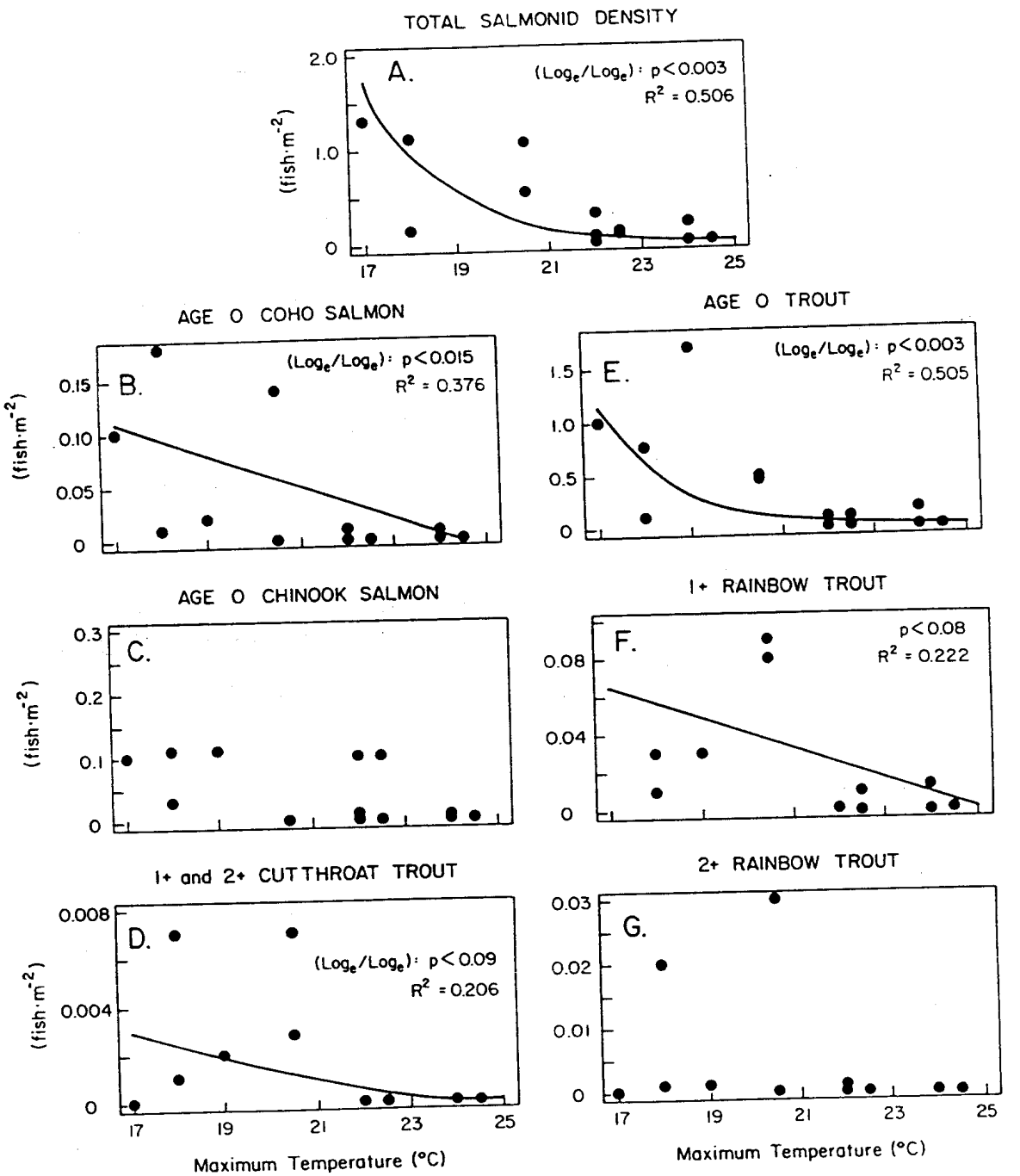
TOTAL SALMONID DENSITY
A.
(Log$_e$/Log$_e$): p<0.003
R^2 = 0.506
(fish·m^{-2})
2.0
1.0
0
17
19
21
23
25
AGE O COHO SALMON
B.
(Log$_e$/Log$_e$): p<0.015
R^2 = 0.376
0.15
0.10
0.05
AGE O TROUT
E.
(Log$_e$/Log$_e$): p<0.003
R^2 = 0.505
1.5
0.5
AGE O CHINOOK SALMON
C.
0.3
0.2
0.1
1+ RAINBOW TROUT
F.
p<0.08
R^2 = 0.222
0.08
0.04
1+ and 2+ CUTTHROAT TROUT
D.
(Log$_e$/Log$_e$): p<0.09
R^2 = 0.206
0.008
0.004
2+ RAINBOW TROUT
G.
0.03
0.02
0.01
Maximum Temperature (°C)
Maximum Temperature (°C)

Figure 25

(Fig. 25F). Like cutthroat, age 2+ rainbow trout were absent or rare in all streams exceeding 21 C (Fig. 6G). The curve for total salmonid density (Fig. 25A) is strongly influenced by age 0 trout, the most abundant group, but is also shaped by the abrupt declines of coho and chinook salmon and 1+ rainbow trout between 21 and 23°C. We could find no obvious relationship between salmonid diversity or density and stream size, distance from the estuary, or measures of within-reach habitat morphology.

