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Salmonid-Habitat Relationships in the Western United States: a Review and Indexed Bibliography

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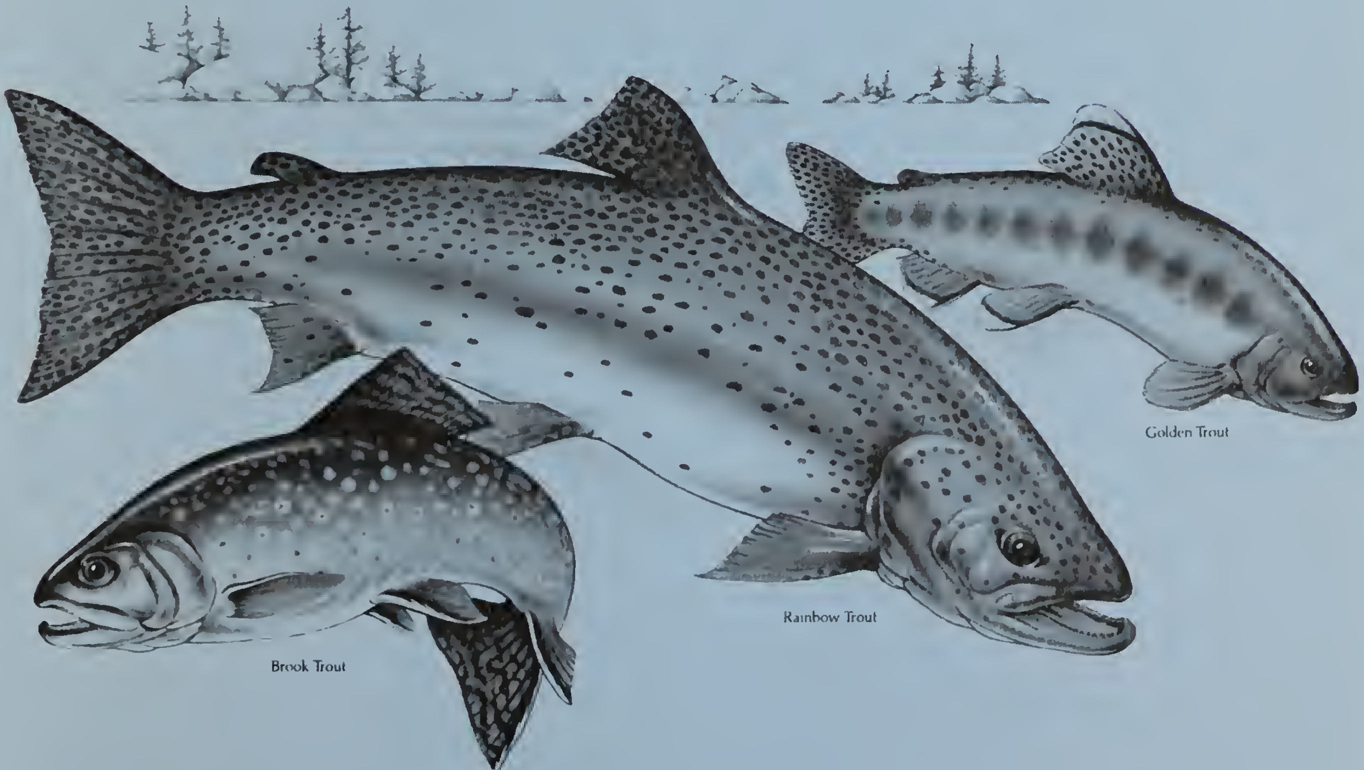

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Abstract

This report includes a general review and analysis of the literature summarizing the available information relevant to salmonid-habitat relationships, particularly as it pertains to the central Rocky Mountains. Also included is a comprehensive indexed bibliography.

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[1] *This report was prepared by personnel of Western Aquatics, Inc., 203 Grand Avenue, Laramie, Wyoming, by agreement with the Rocky Mountain Forest and Range Experiment Station, Laramie, Wyoming. Station headquarters is in Fort Collins, in cooperation with Colorado State University. Michael K. Young is currently a Research Fisheries Biologist with the Rocky Mountain Forest and Range Experiment Station, Laramie, Wyoming.*

Preface

Western lands administered by the USDA Forest Service contain many valuable natural resources, including prominent and highly valued salmonid fisheries. Land management activities can directly affect a large proportion of these fisheries, since most streams inhabited by these fish originate on national forests. But the present Forest Service fish habitat research program is small compared with the extent of the fishery resources on national forests and the technical knowledge required to manage these resources. Recognizing the necessity to augment ongoing research efforts and to expand management options, the Rocky Mountain Forest and Range Experiment Station of the Forest Service requested a literature review on salmonid-habitat relationships in the Western United States.

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EXECUTIVE SUMMARY

This report reviews and analyzes the literature to summarize the available information relevant to understanding salmonid-habitat relationships in the Western United States. Influences by many physical, chemical, and biological variables on salmonid habitats are summarized. Information considered includes the natural range of variation in these variables, how these variables interact, which of these variables are most important, and how forest use affects these natural relationships and impacts the quality of salmonid habitats. Some available management alternatives to mitigate these impacts are briefly discussed. Also, general guides to selected literature are provided on (1) techniques to restore, rehabilitate, and enhance salmonid habitats; and (2) methods to monitor and evaluate habitat quality for salmonids. Important concerns on the development and use of salmonid-habitat models are reviewed. Our discussion draws heavily on the findings and conclusions from many previous reviewers, while it also highlights additional results from some more recent studies. Overall, our literature review shows that proportionally little research has been completed in the central Rocky Mountain region on the relationship of salmonids to their habitats in national forests.

INTRODUCTION

Western lands administered by the United States Department of Agriculture, Forest Service contain many valuable natural resources, including prominent and highly valued salmonid fisheries. Land management activities by the Forest Service can directly affect a large proportion of these fisheries, since most streams inhabited by these fish originate on national forests. Presently, the Forest Service is charged with four minimum fish and wildlife objectives, which are to be met by focusing on habitat manipulation through forest management (Brouha 1987):

- Provide for diversity of plant and animal communities to meet multiple-use objectives and to preserve the diversity of tree species similar to that existing in the region.
- Maintain viable populations of all plant and animal species throughout their existing ranges.
- Accomplish feasible steps to recover threatened and endangered species.
- Maintain and improve habitat carrying capacity for species in public demand.

The existing Forest Service fish habitat research program is small compared with the extent of fisheries resources on national forests and the technical knowledge required to manage these resources. In no Forest Service experiment station are fish habitat needs being fully addressed, although the highly successful research programs at the Pacific Northwest, Pacific Southwest, and Intermountain Stations are helping to meet these needs. Recognizing the necessity to augment ongoing research efforts and to expand available management options, the Rocky Mountain Forest and Range Experiment Station requested a literature review on salmonid-habitat relationships in the Western United States.

The general review and analysis of the literature summarizes available information relevant to salmonid-habitat relationships, particularly as pertaining to the central Rocky Mountains. The common and scientific names for the salmonid species mentioned in the text are shown in table 1. While not all of these species have distributions within the central Rocky Mountains, they are included to help provide insight into the overall biology of salmonid species. Such insight can help guide the formation of appropriate hypotheses for future research in the central Rocky Mountains.

This report does not provide a totally comprehensive review of all work completed that potentially applies to trout-habitat relationships in forest streams of the central Rocky Mountains. This subject is indeed complex and, as later sections show, many good reviews and bibliographies have previously addressed specific facets of these relationships (e.g., Estes 1983, Everest et al. 1985a, Everest and Harr 1982, Leaf 1975a, Platts and McHenry 1988, Reiser et al. 1985, Wydoski et al. 1980). Instead, this report introduces and reviews the principal concerns regarding salmonid-habitat relationships. The discussion draws heavily from the findings and conclusions of the previous reviewers, while also highlighting some findings from more recent studies. Additional reference to much of the earlier literature can be found in these reviews. The indexed bibliography contains over 850 citations of potential interest to fish habitat biologists in the Western United States. These citations were examined but not necessarily cited during our work. (All literature cited is also contained in this bibliography, rather than in a separate literature cited section.)

ENVIRONMENTAL VARIABLES AFFECTING SALMONIDS AND THEIR MANAGEMENT

The quality of salmonid habitats is defined through interactions among a diversity of physical, chemical, and biological variables. Cultural developments have and will continue to affect these natural interactions. Our knowledge of the natural range of variation for these variables, how these variables interact, which of these variables are most important, and how our cultural activities affect these natural relationships and impact the quality of salmonid habitats is continuing to grow through extensive research efforts and experience. But many questions remain.

The following report summarizes much of our present knowledge on these relationships. In discussing specific sources of potential impact, we often provide brief considerations of some available management alternatives to mitigate these impacts on salmonid habitats. Two general guides are included: a guide to some of the techniques available for restoring, rehabilitating, and enhancing salmonid habitats; and a guide to methods available for monitoring and evaluating salmonid habitat quality. Then, the development and use of models for salmonid-habitat relationships are discussed.

NATURAL DETERMINANTS OF HABITAT QUALITY FOR SALMONIDS

Successful survival and reproduction by aquatic organisms is broadly defined by the physical structure in the environment, the quality of the surrounding waters, and interactions with other organisms. In streams of the central Rocky Mountains, the principal characteristics of environmental structure that influence salmonid abundance and population structure include riparian vegetation, channel morphology, streamflows, deposited sediment, and winter snow and ice accumulation. Important water quality characteristics include suspended sediment, temperature, pH, nutrients, and potentially toxic chemicals. Biological influences involve nutrient and energy cycles, interactions with invertebrates, competition with and predation by other fish, and predation by birds and mammals.

Riparian Vegetation

Riparian vegetation consists of nonaquatic vegetation that directly influences the stream (Meehan et al. 1977). Its influence on streams is inversely proportional to stream size, with first- to fourth-order streams tending to be strongly controlled by riparian vegetation (Meehan et al. 1977). Control operates through physical influences on channel structure and chemical influences on organic and inorganic contributions to the stream (Cummins et al. 1984, Lowrance et al. 1984, Sedell and Dahm 1984, Sedell and Froggatt 1984). Specifically, riparian vegetation provides streams with bank stability, trout cover, moderating influences on water temperatures, large organic debris (LOD), and contributions of energy and nutrients (Meehan et al. 1977, Moring et al. 1985, Platts 1983b).

