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Best Management Practices, Cumulative Effects, and Long-Term Trends in Fish Abundance in Pacific Northwest River Systems

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Abstract

Although it is widely believed that forest management has degraded streams and rivers, quantitative relationships between long-term trends in fish abundance and forestry operations have not been successfully defined. In this article we review the difficulties in describing cumulative effects of forest management on fishes of the Pacific Northwest. Despite uncertainties in interpreting long-term trends from catch and escapement statistics as well as widespread programs of hatchery production, many local fish populations are declining. We suggest that trends in the abundance of *individual populations* are often of limited use in identifying the cumulative effects of forest management within a river system. Shifts in the composition and organization of *fish communities* may provide more comprehensive evidence of the extent of environmental alteration. Reduced stream habitat complexity has been one of the most pervasive cumulative effects of past forest practices and probably has contributed to significant changes in fish communities, particularly when accompanied by other land use activities that have led to straightened, confined channels. In simplified streams a few fish species have characteristically been favored while others have declined or disappeared completely. Likewise, fish culture practices have resulted in overall losses of genetic diversity among species. In order to protect channel complexity and biodiversity, best management practices (BMPs) should include measures to preserve physical and biological linkages between streams, riparian zones, and upland areas. Connections must include transfer processes that deliver woody debris, coarse sediment, and organic matter to streams, as these materials are largely responsible for creating and maintaining channel complexity and trophic diversity. Past forest practice regulations have required attainment of individual water quality standards, such as temperature or dissolved oxygen, and have been aimed at protecting certain life history stages of single species (e.g., salmon eggs in spawning gravels). This approach is inadequate to achieve the goal of restoring and maintaining natural levels of complexity at the level of a stream ecosystem. New BMPs

are beginning to address this issue by prescribing riparian management zones with a greater range of vegetative species and structural diversity, thus providing for future sources of large woody debris, floodplain connections, and other linkages important to ecosystem function. Benefits of new BMPs in terms of improved habitat complexity and increased diversity of fishes on the scale of a river basin will require coordinated planning and extensive application, and will take years—perhaps decades—to become apparent.

Key words. Streams, habitat, cumulative effects, fish populations, biodiversity.

Introduction

For many years the cumulative effects of forest management activities on fish populations in river systems of the Pacific Northwest have been of concern (Hicks et al. 1991). Quite often the term "cumulative effects" has been implicitly or explicitly taken to mean the repeated, additive, or synergistic effects of forestry or other land use practices on various components of a stream's environment in time and space (Burns 1991). The term has considerable intuitive appeal, as it suggests that environmental impacts of specific management activities cannot properly be viewed in isolation from a broad perspective of land management at large spatial scales and long time scales. An underlying assumption has been that although individual management actions by themselves may not cause undue harm, taken collectively such land use activities may result in unacceptable stream habitat degradation and long-term declines in fish abundance, particularly when accompanied by heavy fishing pressure and competition with introduced species or hatchery stocks (Cederholm et al. 1981, Salo and Cederholm 1981).

As seemingly logical as this concept is, clear examples of cumulative effects of forest management on stream habitat have been difficult to demonstrate in all but the most severely degraded river systems (Platts and Megahan 1975, Coats and Miller 1981, Tripp and Poulin 1986*a*, *b*; Megahan et al., this volume). Furthermore, establishing unambiguous relationships between abundance of fish populations and cumulative environmental change has been equally difficult, if not more so (Pella and Myren 1974, Platts and Nelson 1988, Holtby and Scrivener 1989). This article reviews these difficulties and describes trends in habitat quality that appear to be common to river basins with histories of forest management and other types of land use activities. We discuss how the concept and implementation of best management practices (BMPs), a term generally taken to mean state-of-the-art environmental protection measures, have both succeeded and failed in attempting to (1) ameliorate adverse cumulative impacts, (2) protect natural interactions between streams and riparian zones, and (3) preserve the integrity of aquatic communities. Although we focus primarily on forest management, the conclusions apply equally to other land uses that cause similar types of environmental change.

Lack of Knowledge About Cumulative Effects and BMPs

Timber harvesting has been practiced in the Pacific Northwest since the early 19th century, but logging and reforestation techniques have changed dramatically. In general, new developments in logging systems have come at approximately 20–30 year intervals, while reforestation technology has evolved much more rapidly over the last 50 years (Figure 7.1). Many timber management techniques considered to be technologically advanced decades ago are viewed as outdated and environmentally destructive today (Franklin, this volume; Oliver et al., this volume). An example is the use of streams and rivers for log transport to mills (Sedell and Luchessa 1982, Sedell and Duval 1985). At one time, water-based log transport was the most practical means of moving very large logs from forested headwaters to downstream processing facilities. This management practice, considered the best in its day, caused a great deal of damage to streams and riparian zones, especially when it involved the use of splash dams (Bisson et al. 1987). Other technological advancements such as high lead and skyline yarding systems, as well as helicopter and balloon logging systems, were developed for harvesting in steep terrain but have proved to be far more environmentally sound on steep slopes than older methods that required skidding logs over forest soils.

The concept of best management practices with regard to environmental protection and restoration was essentially a post-World War II phenomenon and has been applied to land management activities both in Europe and North America (Petts 1990). Public pressure to protect stream habitat in western North America was at least partly responsible for an end to splash damming and log drives in the 1950s, and for some of the first restrictions on yarding across stream channels (Figure 7.1). The first comprehensive long-term forestry related aquatic research program in the Pacific Northwest—the Alsea watershed study—began in 1959 and continued uninterrupted until 1974 (Hall et al. 1987). As a result of findings in the Alsea watershed and elsewhere, most western states and provinces enacted forest practices regulations by the early 1970s. Many of the regulations addressed changes in temperature and fine sediment, two parameters shown to have been increased by logging activities. BMPs were therefore defined mostly in terms of water quality standards (i.e., temperature protection and erosion control).

New concerns about the effects of logging on peak flows and on the abundance of large woody debris in streams began to take shape in the 1970s and resulted in renewed research activity in the 1980s (Salo and Cundy 1987). Forest practice regulations have been revised to accommodate this new research information, yielding BMP regulations that have changed greatly within

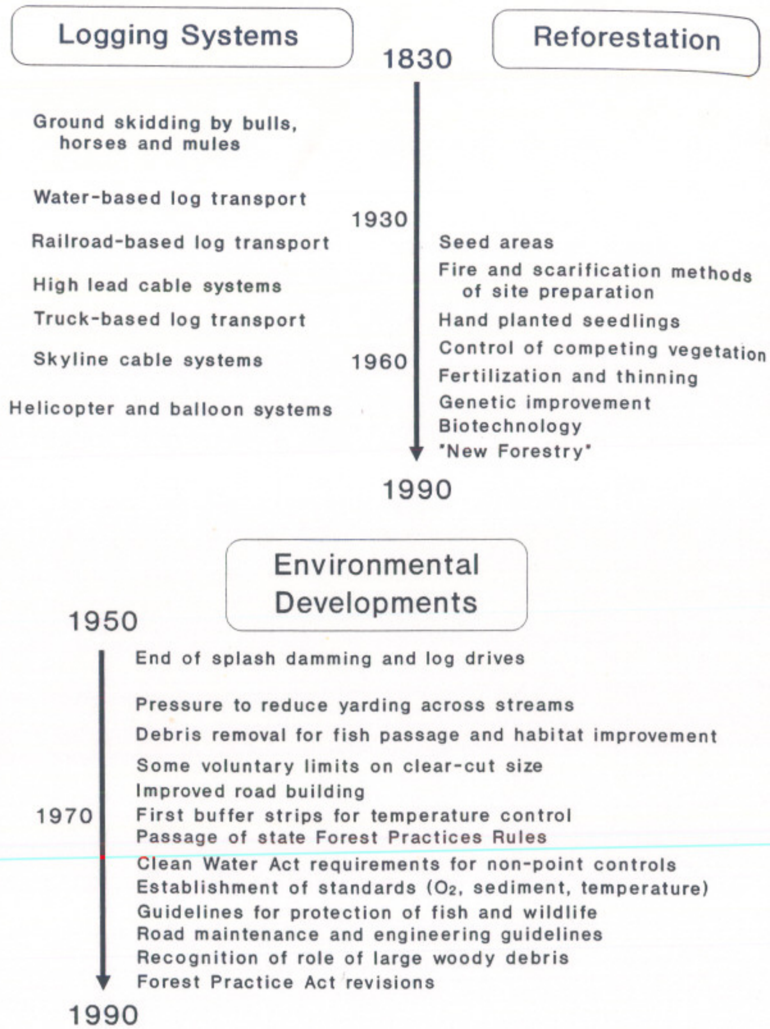


FIGURE 7.1. Diagrammatic chronology of some developments in logging systems, reforestation techniques, and environmental protection for streams and rivers in the Pacific Northwest.

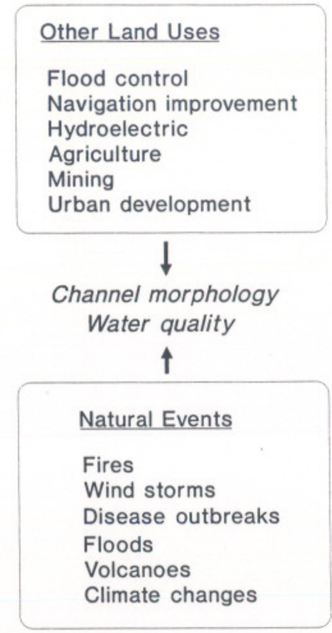


FIGURE 7.2. Some other important types of disturbances caused by various land management activities and natural events that can influence channel morphology and water quality in streams.

a single decade (Bilby and Wasserman 1989). Taken together, changes in logging systems, reforestation techniques, and environmental protection requirements have meant that our concepts of best forest management practices have always been evolving. Understanding the cumulative effects of forest management in light of changing BMPs has posed a difficult challenge.