The pattern in diversity might change somewhat with the addition of the largescale sucker and threespine stickleback. Because these two species appeared to be confined to mainstem habitats and lower-basin habitats, respectively, other geographic factors confound the temperature analysis. For the salmonids, all stream segments we included in this analysis were accessible to and appeared to have the morphological capability to support every age class and species; water temperature appeared to be the environmental variate most strongly influencing assemblage structure.

3.6 Juvenile Behavior and Thermal Heterogeneity

During intensive underwater observations we noticed striking patterns in diurnal activity patterns of salmonids that corresponded with temporal and spatial heterogeneity in water temperature. In mainstem Sixes River near Edson Creek, a reach with a maximum temperature of about 22°C, at about 9 AM each morning we observed large numbers of juvenile chinook salmon and rainbow trout emerging from the interstices of boulder clusters along the shoreline. The fish remained active, feeding in the water column, until about 5-6 PM, when they gradually re-entered boulder crevices. These shifts in behavior corresponded approximately with the time of the minimum water temperature in the

morning, and the time of maximum temperature in the evening; fish were active during the period of warming from morning to late afternoon.

At about 6 PM in a neighboring pool, we observed large groups of fish aggregating in a shallow depression along the slip face of a gravel bar. Hundreds of chinook salmon and rainbow trout of all sizes intermingled densely in an area about 1.5 m^2. The fish lay on or a few mm above the substrate, were not feeding, and showed no evident agonistic behavior. Spot measurements showed the water temperature in the depression within 5 cm of the gravel bed to be 19°C, 2-3°C cooler than the temperature of the water column above. The depression accumulated shallow groundwater which percolates through the gravel bar upstream, creating a small cool lens before mixing with the warmer surface water of the pool. We presume that similar groundwater-fed cold refugia exist within or behind the boulders used by the fish, as these boulders line the bank where the river hugs the toe of a groundwater-rich slump-earthflow landform. The importance of cold refugia may also explain the occurrence of large numbers of fish in a spring-fed backwater pool along the warm mainstem of Sixes River below Dry Creek. We observed fish using other cold pockets formed in accumulations of large woody debris at groundwater inflows, and at depth in thermally stratified, isolated pools in reaches of upper Sixes River and Dry Creek having intermittent surface flow.

4. Discussion

4.1 Temperature and Fish Assemblages

Water temperature appears to be a major factor controlling species distribution and governing the structure of juvenile salmonid assemblages during summer in Sixes River. In warmer rivers whose native fauna are dominated by coldwater species such as salmonids, thermal factors may override or strongly co-determine the effects of habitat morphology, flow, food supply, and behavioral interactions (Smith and Li 1983, Reeves et al. 1987, Dambacher 1991). The relative scarcity of native, warmwater-adapted groups of fishes in many coastal stream systems suggests that in the geologic past, if not the historic past, these rivers were cooler than they are today. In systems open to colonization by warmwater fauna, increased temperature can lead to increased fish diversity, despite the disappearance of salmonid taxa (Barton et al. 1985). Although native salmonids persist in Sixes River, maximum summer temperatures throughout the basin far exceed thermal optima for these species (Beschta et al. 1987), and the ecosystem would likely be highly vulnerable to invasion if exotic, warmwater species of fishes were introduced.

Although all salmonid species in Sixes River appeared to respond negatively to increasing temperature, and species-specific responses generally appeared consistent and predictable, we caution that a simple predictive model of fish assemblage response to stream warming might not be successful. We do not know whether the distribution of fish in relation to temperature reflects mortality or movement; if the primary response to temperature is movement, our data could reflect a change in the distribution of fish with no change in their abundance basin-wide. Thermal refugia, between-basin differences in channel morphology and the spatial arrangement of different kinds of valley segments,

species interactions, food supply, and stock-specific adaptations of life history, behavior, and physiology all complicate biotic response to warming. Survival at egg-to-fry stages may vary dramatically depending on channel stability and flood occurrence (Frissell and Nawa unpublished), and variations in this or other life stages are likely to affect distribution and abundance of juveniles later in their life history. It seems clear, however, that the distribution of juvenile salmonids in Sixes basin is at least partly constrained by summer temperature and that increases in temperature are likely to further stress populations by limiting their movements, possibly capping density or reducing growth in important habitats, and perhaps eliminating key life history options with the imposition of new thermal barriers. Conversely, cooling could remove or relieve such constraints, perhaps result in nonlinear increases in certain habitats and species.

The Siskiyou National Forest (1989) uses a simple linear temperature response model that predicts a 5% decline in salmonid density with each 1°C increase in water temperature, using age 1+ steelhead trout as the indicator species for the assemblage as a whole. Our data indicate that such a linear model may be at least superficially appropriate for age 0 trout, but is not so for other, more thermally sensitive species and life stages. The abrupt loss of thermally sensitive species like coho salmon and cutthroat trout in the range of 21-23°C causes the best fit model for total salmonid density to be a logarithmically declining curve. In other words, there is a more precipitous loss of numbers (and species) between 18-22°C than between 23-27°C. As mentioned earlier, such losses could be a result of migration that does not necessarily lead to mortality. During field work in summer, however, we have frequently observed dead chinook salmon fry and steelhead parr in warmer streams. These individuals rarely showed obvious external evidence of disease, and thermal stress may have caused or contributed their mortality. On the other hand, we also have observed movement of fishes of several kilometers or more during summer (Nawa et al. 1988; Nawa and

Frissell, unpublished data). During repeated seine sampling and snorkel observations of tagged fish we found that some juvenile chinook salmon migrate downstream during summer months. Yearling steelhead (rainbow trout) may leave the mainstem and pack into cooler tributaries during warm periods.