Table 1.—Common and scientific names of salmonids mentioned in this report.[1]

Common name	Scientific name
Golden trout	*Oncorhynchus aguabonita* (Jordan)
Cutthroat trout	*Oncorhynchus clarki* (Richardson)
Lahontan cutthroat	*Oncorhynchus clarki henshawi* (Gill and Jordan)
Yellowstone cutthroat	*Oncorhynchus clarki lewisi* (Simon)
Snake River cutthroat	*Oncorhynchus clarki pleuriticus* (Simon)
Greenback trout	*Oncorhynchus clarki stomias* (Simon)
Pink salmon	*Oncorhynchus gorbuscha* (Walbaum)
Chum salmon	*Oncorhynchus keta* (Walbaum)
Coho salmon	*Oncorhynchus kisutch* (Walbaum)
Rainbow trout	*Oncorhynchus mykiss* (Richardson)
Sockeye salmon (kokanee)	*Oncorhynchus nerka* (Walbaum)
Chinook salmon	*Oncorhynchus tshawytscha* (Walbaum)
Pygmy whitefish	*Prosopium coulteri* (Eigenmann and Eigenmann)
Mountain whitefish	*Prosopium williamsoni* (Girard)
Brown trout	*Salmo trutta* Linnaeus
Bull trout	*Salvelinus confluentus* (Suckley)
Brook trout	*Salvelinus fontinalis* (Mitchill)
Dolly Varden	*Salvelinus malma* (Walbaum)
Lake trout	*Salvelinus namaycush* (Walbaum)
Arctic grayling	*Thymallus arcticus* (Pallas)

[1] *Names follow Baxter and Simon (1970), Robins et al. (1980), and Smith and Stearley (1989).*

Streambank Stability

Li and Shen (1973) stated that vegetation creates roughness that decreases water velocity and reduces the erosivity of overbank flows. Though this decrease in water velocity tends to increase the height of flood peaks, bank erosion still declines (Schumm and Meyer 1979). Decreased water velocity reduces the ability of water to carry sediment; thus riparian zones are sites of sediment deposition (Lowrance et al. 1984). Bank will form through this deposition on the convex shoreline, e.g., inside of the meanders (Platts 1983b). Furthermore, aboveground vegetation reduces rainsplash erosion, animal damage, and ice transport damage on streambanks (Platts 1983b). Roots increase the resistance to substrate erosion of banks (Meehan et al. 1977). Compared with poorly vegetated sites on a Utah stream, a well-vegetated reach on the same stream resisted undesirable changes in morphology related to trout habitat (increased width, increased bank angle, and decreased amount of undercut bank (Platts et al. 1985)).

Cover

Cover has been difficult to define; it relies on the assumption that sites occupied by trout possess cover. Butler and Hawthorne (1968) concluded that cover embraces elements of shade or shadow. Binns and Eiserman (1979) defined cover as any bank or channel feature "that allows trout to avoid the impact of the elements or enemies." Such features include overhanging vegetation, undercut banks, submerged vegetation or objects, water depth, and water turbulence (Reiser and Bjornn 1979). Wesche (1980) suggested a quantitative definition: cover consists of sites deeper than 15 cm with water velocities less than 15 cm/s in the presence of one or more of the aforementioned features. A more specific definition of cover, useful for salmonids in streams of the central Rocky Mountains, employs three components: (1) areas of cobble or boulder (substrate greater than 7.5 cm in diameter) in water at least 15 cm deep; (2) overhead bank cover (undercut banks, overhanging vegetation, logs, or debris jams) at least 9 cm wide and associated with water at least 15 cm deep; and (3) pools with water depths greater than 45 cm (Wesche et al. 1985a, 1987a). Riparian vegetation directly affects the second component.

Field evidence suggests that salmonids respond to changes in cover provided by riparian vegetation. Baldes and Vincent (1969) noted that brown trout tended to occupy shaded locations within a flume. Brown trout and brook trout in a submerged tank occupied sites shaded by artificial overhangs 84% and 72% of the time, respectively (Butler and Hawthorne 1968). DeVore and White (1978) found that brown trout preferred artificial overhangs close to the water surface in conjunction with tactile stimuli similar to submerged, trailing branches. Boussu (1954) reported that brook trout and rainbow trout standing stocks responded to artificially manipulated amounts of overhanging bank cover in a Montana stream. And Wesche et al. (1987a) found that a model using only the amount of overhanging bank cover, compared with models incorporating additional cover components, predicted the greatest amount of variation in brook trout and brown trout standing stocks in small Wyoming streams. The two-variable model predicting the greatest amount of variation in brown trout standing stocks in southeastern Wyoming included overhanging bank cover in one variable (Wesche et al. 1987b).

Nonetheless, the influence of overhanging bank cover on trout standing stocks is not entirely understood. Despite the frequent correlation between overhanging bank cover and trout standing stocks, this relationship can be confounded by inconsistencies associated with trout species and channel type (Kozel 1987). Brook trout and Atlantic salmon selected shaded sites in water less than 25 cm deep, but showed no preference for shade in water over 50 cm deep (Gibson and Power 1975). In a laboratory study, Wilzbach (1985) found that cutthroat trout only associated with artificial overhanging or substrate cover at high food abundance, and she suggested that "cover" may interfere with foraging efficiency. Additionally, Schutz and Northcote (1972) reported that cutthroat trout foraged much less successfully at low light intensities than at high light intensities. Helfman (1981) stated that shaded fish can detect prey more easily than can unshaded individuals, and that shaded fish more readily avoid detection by unshaded predators. But these advantages are lost if both the prey and predator are shaded.

Water Temperature

Brown et al. (1971) suggested that stream water temperature was a function of global and net radiation, convection, conduction, evaporation, and storage. Platts (1983b) stated that direct solar radiation accounts for 95% of the heat input into Rocky Mountain streams during summer at midday. Furthermore, stream temperature is directly proportional to heat input, as affected by solar angle and time of day, and exposed stream surface area, while it is inversely proportional to stream discharge (Meehan et al. 1977). Riparian vegetation can also reduce heat loss to the atmosphere during winter; such loss can cause anchor ice formation and lead to substrate disturbance (Platts 1983b). Thus, shading by riparian vegetation often tends to moderate stream temperatures year-round.

Shading may prevent water from attaining temperatures stressful or lethal to salmonids. Many salmonids tolerate temperatures near 18°C, but higher temperatures may increase stress, and mortality typically occurs near 25°C (Jobling 1981). Riparian vegetation prevented water from incurring stressful temperatures in a Pennsylvania stream during summer, but temperatures often exceeded 21°C after removal of this vegetation (Lynch et al. 1984).

Yet shading influences salmonids in a variety of ways in addition to reducing foraging efficiency. Murphy and Hall (1981) compared clearcut, second-growth, and old-growth sites along Oregon streams, and found that reaches in clearcuts possessed the least shade but contained the greatest densities of coastal cutthroat trout. Hawkins et al. (1983) concluded that unshaded reaches supported greater standing stocks of salmonids in West Coast streams because the increased light intensity increased primary production of algae, which augmented secondary production of benthic invertebrates, thus boosting the number of drifting insects available to trout.

Large Organic Debris

Large organic debris (LOD) has been variously defined as any woody material greater than 2.5 cm in diameter (Harmon et al. 1986); as logs, limbs, and rootwads at least 10 cm in diameter (Keller and Swanson 1979); and as wood at least 2 m long and with one end greater than 30 cm in diameter (Bryant 1985). Whatever its definition, LOD has both physical and biotic impacts on salmonid streams. Physical impacts include changes in stability of streambanks and channels, storage of sediment, dissipation of stream energy, and alteration of channel flows (Bryant 1983, Everest and Meehan 1981a, Harmon et al. 1986).

Streambank cutting, windthrow, ice loading, avalanches, and debris torrents contribute LOD to stream channels; and abrasion, biological decomposition, debris torrents, and flotation at high flows remove LOD (Keller and Swanson 1979). Density of LOD tends to be highest in first-order streams because the wood-contributing area to stream area ratio is the highest, and because stream power is lowest (Keller and Swanson 1979). Stream power also determines distribution of LOD within streams; LOD occurs at random in first- and second-order streams, is redistributed within the channel in third- to fifth-order streams, and develops patchy distributions of debris jams in larger streams (Harmon et al. 1986).

When LOD is deposited in streams, several variables determine whether the debris will be stably incorporated into the channel or displaced downstream during high flows. Bilby (1984) noted that pieces larger than 10 m long and 75 cm in diameter, with both ends and one "face" buried, were the most stable. Further, debris that had been in place over 5 years, resisted decomposition, and bridged the entire channel generally remained stable, but usually only if stream gradient was less than 10% in high-order streams (Bryant 1983). Residence of stable LOD may exceed 100 years (Swanson and Lienkaemper 1978).

Unstable LOD may cause erosion during high flows by abrasion, and unstable accumulations may result in debris torrents, which produce relatively smooth, U-shaped channels that terminate in huge debris jams in low-gradient reaches (Bryant 1983, Swanson and Lienkaemper 1978). Alternatively, the removal of stable LOD also induces erosion. Bilby (1984) reported that debris removal from a fourth-order Washington stream reduced bed elevation by an average of 25.4 cm after the first high flow. After debris removal from an Oregon stream, mean scour depth was 90 cm (Beschta 1979).

Single logs may block the entire channel and form steps. These log steps reduce water velocity upstream and trigger sediment deposition (Heede 1985). LOD-influenced sediment storage is more important in high-gradient streams lacking hydraulic storage of sediment (Keller and Swanson 1979). Swanson and Lienkaemper (1978) suggested that less than 10% of the sediment stored in streams with abundant LOD is transported during yearly high flows. Beschta (1979) estimated that 21 m^3 of fine sediment per meter of stream were stored above log steps. Prior to debris removal, log steps stored 87% of the fine sediment in a New Hampshire stream, but after cleaning, the stream lost 83% of its sand and silt (Bilby 1981).

Small waterfalls form plunge pools on the downstream side of log steps. This stepped gradient causes much of the stream channel to have a lower gradient than the valley bottom and dissipates stream energy (Swanson and Lienkaemper 1978). Heede (1981b) found water velocities of less than 12 cm/s in plunge pools, despite velocities up to 61 cm/s above and below these pools. In several western Oregon streams, LOD influenced 30–80% of the drop in stream elevation (Keller and Swanson 1979). Debris dams were responsible for 52% and 46% of the drop in first- and second-order streams, respectively, in a New Hampshire forest (Bilby 1981). Heede (1985) noted that log steps and transverse gravel bars both act to dissipate stream energy. If the former are removed or absent, the stream will transport bedload to form the latter. But if adequate substrate is unavailable, the stream width will increase, meandering will decrease, while longitudinal distance and, hence, downcutting will increase (Heede 1985).

The orientation and size of LOD in streams can increase channel heterogeneity by altering flow direction and velocity (Everest and Meehan 1981a). Following the first high flows after a Washington stream was cleaned, the number, area, and volume of pools decreased, while the area and volume of riffles increased (Bilby 1984). Channel heterogeneity relates to the biotic impacts of LOD, because salmonids preferentially occupy main channel, near shore, and side channel habitats influenced by LOD (Harmon et al. 1986).