Another reason for the poor understanding of cumulative effects of forest management on streams is that disturbances unrelated to forestry operations from both natural events and other land use activities have occurred concurrently with logging and reforestation (Figure 7.2). Naturally occurring events have taken place throughout drainage systems, although some (wildfires, windstorms) have probably had greater impact in forested headwaters than in nonforested lowlands (Keller and Swanson 1979). Many of the changes resulting from other types of land use have taken place in larger river systems, although agriculture, urban development, and mining have all affected small streams. Different types of disturbances can cause characteristic changes in stream habitat, but in some cases the environmental impacts of natural events or other land use activities may be relatively similar (Hicks et al. 1991, Schlosser 1991). For example, increased sediment deposition may

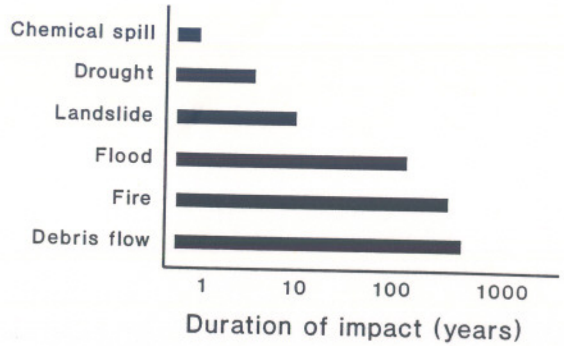


FIGURE 7.3. Hypothetical duration of impact of various disturbances on stream ecosystems in the Pacific Northwest, based in part on data from Swanson and Lienkaemper (1978), Grant (1988), and Niemi et al. (1990). Variation in the resilience of stream ecosystems after different disturbances, not shown in this figure, can be considerable.

accompany agriculture, mining, or urban development, as well as natural wildfires or floods. Although the timing and amount of inputs from these sources may vary (Poff and Ward 1990), it can be difficult to distinguish sediment produced by forestry practices from sediment produced from other types of disturbance (Everest et al. 1987). Effects of other disturbances may therefore obscure or mask cumulative environmental change attributable to forestry operations.

Recovery rates of the physical environment from different types of environmental disturbance are highly variable (Swanson and Lienkaemper 1978, Grant 1986). Likewise, the recovery of stream biota is highly variable, depending both on the nature of the disturbance (i.e., its spatial extent and temporal duration; Poff and Ward 1990) and the assemblages of plants and animals in the stream (Niemi et al. 1990). The duration of physical and water quality impacts from disturbance (Figure 7.3) may range from very short (a few days) to very long (several hundred years). Stream channels reflect disturbances that took place recently as well as changes that occurred decades or even centuries ago (Gregory et al. 1987, Grant et al. 1990). Reconstructing the disturbance history of a stream from surveys of existing conditions often requires subjective interpretation of cause and effect. Nevertheless, some recent procedural advances have improved our ability to interpret changes caused by past catastrophic disturbances (Grant 1988). Recovery rates of stream biota may depend, among other things, on the biota's "preadaptedness" to a particular type of disturbance (Poff and Ward 1990).

All of these factors make quantification of the effects of past disturbances based solely on recent physical and biological surveys very difficult. Thus

long-term records of environmental conditions are essential if habitat change is to be associated with specific natural events and land use activities (Sedell and Luchessa 1982). Unfortunately, long-term habitat monitoring has not taken place in the majority of Pacific Northwest river systems. Very early habitat assessments were in the form of verbal descriptions of river valleys (Sedell et al. 1988, Gonor et al. 1988), and it was not until the early 20th century that biologists first attempted to quantify habitat conditions (Pacific Fishery Management Council 1979). By this time many rivers had been changed by logging and other land use practices (Chapman 1986).

Many of these original habitat surveys were poorly archived or never published in a widely available format; in most cases original data have been lost. Changing inventory methods and personnel transfers have also been a barrier to monitoring continuity. In rare instances where earlier habitat surveys could be related to modern conditions, we have been able to obtain a much more accurate picture of long-term trends in habitat quality (Sedell and Everest 1991). But these instances are rare, and may be limited to portions of river basins.

Long-Term Trends in Fish Abundance

Long-term trends in the abundance of Pacific Northwest fish populations, especially anadromous salmonids, have often been determined through examination of commercial, sport, and Native American catches, as well as estimates of the escapement of adult fish to spawn. Although catch and escapement records occasionally extend back into the late 19th or early 20th century, they are prone to measurement errors that can confound actual trends in stock abundance. Estimates of historical run size often require numerous assumptions and conversions, many of which cannot be verified but which nevertheless are necessary (Chapman 1986). Several different types of errors are associated with catch statistics. Unreported catches can be significant where catch monitoring efforts are small and where harvest records are supplied by fishermen themselves. Significant changes in fishing gear and other methods of harvest can result in significant increases or decreases in catch unrelated to stock abundance. Likewise, changes in size restrictions and fishing season openings can strongly affect the number of fish caught. Fishing pressure may change from year to year depending on weather, economic conditions, and a variety of political considerations. Finally, mixed-stock catches, which are the rule in most ocean fisheries, often prevent the separation of commercial and sport catches into component stocks from different drainage systems. Many of these problems, and efforts to circumvent them, are discussed by Healey (1982).

Although there are fewer types of errors associated with escapement estimates, the magnitude of the errors can be great. Where escapement is based on counts from viewing windows at fish ladders or on fish traps at impassable barriers, estimates can be relatively accurate, provided that no fishing or other significant mortality occurs between the counting location and spawning areas. However, these techniques require proper functioning of equipment at all times, including periods of high streamflow. But even ladder counts can be very inaccurate. For example, U.S. Army Corps of Engineers Annual Fish Passage Reports for the Columbia River often record substantially more anadromous shad (*Alosa sapidissima*) at The Dalles than at Bonneville Dam (ASACE 1989). In 1988, 2.01 million shad were counted at The Dalles Dam while only 1.16 million were counted from Bonneville Dam, which is downstream and would have had to pass at least as many fish as were counted at The Dalles.

Counts of adult fish on spawning grounds are often used to gauge run size, but such counts require frequent surveys during the period of spawning, and water quality must be conducive to fish viewing (Beidler and Nickelson 1980). Quite often these requirements are not met. Surveys of entire drainages may be logistically difficult, bad weather may hamper viewing, fish may be counted twice or more, and aerial surveys may be obscured by riparian vegetation (Neilson and Geen 1981). Changes in survey methods or locations of index areas often add unknown errors to escapement estimates. Finally, indices of adult abundance such as spawning counts from designated index sites or redd counts that assume a given number of redds per female—and in turn a ratio of females to males—require careful, local verification (Solazzi 1984, Nickelson et al. 1986); where these assumptions are not verified, escapement estimates may be subject to bias.

Catch statistics by themselves are often of limited use in determining the causes of advances or declines in stock abundance. Anadromous salmonids experience a variety of environmental conditions over the period of freshwater and marine rearing prior to being taken in a fishery. Because so many potentially limiting factors are encountered before capture (Figure 7.4), it is often impossible to determine the reasons for annual changes in run strength. For example, McDonald and Hume (1984) reported over tenfold variation in marine survival of Babine Lake sockeye salmon (*Oncorhynchus nerka*) between 1961 and 1977, and fivefold differences in marine survival between consecutive years. A variety of correlation approaches have been attempted in order to relate catch statistics to freshwater and marine habitat parameters such as streamflow and ocean upwelling (Neave 1949, Smoker 1955, Scarnecchia 1981, Nickelson 1986). While these authors claim varying degrees of success in predicting annual adult abundance at broad geographical scales, all have acknowledged that identification of specific environmental factors governing abundance using statistical correlation is very difficult. Furthermore, statistical design becomes crucial to the interpretation of correlation analyses and may lead to inaccurate conclusions if improper design is employed (Walters et al. 1988, Peterman 1989).

An illustration of the difficulty in relating catch statistics to land use is shown by a comparison of commercial catches (Mullen 1981) of coho salmon (*O. kisutch*) in two nearly adjacent drainages of similar characteristics on

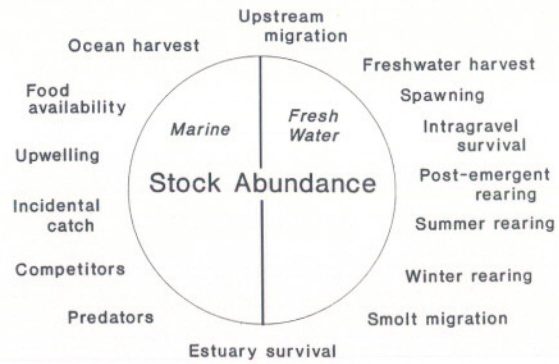


FIGURE 7.4. Some of the factors known to exert an influence on stock abundance in freshwater and marine environments.

the Oregon coast (Figure 7.5). The catch records cover a period of more than 30 years when both basins were being actively logged and relatively little protection was given to streams. The fishing methods in both cases were terminal nets located near the river mouths, and the records describe the catches of wild fish. Salmon hatcheries in this area began production in the mid-1950s. In the Alsea River Basin, catches of coho salmon generally increased to peaks in the 1930s and early 1940s, after which there was a decline. In the Siletz River Basin there was a steady decline over the entire period. Because these nearby basins were being logged at approximately the same time, it is difficult to explain why the catch record for the Alsea River appeared to be dome-shaped while that of the Siletz River trended consistently downward. In all likelihood, factors other than or in addition to habitat damage associated with timber harvest were influencing the commercial catch of coho salmon in the rivers. Determination of the causes of increased or decreased catch in such instances becomes statistically intractable without a comprehensive long-term knowledge of both freshwater and marine conditions, as well as thorough records of fishing methods and pressure.