In Sixes River 1+ steelhead (rainbow trout) were apparently less sensitive and less consistent than other fishes in their response to thermal stress, and are therefore a poor indicator of the salmonid community as a whole. A more sensitive indicator species might be cutthroat trout, which appear to decline most rapidly with increasing water temperature. The number of salmonid species present (e.g., in a standard reach of 100 m^2 sampled for one hour) would make a much more direct, information-rich, and easily measured indicator for monitoring the biotic integrity of native assemblages (sensu Miller et al. 1988) and, by inference, the condition of their habitats. If patterns in Sixes River reflect those in other coastal streams of the Klamath Mountains, species richness should generally be correlated with overall abundance of salmonids. It should be relatively easy to test this prediction with field data, and if it holds, it could considerably facillitate the assessment and monitoring of habitat quality.

4.2 Physical and Biological Recovery

Over time Sixes basin has seen local thermal recovery and some expansion of juvenile chinook and coho salmon into tributaries that were formerly too warm to hold these species through summer. These gains, however, are substantially offset by the disappearance of the same species from tributaries elsewhere in the basin and loss of chinook salmon from portions of the mainstem. The lack of cooling at most mainstem stations, and the fact that despite 3 C cooling since 1972 the mainstem Sixes River

above Little Dry Cr. remains too warm to hold most species, indicate that the lower mainstem river, the primary fluvial habitat of chinook salmon in summer, has not markedly improved over time and may rather have deteriorated somewhat. The scarcity of cutthroat trout in upper and middle Sixes basin is consistent with other studies (Hall et al. 1987, Schwartz 1991) showing that coastal cutthroat trout populations depressed by logging have not recovered more than 20 years after the disturbance. Cutthroat trout in Sixes basin were most abundant in Dry Creek and Edson Creek, two lower-basin tributaries which have seen relatively limited riparian logging. Despite local habitat recovery in some tributaries, persistent factors such as thermal barriers in the mainstem could prevent the re-emergence of mainstem-dependent juvenile life history patterns that under historical conditions might have sustained much of the productive capacity of salmon and trout populations.

Because alluvial valleys at low elevations were logged and cleared early in the century and have experienced extensive disturbance from both upstream and riparian sources (Frissell et al. 1992a), we are uncertain whether the temperature of lower mainstems of rivers such as the Sixes has changed in recent decades. Regardless, we know that along large, wide rivers, especially in alluvial valleys such as lower Sixes River where bedrock, boulders, and other non-vegetative roughness elements are lacking, the role of forests in maintaining channel complexity probably greatly outweighs their role in providing shade. Shade provided by riparian forests has a relatively minor effect on the temperature of large, wide rivers (Theurer et al. 1985). Floodplain forests provide woody debris, help stabilize banks, bars, and secondary channels, and they may encourage the recharge of hyporheic zones, all processes that create complex, thermally heterogeneous habitats in alluvial streams (Sedell and Swanson 1982, Sedell et al. 1990).

The cumulative, downstream-propagating effects of logging-related erosion in basins like Sixes River are probably more important for fishes like chinook salmon than

are the direct impacts of human activities in riparian areas and channels (Frissell 1991, Frissell et al. 1992a, 1992b). Lichatowich (1989) suggested that coho salmon may be more vulnerable than chinook salmon to long-term impacts of logging and other activities, because mainstem river habitats are recovering from historic disturbances while tributary streams important for coho continue to be disturbed. However, our study suggests that, due to thermal limitations, in southwest Oregon the capacity for recovery is greatest in tributaries, and least in mainstem segments. Although narrow canyons and alluviated canyon segments in tributaries can be highly vulnerable to heating if riparian areas are logged or destroyed by landslides or debris flows, they recover shade relatively rapidly with riparian revegetation (Frissell et al. 1992a). Whereas coho salmon appear to have responded at least locally to recovery of riparian zones and cooling of headwater tributaries of Sixes River, there is no evidence that the most important summer habitats of fall chinook salmon in the lower mainstem have improved in the past two decades.

Alluvial valleys of mainstem rivers, by contrast, are less affected by loss of riparian shade; their thermal regime can be buffered by contact with a large hyporheic reservoir underlying floodplains and terraces. The cooling evident in the lower kilometers of Sixes River, which Reimers (1971) attributed to cooler climatic conditions along the coastline, may in fact stem from mixing of surface waters with shallow groundwater in the alluvial valley. Dry Creek, a major tributary which contributes no surface water to Sixes River during summer, probably does constitute a major hyporheic plume that cools mainstem Sixes River. It seems likely that long-term depletion of large woody debris, channel aggradation and straightening resulting from the downstream transfer of sediment from landslides and other erosion sources (Kelsey 1980, Lisle 1982, McHugh 1987, Ryan-Burkette 1989, Frissell 1991), and subsequent artificial revetment of unstable reaches to protect roads and farmland, have worsened conditions for chinook

salmon, and possibly steelhead, by reducing the availability of thermal refugia, and perhaps reducing contact between surface and hyporheic waters in the alluvial valley of Sixes River.