LOD can reduce the habitat quality of salmonid streams by decreasing channel stability (as described above), by reducing water quality, or by blocking migration. Peters et al. (1976) reported that some organic chemicals in the heartwood (tropolones) and foliage (terpenes) of western red cedar (*Thuja plicata*) were toxic to fry of coho salmon, and that lethal concentrations developed in streams contaminated with logging wastes. Yet, Harmon et al. (1986) suggest that oxygen depletion will occur prior to reaching toxic concentrations of conifer leachates. Sedell and Luchessa (1982) noted that excessive woody debris can block the upstream migrations of anadromous fish, particularly in low-order streams. But debris dams rarely completely block fish passage, and such blocks may only be seasonal (Bryant 1983).

Many juvenile anadromous salmonids, especially coho salmon, heavily use cover created or influenced by LOD. Side channels created by LOD in large rivers are used far in excess of their availability by coho salmon smolts (Sedell and Luchessa 1982). M. D. Bryant (1985) found that age 0 and 1 coho salmon were more abundant in reduced-velocity channel edges controlled by LOD than along edges lacking LOD. Midwinter densities of juvenile coho salmon were significantly positively correlated with debris volume, and coho microhabitats were always associated with LOD (Tschaplinski and Hartman 1983). After debris removal from a southeastern Alaska stream, the abundance of age 1 coho salmon declined (Bryant 1982). Lister and Genoe (1970) stated that recently emerged coho and chinook salmon associated with cover provided by LOD. Following selective removal of LOD from two Alaska streams, densities and production of age 0 and 1 coho salmon and age 1 and 2 Dolly Varden decreased (Dolloff 1986). During winter at water temperatures below 9°C, age 0 and 1 coho salmon and age 1 steelhead occupied microhabitats within 1 m of LOD (Bustard and Narver 1975b).

Other anadromous and resident species also respond to the presence of LOD. Bisson et al. (1982) noted that age 0, 1, and 2 cutthroat trout associated with woody debris in pools, and that age 0 and 1 steelhead were found near LOD in riffles. The mortality of brown trout fry was greater in a stream subjected to weed and debris removal than in a nearby control stream (Mortensen 1977). Lestelle and Cederholm (1984) discovered that cutthroat trout populations in a cleaned stream did not decline until winter, and the population recovered within 1 year. However, the volume of instream LOD had also recovered to pretreatment conditions.

Nutrient and Energy Effects

Riparian vegetation can act as a sink that removes nutrients and particulate sediment from water prior to entry into streams (Karr and Schlosser 1977, Lowrance et al. 1984). But perhaps the more important role of riparian vegetation is in providing the organic materials that provide the principal energy base for instream biota. Often, the morphological characteristics of allochthonous detritus (i.e., detritus input from sources external to the stream channel) define the microbial and invertebrate communities and, consequently, the energy bases for fish inhabiting these waters (Cummins 1973, 1974; Cummins et al. 1984; Minshall et al. 1982; see Short et al. 1980 for results from the central Rocky Mountains).

Among the characteristics of debris that are important determinants of community composition are the types, sizes, and physical and biochemical compositions of riparian material inputs. These characteristics in turn help define the rates at which debris from the riparian vegetation will decompose in the channel. Note, however, that not only are plant materials from riparian vegetation important contributors to the energetics of stream communities, but terrestrial insects flying or falling from these plants also provide important contributions to the nutrient and energy inputs to stream communities (Cada et al. 1987a, Meehan et al. 1977).

Allochthonous inputs can contribute up to 100% of the organic material ingested by some invertebrate taxonomic groups in streams (Minshall 1967). In some running water systems, however, with limited growths of riparian vegetation (e.g., some high alpine, sagebrush, and grassland areas) autochthonous productivity (i.e., productivity by sources within the stream) can dominate these systems, at least during some times of the year (Minshall 1978). (See the section Natural Determinants of Water Quality and Energy Dynamics in Streams for additional discussion of the influence of riparian vegetation on stream energetics, and the section Instream Biological Relationships for additional information on fish-invertebrate interactions.)

Channel Morphology

Physical features in stream channels are a primary determinant of the types and quality of fish habitat. These physical habitat features include streambed gradient, water depth, water velocity, substrate, and cover (Stalnaker and Arnette 1976). This section discusses the relationships affecting the morphology of stream channels.

The morphological and hydraulic characteristics of stream channels are determined by the flow regime, and environmental factors such as geology, climate, and vegetation (Heede 1980). Several authors have documented changes in channel size and shape with changes in the flow regimes (Bray and Kellerhals 1969, Petts 1977, Williams 1978, Williams and Wolman 1984). Others have shown the effects of environment on channel morphology (Hynes 1970, Leopold and Maddock 1953, Osterkamp and Hedman 1982, Richardson et al. 1975). Andrews (1980) concluded that perennial stream channels are naturally adjusted to their bankfull flows. Paraphrasing Hynes' (1970) and Heede's (1980) summaries on the dynamics of stream channels:

> The main channel tends to swing from side to side, even in relatively straight reaches of stream, causing the channel pattern to continually change. When a stream rounds a curve, the main current hugs the outside, concave bank and the momentum of the water causes the stream surface to become banked. Here, the water in contact with the bank is slowed by friction. As the stream slows, it flows downward and inward, eroding the concave bank downward and outward. Meanwhile, material is being deposited on the inside, convex bank of the curved channel. The wave length of meanders are generally 7 to 10 times the width of the stream, while the actual path measured along the channel itself is about 11 to 16 times the width. Riffles tend to occur at a frequency of about 5 to 7 times the width, particularly in gravel-bed streams, and tend to remain rather stationary. They rarely occur in sandy bottomed streams. Where the channel substrate is loosely consolidated (e.g., in glacial outwash gravels), braided or reticulate channels can form.

Steep drainages in mountain regions of the Western United States pose potentials for high erosion and production of deeply incised channels and greatly steepened valley slopes (Heede 1980). To counter these potentials, natural mechanisms exist that allow streams to adjust channel slopes, which help to protect streambeds. These mechanisms include (1) bed armoring by gravel and boulders, (2) gravel bars that form transverse to streamflows, and (3) log steps that incorporate fallen timber and associated debris into the streambed. Transverse gravel bars and log steps often create longitudinal profiles with steps spaced at regular intervals; and when log steps are lacking the incidence of transverse gravel bars tend to increase. Both of these mechanisms essentially stop the flow in places. Heede (1980) suggested that while such natural flow adjustments were temporary, their continual, natural replacement created a "dynamic equilibrium" between the stream's erosive forces and the stream's beds and banks. By manipulating the number of dead and dying trees in the streamside forest, the manager can influence the hydraulic nature of small streams. Lisle (1986b) found that most pools and upstream gravel bars are associated with large streamside obstructions and bends; based on his observations he developed a general model to define these relationships.

The relation of trout standing stock to channel stability is equivocal. Using the Stream Reach Inventory and Channel Stability Evaluation (SRICSE), developed by Pfankuch (1975) to assess overall channel stability for streams in the central and northern Rocky Mountains, Eifert and Wesche (1982) noted that the mid to high

SRICSE scores and six of its variables (bank cutting, deposition, substrate brightness, particle packing, stable materials, and scouring and deposition) had significant negative correlations with standing stocks of brown trout in two southeastern Wyoming streams. That is, as channel stability (as indicated by SRICSE scores) increased, actual channel stability and standing stocks decreased. The abundance of young Dolly Varden was positively correlated with stream channel stability in an Alaskan stream (Murphy et al. 1986). Nonetheless, Dunham and Collotzi (1975) considered moderate SRICSE scores optimal for trout habitat. And Kozel (1987) found that overall SRICSE scores and individual variable scores were inconsistently related to standing stocks of brook and brown trout.

Rosgen (1985) has developed a stream classification system that categorizes various stream types by morphological characteristics, including stream gradient, sinuosity, width-depth ratio, channel materials, entrenchment, confinement, and soil/landform features. Using Rosgen's classification in the Medicine Bow National Forest, Wyoming, Kozel (1987) found that in B channels brook trout habitat characteristically had narrow channels with abundant overhanging cover and back pools with aquatic vegetation; brown trout habitat was typified by deep, dammed pools ($\geq$15 cm). Though both brook and brown trout typically inhabited narrow channels in C channels, abundant aquatic vegetation providing pool cover characterized brown trout habitat. Overall, C channels had greater standing stocks of trout than did B channels. In another study from southeastern Wyoming, Chisholm and Hubert (1986) found that stream gradient was negatively correlated with standing stocks of brook trout. And, Lanka et al. (1987) found that using geomorphic variables for Wyoming watersheds alone predicted standing stocks of trout as well as did stream habitat variables; the best predictor of trout abundance used both geomorphic and instream variables.

Streamflows

Runoff volumes to streams usually follow seasonal patterns of precipitation, generally with great overall variation (e.g., Parrett and Hull 1985). In mountainous headwater streams of the West, snowmelt provides most of the annual streamflow, with flow peaking from May to July. Minimum streamflow occurs during the fall and winter, and consists largely of groundwater influxes.

Few continuously recording streamflow gaging stations have been established in small mountain drainages having less than 260 km^2 (100 mi^2). In one approach to estimate stream discharge for such systems, both Hunt (1963) and Riggs (1969) determined that annual discharge at an ungaged site could be estimated with an error of 10% using ratios between point measurements for ungaged streams and measurements for the same day from a nearby gaged stream site. Lowham (1976) developed empirical equations using channel size or watershed characteristics to predict mean annual and peak flows for small watersheds in Wyoming.

In an alternative approach, Parrett and Hull (1985) used regression to relate both mean and long-term annual discharges to drainage area and annual precipitation. They also developed equations to predict probabilities of flows that would exceed predicted annual flows. While equations developed by such approaches tend to have localized applications, similar methods can be used to estimate streamflows in other ungaged drainages.

Streamflows may be thought of as having "subcritical" and "supercritical" velocities (Heede 1980). Subcritical flows exert relatively low energies on banks and beds, while supercritical flows can produce highly erosive force and cause channel damage. Standing waves are commonly associated with supercritical flows.

Periodic high streamflows that flush fine sediment from the deeper bed layers are necessary to maintain the channel and riparian habitats (Reiser et al. 1985, 1987). Such flows prevent vegetative encroachment into the channel and encourage plant succession in riparian zones, thereby maintaining and enhancing fishery habitat.

Fine Sediment

Though researchers do not agree on the exact size of fine sediment, it is generally less than 6.3 mm in diameter (Chapman 1988). Sediment transport in streams is a very complex relationship involving at least 30 variables (Heede 1980). But, in general, transport of fine sediment may be via saltation along the stream bottom or suspension in the water column, with discharge and channel slope proportional to the quantity and size of transported sediment (Hasfurther 1985). Typically, transport of sediment is greater on the ascending limbs of storm hydrographs, but this is due more to the supply of sediment rather than the hydraulics of the flows (Sidle 1988, Sidle and Campbell 1985).