Over the past several decades the output of smolts from hatcheries has risen along the Pacific Northwest coast. Nickelson (1986) notes that coho salmon smolt releases in the Oregon Production Area increased from fewer than 1 million during the 1950s to over 30 million by 1970. After stabilizing at approximately 35 million smolts in the mid-1970s, large-scale releases of coho from privately owned facilities increased total hatchery production to 62 million fish by 1981, although hatchery smolt releases have since declined. Light (1992) found that production of steelhead (*Oncorhynchus mykiss*) smolts throughout the Pacific coast rose by a factor of 10 from approximately 3 million in 1960 to 30 million by 1987 (Figure 7.6A). Similar increases in hatchery output of other anadromous salmonids have occurred

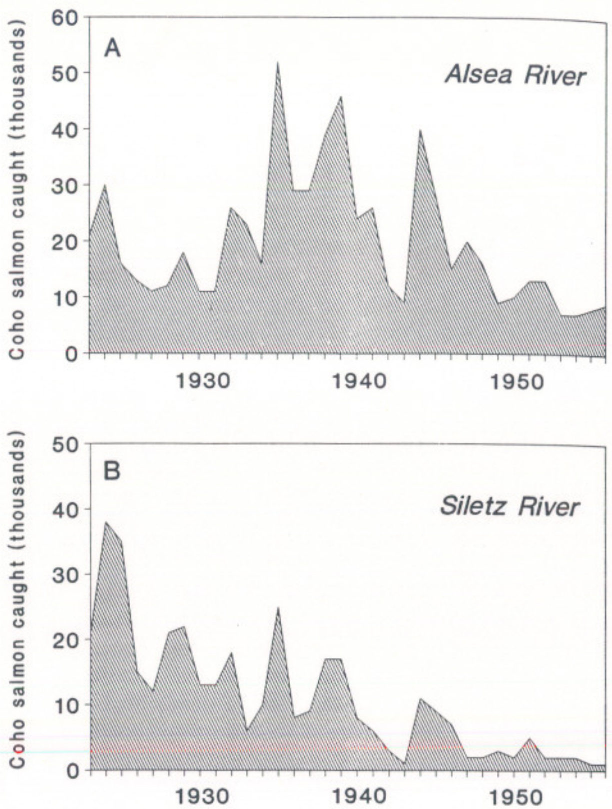


FIGURE 7.5. Number of adult coho salmon caught in (A) the Alsea River and (B) the Siletz River, Oregon, from 1923 to 1956. Data from Mullen (1981).

in response to greater demands by commercial, sport, and Native American fishermen, in addition to mitigation requirements for habitat loss. As a result of extensive hatchery production, overall catches may remain stable even though the abundance of naturally produced fish has declined (Figure 7.6B). It is possible too that hatchery fish have had a directly negative impact on wild stocks through competition for limited freshwater or marine resources (Nickelson et al. 1986, Lichatowich and McIntyre 1987, Hilborn 1992) or through genetic introgression of nonadaptive traits (Leider et al. 1984, Chilcote et al. 1986). These factors have often not been considered when transferring salmonid eggs and fry between river systems. Furthermore, an intense fishery targeting a large hatchery run can incidentally depress the escapement of wild fish (Nehlsen et al. 1991, Hilborn 1992).

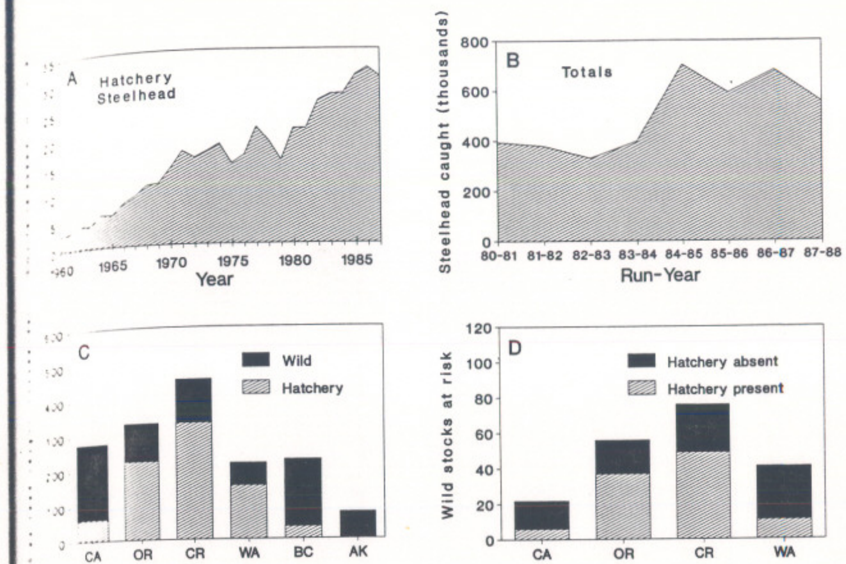


FIGURE 7.6. (A) Total number of steelhead smolts released from hatcheries along the Pacific coast from 1960 to 1987. (B) Total catch of adult steelhead on the Pacific coast from run-years 1980–81 through 1987–88. (C) Relative contribution of hatchery and wild steelhead to the total catch by region. CA=California, OR=Oregon coastal rivers, CR=Columbia River Basin, WA=Washington coastal and Puget Sound rivers, BC=British Columbia coastal rivers, AK=Alaska coastal rivers. (D) Number of wild stocks of *Oncorhynchus* considered to be at risk of extinction in river systems with and without hatcheries producing the same species. Locations as in C. Data for A-C from Light (1992); data for D from Nehlsen et al. (1991).

The heavy contribution of hatchery fish to total harvest for some species is illustrated by the relative catches of hatchery and wild steelhead over the range of steelhead distribution in western North America (Figure 7.6C). In the geographical center of the range, which includes coastal Washington, Oregon, and the Columbia River Basin, hatchery steelhead comprise about 70–80% of the total catch. Thus hatchery fish have largely supplanted wild steelhead in the most productive part of the range. Additionally, Nehlsen et al. (1991) note that substantial fractions of the anadromous salmonid stocks at risk of extinction in the Columbia River basin and in rivers on the Oregon coast occur in drainages where hatcheries are propagating the same species (Figure 7.6D).

Many of the nonnative fishes that have become established in Pacific Northwest river systems were introduced to provide a wider variety of sport fishing opportunities than existed with the native fauna. The majority of introduced species have belonged to the Centrarchidae, Percidae, and Ictaluridae, but some species have been imported for other reasons. For example, mosquitofish (*Gambusia affinis*) were imported to control mosquitoes, the vectors of malaria. Some of the introductions have been extremely successful; the American shad (*Alosa sapidissima*) has become the most abundant large anadromous fish in the Columbia River, with adult shad runs sometimes exceeding the combined run totals of all native anadromous salmonids. Recently, sterilized Asian grass carp (white amur, *Ctenopharyngodon idella*) and tiger musky (a sterile muskellunge x northern pike hybrid) have been introduced into the region on a limited basis to control rooted aquatic vegetation and northern squawfish (*Ptychocheilus oregonensis*), respectively. In addition, some invertebrates (e.g., opossum shrimp, *Mysis relicta*) have been released into lakes and have been able to establish viable populations, and many species of exotic marine invertebrates are released into Pacific Northwest waters through ballast water discharges.

The overall impact of exotic species on native fishes is poorly known, but exotics are potentially able to prey upon and compete with the native fauna both as juveniles and adults. Li et al. (1987) have summarized the food web of the middle reaches of the Columbia River (Figure 7.7), where introduced species now dominate many of the trophic pathways. Although the effects of exotic species introductions are most likely to be felt in large rivers and in lakes, many native species use these areas at some point in their freshwater life cycle. In an examination of the extinctions of North American freshwater fishes during the 20th century, Miller et al. (1989) found that 67% of the extinctions occurred in areas with established populations of nonnative species. It seems highly likely that exotic species have contributed to the decline of certain stocks of Pacific Northwest fishes, but the extent to which these negative interactions can be separated from those involving competition with hatchery fish or from the impacts of habitat damage and overfishing has not been quantitatively determined.

Because of their great importance to the region, the majority of research on fish population abundance in the Pacific Northwest has focused on salmon and trout. Very little is known about the effects of cumulative habitat changes on the abundance of most nonsalmonid species, particularly those that do not contribute directly to sport or commercial fisheries. Some of these species may be more sensitive to habitat change than anadromous salmonids because they spend their entire lives in freshwater and may be associated with a specific type of habitat.

We are aware of no studies that have attempted to assess the abundance of nonsalmonid populations at the scale of a drainage basin in the Pacific Northwest. Furthermore, there are few if any long-term records of nonsalmonids at index sites, where only records of salmonid abundance tend to be maintained. There are several reasons for the paucity of information on nonsalmonids, apart from their lack of commercial or recreational significance. Sampling gear for streams and rivers is usually designed to capture salmonids and other midwater species; it is often very inefficient at sampling

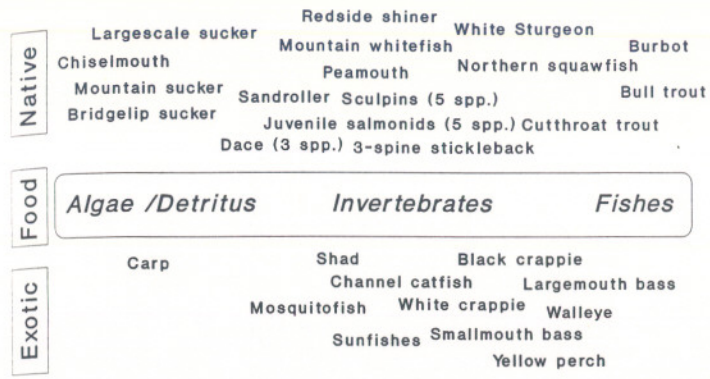
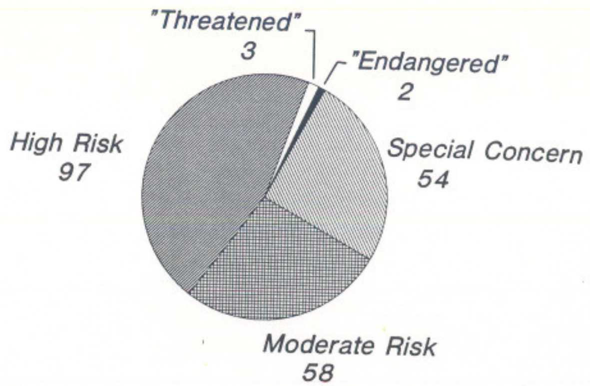


FIGURE 7.7. Native and exotic fishes of the middle and lower Columbia River, arranged according to their approximate preferences for different food types. Based on a diagram in Li et al. (1987). White sturgeon=*Acipenser transmontanus*, American shad=*Alosa sapidissima*, mountain whitefish=*Prosopium williamsoni*, juvenile salmonids=*Oncorhynchus* spp., cutthroat trout=*Oncorhynchus clarki*, bull trout=*Salvelinus confluentus*, chiselmouth=*Acrocheilus alutaceus*, carp=*Cyprinus carpio*, peamouth=*Mylocheilus caurinus*, northern squawfish=*Ptychocheilus oregonensis*, dace=*Rhinichthys* spp., redside shiner=*Richardsonius balteatus*, bridgelip sucker=*Catostomus columbianus*, largescale sucker=*Catostomus macrocheilus*, mountain sucker=*Catostomus platyrhynchus*, channel catfish=*Ictalurus punctatus*, sandroller=*Percopsis transmontana*, burbot=*Lota lota*, mosquitofish=*Gambusia affinis*, threespine stickleback=*Gasterosteus aculeatus*, sunfishes=*Lepomis* spp., smallmouth bass=*Micropterus dolomieui*, largemouth bass=*Micropterus salmoides*, white crappie=*Pomoxis annularis*, black crappie=*Pomoxis nigromaculatus*, yellow perch=*Perca flavescens*, walleye=*Stizostedion vitreum*, sculpins=*Cottus* spp.

fishes living on or in the benthos. Some taxa are difficult to identify and are often assigned simply to genera (e.g., *Cottus* spp.). Other species do not live in areas of the drainage inhabited by salmonids, and are likely to be overlooked in fish surveys. All of these factors have contributed to the general absence of knowledge of the status of nonsalmonids in the region.