4.3 Habitat Refugia and Persistence of Fish Populations

Thermal refugia appear to be critical in shaping the salmonid assemblage of Sixes basin at two different spatio-temporal scales (Sedell et al. 1990). On the scale of the stream system as a whole, we hypothesize that one important factor was the persistence of un-logged, debris-rich, cool river segments in the Dement Ranch in upper Sixes River, a portion of the basin that otherwise was heavily logged and extensively warmed during the 1960's-1970's (Stein et al. 1972, Reimers 1971). The Dement Ranch segments may have provided refuge several kilometers long from which salmon colonized other upper-basin tributaries as logged streams revegetated and cooled. This refuge may have sustained coho salmon and the upper-basin runs of chinook salmon and cutthroat trout. The absence of any productive, cool refuge in the middle portion of Sixes basin could help explain the apparent extripation of salmon populations from that portion of Sixes basin.

At a finer scale, thermal micro-refugia are critical to sustain salmonids in otherwise warm surface waters within alluvial valley segments. Protected cold pockets were heavily used by juvenile salmonids during the warmest hours of the day, and these refugia allowed persistence of thermally sensitive species in some warmer segments of Sixes River. Similar phenomena have been observed in other studies (Gibson 1966, Kaya et al. 1977, Bilby 1989, Ozaki 1987), but the overall significance of thermal

microrefugia for fish production and species diversity has not been adequately investigated. Thermal refugia in Sixes River were associated with large woody debris accumulations, boulder banks along the toe of large slump-earthflow slopes, deep pools, and spring-fed side channels or backwaters.

As in Redwood Creek, a highly aggraded stream in the Klamath Mountains of northern California (Ozaki 1987), thermal refugia in lower Sixes River are associated with interannually transient geomorphic features that are easily disrupted by moderate-sized floods. In damaged watersheds, floods inhibit the recovery of riparian forests along aggraded channels by eroding streambanks and toppling or burying floodplain trees (Lisle 1981, Frissell et al. 1992b). During sediment-charged floods, secondary channels and backwater pools are rapidly filled with bedload deposits, making them dry in summer (Frissell, personal observation). Significant improvement of habitat in the alluvial valley segments of Sixes River and other southwest Oregon streams will probably not occur without stabilization of the watershed and consequent reduction of sediment load from upstream sources, together with restoration of fully-functioning floodplain forests (Lisle 1981, Frissell and Nawa 1992).

We suggest that long-term persistence of coho salmon, chinook salmon, and other salmonids in the Klamath Mountains region, where most stocks are currently seriously depressed (Nehlsen et al. 1991), will require special protection of areas that serve as refugia and sustain both downstream and riparian ecological functions (Moyle and Williams 1990, Frissell 1991, Johnson et al. 1991). Such forested refugia appear to be critical for survival of sensitive amphibians in coastal basins (Corn and Bury 1989, Welsh 1990) and there is increasing evidence that this is true for native fishes as well (Frissell 1991). To sustain the full complement of native aquatic species, effective refugia must protect both watersheds and corridors of lower-elevation, alluviated canyon or alluvial valley segments having forested floodplains.

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Chapter 5

Incidence and Causes of Physical Failure Of Artificial Habitat Structures in Streams of Western Oregon and Washington

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ABSTRACT

In recent years an increasing share of fishery management resources has been committed to alteration of fish habitat using artificial stream structures. We evaluated rates and causes of physical impairment or failure for 161 fish habitat structures in 15 streams in southwest Oregon and southwest Washington, following a flood of between 2-10-y recurrence interval. The incidence of functional impairment and outright failure varied widely among streams, with a median failure rate of 18.5% and median damage rate (impairment plus failure) of 60%. Modes of failure were diverse, and bore no simple relationship to structure design. Damage was frequent in low-gradient stream segments, and widespread in streams showing signs of recent watershed disturbance, high sediment loads, and unstable channels. Comparison of estimated 5-10-year damage rates from 46 projects from throughout western Oregon and southwest Washington, showed high but variable rates (median 14%, range 0-100%) in regions where 10-y recurrence interval unit peak discharge of streams exceeds 1.0 $m^3 \cdot sec^{-1} \cdot km^{-2}$ (92 $cfs \cdot mi^{-2}$). Results suggest that commonly prescribed structural modifications are often inappropriate and counterproductive in stream systems with high or elevated sediment loads, high peak flows, or highly erodible bank materials. Restoration of 4th-order and larger alluvial valley streams, where most potential fish production exists in the Pacific Northwest, will require re-establishment of natural watershed and riparian processes over the long term.

1. Introduction

During the past decade, popular demand and financial support for restoration of fish habitat in North American streams have increased dramatically. Restoration or "enhancement" activity in the west has been concentrated on direct modification of streams by installing artificial structures, such as log weirs and gabions. Despite numerous pleas for careful scientific evaluation (e.g., Reeves and Roelofs 1982; Hall and Baker 1982; Everest and Sedell 1984; Hall 1984; Klingeman 1984; Platts and Rinne 1985), large and costly projects continue to be planned and implemented by federal and state agencies with little or no analysis of their effectiveness.

During the 1980's, habitat management programs of federal agencies became increasingly dominated by artificial structure programs. For example, according to the Bureau of Land Management (1989), even as the number of fishery biologists in the agency dropped by more than half between 1980 and 1987, budgets increased for fish "habitat development" and "project maintenance"-- a program dominated by artificial structures. The Bonneville Power Administration spends more than $5 million annually on stream structures and related projects in Idaho, Oregon, and Washington in attempts to mitigate for hydropower impacts on wild fish (Bonneville Power Administration, unpublished data). In Fiscal Year 1987 the U.S. Forest Service built more than 2,400 fish habitat structures in the Pacific Northwest Region , and the budget for this program far

exceeded funds available to protect, monitor, and rehabilitate soil and watershed resources (U.S.D.A. Forest Service, unpublished data).