Everest et al. (1987) suggested that some fine sediment may be beneficial to salmonids by contributing to increased invertebrate productivity, and that the adverse consequences of fine sediment introduction to trout streams have been overstated. Nonetheless, the transport and deposition of fine sediment frequently are assumed to deleteriously affect survival throughout the life history of salmonids.

Turbidity is a measure of the scattering and absorption of light by dissolved and particulate matter in water (Lloyd 1987). Usually, turbidity and suspended sediment concentration are highly correlated; thus, turbidity can provide an index of suspended sediment concentrations (SSC) (Lloyd et al. 1987). Because murky water absorbs more heat than clear waters, increased suspended sediment loads can cause water temperatures to increase (Hynes 1970). Water with a temperature of 5°C is able to carry 2 to 3 times more sediment than 27°C waters (Heede 1980).

SSC may directly or indirectly influence the survival of salmonids (Iwamoto et al. 1978). SSC can affect fish directly by clogging and damaging respiratory organs; laboratory and field studies have shown that the extent of this impact depends highly on the size and composition of the suspended material and on the individual species of fish (e.g., Branson and Batch 1972, Everhart and Duchrow 1970, Herbert and Richards 1963). Redding and Schreck (1980) and Redding et al. (1987) found that high SSC elevated several physiological measures of stress in juvenile coho salmon and steelhead. Over 50% of juvenile coho and chinook salmon died after a 96-hour exposure to water containing about 500 mg/L of SSC (Stober et al. 1981). SSC above 100 mg/L have reduced survival of juvenile rainbow trout (Herbert and Merkens 1961). Reductions in growth or feeding of salmonids were associated with turbidity over 25 nephelometric turbidity units (NTU) (Olson et al. 1973, Sigler et al. 1984, Sykora et al. 1972). Since salmonids are considered to be sight-feeders, the reduction in light transmission caused by high turbidity may result in less feeding and decreased growth (Berg 1982). In response to turbidity, salmonids may change their use of cover or reduce territoriality (Berg and Northcote 1985, Gradall and Swenson 1982). When given the opportunity, juvenile coho salmon avoided turbid water (Bisson and Bilby 1982). Despite these impacts, salmonids often successfully inhabit streams with seasonally high turbidities, perhaps due to behavioral modifications and to limited exposure to concentrated suspended sediments.

Deposition of fine sediment can acutely affect survival of salmonids (1) during intragravel incubation of eggs and alevins; (2) as fingerlings; and (3) throughout winter (Chapman and McLeod 1987). Timing, source, and quantity of deposited sediment can affect survival. Winter peak flows, and thus sediment transport and deposition, correspond with the incubation of eggs and alevins of salmon and steelhead in the Pacific Northwest (Meehan and Swanston 1977). But cutthroat and rainbow trout in the central Rockies spawn after the spring peak flows, and redd substrates may change very little throughout incubation (Young et al. 1988).

Duncan and Ward (1985) demonstrated that the percentage of fine sediment in 12 southwestern Washington streams was more closely related to the percentage of the watershed composed of sedimentary rock than to the percentage of watershed area in roads. They noted that a stream draining a heavily roaded watershed composed of soils derived entirely from volcanic rock had a lower proportion of fine sediment in the stream channel, compared with channels in watersheds largely composed of sandstones and siltstones.

Increasing proportions of fine sediment in substrates have been associated with reduced intragravel survival of embryonic brook trout (Hausle and Coble 1976), brown trout (Witzel and MacCrimmon 1983a), cutthroat trout (Irving and Bjornn 1984), rainbow trout (Witzel and MacCrimmon 1981), steelhead (Tappel and Bjornn 1983), and various species of Pacific salmon (McNeil and Ahnell 1964, Phillips et al. 1975). But increases in fine sediment can directly limit survival-to-emergence only by entrapping alevins (Koski 1975); the potentially greater influence on survival by increased sediment deposition is the decrease in dissolved oxygen concentration coupled with reduced intragravel water flow (Chapman 1988). For example, Sowden and Power (1985) found no correlation between the amount of fine sediment in redds and the survival-to-emergence of rainbow trout, but they did report that reduced survival in redds was significantly correlated with both reductions in intragravel flows and in dissolved oxygen concentrations. Furthermore, most studies evaluating the impacts of fine sediment on embryonic survival have been conducted in the laboratory; few or no field studies have satisfactorily quantified actual impacts (Chapman 1988).

Fingerling density has often been associated with low concentrations of fine sediment deposited between and on the surface of larger substrate particles. This is defined as embeddedness (Burns and Edwards 1985). Crouse et al. (1981) found that production of coho salmon increased as embeddedness decreased. After the installation of a sediment trap, the abundance of juvenile brown and rainbow trout in a Michigan stream increased by 40% (Alexander and Hansen 1983). Following the experimental addition of fine sediment to a Michigan stream, brook trout densities declined by more than 50% (Alexander and Hansen 1986). Klamt (1976) suggested that fine sediment filled in pools and interstices between cobble, thus reducing the amount of habitat available to fingerling and adult salmonids. Nonetheless, Chapman and McLeod (1987) reported that the relation between the rearing densities of salmonids and fine sediment was equivocal, citing several studies demonstrating no or a positive relation between fingerling abundance and embeddedness. They suggest that changes in stream morphology caused by fine sediment may outweigh the effects of embeddedness on fingerling survival.

As indicated in a later section, declining water temperatures in winter may cause salmonids to seek refuge within the interstitial spaces of the substrate. Bjornn (1971) noted that more juvenile steelhead trout left experimental channels containing large amounts of fine sediment than those containing little or no fine sediment. Deposition of fine sediment could also restrict winter cover for adult fish by filling in low-velocity habitats, e.g., pools and undercut banks (Bjornn et al. 1977a). Very little additional research has been conducted to quantify the effects of fine sediment on winter habitat in the western United States.

The ongoing Sediment-Fish Response Project at the University of Wyoming is investigating the impact of fine sediments on embryonic salmonids in the intermountain Rocky Mountains (Young et al. 1988). Building on previous work completed in the northeastern and northwestern United States, this project is focusing on estimating survival-to-emergence (STE) for embryonic salmonids in the laboratory under realistic conditions; on quantifying the spatial and temporal differences in the substrate between egg pockets, redds, and undisturbed sites; on defining the best field measures for evaluating substrate and intragravel flows; on developing and refining a model for STE; and on validating this model with field investigations.

Snow Cover and Ice

Winter alters trout behavior and affects survival by changing the physical habitat of trout. Among the most obvious changes are reduced water temperature and increased ice formation and snow cover. Declining temperatures in autumn may cause a variety of responses in salmonids. Bjornn (1971) noted that juvenile salmon and trout moved downstream as water temperature decreased, and he attributed this to a lack of suitable substrate (rubble with large interstices between particles); significantly more fish left troughs containing gravel than left troughs containing rubble. In a field experiment, Chapman and Bjornn (1969) found that 7% of the juvenile steelhead left a trough filled with coarse rock, but 35% left a trough filled with gravel. While most steelhead entered the substrate when temperatures fell below 5°C, juvenile coho salmon apparently did not enter the substrate during winter. Yet Rimmer et al. (1983) found that over 90% of immature Atlantic salmon entered the substrate when temperatures dropped below 10°C. And Hillman et al. (1987) stated that winter rearing densities of chinook salmon increased over eightfold after cobble was added to a section of an Idaho stream, presumably due to the increased availability of interstitial space. In fact, juvenile steelhead have been found up to 15 cm deep in substrates during winter (Edmundson et al. 1968).

As winter progresses, one or more types of ice may develop. Shelf ice forms along the streambank and may eventually cover the entire stream (Johnson et al. 1982). Anchor ice forms on the stream bottom when a streambed radiates energy to the sky (usually during clear conditions) causing water temperatures to decline below 0°C (Benson 1955). Frazil ice consists of smaller crystals suspended in the water column (Maciolek and Needham 1952).

Salmonids occupy sites beneath shelf ice, suggesting it provides suitable overhanging cover (Logan 1963, Maciolek and Needham 1952). Cunjak and Power (1987) noted that both brown trout and brook trout occupied areas beneath submerged artificial overhangs in an ice-free Ontario stream. In an ice-free Cascade Mountain stream, rainbow trout occupied pools during the day, but tended to enter the spaces in the substrate as temperatures declined from 13°C to 8°C (Campbell and Neuner 1985). But these fish reappeared at night in shallow, near-shore sites and remained there until approximately 1 hour before dawn, when they reentered the substrate. Needham and Jones (1959) also observed more trout in the water column at night than during the day in the winter in a California stream. Thus cover, as shelf ice, overhanging vegetation, substrate, or darkness, apparently influences winter behavior of salmonids.

Both brown and brook trout have been found to occupy low-velocity sites ($<$15 cm/s) under complete ice or snow cover (Chisholm et al. 1987, Wichers 1978). However, ice cover reduces available habitat; Chisholm et al. (1987) noted that ice excluded from 64% to 84% of the cross-sectional area of one section of a southeastern Wyoming stream. Streams completely covered with snow have little or no ice formation and provide very stable habitat conditions. Johnson et al. (1978) devised a model to predict habitat excluded by ice based on the days since the winter solstice, solar radiation, mean water depth, and mean water velocity.

Ice or snow may be associated with mortality in other ways. As water temperature increases, anchor ice may detach from the stream bottom to form ice dams; flows continually redirected by these dams may eventually strand trout (Maciolek and Needham 1952, Needham and Jones 1959). This detachment may also dislodge aquatic insects and increase turbidity (Maciolek and Needham 1952). Needham and Slater (1944) reported that the collapse of a snowbank killed over 300 trout. Ice forming in the substrate may cause mortality of the embryos of fall-spawning salmonids (Benson 1955, Reiser and Wesche 1979). However, Olsson (1981) reported that most benthic invertebrates survived intragravel freezing, and he suggested that freeze tolerance removed the need to seek flowing water that might expose invertebrates to predation.

Finally, winter conditions affect the feeding behavior of trout and, consequently, their physiology. Reimers (1957) found that wild brown trout starved for 180 days suffered only 8% mortality. Also, these fish required up to 70 hours to assimilate a 5-g meal at 0°C. Hatchery trout tended to die in early spring as water temperatures increased, apparently because they lacked energy reserves to meet the increase in metabolic activity (Reimers 1963). Hunt (1969) noted that the overwinter survival of fingerling brook trout increased as fingerling size increased, and he attributed this to a greater resistance to temperature-induced physiological stress in larger individuals.