Decline of Wild Salmonid Stocks

Recent analysis of the status of anadromous salmonid stocks (genetically distinct populations native to particular drainage systems) in the western United States has revealed over 200 stocks that are currently at some level of risk of becoming extinct (Figure 7.8). Nehlsen et al. (1991) have separated these stocks into three risk categories. Those with a high risk of extinction include stocks with consistent and significantly declining spawning escapements or those where the total adult population is believed to be less than 200 indi-



High Risk of Extinction	Moderate Risk of Extinction	Stocks of Special Concern
Spawning escapements significantly declining	Population stable after period of major decline	Vulnerable to minor disturbance
Fewer than 200 adults remain	More than 200 adults remain	Population trends suggest decline
		Interbreeding with non-native fish
		Unique character

FIGURE 7.8. Native stocks of anadromous Pacific salmon (*Oncorhynchus*) from California, Oregon, Idaho, and Washington considered by Nehlsen et al. (1991) to be at risk of extinction, and their criteria for classifying stocks in different risk categories. Some stocks have recently been classified as "Endangered" or "Threatened" under the U.S. Endangered Species Act.

viduals. Those with a moderate risk of extinction have relatively stable populations after a period of significant decline and have more than 200 spawning adults. Those stocks considered to be of special concern include populations that are vulnerable to minor disturbances, populations whose trends in abundance suggest a consistent pattern of decline, and populations in which there is significant interbreeding with nonnative fish.

Nearly half of the stocks considered to be at risk by Nehlsen et al. (1991) were placed in the high risk category (Figure 7.8). Of these, the greatest number occurred in the Columbia River Basin, although approximately 20 high risk stocks occurred respectively in Oregon coastal streams and in Washington coastal and Puget Sound streams (Figure 7.9A). Stocks that may already be extinct were identified from California, the Columbia River, and Washington. The greatest number of stocks classified at moderate risk of extinction occurred in Oregon. Without exception, habitat damage (Figure

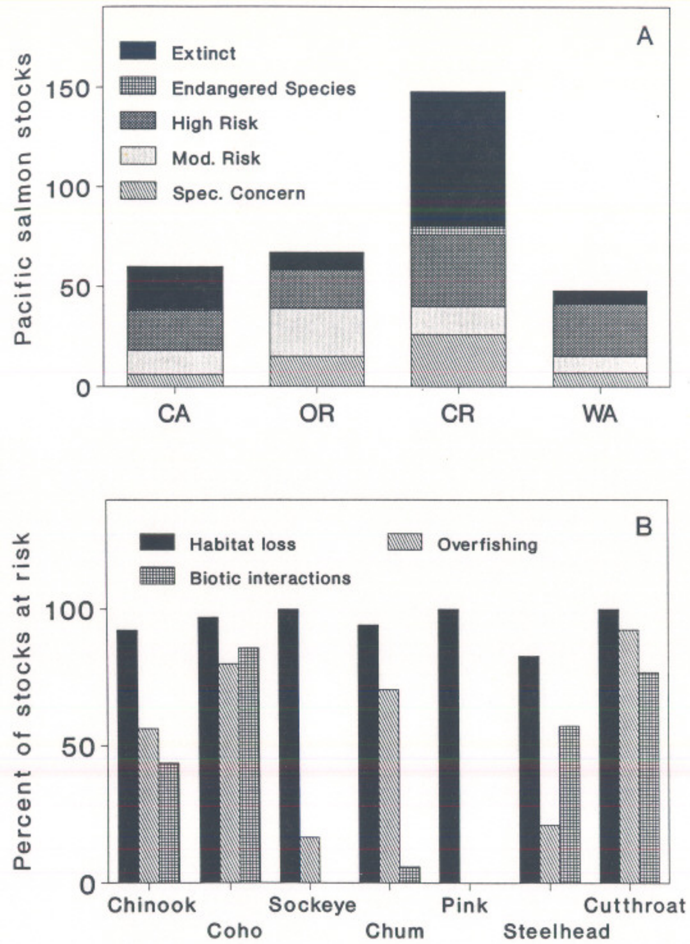


FIGURE 7.9. (A) Distribution of stocks of anadromous Pacific salmon (*Oncorhynchus*) in different extinction risk categories within various regions of the Pacific coast. (B) The percentage of stocks in which habitat damage, overfishing, and harmful biotic interactions have been implicated in declines of stock abundance. Data from Nehlsen et al. (1991).

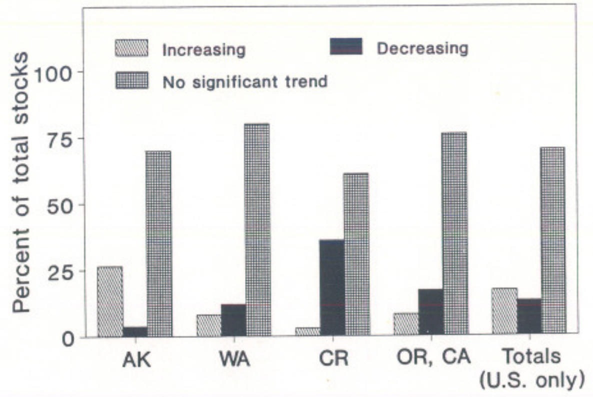
7.9B) was associated with declines of each of the seven Pacific salmon examined by Nehlsen et al. (1991). More important, for each species habitat destruction was accorded a significance equal to or greater than either overfishing or the negative effects of biotic interactions such as competition, predation, and disease. No attempt was made to separate habitat loss caused by forest management from losses caused by other land use activities; however, an important conclusion from this analysis was that declines in the abundance of many native stocks were caused, at least in part, by the cumulative effects of freshwater habitat damage.

Regionally, not all species are declining at the same rate, and in some areas populations are actually increasing (Konkel and McIntyre 1987). For example, more Alaska populations of chinook salmon, coho salmon, and steelhead for which long-term escapement records are available have significantly increased than have decreased between 1968 and 1984 (Figure 7.10). In other regions, notably the Columbia River Basin, declining stocks far outnumber increasing stocks for all salmonid species. On a coastwide basis the majority of chinook, coho, chum (*Oncorhynchus keta*), and steelhead populations examined by Konkel and McIntyre (1987) were found to have demonstrated no statistically significant trends over the period of evaluation, but there were more declining stocks than increasing stocks of each species. Again, the main conclusion was that a number of important wild populations of anadromous salmonids have declined over the last two decades.

Trends in Single Species Abundance in Relation to Forest Management Practices

Difficulties in ascribing long-term declines in populations of Pacific Northwest fishes, particularly anadromous salmonids, to specific management actions or to the combined effects of multiple activities indicate that a change in the abundance of a single species may not be a useful measure of the cumulative effects of forest practices on fish populations in a river system. We are inclined to reject the idea that trends in designated "indicator species" can be used to gauge the cumulative effects of forestry operations, unless it can be convincingly demonstrated that those species are not impacted by other land use activities or overfishing. Even where it can be shown that stocks are undergoing severe declines, it is usually impossible to determine with reasonable certainty the relative effects of habitat degradation, fishing pressure, and biotic interactions such as competition, predation, and disease. For most populations, declines have resulted from a combination of several factors, the relative importance of which may change from year to year.

A potentially more powerful approach is to examine the relationship between forestry-related habitat changes and the structure of fish communities in streams and rivers. Community-level studies have been used successfully



Region	Chinook UP	Chinook DN	Coho UP	Coho DN	Chum UP	Chum DN	Steelhead UP	Steelhead DN
Alaska	43%	1%	15%	11%	3%	13%	17%	0%
Coastal WA	12%	32%	9%	0%	6%	15%		
Columbia R. Basin	3%	39%	0%	45%	0%	33%	8%	25%
Coastal OR, CA	19%	12%	2%	17%	11%	11%	20%	40%
U.S. Total	20%	22%	6%	17%	4%	14%	11%	23%

FIGURE 7.10. Trends in the abundance of wild stocks of chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), chum salmon (*O. keta*), and steelhead (*O. mykiss*) from river systems along the Pacific coast. UP = percentage of stocks significantly increasing, DN = percentage of stocks significantly decreasing. Data from Konkel and McIntyre (1987).

elsewhere in North America to detect the effects of persistent environmental disturbances on fish assemblages (Karr 1981, Berkman and Rabeni 1987, several papers in Matthews and Heins 1987, Fausch et al. 1990) but there have been relatively few studies of this type in the Pacific Northwest (Li et al. 1987, Pearsons et al. 1992). One reason for lack of information on forestry-related environmental disturbance and fish communities in the region has been the almost complete absence of studies on the relationship between forest management and nonsalmonid species. A second reason is that fish assemblages are structured according to functional groups that include trophic, habitat, or reproductive guilds (Berkman and Rabeni 1987), but little is known about the food habits and spawning behavior of many native Pacific Northwest species. A third reason is that, unfortunately, comprehensive faunal surveys have not been carried out in most Pacific Northwest rivers. A fourth reason is that many Pacific Northwest river systems are faunally depauperate due to recent glaciation and lack of zoogeographic access (Moyle and Herbold 1987), and this has been a barrier to the successful application of indices of community integrity, which require at least moderate species richness (Fausch et al. 1990). Identification of significant changes in fish community structure often requires sampling designs that extend beyond intensive studies of limited stream reaches (Angermeier 1987). Sampling approaches to fish community characterization have been reviewed by Fausch et al. (1990), who recommended multivariate statistical approaches to characterizing community structure (species richness) and function (guild analysis).