An illustration of the new reliance that resource managers are placing on artificial fish habitat technology is the Siskiyou National Forest Management Plan (U.S.D.A. 1989), which prescribes fish habitat structure projects costing more than 1.7 million dollars over three years. In the computer model used by the Siskiyou National Forest to assess the economic effects of its activities planners assumed, without supporting evidence, a net gain of three to four pounds of anadromous fish annually for each dollar spent on artificial structures. Logging in riparian areas and a projected influx of many tons of sediment annually caused by new roads and logging was assumed to have no significant adverse effect on fishery values (U.S.D.A. 1989). The Forest Service assumed that any adverse effects on fish habitat and water quality would be more than compensated by fish habitat created from new artificial structures.

On-going evaluation of failures, as well as successes, is necessary to ensure that a program is achieving its objectives, without costly mistakes or unintended side effects. The few evaluations of artificial structure projects in the Pacific Northwest have shown mixed results. Hall and Baker (1982) and Hamilton (1989) summarized published and many unpublished evaluations of the effectiveness of fish habitat modification projects in streams. Although studies of apparently successful projects (e.g., Ward and Slaney 1981; House and Boehne 1986) have been cited widely, Hamilton's review (1989) suggests that studies which showed neutral or negative biological effects have been published less frequently than those with favorable results.

Several studies have indicated that structural modifications can be ineffective, or sometimes damaging. For example, Hamilton (1989) observed reduced trout abundance in a northern California stream reach with artificial boulder structures, compared to an adjacent unaltered reach. A large-scale habitat modification program

interval within the first few years after construction provided an excellent opportunity to evaluate how well these projects could be expected to survive and function for their projected life spans. We examine the incidence of structure impairment and failure in relation to design, stream characteristics, and regional hydrologic conditions, and discuss the implications for fish habitat management in the Pacific Northwest.

2. Methods

2.1 Study Sites

In the summer of 1986 we evaluated the incidence of physical impairment or failure in artificial structure projects on 8 streams in southwest Oregon and 7 streams in southwest Washington (Fig. 26, Table 15). The sample included 161 structures built by the state of Oregon's Salmon and Trout Enhancement Program and the U.S. Forest Service between 1981 and 1985.

South coastal Oregon has intense winter precipitation, flashy streamflow, and very high sediment yields, particularly from heavily logged watersheds. In southwest Oregon, projects were intended to increase spawning habitat for fall chinook salmon Oncorhynchus tshawytscha by stabilizing gravel and providing cover for adults, and to improve rearing habitat for juvenile chinook salmon, steelhead O. mykiss, and cutthroat trout O. clarki, by increasing area, depth, and complexity of pools (Johnson 1984; U.S.D.A. 1989; G. Westfall, Oregon Department of Fish and Wildlife, unpublished data). Structures in southwest Oregon included lateral log deflectors, cross-stream log weirs, multiple-log structures, and cabled natural debris jams. Benefit-cost projections were based on a life span of 20 to 25 years for all structures (Johnson 1984; U.S.D.A. 1989).

Figure 26. Location of stream structure projects evaluated in southwest Oregon and southwest Washington during 1986. 1) Bear Cr; 2) Silver Cr; 3) Shasta Costa Cr; 4) Foster Cr; 5) Euchre Cr; 6) Crooked Bridge Cr; 7) Outcrop Cr; 8) Boulder Cr; 9) Lower Trout Cr; 10) Layout Cr; 11) Upper Trout Cr; 12) Wind R; 13) Trapper Cr; 14) Falls Cr; 15) Rush Cr.