Natural Determinants of Water Quality and Energy Dynamics in Streams

The natural physical and chemical qualities of waters in streams are determined by (1) volume, frequency, and chemical nature of precipitation; (2) physical and chemical nature of the underlying parent rock and soil that these waters flow over and through; (3) chemical changes to the waters in the watershed produced via nutrient uptake and decomposition by terrestrial vegetation and microbes; (4) the degree of bank stability and physical shading provided by riparian vegetation; and (5) nutrient uptake by and decomposition of aquatic organisms. This section discusses some of the intensive studies that have been conducted on the relationship between streams and their watersheds, and then briefly reviews two paradigms that currently summarize much of our understanding of nutrient and energy dynamics in flowing water systems. These are "material spiralling" and the "River Continuum Concept."

Stream-Watershed Relationships

For many nutrients and particulate materials, streams transport or accumulate virtually all losses from watersheds, except for often relatively minor losses to the atmosphere. For example, all measurable particulate and dissolved losses of nitrogen from one small watershed passed through or accumulated in the stream, which comprised <1% of the watershed area (Triska et al. 1984). That study further reported that biotically derived inputs, including litterfall, throughfall, lateral movement, dissolved organic carbon in groundwater and nitrogen fixation, constituted >90% of the nitrogen input to the stream. Leaves, bark, and wood supplied to the stream tended to rapidly leach nitrogen due to physical processes. Subsequently, nitrogen associated with these particles increased due to nitrogen fixation by the stream microflora (Buckley and Triska 1978). Leaf packs having the greatest associated nitrogen contents had the greatest densities of invertebrates. Thus, the capacity of litter microflora to control nitrogen uptake and release apparently influenced directly the production of litter-consuming invertebrates (Triska and Buckley 1978). Microbial communities similarly can be the primary regulator of phosphorus movement in streams (Gregory 1978). In tundra areas the carbon cycle of rivers can be dominated by inputs of eroding particles and leaching of dissolved organic carbon from peat (Peterson et al. 1986).

Intensive studies conducted in cooperation with the USDA Forest Service's Northeastern Forest Experiment Station on the Hubbard Brook Watershed of New Hampshire provide a benchmark for future watershed-stream ecosystem investigations (Likens et al. 1977). These studies found that, while evapotranspiration reduced the volume of water flowing from watersheds, annual evapotranspiration rates were essentially constant over a wide range of precipitation and environmental conditions. Stream water chemistries in the undisturbed forests were highly predictable: concentrations of sodium and silica were diluted by streamflows, while concentrations of aluminum, nitrate, hydrogen, potassium, and dissolved organic carbon increased as streamflow increased. Seasonal biological activities strongly affected the stream concentrations of nitrate and potassium. Atmospheric inputs to the watershed were the major sources for sulfur, nitrogen, chloride, and phosphorus; weathering was the major source for calcium, magnesium, potassium, and sodium; biological activities were the major watershed contributors of carbon and nitrogen; and terrestrial plants served as important impaction surfaces for atmospheric sulfur. Overall, the watershed accumulated nitrogen, sulfur, phosphorus, and chloride, while it lost silica, calcium, sodium, aluminum, magnesium, and potassium. Additionally, Vitousek (1977) found that forest successional stages are important determinates of nutrients dynamics in and losses from watersheds.

For a watershed in the Rocky Mountains of northern Colorado, Lewis and Grant (1979) found that concentrations of bicarbonate, nitrate, calcium, magnesium, and sodium decreased with increasing stream discharge; concentrations of dissolved organic carbon, hydrogen ions, and phosphate increased with increasing discharge; while ammonium, dissolved organic phosphorus and nitrogen, potassium, and sulfate concentrations showed no trend of change with streamflows. In a related study, Lewis and Grant (1980) also found that decreased snowpack was associated with greater areas of frozen ground and significantly greater runoff of calcium, magnesium, potassium, phosphate, nitrate, dissolved organic phosphate, bicarbonate, and hydrogen ions. Sulfate, sodium, nitrate, ammonia, dissolved organic carbon, and dissolved organic nitrogen displayed no significant relationship to snowpack.

During their analysis of water quality in streams of the Fraser Experimental Forest in Colorado, Stottlemyer and Troendle (1987) found that major cation and bicarbonate concentrations were much greater than those reported by Lewis and Grant (1979, 1980). Stottlemyer and Troendle also found that trends of change for cations associated with stream discharges were similar to those reported by Lewis and Grant, but these patterns tended to be weaker and there was no significant change in anions with discharge. The differences were attributed to differences in watershed geologies, which led them to caution against making generalizations about "representative" watersheds, their functions, and their responses.

Material Spiralling in Streams

In flowing waters, nutrients, energy, and other materials tend to cycle from organisms to sediments to suspension and then back to organisms again at "progress intervals" downstream. This contrasts with nutrients cycles in lakes and terrestrial ecosystems where these cycles often are completed within relatively close spatial proximity of each other. Thus, in rivers and streams the continual downstream movement of these cycles has been termed "spiralling" (Webster and Patten 1979).

While travelling downstream, materials can be transferred among environmental compartments where nutrients can be "stored" for varying periods, effectively altering rates of downstream transport. Storage times for different compartments can be relatively long or short. For example, some nutrients incorporate mostly into tissues of organisms, from which they are released back into the water at relatively slow rates; other nutrients can be rapidly excreted by organisms into the water, where they are readily available for use by other organisms. Such differences produce spirals of different spatial lengths for different nutrients, different streams, or different reaches of the same stream (Newbold et al. 1981).

Under this concept, "tighter" nutrient spirals are associated with constancy in stream ecosystems, increased biomasses of resident biota, spatial heterogeneity through the stream continuum, resistance to external stresses, and ability for rapid recovery from perturbations. Disturbances tend to disrupt storage mechanisms, increase losses of dissolved nutrients and/or nutrients in the sediment, and increase spiral lengths within streams (Newbold et al. 1981, Webster and Patten 1979).

River Continuum Concept

Much of the current understanding about the total nutrient and energy dynamics in flowing water systems has been summarized and synthesized within the "River Continuum Concept," which describes changes in structure and function through lengths of river systems (Minshall et al. 1983, 1985; Sedell et al. 1978; Statzner and Higler 1985; Vannote et al. 1980). Most simply this theory proposes that biological systems occupying natural, undisturbed river systems are shaped fundamentally by the physical forces present in watersheds. Progressive downstream changes in these physical characteristics are suggested to produce a continuum of change in sources of energy, sizes of organic (food) particles, and types of organisms.

Upstream, headwater reaches tend to be dominated by riparian vegetation, which reduces production of instream plants by shading and contributes large amounts of allochthonous materials to the system. These contributions cause carbon stores to be dominated by coarse particulate organic materials (Speaker et al. 1984). Consequently, invertebrate communities in headwater streams will tend to be dominated by organisms that shred coarse matter into fine particles, and by those that collect these drifting particles. Due to the preponderance of materials and energy originating from outside of the system, the instream ratio of community photosynthesis to respiration (P/R ratio) is generally less than 1, i.e., more carbon compounds are used by the community than are produced by it.

Downstream, in medium-sized streams, algae and other aquatic plants have greater importance. With the increase in autotrophic production, P/R ratios often can exceed 1. This shift in the energy base for these streams favors the dominance of invertebrates capable of collecting particles released by upstream processes, and of those able to graze on the plants growing in the streams.

Further downstream, the energetics of large rivers are increasingly dominated by fine particles released by upstream sources. Here the rivers again have P/R ratios of less than 1 and the invertebrate communities tend to be dominated by collectors.

Through the River Continuum Concept it is suggested that streams having low physical variability tend to have low biological diversities and high ecological stabilities. Alternatively, streams with high physical variability tend to have high biological diversities and complexities, while also tending to maintain ecological stabilities. Stabilities developed by biological communities in stream systems may be thought of as a strategy through which energy or nutrient loss downstream is minimized.

The River Continuum Concept was developed using research based largely on flowing water systems in the wet coniferous forest of the Northwest and in the deciduous and coniferous forests of the East and Midwest. Of course, patterns of riparian vegetation that differ from those generalized above can produce variations in the other inherent aspects of the adjacent stream (Conner and Naiman 1984, Gurtz et al. 1988, Minshall et al. 1985, Statzner and Higler 1985). Also, disturbances within stream systems can disrupt the overall stream continuum by changing conditions over the disturbed reach to conditions more similar to those occurring either upstream or downstream of the disturbance (Vannote et al. 1980, Ward and Stanford 1983). Little is known on how the streams and rivers of the central Rocky Mountains conform to or deviate from the River Continuum Concept.

Water Quality Criteria for Fish and Other Aquatic Life

Beyond the natural determinates of water quality, numerous cultural activities can alter the quality of surface waters and affect aquatic habitats, as are more fully detailed in later sections. For example, clearcuts or other watershed disturbances tend to increase the loss of ions (Dillon and Kirchner 1975; Prairie and Kalff 1988a, 1988b; Vitousek 1977). Various forest use activities can alter water qualities and potentially produce conditions that violate water quality criteria established by the U.S. EPA to protect fish and other aquatic life. This brief overview of the U.S. EPA's present water quality criteria for suspended sediment, temperature, pH, nutrients, and various additional potentially toxic chemicals is based largely on the ''Quality Criteria for Water'' produced by the U.S. EPA (1986) and on an earlier review completed by the American Fisheries Society (Thurston et al. 1979).

Suspended Sediment

The European Inland Fisheries Advisory Commission concluded that suspended sediment can affect aquatic organisms by killing them directly, by reducing growth rates and resistance to disease, by preventing successful development of eggs and larvae, by modifying natural movement or migration patterns, or by reducing the natural availabilities of food (U.S. EPA 1986). A review completed by the National Academy of Science suggested that a limit of 25 mg/L of suspended sediment would provide high, 80 mg/L moderate, 400 mg/L low, and over 400 mg/L very low levels of protection for aquatic organisms (Thruston et al. 1979). The present water quality criterion established by the EPA for fish and other aquatic life is based on the depth in the water column at which planktonic photosynthesis equals respiration; hence, it does not apply to most salmonid habitats in the central Rocky Mountains. Lloyd (1987) suggested, based on his review of turbidity studies in Alaska, that a water quality standard that permitted an increase of 25 NTUs (nephelometric turbidity units) above ambient would provide moderate protection for clear, coldwater stream habitats. It remains to be learned whether such a standard would be applicable to salmonid streams in the central Rocky Mountains.

Temperature

Most biological and chemical processes in aquatic environments ultimately are regulated by water temperature. Fish and essentially all other aquatic animals are cold blooded (poikilotherms); thus, their metabolism, reproduction, development, and scope for activity is largely controlled by environmental temperatures. Similarly, aquatic plant photosynthesis and respiration, chemical reaction rates, gas solubilities, and microbial mediated processes including decomposition and nutrient cycling are also temperature dependent. In fact, the

Federal Water Pollution Control Administration in 1967 described temperature as "a catalyst, a depressant, an activator, a restrictor, a stimulator, a controller, a killer, one of the most important and influential water quality characteristics to life in water" (U.S. EPA 1986).