Although characterization of fish communities in streams and rivers in the Pacific Northwest has rarely been attempted, community organization may yield important clues about the nature and long-term effects of cumulative environmental change. For example, temperature increases resulting from forest canopy removal can alter the outcome of competitive interactions between cyprinids and salmonids, resulting in a redistribution of species dominance along thermal gradients (Reeves et al. 1987). Deposition of fine sediment may reduce fish species diversity by eliminating spawning habitat and altering invertebrate food resources (Karr and Schlosser 1978, Karr et al. 1985, Berkman and Rabeni 1987). Separate age classes of many salmonids have particular habitat requirements (Bustard and Narver 1975, Bisson et al. 1988, Moore and Gregory 1988*a*) and can be considered functionally distinct members of stream communities (Schlosser 1991); therefore, they are often unequally impacted by habitat alteration. Stream fishes tend to be habitat specialists (Gorman and Karr 1978) and are affected by an increase or loss of preferred habitat types (Moore and Gregory 1988*b*), but a transformation of one habitat type to another (e.g., conversion of a pool to a riffle) may not lead to an overall reduction in fish density; rather, the effect may be expressed as a shift in species or age class composition (Schlosser 1991). Although factors other than anthropogenic habitat disturbance can influence community structure (Grossman et al. 1982, 1990; Herbold 1984, Schlosser 1985, Schlosser 1987, Power 1990), persistent changes in stream habitat are more likely to be detectable through analysis of fish assemblages than through interpretation of long-term trends in the abundance of individual species. In the Pacific Northwest, we conclude that the strong research emphasis on salmon and trout to the exclusion of other fishes has become a significant obstacle to defining the cumulative effects of forest management on stream ecosystems.

To fully understand how stream fish communities might be altered by cumulative habitat change, it is necessary to identify general environmental trends associated with forestry operations. In the following section, evidence that past forestry practices have led to simplification of stream channels and truncation of natural linkages between streams and riparian zones is reviewed. We conclude with conceptual recommendations for future BMPs that will provide a basis for protecting these complex linkages in a way that conserves the natural biodiversity of stream communities.

Past Forest Management Practices

Specific changes in stream environment caused by past forest practices in the Pacific Northwest vary according to logging and reforestation history, watershed geology, regional climate, and the degree of protection given to riparian zones during management activities. The one change that appears to be consistent over all areas in which the effects of forest management on streams have been studied is a trend toward simplification of stream channels and a loss of habitat complexity (Bisson and Sedell 1984, Grant 1986). This trend has resulted from a combination of management and regulatory actions, and represents a cumulative effect upon which there appears to be general scientific agreement (Hicks et al. 1991, Sedell and Beschta 1991). Simplification of stream channels involves loss of hydraulic complexity (i.e., caused by variation in current velocity and depth, and structural obstructions to flow), elimination of physical and biological interactions between a stream and its floodplain (see Naiman et al., this volume), reduction of structures that serve as cover from predators, an increase in the dominance of one particular substrate type, and loss of sediment and organic matter storage capacity (Sullivan et al. 1987).

Stream simplification and loss of complexity is perhaps most evident in the changes in frequency, size, and location of different types of habitat units within the channel. Over the last decade, physically based systems of channel unit classification (Bisson et al. 1982, Frissell et al. 1986, Sullivan 1986, Cupp 1989, Grant et al. 1990) have increased our ability to resolve stream morphology at a scale that is meaningful to understanding the distribution and abundance of fishes (Bisson et al. 1988). The most pervasive change has been a reduction in the frequency and size of pools, particularly large plunge and scour pools that constitute preferred habitat of certain species and age classes (Bisson and Sedell 1984, Sedell and Everest 1991). There have been two principal causes of pool reduction in Pacific Northwest streams: the filling of pools by sediment (Megahan 1982) and the loss of pool-forming structures such as boulders and large woody debris (Bryant 1980, Sullivan et al. 1987, Meehan 1991). Although both causes have been directly related to forest management activities, woody debris removal has been practiced in streams and rivers for navigation improvement, to aid water-based log transport, and to promote fish passage (Sedell and Luchessa 1982).

Examples of studies that have demonstrated relationships between forestry and pool frequency are shown in Figure 7.11. In ten Oregon coastal streams, Hicks (1990) found that the number of scour pools associated with large woody debris decreased in proportion to the percentage of the drainage basin that had been logged, and that the decrease occurred in drainages with both basalt and sandstone parent rock. Bilby and Ward (1991) found that streams in old-growth forests held more pools for a given channel width than streams in clearcuts or in second-growth forests. Bisson et al. (1987) cited numerous studies that have associated declines in fish abundance with loss of pools and woody debris in Pacific Northwest streams.

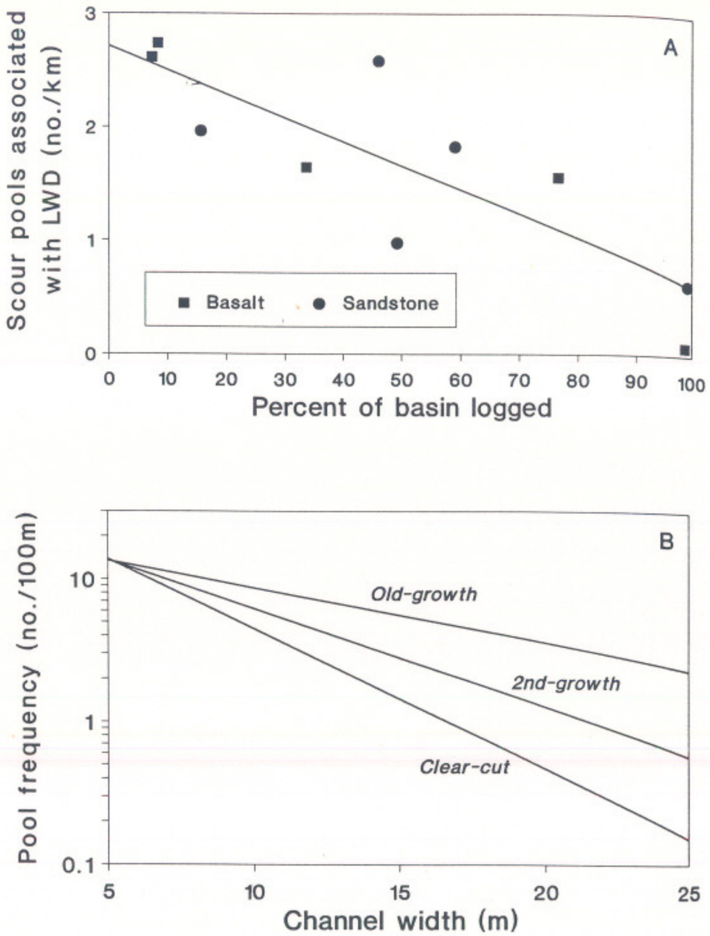


FIGURE 7.11. (A) Frequency of pools associated with large woody debris in ten Oregon coastal streams with different logging histories. Redrawn from Hicks (1990). (B) Relationship between pool frequency and stream channel width in old-growth, second-growth, and clearcut streams in western Washington. Redrawn from Bilby and Ward (1991) with permission of the *Canadian Journal of Fisheries and Aquatic Sciences*.

In addition to reductions in the number and size of large scour and plunge pools, forestry and other land use practices have led to stream channel simplification by eliminating edge habitat along stream margins. Complex channel margins are used extensively by underyearling fishes of many species, as these areas provide breaks in the stream current for resting and feeding. Edge habitat can be destroyed by reducing the abundance of large flow obstructions such as logs and boulders near the stream's margin and by isolating a stream from its floodplain through channelization, streambank stabilization, and other measures designed to confine the flow to a single channel. Experimental removal of eddy pools and backwater areas along the margin of a small stream greatly reduced the carrying capacity for young-of-the-year trout (Moore and Gregory 1988*b*). Complexity of channel margins is further reduced when management activities cause streambanks to collapse or when timber harvest in riparian zones results in destruction of root systems and eliminates future sources of large woody debris (Toews and Moore 1982, Bryant 1983).

Long-term reductions in the supply of large woody debris as the result of timber harvest have affected other important processes within stream ecosystems (Harmon et al. 1986). Small headwater streams serve as temporary storage sites for both sediment and fine particulate organic matter (FPOM) from the surrounding forest (Keller and Swanson 1979, Triska and Cromack 1980). Loss of sediment and FPOM storage capacity in small streams caused by reduced debris frequency greatly lessens the capacity of the streams to biologically process organic matter and ultimately make the energy of terrestrial plant materials available to fishes (Triska et al. 1982, 1984, Gregory et al. 1987). Because their storage and processing capacities are greatly diminished, streams with simplified channels route sediment and organic matter much more quickly downstream to larger streams (Naiman and Sedell 1980, Sedell and Beschta 1991). In some cases, rapid transport of sediment can overwhelm larger stream systems (Megahan and Nowlin 1976; Megahan, this volume), resulting in lower biological productivity (Platts and Megahan 1975) and reduced diversity of species requiring clean gravel substrate for spawning (Berkman and Rabeni 1987).

Large events of a catastrophic nature such as major floods or landslides can trigger debris flows that cause extensive scouring and simplification of headwater streams (Benda 1990, Lamberti et al. 1991). In the Queen Charlotte Islands, British Columbia, Rood (1984) found that the frequency of landslides increased over 30-fold after logging on geologically unstable hillslopes, and the frequency of debris flows in logged and roaded areas increased 40–76 times. Tripp and Poulin (1986*a*) investigated morphological changes in streams that had undergone massive debris flows and found that average pool depth was reduced by 20–24%, average pool area was reduced by 38–45%, large woody debris in the channel was reduced by 57%, and undercut bank cover was reduced by 76%. They found that riffle area was increased by an average of 47–57% and average channel width increased

48–77%. Large debris flows have also been found to strongly influence the species composition and structural characteristics of riparian vegetation in western Oregon (Gecy and Wilson 1990). Lamberti et al. (1991) have suggested that debris flows diminish stream ecosystem stability, leading to large annual fluctuations in fish populations.

Simplification of streams and rivers has also resulted from other types of land use activities, including agriculture (Schlosser 1982, Karr et al. 1983), grazing (Chapman and Knudsen 1980, Platts and Nelson 1985), and urban development (Leidy and Fiedler 1985). Habitat changes associated with these land uses are usually located downstream from forestry operations, but the net effect of combined management activities can be extensive loss of channel complexity from headwater streams to river mouths (Sedell et al. 1988, Gonor et al. 1988).