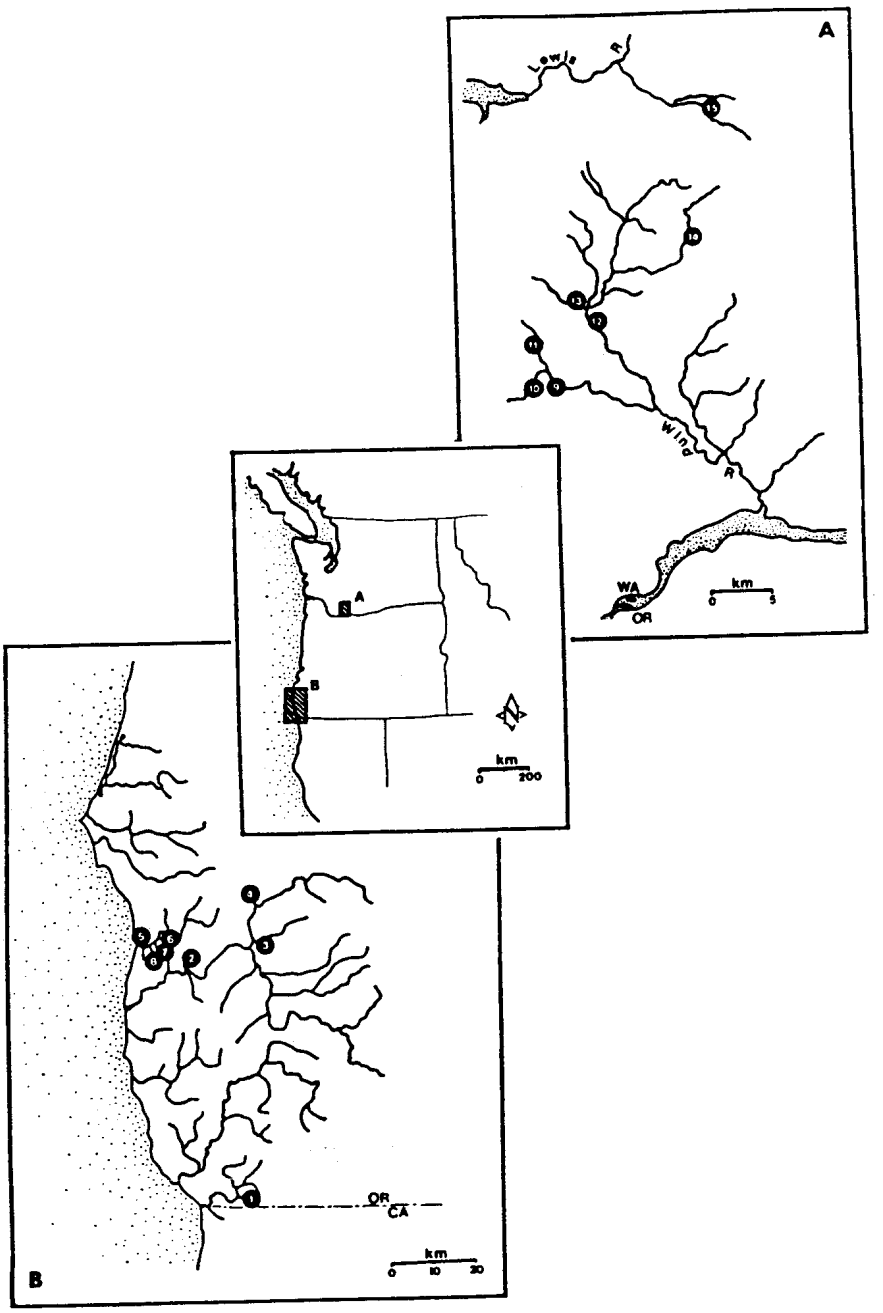


Figure 26

Table 15. Physical characteristics of study sites. Valley segment types are large-scale geomorphic units, following Frissell et al. (1992a) and Cupp (1989). Valley segment codes are: AV = alluvial valley; AFV = alluvial-fan-influenced valley; TBV = terrace-bound valley; AC = alluviated canyon; IUH = incised U-shaped valley, high gradient; slash between two codes means both valley types occured within project area.

Stream	Elev. (m)	Drainage Area (km^2)	Mean Channel Slope (%)	Mean Active Channel Width (m)	Mean Active Channel Depth (m)	Valley Segment Type
Southwest Oregon:						
Bear Cr	50	22.9	2.0	10.9	0.7	AC
Foster Cr	60	30.6	1.5	9.6	0.8	TBV
Silver Cr	20	25.1	1.0	8.9	0.4	AFV
Shasta Costa Cr	50	46.0	1.0	18.2	0.9	TBV
Euchre Cr	25	51.4	1.0	30.0	1.0	AV/AFV
Crooked Bridge Cr	25	2.7	2.5	6.0	0.7	AV
"Outcrop" Cr	25	1.2	4.0	5.5	0.5	AFV
Boulder Cr	25	5.9	2.0	12.0	0.7	AFV/AV
Southwest Washington:						
Rush Cr	945	17.8	2.0	10.4	0.7	AV
Falls Cr	830	24.8	6.0	8.1	0.6	IUH
Layout Cr	540	14.8	1.0	15.6	0.7	AV
Upper Trout Cr	565	10.8	2.0	9.3	0.6	AV
Lower Trout Cr	535	62.7	1.0	20.0	1.0	AV
Wind R	335	632.0	1.0	31.2	1.0	AV
Trapper Cr	340	28.8	1.5	25.6	0.8	AV

In southwest Washington, a region of moderately high sediment yield and high peak flows from winter rain-on-snow events, projects were intended to increase pool area for rearing of juvenile salmonids (U.S.D.A. 1987). Steelhead trout, brook trout Salvelinus fontinalis, and spring chinook salmon occurred in project streams. The projects included log weirs, diagonal log deflectors, multiple-log structures, cabled natural woody debris jams, and single and clustered boulders.

2.2 Flood Peak Estimation

Because none of the project streams were gauged, we used several methods to estimate recurrence interval of the February 1986 flood, the primary event affecting our study. Gauged streams in southwest Oregon experienced an instantaneous peak flow of about 2 year recurrence interval (Geological Survey Water Resources Data for Washington and Oregon, Water Year 1986; Friday and Miller 1984). However, this flood was unusual in its duration, with high flow for several consecutive days. After adjusting for duration the estimated recurrence interval is 5-7 years, based the estimates of Friday and Miller (1984) for Chetco River near Brookings and South Fork Coquille River at Powers. McGavock et al. (1986) estimated the recurrence interval of the February 1986 flood in gauged southwest Washington streams at 3-5 years.

To assess variation in the February 1986 peak flow among the project streams in southwest Oregon, we surveyed cross sections at the project sites and reconstructed flood crests based on flotsam lines. Using the Manning equation (Thorne and Zevenbergen 1982; Richards 1982) with roughness estimated visually (Barnes 1967), we estimated peak flows for each stream. We then estimated flood recurrence intervals (for instantaneous peak flow) following three regional prediction procedures (Harris et al.