The present criterion to protect freshwater aquatic life is based on "the important sensitive species" resident during the time of concern and consists of two upper temperature limits (U.S. EPA 1986). The first limit is based on short (i.e., over durations of minutes) exposure, is computed using an equation presented in the EPA criterion, and uses data presented in a National Academy of Sciences document. The second limit is based on a weekly maximum average temperature, which changes with season, with reproductive stage present, to maintain species diversities, or to prevent nuisance growths of organisms. For rainbow and brook trout adults and juveniles, the maximum weekly average temperature for growth during the summer is listed as 19°C, and the short-term maximum temperature limit for survival during summer is 24°C (U.S. EPA 1986). This report also lists 9°C as the average weekly maximum temperature reported for spawning by these species, and 13°C as the short-term maximum reported for survival of their embryos. The present temperature criterion presents numerous interpretational problems with respect to defining "important sensitive species" and "short term" (Thurston et al. 1979).

pH

In natural waters, pH is primarily regulated by the solution of atmospheric carbon dioxide, which reacts with water to form carbonic acid and then disassociates to hydrogen and bicarbonate ions. Distilled water at equilibrium with atmospheric CO_2 has a pH of 5.6 at sea level. But natural waters contain various dissolved salts and organic chemicals derived from watershed rocks, soils, and organisms that tend to buffer the natural acidities and raise the pH of surface waters.

As with temperature, the concentration of hydrogen ions is an important regulator of many chemical and biological processes in aquatic environments. (Most accurately, it is the chemical activity of hydrogen ions, rather than their concentration, that is reflected by measured pH levels.) For example, pH primarily defines the chemical natures of dissolved ions in waters, the directions of chemical reactions, the adsorption of chemicals onto organic and inorganic particles, plus the availability and toxicity of chemicals to organisms. Similarly, uptake and release rates for ions across gills, the primary method of ion regulation for aquatic animals, is at least partly pH dependent. Environmental conditions beyond their natural pH limits can produce stress and cause mortality of organisms.

The criteria range to protect freshwater aquatic life is pH 6.5 to 9.0 (U.S. EPA 1986). Many fish, however, are well able to survive and reproduce at pH levels outside of this range. Also, not all species and not all life stages of most species are equally sensitive to pH changes. For example, as acidity levels increase, brook trout are generally less sensitive than brown trout, which are in turn less sensitive than rainbow trout; hatching and larval stages are the life stages for these species that are most sensitive to acidity (Marcus et al. 1986). Brook trout populations frequently have been found to inhabit waters that have pH levels less than pH 5.5 (e.g., Schofield and Trojnar 1980).

Nutrients

The two nutrients of greatest potential concern in aquatic systems are nitrate and phosphate. These nutrients are the two most related to the eutrophication of surface waters, the associated nuisance growths of algae, and the development of other noxious conditions. No criterion is provided by EPA for either of these two nutrients with respect to the control of eutrophication. The U.S. EPA (1986) does discuss the toxic potential for nitrates to fish. This report concludes that nitrate-nitrogen concentrations at or below 90 mg/L should be protective for warmwater fishes, while concentrations at or below 0.06 mg/L should be protective for salmonid fish. This guideline for salmonids is based on very limited data, and many natural salmonid waters have nitrate concentrations exceeding this level.

The U.S. EPA (1986) suggests as a guideline to prevent nuisance algal growths and limit cultural eutrophication that total phosphates as phosphorus should not exceed 0.1 mg/L in any stream or other flowing water, exceed 0.05 mg/L in any stream at the point where it enters a lake or reservoir, or exceed 0.025 mg/L in any lake or reservoir. In general, eutrophication may occur in surface waters that have nitrate-nitrogen concentrations above 0.3 mg/L and phosphate-phosphorus concentrations above 0.02 mg/L (Golterman 1975). Since algae on an average contain nitrogen and phosphorus in ratio of 16:1, this ratio in natural waters is commonly cited as an important indicator of relative nutrient limitations by these two nutrients (Stumm and Morgan 1970). Higher ratios indicate possible phosphorus limitation, while lower ratios indicate possible nitrogen limitation. Knowledge of this ratio for surface waters can help to target potential sources of nutrients and to guide corrective management actions aimed at reducing possible eutrophication problems.

Potentially Toxic Chemicals

Fisheries can be affected by an increasing diversity of potentially toxic chemicals. A review of the toxicity and potential impacts of all of these chemicals is beyond the scope of this report. But, table 2 lists the lowest observed effects levels obtained from short-term acute toxicity and long-term chronic toxicity tests for a variety of toxicants potentially encountered by salmonids in streams of the central Rocky Mountains; water concentrations less than the indicated levels may not affect fish. The reported levels are summarized from the U.S. EPA's 1986 water quality criteria, and provide a guideline to determine

Table 2.—1986 quality criteria for water (U.S. EPA 1986).

Chemical	Freshwater concentration in µg/L	
	Acute criteria	Chronic criteria
Aldrin	3.0	
Ammonia	Criteria are pH and temperature dependent	
Antimony	9,000[1]	1,600[1]
Benzene	5,300[1]	5,100[1]
Beryllium	130[1]	5.3[1]
Cadmium	3.9[2]	1.1[2]
Chlordane	2.4	0.0043
Chlorine	19	11
Chloroform	28,900	1,240
Chromium (Hex)	16	11
Chromium (Tri)	1,700[2]	210[2]
Copper	18[2]	12[2]
Cyanide	22	5.2
DDT	1.1	0.0010
DDT metabolite (DDE)	1,050[1]	
DDT metabolite (TDE)	0.6[1]	
Dieldrin	2.5	0.0019
Endrin	0.18	0.0023
Iron		1,000
Lead	82[2]	3.2[2]
Malathion		0.1
Mercury	2.4	0.012
Nickel	1,800[2]	96[2]
Parathion		0.04
pH		6.5–9.0
Phenol	10,200[1]	2,560[1]
Selenium	260	35
Silver	4.1[2]	0.12
Sulfide-hydrogen sulfide		2
Thallium	1,400[1]	40[1]
Toluene	17,500[1]	
Zinc	320[2]	47

[1] *Insufficient data to develop criteria. Value presented is the Lowest Observed Effect Level (LOEL).*

[2] *Hardness dependent criteria (100 mg/L hardness level used for reported value).*

potentially hazardous conditions for salmonids in the central Rocky Mountains. For a specific water, actual toxic concentrations are often likely to be either greater or less than the reported values. For example, the toxicity of ammonia increases as either temperature or pH of the water increases (Thurston et al. 1979).

As another example, the toxicity of many metals and other chemicals is affected by hardness. That is, actual instream toxicities of metals to aquatic organisms does not depend solely on the total concentrations of the metals in the water. A variety of studies, including those from the Rocky Mountains, show that water quality characteristics, especially hardness, alkalinity, pH, and chelating organic materials, can affect the aquatic chemistries, toxicities, and bioavailabilities of metals (Black et al. 1975, Marcus et al. 1983, Parkhurst et al. 1984). Studies also reveal that not only can calcium hardness, in particular, affect the instream chemistry of many metals, but it can affect the physiological susceptibilities of the organisms to the potential toxicities of many metals (Davies and Woodling 1980, Mount et al. 1988, Parkhurst et al. 1984, Pascoe et al. 1986).

In addition, potentially toxic chemicals are rarely present singularly; toxicities derived from multiple chemical sources may be additive, subtractive, or multiplicative. Therefore, when evaluating the potential toxicities to fish by chemicals in surface waters, evaluations must include careful determination of and potential interactions with other chemicals present.

Instream Biological Relationships

Various interactions occur among fish and other aquatic organisms inhabiting streams. These include competition for space and food resources, natural predation, and interactions with beaver.

Competition for Space and Food Resources

When resource demand exceeds resource availability, fish species may adjust their use of those resources, i.e., undergo niche shifts. Thus, competition may structure fish communities, depending on the ability of each species to adapt to competition. However, Hearn (1987) noted that factors limiting fish densities may reduce or eliminate the impact of competition on fish communities. And one salmonid species may replace another by mechanisms other than competition (Robinson and Tash 1979). These include predation, disease, and habitat alterations to the disadvantage of a species. Nonetheless, a variety of experiments demonstrate niche shifts by salmonids that have been attributed to competition (Hearn 1987); many of these species inhabit streams in the central Rocky Mountains.

Fausch and White (1981) noted that brown trout apparently excluded brook trout from certain stream microhabitats; they concluded that brown trout occupied the most favorable feeding sites (those sites possessing low velocity water adjacent to food-carrying high velocity currents). Habitat perturbation may exacerbate the advantage of brown trout over brook trout (Waters 1983). When sympatric with brown trout, rainbow trout shifted to microhabitats with greater water velocities, over coarser substrates, farther from overhead vegetation, and with less shade than when allopatric (Gatz et al. 1987). Brook trout apparently displace cutthroat trout (Behnke and Zarn 1976), though Griffith (1972b) suggested that these species prefer different habitats in allopatry and sympatry. Due to the interaction with rainbow trout, cutthroat trout in a British Columbia lake shifted from midwater areas when allopatric to littoral zones when sympatric (Nilsson and Northcote 1981). Finally, rainbow trout tend to replace brook trout, possibly due to competition for food immediately after swim-up by larval rainbow trout (Rose 1986). Nonetheless, the existence of competition between any of these species has not been directly verified in the central Rockies.

As fish grow, their microhabitat preferences may shift. Everest and Chapman (1972) noted that age 1 steelhead occupied deeper, faster sites than did age 0 steelhead. But greater size is also associated with dominance in salmonids (Newman 1956). Jenkins (1969) reported that size strongly correlated with dominance among brown and rainbow trout in two California streams. Therefore, if different age classes of a single species prefer the same microhabitat, the larger individuals should occupy the most favorable sites and displace the smaller fish to suboptimal sites (interactive habitat selection; Fretwell 1972). Larger age 1 brown trout aggressively displaced younger brown trout from certain microhabitats (Symons and Heland 1978). Alternatively, Baltz and Moyle (1984) suggested that habitat use was related to developmental changes, not to dominance (selective habitat segregation). In any case, the interaction between ontogenic shifts in microhabitat preference and intraspecific competition is not well understood.

Invertebrates serve as the primary food for many populations of salmonids, though they occasionally act as predators on embryonic or juvenile salmonids (Claire and Phillips 1968). Though invertebrates may be captured from the substrate, salmonids generally feed in the water column (Elliott 1970). The presence of invertebrates in the water column of streams is known as *drift*. Invertebrate drift tends to peak shortly after dusk or during physical disturbances (e.g., floods), but drift persists continuously at low rates in most streams (Waters 1969).