Simplified Streams and Biodiversity

The extent to which habitat simplification has led to an increase in the number of Pacific salmon stocks either extinct or at risk of extinction (Nehlsen et al. 1991) is not known. However, the large number of cases in which habitat degradation has been cited as a factor in stock declines (Figure 7.9B) suggests that loss of critical habitat has played an important role, particularly with species spending extended periods in fresh water and undertaking extensive seasonal movements within the drainage system. Severe reductions in stock abundance or outright extinctions have led to losses of genetic diversity within species, with one possible outcome being limited ability to maintain viable populations under unusual conditions. Potential consequences of the interactions between habitat simplification (Schlosser 1991) and genetic "simplification," whether due to loss of locally adapted stocks or to fish culture practices such as widespread planting of fry from a single hatchery population (Hilborn 1992), have not been adequately investigated.

At present there is little direct evidence that diversity of fishes has been reduced in simplified streams in the Pacific Northwest, because few studies have attempted to relate fish community composition to habitat characteristics. Most investigations have emphasized only salmon and trout; however, habitat simplification has been shown to alter the proportions of different salmonid species and age classes. Bisson and Sedell (1984) found that streams in western Washington in which logging debris had been removed had fewer pools and longer riffles than streams in old-growth forests. Although total salmonid biomass was greater in logged and cleaned streams than in old-growth sites, the communities were dominated by underyearling trout and there were proportionately fewer age one and older trout. Additional data of P. Bisson (cited by Sullivan et al. 1987, Hicks et al. 1991, and Naiman et al., this volume) indicate that conversion of pool to riffle habitat favors species and age classes that utilize riffles at the expense of those that prefer pools. The latter habitat type is usually preferred by juvenile coho salmon

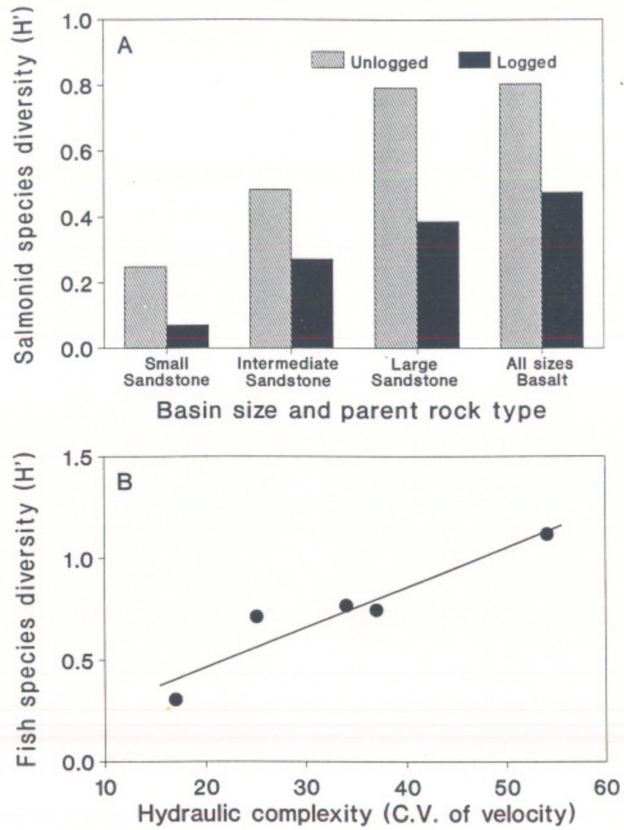


FIGURE 7.12. (A) Diversity of salmonid fishes expressed by H', the Shannon-Wiener index (Pielou 1969), in logged and unlogged Oregon coastal streams with different parent rock types. Unpublished data of G.H. Reeves, F.H. Everest, and J.R. Sedell. (B) Relationship between fish species diversity (H') and water current variability (coefficient of variation of velocity as measured by an inert dye tracer) in pools possessing different levels of hydraulic complexity in Huckleberry Creek, Washington. Different salmonid age classes were treated as separate taxa to reflect differences in habitat utilization (Bisson et al. 1988). Unpublished data of P.A. Bisson and B.R. Fransen.

and by older, larger trout. In a survey of Oregon coastal streams, G. Reeves, F. Everest, and J. Sedell (unpublished data) found that diversity of salmonids was lower in streams in which timber harvest had taken place than in similar streams in unlogged basins (Figure 7.12A). Diversity was lower in harvested sites regardless of drainage basin size and parent rock type.

Geologic conditions may influence the recovery of salmonid diversity after logging. Hicks (1990) suggested that diversity would be more likely to return to normal levels after logging in areas dominated by basalt than in areas dominated by sandstone. He found that basalt streams contained larger substrate, were inherently more complex, and were more resistant to summer flow reductions than sandstone streams. Basalt streams in Oregon that had previously been logged possessed higher salmonid species richness and evenness than did previously logged streams in sandstone basins.

Bisson and Fransen (unpublished data) estimated the hydraulic complexity of different habitat types in a small western Washington stream using controlled releases of an inert tracer dye to measure variability in current velocity. They found that habitat units with greater flow variability contained more diverse fish assemblages than habitat types with uniform flow characteristics (Figure 7.12B). The more complex habitats possessed woody debris, boulders, and undercut banks, while less complex habitat units possessed few structural obstructions to streamflow. This study included all fish species present in the stream and treated different salmonid age classes as separate taxa.

Several investigations in the Pacific Northwest have shown that timber harvest can result in increased salmonid productivity, chiefly by enhancing autotrophic production within streams (Murphy et al. 1981, Hawkins et al. 1983, Bilby and Bisson 1987, 1992). It is important to note that habitat simplification does not necessarily cause total fish production to decline. Rather, loss of complexity, if accompanied by increased light and dissolved nutrients, is likely to result in productivity increases concentrated in only a few taxa that directly benefit from the changes (Schlosser 1991). Where there are no potential competitors present, younger age classes of some salmonids appear to prosper in shallow, riffle-dominated streams with open vegetative canopies (Bisson and Sedell 1984, Bisson et al. 1988, Lamberti et al. 1991). However, where potential competitors exist, underyearling salmonids may be at a competitive disadvantage to species that are better adapted to warmer water that accompanies forest canopy removal (Reeves et al. 1987). The result may be loss of salmonid species and a large increase in the abundance of certain nonsalmonids, particularly cyprinids.

A similar pattern has been observed in the structure of aquatic invertebrate communities after logging. Erman et al. (1977) found that the density of benthic invertebrates was greater in northern California streams logged without buffer strips than in unlogged streams. Invertebrate species diversity, however, was greater in the unlogged streams (Figure 7.13). A few taxa (Ephemeroptera, Chironomidae) were much more abundant after logging while others disappeared from the streams. Results suggested that the pattern of increased production of a few taxa accompanied by a reduction in overall biodiversity may be common to all consumer trophic levels in streams where habitat has been simplified but light and nutrients are more plentiful (Gregory et al. 1987). Bilby and Bisson (1992) found that logging riparian veg-

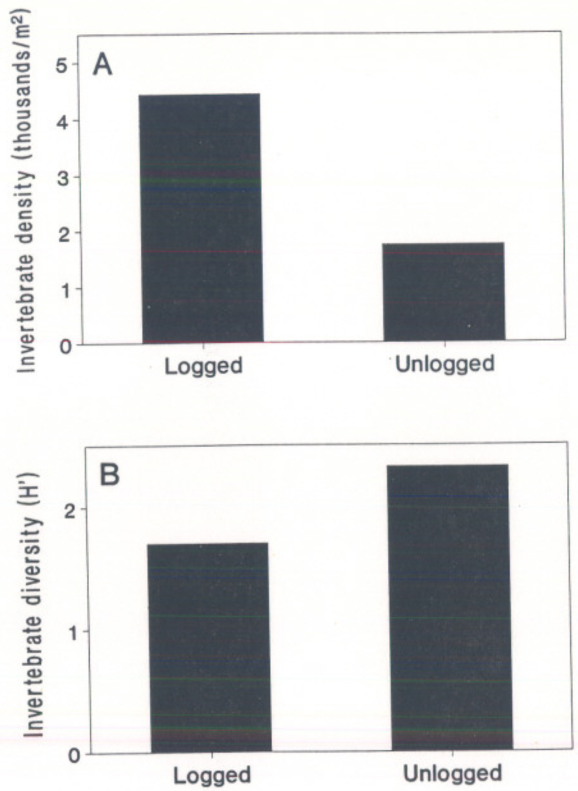


FIGURE 7.13. Density (A) and diversity (B) of aquatic invertebrates in logged and unlogged streams in northern California. Data from Erman et al. (1977).

etation also led to reduced diversity in the forms of terrestrial organic matter (leaves, needles, and other plant materials) entering a small stream, but that increased autotrophic production in the open channel led to greater availability of invertebrate species that were directly utilized by salmonids.

Restoration of Healthy, Productive River Systems

Managing for Large Woody Debris

Many streams in the Pacific Northwest were logged to the edge of the channel prior to enactment of the first state and provincial forest practices laws in the early 1970s. Because initial stream protection regulations were concerned primarily with temperature and erosion control, early BMP guidelines called for leaving enough vegetation next to streams to protect streambank integrity and provide shade. Forest managers usually complied with the new guidelines by leaving buffer strips of unmerchantable trees, chiefly hardwoods, along streams. These buffer strips were helpful in controlling streambank erosion and providing shade during the summer, but postlogging riparian zones differed in vegetative composition from what had existed originally (Kauffman 1988). New riparian zones in watersheds along the Pacific coast and on the west slope of the Cascade Range where logging to the stream edge had occurred were often dominated by red alder (*Alnus rubra*), with relatively few coniferous species present (Oliver and Hinckley 1987, Gregory et al. 1991). Forest management along with other land use activities such as grazing has thus transformed the structure and composition of riparian zones throughout entire river basins (Oakley et al. 1985, Kauffman 1988).

The transformation of riparian zones by previous forest practices has altered and simplified the form and inputs of organic material to streams (Gregory et al. 1987, Schlosser 1991). Not all of these changes have been detrimental; for example, nitrogen-rich alder leaves are an excellent food source for invertebrate shredders (Triska et al. 1984). However, one of the most significant changes has been a reduction in the input rate of large conifer debris (Swanson et al. 1976, Swanson et al. 1982). Removal of the sources of future large conifer debris combined with stream clearance programs for fish passage has left many streams severely lacking in these very important storage and roughness components (Harmon et al. 1986, Bisson et al. 1987, Bilby 1988).