1979; Campbell et al. 1982; and Andrus et al. 1989). Final estimates for each stream are the averaged results of the three procedures, except for two watersheds of less than 5 km^2 drainage area, where only the Andrus et al. (1989) method appeared to provide reasonable estimates. Because peak discharge estimates often err by as much as 30 percent (Thorne and Zevenbergen 1982), and additional error exists in recurrence interval predictions, these estimates, which varied from slightly less than 2 years to 10 years among the Oregon streams (Table 2), should be viewed as rough approximations.

2.3 Definitions

We classified structures into three different categories, depending on their physical condition and function. A structure that had been washed downstream, severely fragmented, or grossly dislocated such that it had little or no contact with the low flow channel or was otherwise incapable of achieving its intended physical objective (e.g., creating or enlarging a pool) was classified as a failure. Where the structure remained in its original location, but due to alteration of the structure or changes in the stream channel, no longer functioned in the intended mode or appeared to be at least temporarily ineffective, it was classified as impaired. Structures which had been buried under bedload deposits were considered impaired. Structures not visibly damaged or debilitated were categorized as "functioning roughly as intended," or successful.

We define damage rate as the proportion of structures of a project in either the failed or impaired performance categories, i.e., the proportion of structures which were not successfully meeting physical objectives. Failure rate is defined as the proportion of structures of a project in the failure category only, i.e., structures that were lost or completely dysfunctional. Based on the time since installation and the estimated

recurrence interval of the February 1986 flood, we assume these rates reflect the incidence of damage and failure to be expected over a 5-10-year time span.

Some subjectivity is obviously involved in judgments about impairment, and to a lesser extent failure, particularly where the intent of the designer is not immediately clear. We based our determination of whether structures achieved design objectives primarily on the general physical objectives outlined in project plans (e.g., "create new rearing pools"), but criteria varied somewhat depending on structure type. For example, whereas a log weir would be expected to produce a plunge pool, a single boulder was probably intended only to create a small scour hole; a cabled natural debris jam would be expected to stay in place and maintain pre-existing pool and cover conditions. Within these limitations, damage rate is a useful indicator of the effective life of a project, maintenance requirements, the importance of unintended side effects, and the likelihood of future failure.

2.4 Structural Evaluation

As we surveyed a stream, we recorded the location and type of each structure. We measured reach slope with an Abney level, measured width of the active (unvegetated) channel with a meter tape, and measured the depth and surface area of the pool associated with each structure. We recorded processes or events contributing to impairment or failure of the structure, and in some cases drew a small sketch map. Our previous knowledge of structure design and placement at many of the projects helped us reconstruct failure processes, although we avoided speculation where no physical evidence of failure mode remained. Because failed structures sometimes wash away and leave no trace, we undoubtedly underestimated the number of structures

originally present in some projects, making our estimates of failure rates conservative. We recorded information on stream bank materials and riparian landform in the field, and compared these data with topographic maps to classify stream segments following Frissell et al. (1992a) and Cupp (1989). We calculated failure and impairment rates for each structure type and each stream, and compared them with stream-specific data on flood flow magnitude, mean channel width, slope, drainage area, and stream segment type (Table 15).

2.5 Inter-regional Comparisons

To set our results in broader context, we compared our summary data with unpublished information on other projects constructed during 1981-1985 by the U.S. Forest Service (B. Higgins and H. Forsgren, Mt. Hood National Forest, unpublished data; D. Hohler, Mt. Hood National Forest, unpublished data) and Bureau of Land Management (House et al. 1989). Although we find that our impairment or failure estimates are often somewhat greater than those of agency biologists responsible for the projects, we believe these data are comparable as a rough approximation of regional patterns. Since most of these projects had experienced a flood of between 5- and 25-year recurrence interval during the 2-8 years they had been in place, we used the data to approximate average failure and impairment rates.

Because climatic and geomorphic conditions in western Oregon are diverse, we grouped the data for all projects into five regions, defined by geology, topography, elevation, climate, and streamflow patterns. We examined stream flow statistics for gauged streams in each region (Friday and Miller 1984) and used these to characterize regional peak flow magnitudes.

3. Results

3.1 Damage Rates in Relation to Stream Characteristics

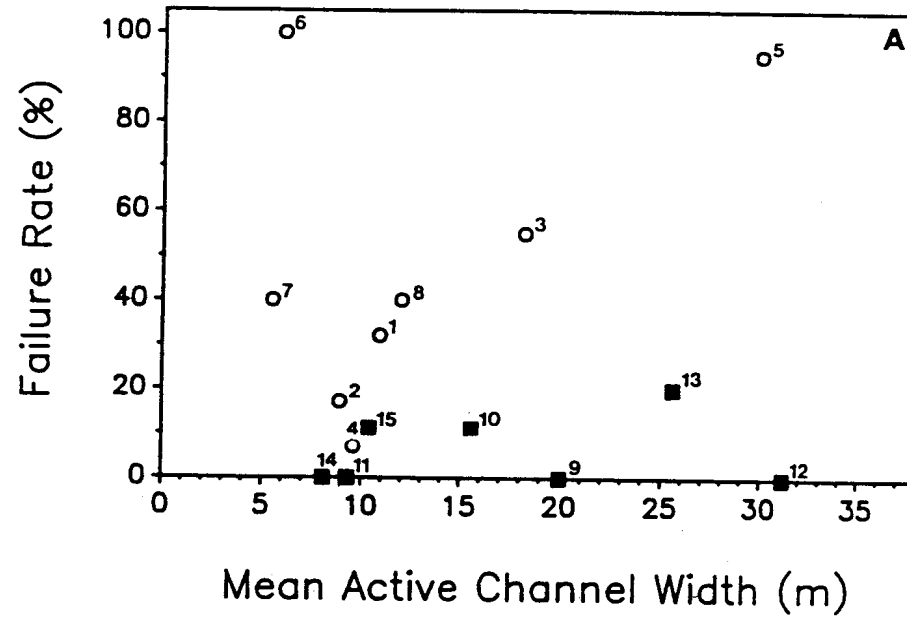
The incidence of structure failure and damage varied widely among streams (Table 16). Overall, failure rates were higher in southwest Oregon streams (median 40%, mean 48%, range 7-100%) than southwest Washington streams (median 0%, mean 6%, range 0-20%). Rates of overall damage were less disparate, but appeared to be higher in southwest Oregon (median 70%, mean 67%, range 27-100%) than southwest Washington (median 42%, mean 46%, range 0-89%).