Drift may be critical in determining microhabitat choice by salmonids. Chapman (1966) suggested that territorial defense by salmonids is a substitute for scramble competition for food. Fausch (1984) compared positions occupied by juvenile coho salmon that had formed a dominance hierarchy, and found that the ranking of sites by potential net energy gain (energy from food less energy for maintaining position in the water column) was positively correlated with the rank of fish in the dominance hierarchy. Densities of rainbow trout fry in experimental channels peaked and aggressive behavior declined at high drift densities, but densities decreased and aggression increased at low prey abundance (Slaney and Northcote 1974).

Sympatric salmonids frequently consume the same foods; the invertebrate diets of brown and rainbow trout (Elliott 1973), brown and brook trout (Allan 1978), and cutthroat trout and coho salmon (Glova 1984) overlapped greatly. Griffith (1972a) reported that age 0 brook and cutthroat trout consumed similar foods, but the diets of these species differed among older fish. Typically, the proportion of invertebrates in salmonid diets tends to reflect the abundance of aquatic and terrestrial invertebrates in the drift (Cada et al. 1987a, 1987b). But invertebrate predation by salmonids is size-selective, i.e., the largest individuals of invertebrate species are preferred (Allan 1978, Dunbrack and Dill 1983, Newman and Waters 1984) and small salmonids tend to capture smaller prey than do large salmonids (Glova 1984). Additionally, individual fish may specialize in capturing certain invertebrate species despite the presence of more abundant invertebrates (Dill 1983), or salmonids may avoid certain abundant invertebrates (Ringler 1985). Finally, salmonids may alter the abundance of various aquatic invertebrates; troutless pools in a California stream contained significantly more odonate naiads, coleopteran larvae, and hemipteran adults than did pools containing trout (Hemphill and Cooper 1984).

Predation

The impact of natural (nonfishing) predation on salmonid populations in the central Rockies is largely unknown. Bowlby and Roff (1986) noted that trout biomass was related to the presence of northern pike in southern Ontario streams, but the occurrence of this piscivorous (fish eating) fish accounted for very little of the variation in trout biomass. Burgess (1980) noted that mink fed little on trout but instead preferred crayfish. However, Alexander (1976) believed mink control would improve salmonid populations in Michigan streams. Squawfish (*Ptychocheilus lucius*) predation on migrating smolts is believed to be a significant source of salmonid mortality in the Columbia River (B.E. Rieman, Oregon Department of Fish & Wildlife, personal communication). Dippers (*Cinclus mexicanus*) appear to consume large numbers of recently emerged fry of Colorado River cutthroat trout in the North Fork Little Snake River drainage in Wyoming (M. Bozek, Department of Zoology and Physiology, University of Wyoming, personal communication). Nonetheless, despite the variety of birds, mammals, and fishes that are piscivorous, the magnitude of this predation has rarely been quantified.

Beaver-Trout Interactions

Beaver-trout relations in the United States have been studied since the 1940s (Huey and Wolfrum 1956, Patterson 1950, Rasmussen 1941, Rupp 1954). Several surveys of trout in beaver ponds in the central Rocky Mountains have been conducted (Call 1960, 1970; Gard and Seegrist 1972; McDowell 1975; Rabe 1970).

Since the early studies, a dichotomy appears to exist between beaver-trout relations found in the eastern United States versus the Rocky Mountains (Allen 1956). In the East, beaver impoundments are generally detrimental to trout populations due to their tendency to increase water temperatures above tolerance limits for the resident salmonids, to cover spawning gravels with silt, to remove bankside cover, and/or to entrap predators within the ponds. In the Rockies, however, beaver ponds generally appear to benefit salmonids. Increased water temperatures in the ponds is often correlated positively with trout growth, and siltation of spawning areas is generally precluded by greater stream gradients, which also prevents beavers from impounding the entire streamcourse (Retzer et al. 1956). Also, beaver ponds can enhance baseflows in streams. While removal of bankside vegetation by beaver tend to increase water temperatures in beaver ponds, this potential benefit relative to the negative impact from losing overhead cover for trout has not been quantified.

Despite the general benefits provided salmonids by beaver in the West, they can adversely affect trout in two ways. First, large numbers of dams over relatively short stream distances can prevent or slow the movement of spawners to suitable sites. During the spring, however, high water levels in the ponds and over sections of the dam probably mitigate this potential impact. Second, siltation of spawning sites can occur in some western streams. While the extent of this siltation has been shown to be negligible under normal circumstances or over short time periods (Rasmussen 1941), dams are more prone to washout when built where erosion-resistant substrates or structures (e.g., rock outcrops and moraines) are lacking. Then, potentially large masses of previously trapped sediment can be released to cover downstream spawning gravels and limit spawning success.

In total, beaver ponds in the Rocky Mountain West tend to support greater production of aquatic invertebrates, as well as more and larger trout than undammed sections of the same stream (Huey and Wolfrum 1956, Naiman et al. 1984). This, however, depends somewhat on the age of the pond; often older ponds support large biomasses of stunted trout (Rabe 1970). Generally, invertebrate prey for salmonids in beaver ponds have greater densities and masses, apparently due to the higher alkalinities, warmer water temperatures, and greater productivities found in beaver ponds. Also, recent work has shown that beaver ponds can be a sink for nutrients from influent streams, which may further contribute to the increased productivity in these ponds (Maret et al. 1987).

Recently, a method has been devised that predicts trout abundance in beaver ponds by relating habitat attributes to trout abundance (Winkle 1988). The habitat variables most significantly correlated with trout productivities were surface area of the pond, mean depth, potential spawning habitats, downstream pond density, and spring to fall water level fluctuation. Additional study and validation of this model is required before it can be generally applied.

POTENTIAL IMPACTS TO SALMONID HABITATS

Cultural developments in forests can have severe consequences on trout habitats. These activities include road construction and use; logging and reforestation; fire; livestock grazing; water developments; mining and mine reclamation; recreation, including fishing, fish management, and fish stocking; and urbanization. These activities can affect salmonid habitats individually and cumulatively.

Impacts Associated with Roads

Roads built in forests can impact salmonids and their habitat primarily through two mechanisms. First, forest roads increase erosion rates and sediment loads in streams, potentially affecting salmonids by reducing respiration and ion exchange rates across gills and by clogging spawning areas. Second, improperly constructed road culverts and bridges can block migration routes. In addition, there is some evidence to suggest that road networks alone can accelerate peak flows in small watersheds because roads effectively increase drainage networks in watersheds (Chamberlin 1982); but, the importance of this potential impact to salmonids will not be discussed here.

Erosion and Sediment

Many studies have associated increased stream-suspended solids and channel sedimentation load with roads (e.g., Cline et al. 1983, Fredriksen 1970, Krammes and Burns 1973, Yee and Roelofs 1980). In fact, roads can be the major source of sediment to forest streams (Packer 1966). These studies show that large releases of sediment have been associated with failures of cutbanks and fill-slopes, road surface erosion, ditch erosion, and failure of drainage structures. Also, landslides, slump earthflows, and debris torrents are mass erosion events frequently associated with road construction (Everest et al. 1985a). A study in the Cascade Mountains found that landslides within road cut corridors were 27 times more frequent than the frequency of landslides in forested areas without roads (Lyons and Beschta 1983). Of course, landslide frequencies also largely depend on the nature of the underlying geologies and the grades of cut slopes, in addition to the nature of the road (Duncan and Ward 1985). Debris torrents can leave deep accumulations of boulders, large woody debris that blocks fish passage, and fine sediment that clogs spawning and rearing habitats.

A study in steep-sloped forests in Idaho suggested that skyline logging alone (without the influence of roads) increased sediment erosion by a factor of about 0.6 over natural rates, while roads associated with jammer logging increase sediment production by over 750 times during the 6 years after road construction (Megahan and Kidd 1972). When roaded areas exceed 2% to 3% of the watershed, accumulations of sediment in the stream gravels can markedly increase. Paved roads released less that 1% of the sediment that was released from heavily used roads with gravel surfaces, and sediment production increased with the use intensity of roads (Reid and Dunne 1984). Heavy use (greater than four loaded logging trucks per day) increased sediment production 750% over that occurring when the roads were not used; occasional use by light vehicles limited sediment loss to less than 1% that occurring with heavy use. Effects of forest roads on watersheds in mountainous areas are reviewed by Megahan (1985, 1987).

Approaches to control erosion on forest roads have been presented by Everest et al. (1985a), Kochenderfer (1970), Megahan (1985, 1987), and Yee and Roelofs (1980). Both Leaf (1974) and Reid and Dunne (1984) present methods to estimate sediment production from forest roads. Briefly, sediment impacts due to roads can be at least partly mitigated to protect salmonid habitats by careful planning of their location and design, use of lower impact construction methods, and, because deteriorating roads can contribute greatly to instream sediment loads, establishing programs of regular road maintenance.

Culverts and Fish Passage

Culvert and bridge placement can cause loss of habitat at least through the reach of the placement, plus through any area above and below that is necessary to stabilize the structure. Road culverts can also produce serious potential threats to fish migration because of outfall barriers, excessive or insufficient flows through culverts, lack of resting or jump pools, or a combination of these factors (Evans and Johnston 1980, Everest et al. 1985a, Yee and Roelofs 1980). The resulting blockage can cause loss of important spawning or rearing habitats in streams above improperly designed and installed bridges and culverts.

Because bridges generally disturb the stream less than culverts, construction of bridges is often preferable to installation of culverts, but often culverts are more economical. Yee and Roelofs (1980) suggest a diversity of considerations necessary to minimize potential effects of forest road culverts on salmonids. These include (1) establishment of a resting pool immediately below the culvert that permits fish to conserve energy and obtain a good start to overcome the obstacle; (2) minimizing the jump necessary to enter the culvert to generally less than 1 foot, or to less than 0.5 foot if a series of jumps is required; (3) if swimming distances through culverts or bridge aprons are greater than 50 to 100 feet, resting pools may be required en route; and (4) a second resting area should be available upstream. Finally, culverts are best placed where there are no sudden increases in water velocities above, through, or below culverts; where the stream alignments are similar above and below the culvert; and where the culvert can be installed as close to a zero gradient as possible. Additional approaches to mitigate the effects of road construction have been suggested by the AFS Riparian Habitat Committee (American Fisheries Society 1982), and an annotated bibliography of literature associated with fish passage and road crossings has been prepared by Anderson and Bryant (1980).

Impacts Associated with Forest Harvest and Reforestation Practices

In addition to impacts directly associated with road construction, forest harvest and reforestation practices can also affect salmonid habitats by altering patterns for erosion and deposition of sediment, streamflows, fish migrations, structural habitat cover, water temperatures, nutrient cycles, and potentials for exposure to toxicants. Salmonid populations naturally encounter fluctuations in many of these patterns, and they have evolved natural strategies to compensate for such environmental changes. But the magnitude and frequency of such changes often increase under timber management activities. This can produce excessive stress on resident salmonid populations and lead to long-term population declines.