Recent calls for revision of forest practice laws have recognized the importance of identifying and protecting conifers in managed riparian zones to provide a future source of large woody debris (Murphy and Koski 1989, Bilby and Wasserman 1989, Robison and Beschta 1990, Gregory et al. 1991) as well as for wildlife habitat (Raedeke 1988). A survey of western Washington riparian zones after enactment of the Timber, Fish and Wildlife agreement and incorporation of its provisions into the Washington Forest Practices Act in 1989 has shown that, while red alder remains the dominant tree species, about one-third of the remaining trees are now coniferous species (Figure 7.14). Similar measures have been adopted by other western states (Oregon Department of Forestry 1987) and in internal forestry planning and operating guidelines within the U.S. Forest Service (Hemstrom 1989), Bureau of Land Management (Oakley 1988), and Indian tribes (Bradley 1988). The new riparian zone prescriptions attempt to promote riparian communities like those in naturally (not anthropogenically) disturbed watersheds (Agee 1988) and will differ considerably from earlier managed riparian zones consisting of nonmerchantable timber.

Water Quality Standards

Attempts to regulate cumulative effects in forested watersheds have often relied on determining if water quality standards, here taken broadly to mean fixed levels of chemical constituents, temperature, water clarity, and both

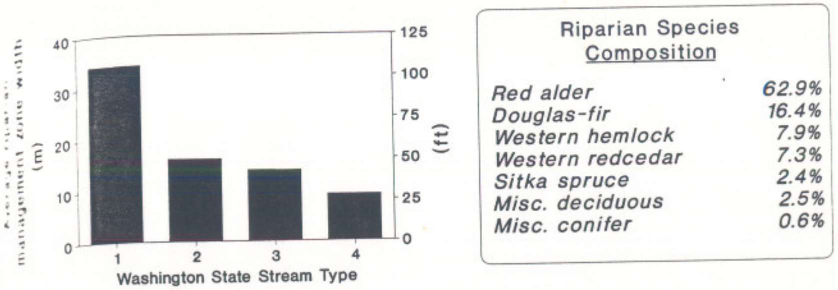


FIGURE 7.14. Average width and tree species composition of riparian vegetation left during logging after the revised 1989 Washington state riparian management requirements went into effect. The Washington State Stream Type refers to a regulatory classification system where Type 1 streams are large and Type 4 streams are small; generally, Type 1–3 streams contain fishes while Type 4 streams do not. Data from Quinault Indian Nation (1990).

suspended and deposited sediment, have been exceeded as the result of land management activities (Coats and Miller 1981). We wish to emphasize that while individual water quality standards, usually expressed as potentially harmful threshold levels, may serve useful functions as measures of relative risk to certain life history stages of individual species, their application to field situations in forested watersheds of the Pacific Northwest has been largely unsuccessful at either diagnosing or preventing cumulative environmental change. In many instances, difficulties have resulted from attempting to establish baseline levels of the parameter of interest, from attempting to extrapolate from laboratory experiments to field situations, or from attempting to extrapolate findings from one region to another (Burns 1991).

Although some water quality parameters are more easily quantified than others, ease of measurement does not guarantee predictable biological responses to cumulative disturbance. Quite often the characteristics of the stream system will influence the degree of impact. For example, Baltz et al. (1987) noted that cumulative temperature changes influenced microhabitat selection by trout in a California stream, but Modde et al. (1986) did not observe a clear impact of temperature modification on a trout population in a North Dakota stream. Berman and Quinn (1991) found that adult spring chinook salmon (*O. tshawytscha*) were able to locate cool water pockets for holding in a river during warm summer months. These cool water areas were generally undetectable from the streambank and would have gone unnoticed and unmeasured had the fish not been fitted with temperature-sensitive radio transmitters. The finding that adult salmon were capable of behavioral thermoregulation suggests that impacts of cumulative temperature change may be mediated by the presence of thermal refugia; therefore, understanding the distribution and abundance of cool water areas becomes critical to predicting the response of fishes and other organisms to temperature increases.

Although numerous laboratory studies have defined negative relationships between the percentage of fine sediment in spawning gravels and the survival of salmonid eggs and alevins, a thorough review of laboratory and field studies (Chapman 1988) concluded that extrapolation of laboratory results to natural stream conditions was currently impossible without better sampling techniques, and that establishing thresholds was not yet feasible without more carefully controlled field experimentation. These examples highlight a few of the formidable sampling problems associated with trying to relate measurements of individual parameters to the abundance of species of interest. Other problems encountered in applying fixed thresholds include adaptation of populations to local conditions (e.g., sediment rich glacially fed streams) and the possibility that harmful threshold levels may vary among species or even among life history stages of a single species (Davis 1975, Noggle 1978). We further believe that water quality standards and thresholds considered individually are not readily applicable to the goal of protecting biodiversity. On the other hand, we support the development of appropriate measures of ecosystem health focused on defining goals for the maintenance of important physical and biological processes that preserve the integrity of stream communities. With the exception of recent regulatory guidelines for maintaining trees in riparian zones for future recruitment of large woody debris, such goals remain largely unexplored.

BMPs and Stream Enhancement Programs

Restoration of stream habitat damaged by natural catastrophes, past forest practices, and other management activities has been undertaken by many federal, provincial, state, and tribal organizations. Some stream enhancement programs, including those of the Forest Service, have been extensively funded and cover large geographical areas, while others have been much more modest and local in scope. What have results from these programs taught us about the response of fish populations to habitat improvement, and what conclusions can we draw with regard to the prospects for recovery of altered stream systems with improved riparian management?

First, we have learned that correction of large and obvious problems can lead to measurable increases in population size, but recovery of stocks that are heavily fished is likely to take many years. In one of the longest continuous records of relative stock abundance on the Pacific coast, the abundance of sockeye salmon (*O. nerka*) in the Fraser River, British Columbia, declined precipitously when large rock slides created nearly impassable conditions in the Hell's Gate portion of the Fraser River canyon in 1913 and 1914, and splash dams in the Adams River prevented access to important spawning areas (Thompson 1945). Sockeye catches remained at only a small fraction of historical levels until completion of the Hell's Gate fishways in

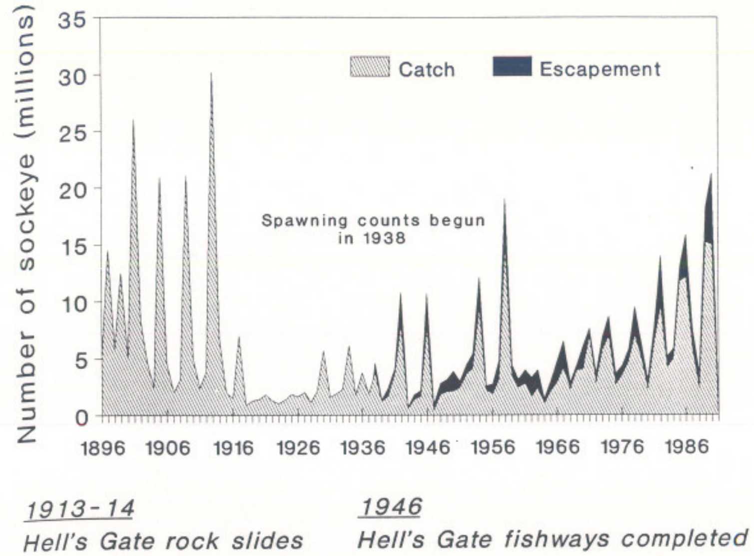


FIGURE 7.15. Catch and escapement of adult sockeye salmon in the Fraser River, British Columbia. Data of Thompson (1945), Ricker (1987), and annual reports of the International Pacific Salmon Fisheries Commission and the Pacific Salmon Commission.

1946, after which four-year cyclic peaks in the sockeye run gradually but steadily rose (Figure 7.15) despite a high exploitation rate. By 1989, sockeye catch plus escapement had almost reached the historically large run sizes of the early 1900s. This example demonstrated that a single restoration project could make a significant contribution toward rebuilding a depleted stock, but it has taken approximately 50 years for the population to recover.

Second, we have learned that carefully regulated fishing may be required to accompany habitat restoration in order to see an improvement in population abundance, particularly where populations are depressed to very low levels. In the Yakima River of central Washington, the population of spring chinook salmon declined from an estimated run of about 200,000 adults to 12,000 in 1920 and then to near extinction in the mid-1970s (Figure 7.16). There were a number of causes of the decline, but among the most important over the last 50 years have been the commercial and sport chinook salmon fisheries in the Columbia and Yakima rivers, completion of four hydroelectric dams downstream on the Columbia River between 1936 and 1968, and extensive water withdrawals from the Yakima River for crop irrigation (Fast et al. 1988). Droughts during the 1976 and 1977 summers further depleted the number of successfully spawning adults. When it became clear

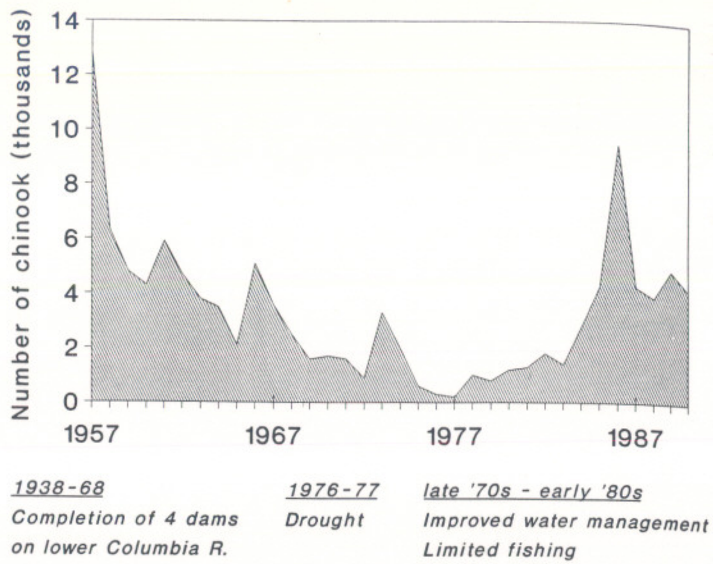


FIGURE 7.16. Estimated number of adult spring chinook salmon entering the Yakima River, Washington, from 1957 to 1990. Data from Fast et al. (1988) and S. Parker (pers. comm.).

that the population faced imminent extinction, severe limitations were placed on chinook salmon harvest in the river and an aggressive program of improved streamflow management was implemented. The population has since begun to increase but at a gradual rate. Fishery managers in the Yakima Basin believe that initial recovery of the run would not have been possible without both habitat improvement and reduced fishing intensity (Fast et al. 1988). This is a key point, because if survival in fresh water is reduced and marine survival remains fairly constant, fishing must be limited if stocks are to recover. However, even where freshwater survival has improved owing to habitat enhancement, continued fishing may lead to reductions in body size and, consequently, fecundity (Ricker 1980). Thus it is possible for exploited stocks to continue to decline due to reduced egg production per female in situations in which survival in streams and rivers has increased but average size of adults has become smaller. Complex interactions such as these between freshwater survival, marine survival, fishing intensity, and methods of harvest (i.e., seining, trolling, gillnetting) have surely contributed to the lengthy recovery period shown by several stocks in the Pacific Northwest.