Rates of damage were higher in larger and wider streams (Fig. 27a). Projects in streams with active channel widths wider than 15 m had a median damage rate of 79% (range 50-100, n=6), whereas those with active channels narrower than 15 m were highly variable and had a median damage rate of 50% (range=0-100, n=9). Southwest Oregon data suggested a roughly linear increase in failure rate with stream width (Fig. 27b). In southwest Washington, failure rate was not apparently correlated with stream width, although impairment and therefore damage rate was correlated with stream size. There was no clear relationship between drainage basin area and failure or damage rates. Because climatic and hydrologic characteristics of individual streams vary within a region, active channel width is a better site-specific, integrated measure of streamflow and associated hydraulic stresses than is basin area. Channel width is influenced by bank material erodibility (Schumm 1960; Richards 1982), which also affects structure performance (see Mode of Failure).

Table 16. Flood magnitude estimates and rates of total damage and failure for fish habitat structure projects surveyed in 1986. Flood peak is estimated peak discharge in m^3 per second, and estimated recurrence interval in parentheses. Dash indicates discharge data not available; recurrence interval at these sites estimated from nearby streams or regional analyses by US Geological Survey.

Stream	1985-86 Flood Peak	Number of Structures	Damage Rate (%)	Failure Rate (%)
Southwest Oregon:				
Bear Cr.	28 (2 y)	19	79	32
Foster Cr.	30 (2 y)	15	27	7
Silver Cr.	17 (2 y)	6	50	17
Shasta Costa Cr.	45 (<2 y)	18	83	55
Euchre Cr.	92 (5 y)	19	100	95
Crooked Bridge Cr.	12 (10 y)	6	100	100
"Outcrop" Cr.	7 (5 y)	5	40	40
Boulder Cr.	- (5 y)	5	60	40
Southwest Washington:				
Rush Cr.	- (<2 y)	9	22	11
Falls Cr.	- (<2 y)	6	0	0
Layout Cr.	- (3-5 y)	9	89	11
Upper Trout Cr.	- (3-5 y)	19	42	0
Lower Trout Cr.	- (3-5 y)	5	40	0
Wind R.	- (3-5 y)	10	70	0
Trapper Cr.	- (3-5 y)	10	60	20

Figure 27. Failure (A) and damage (B) rates of projects in southwest Oregon (open circles) and southwest Washington (solid squares) in relation to active channel width. Stream numeric codes are same as Figure 1. Damage rate includes both failed and impaired structures (see text).


A
Failure Rate (%)
Mean Active Channel Width (m)
0
20
40
60
80
100
0
5
10
15
20
25
30
35

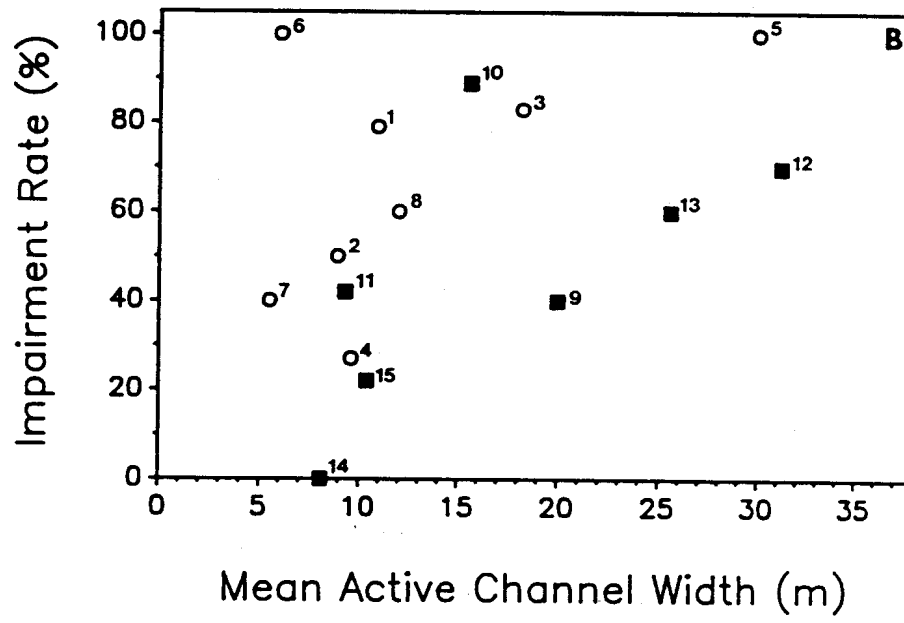

B
Impairment Rate (%)
Mean Active Channel Width (m)
0
20
40
60
80
100
0
5
10
15
20
25
30
35

Figure 27