Third, we have learned that adding large structural roughness elements such as logs or boulders to small streams can increase pool frequency and

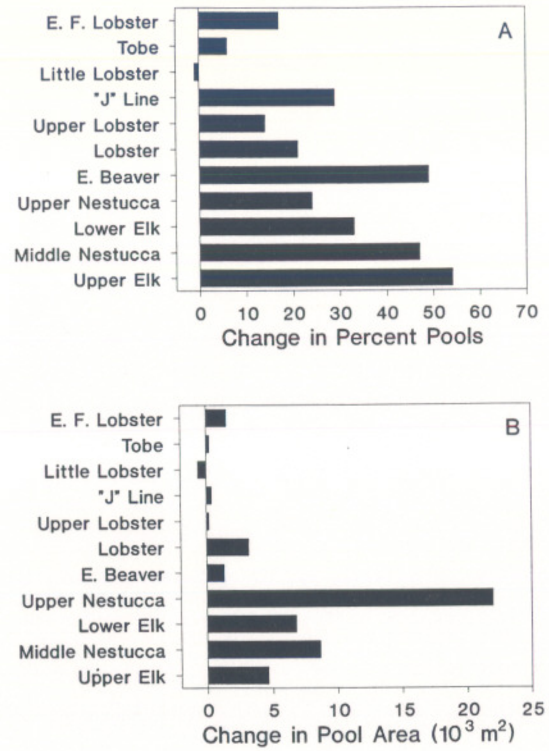


FIGURE 7.17. (A) Change in the percentage of stream area in pool habitat after addition of logs, boulders, and gabions in 11 Oregon coastal streams. (B) Change in the total area of pool habitat after enhancement. Data from House et al. (1989).

create additional habitat diversity (Figure 7.17) but the costs of such projects often limit habitat restoration to selected stream reaches, and would be prohibitively expensive at the scale of an entire drainage. Structure additions to small streams usually involve boulder, large woody debris, or gabion placement, and the purpose is usually to create large rearing pools or to trap gravel suitable for spawning (House and Boehne 1985). In terms of numbers of projects, structure additions constitute the most common form of stream enhancement in the Pacific Northwest. Yet the amount of stream habitat improved by these projects still represents only a tiny fraction of the total length of fish-bearing streams and rivers in the region (Sedell and Beschta 1991). Of course, not all streams need additional roughness elements, but the scale upon which structural enhancement of small streams has been practiced has been insufficient in most cases to determine if these activities significantly elevate total population abundance (also see Peterman 1990). This conclusion suggests that BMPs for land management will need to be implemented throughout entire drainages in order to effectively improve habitat and increase fish populations.

Finally, we have learned that some of the most effective enhancement projects were undertaken only after research had identified the most probable factor limiting the production of a fish species of interest (Reeves et al. 1989). In the Clearwater River on Washington's Olympic Peninsula, life history studies of juvenile coho salmon revealed that the area of riverine ponds suitable for overwintering was the habitat resource most likely to limit the production of coho salmon smolts from this coastal rain forest river basin (Peterson 1982, Peterson and Reid 1984, Cederholm and Reid 1987). To increase the amount of available off-channel pond habitat, a series of small interconnected ponds was created along an alluvial terrace tributary and access to and from the lower river was maintained during winter and spring (Cederholm et al. 1988). Several thousand juvenile coho entered the ponds each winter (Figure 7.18), and both numbers of smolts and average smolt size increased over preenhancement levels. Cederholm et al. (1988) estimated that this single project, on only one tributary, resulted in a 2.8% increase in total smolt yield from the basin, and that the benefit-cost ratio of the project had been favorable. The implication of successful enhancement projects for improved BMPs is that management requirements should be sufficiently flexible to recognize the need for special protection or possibly active environmental manipulation in areas that have been designated as critical habitat or "hot spots" for populations or communities. Identification of these critical areas should be a high priority in river basin habitat inventories (Hankin and Reeves 1988).

Patience, Patience, Patience

We conclude this review with the observation that restoration of naturally complex channels and unimpeded connections between streams and riparian zones is a formidable task requiring an unprecedented level of cooperation and willingness to alter current land use practices (Gregory et al. 1991, Sedell and Beschta 1991). Although we have freely used the term *best* management practice, we concur with Petts (1990) that management alternatives will continue to evolve as our knowledge increases. The BMPs of tomorrow will be better than those of today, and there is every reason to believe that there will always be room for environmental improvement. Recovery of habitat complexity and biological diversity in streams, even with the most benign of land use influences, will follow a trajectory dictated by landscape patterns and natural climatic regimes within the region (Schlosser 1991). Coupled with continued industrial development and fishing pressure, recovery rates of naturally produced aquatic resources to some desired but usually unspecified levels will not be rapid in any event. We wonder, for example,

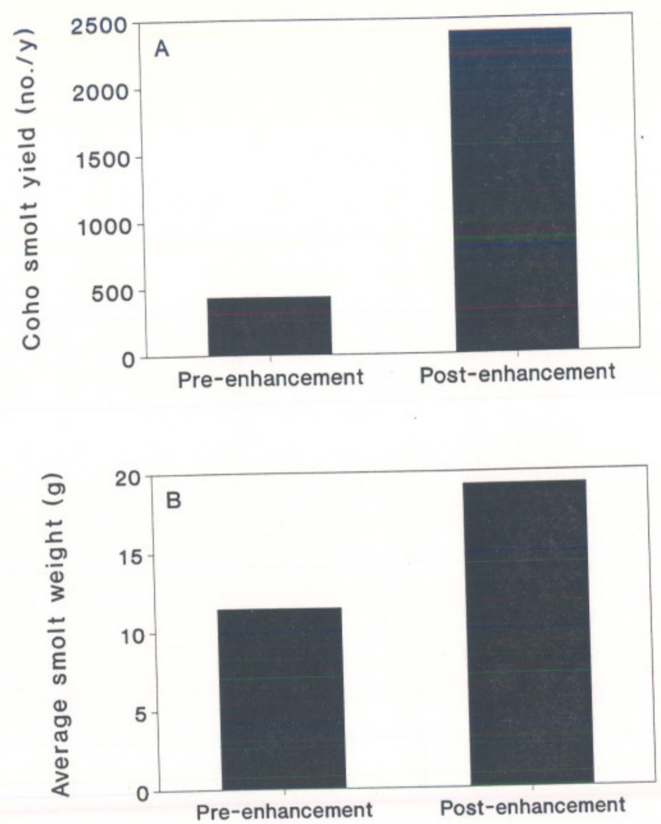


FIGURE 7.18. Average number of coho salmon smolts and average smolt weight after creation of additional winter rearing habitat in a small tributary of the Clearwater River, Washington. Data from Cederholm et al. (1988).

if programs intended to double the number of salmon returning to the Columbia River by the year 2000 are based on realistic assumptions and expectations.

But some of the most valuable aquatic resources in the Pacific Northwest are in jeopardy, and decisive action is needed. In many instances the need to take decisive action has led management organizations to adopt restoration approaches based more on *mitigation* of losses than on *protection or restoration* of natural ecological processes that have created and maintained diverse and productive stream habitat. Mitigation approaches seek rapid increases in numbers of harvestable fish rather than investments in long-term natural productivity that yield gradual but sustained improvement. However well-intentioned, many hatchery and stream enhancement projects fall into this category. Mitigation is costly and its effectiveness is too often questionable; the results of years of cumulative environmental damage cannot easily be reversed in a short time. Within the last decade, ecologically based concepts that emphasize the importance of ecosystem complexity and biodiversity have begun to revolutionize forest management. Included in the new paradigm are techniques for maintaining key elements or "legacies" that provide the foundation for the establishment of natural forest communities. It is time for a similar revolution in the way streams and riparian zones are managed for protection of aquatic habitat. This new way of thinking about BMPs as procedures to preserve stream ecosystem integrity and not individual fish populations must begin with the realization that benefits of improved practices will not be immediate, but will require patience and a willingness to incorporate new knowledge as it becomes available.

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8

Sensitivity of the Regional Water Balance in the Columbia River Basin to Climate Variability: Application of a Spatially Distributed Water Balance Model

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Abstract

A one-dimensional water balance model was developed and used to simulate the water balance for the Columbia River Basin. The model was run over a 10 km digital elevation grid representing the U.S. portion of the basin. The regional water balance was calculated using a monthly time step for a relatively wet year (1972 water year), a relatively dry year (1977 water year), and a double ($2xCO_2$) climate scenario. Input data, spatially distributed over the grid, included precipitation, maximum soil moisture storage capacity, potential evapotranspiration (PET), and threshold baseflow. The model output provides spatially distributed surfaces of actual evapotranspiration (ET), runoff, and soil storage. Model performance was assessed by comparing modeled ET and runoff with the input precipitation data, and by comparing modeled runoff with measured runoff. The model reasonably partitions incoming precipitation to evapotranspiration and runoff. However, modeled total annual runoff was significantly less than measured runoff, primarily because precipitation is underestimated by the network of measurement stations and because of limitations associated with the interpolation procedure used to distribute the precipitation across the grid. Estimated precipitation is less than measured runoff, a physical impossibility. Under warmer $2xCO_2$ climate conditions (January 4.0°K warmer, July 6.5°K warmer), the model predicts that PET increases by about 80%, ET increases, and runoff and soil moisture decrease. Under these climate conditions, the distribution and composition of forests in the region would change dramatically, and water resources would become more limited.

Key words. Regional water balance, runoff, evapotranspiration, soil moisture, climate change.