Several comprehensive reviews are available that discuss the effects of forest management and forest use on fishery habitats. For example, Brown (1974) examined how multiple use affects the hydrologic cycle and water quality in forest streams. Chamberlin (1982) overviewed the effects of timber harvesting on stream ecosystems, while Everest and Harr (1982) reassessed the effects of silvicultural treatments. Everest (1984) compiled a list of suggested guidelines for coordinating wildlife and fishery habitat concerns with forest management activities. And, Everest et al. (1985a) recently reviewed the effects of timber management activities on salmonid habitat. Because such recently completed comprehensive reviews exist, we will only highlight some of the more important relationships. In general, most timber management activities, including harvest schedule, cutting systems, felling, yarding, and other silvicultural activities, have considerably less impact on fisheries habitat than do roads (Everest et al. 1985a).

It is important to emphasize, however, that most of the information available on the relationships between timber harvest and salmonid habitats has been obtained from studies focusing on anadromous fish species in forests of the Pacific Northwest. Little published information is available that relates forest harvest and reforestation directly to salmonid habitats in the central Rocky Mountains. Accordingly, the relationships discussed in the following paragraphs require, for the most part, additional study to evaluate their significance for salmonid habitats in the central Rocky Mountains.

Erosion and Deposition of Sediment

Sediment production from watersheds can increase dramatically when old-growth forests are logged. In fact, sediment levels in streams are likely to establish new equilibria above base levels that will remain as long as the watershed is intensively managed for timber (Everest et al. 1985a). The extent of the sediment related impact, however, is directly related to precipitation input volumes.

Logging can reduce the stability of slopes and cause mass erosion events (e.g., landslides, debris torrents), providing temporary or long-term barriers to migration (Everest et al. 1985a). One study in the Cascade Mountains found that landslides associated with clearcuts were 23 times more frequent than in forested areas (Lyons and Beschta 1983). When the slopes are near the limit of their safety factor, timber harvesting can accelerate rates of mass movement, especially in response to large storms and other major events. The extent or rate of mass movement, however, is hard to predict. If these events occur near or in stream channels, the impacts can be substantial. Recovery of slopes from mass movement and the impacts of their sediment on streams can continue for several decades (Chamberlin 1982). To avoid such impacts, managers must identify and avoid harvesting timber on slopes above or near stability margins (Chamberlin 1982).

When logging removes riparian vegetation, the stability of streambanks is decreased. This can substantially decrease the buffering effect produced by riparian vegetation and increase sediment delivery to streams (Everest et al. 1985a). This also can lead to stream channels becoming wide, shallow, and unstable, often with braided channels. As pools and gravels fill with sediment, they become less suitable habitat for trout. In small streams, channel aggradation produced by accelerated erosion can produce underground flows through deeper gravel layers during the summer and block migration routes. Barton et al. (1985) demonstrated the importance of buffer strips to trout in small Ontario streams.

Harvest techniques producing large bare areas, including tractor yarding and skidding, and harvest schedules that allow entire watersheds to be cut within a few years, can increase sediment production sufficiently to degrade salmonid habitat (Everest et al. 1985a). But felling has little effect on salmonid habitat, unless timber is felled directly into streams. Cable yarding and skyline systems produce little effect if riparian and floodplain areas are not damaged.

Various techniques can be used to reduce logging remnants and ready sites for reforestation. These include broadcast burning, brush raking or blading to pile slash and remove brush, and ripping compacted soils under landings and roads. The potential for erosion from these areas is directly proportional to the amount of soil exposed, extent of soil disturbance, slope gradient, precipitation input, stability of the exposed soil aggregates, and the proximity of the site to a stream (Everest and Harr 1982). Ripping compacted areas can actually reduce the long-term potential for erosion by enhancing conditions that speed revegetation (Everest and Harr 1982).

Despite the potential for substantially increased erosion as a consequence of logging, timber harvest operations in mixed conifer forests do not markedly increase erosion rates if the timber operation is done professionally (Heede 1987). Everest et al. (1985a) presented a number of management considerations that can help to minimize soil erosion. Briefly, these include approaches to minimize soil disturbances during logging and measures that can help to avoid impacting stream channels and riparian areas directly. Megahan and King (1985) also presented important considerations for identifying critical areas of forests for control of nonpoint sources of sediments.

Streamflow Alterations

Various studies in small, experimental watersheds have shown that logging can increase streamflows in both the Rocky and the Cascade Mountains, with the degree of increase dependent on characteristics of the harvest in relationship to the watershed topography and wind patterns (e.g., Harr et al. 1979; Leaf 1975a, 1975b; Troendle 1980; Troendle and King 1987). But other studies in both areas indicate that timber removal can result in no increase or even a decrease in seasonal runoff volumes, and that annual changes in runoff volumes may be associated more with annual differences in precipitation than with the percentages of forest harvested (Fowler et al. 1987, Harr 1980, Johnston 1984, Troendle and King 1985). Similarly, while various studies show that logging can increase flows from the small, experimental tributary subdrainages, these increases often appear to produce relatively minor changes in the main channels downstream (Harr et al. 1979, Troendle and King 1987). Indeed, the effects of operational logging on water flows from larger watersheds or regionally are unclear (Chamberlin 1982, Ziemer 1987). In general, however, it appears that vegetation management options will have little relationship with streamflows where precipitation is less that 15 inches per year, and some, but minimal, effect where they are less than 20 inches per year (Ziemer 1987).

The experimental studies show that small harvest openings (up to 8 tree heights) often appear to be more effective at trapping snow (Chamberlin 1982; Leaf 1975a, b). Due to this and to reduced evapotransporation because of the removed forest canopy, soils under logged forests tend to be more saturated with water in the spring. Consequently, runoff volumes tend to occur earlier in the spring (by as much as 1 month) and in greater volumes. In the central Rocky Mountains, this phenomenon can persist without lessening for up to 20 years, and can continue for at least 50 years (Chamberlin 1982, Troendle 1980). The extent and duration of the modification in runoff patterns depend on the distributions of the cuts, and their aspects, elevations, and distances from stream channels. It may be possible to manage these features of forest cuts to manipulate the delivery of water to stream channels for fisheries (Chamberlin 1982).

The amount of increase in spring runoff due to most timber cutting does not generally appear great enough to cause abnormal increases in streambed scouring. There is circumstantial evidence, however, that some streambed deterioration can occur when rain-on-snow events occur, or when very large percentages or entire watersheds are cut (Chamberlin 1982).

There is a prevailing belief that logging will benefit soils and water resources by desynchronizing the runoff from subcatchments; but, while timber removal can desynchronize subcatchment flows, it can, with equal likelihood, cause synchronization of previously desynchronized flows (Harr 1987). Whether such synchronization or desynchronization actually benefits salmonids and their habitats appears to depend on the specific watershed under discussion.

Alterations in Structural Habitats

Logging can leave large accumulations of large organic debris in streams (e.g., Duncan and Brusven 1985a). While this woody debris often can have a positive effect on salmonid habitat (e.g., rootwads can trap gravel, stabilize channel spawning areas, and create resting pools and cover), these accumulations also can provide temporary or long-term barriers to migration. Additionally, accumulations of logging slash can clog spawning gravels with fine organic particles, scour channels and cause instability during high flows, and cause routing and redeposition of gravels around debris accumulations (Everest et al. 1985a, Lisle 1986a). Finally, logging slash can also impede fishing access and other recreational opportunities (Chamberlin 1982).

As a consequence of debris torrents associated with accumulations of logging slash, numbers and biomasses of fish have been observed to be reduced by up to 90% in smaller streams and up to 50% in large streams, with little recovery occurring after 3 years (Everest and Meehan 1981a). Debris torrents also can degrade spawning habitats by increasing the percentage of fine sediments by over 90% in spawning gravels and cause at least temporary declines in densities and biomasses of aquatic invertebrates. But debris torrents also have been found to improve habitats for salmonids by creating new resting pools and new deposits of spawning gravels (Everest and Meehan 1981a).

Riparian vegetation removal, canopy removal, and excessive stream cleanup can reduce the amount of large woody material entering streams, reducing the structural complexity in the stream. Streams with varied channel morphologies, stable instream debris, and a variety of substrate sizes provide better cover for fish (Binns and Eiserman 1979). The presence of large woody debris recruited from riparian vegetation tends to create a series of check dams and plunge pools, which helps to collect sediment and provide protective cover for fish (Everest et al. 1985a, Lisle 1986a, Tschaplinski and Hartman 1983).

Harvest of large-diameter, old-growth forests predictably yields small-diameter second-growth. Little is known on how the resulting smaller sized logs entering streams will affect stream dynamics. Rates of sediment transport may increase, producing streambeds that are less stable and less productive over extended reaches (Everest et al. 1985a).

Clearcutting to the edge of streams can produce unstable streambanks that collapse during periods of high streamflows and decrease the suitability of the habitat for salmonids (Tschaplinski and Hartman 1983). Buffer strips riparian vegetation left along streambanks can substantially lessen logging impacts on streams by stabilizing banks and helping to maintain or increase debris important to salmonid winter habitat needs (Murphy et al. 1986).

Newbold et al. (1980) found that buffer zones also can have major influences on invertebrate populations important as food for fish. In comparisons to stream samples from uncut forested sites in northern California, benthic invertebrate samples from commercially logged stream sites had dissimilar compositions; had lower taxonomic diversities; and showed higher densities for total macroinvertebrate fauna, *Chironomidae*, *Baetis*, and *Nemoura*. In comparison, where streams in timber cuts had buffer strips greater than 30 m, the compositions and diversities of the benthic invertebrate communities were similar to those in uncut forest streams. (There are no similar studies on the effectiveness of buffer strips in the central Rocky Mountains.)

Decisions that can benefit fisheries habitats during reforestation include selection of suitable vegetation and appropriate planting distributions along the streams (Everest and Harr 1982). Planting conifers instead of deciduous species can reduce the contribution of allochthonous organic materials to streams and reduce salmonid productivities. When compared with closed canopies, open and semiopen stream reaches often are more productive. Research has shown that stream productivities may be enhanced by maintaining open streamside conditions, as long as solar heating does not become a problem. Where heating of the stream was a potential problem, plants of fast-growing species helped to lessen the problem. Also, reforestation programs that included long-term plans for the recruitment of large woody debris to streams can enhance salmonid habitats (Everest et al. 1985a).

Alterations in Water Temperatures and Incident Light

Removal of the riparian canopy adjacent to and within riparian zones produces an increase in summer temperatures and a decrease in winter temperatures. The extent of temperature change appears to increase as stream sizes become smaller, gradients less, and time after harvest shorter (Chamberlin 1982, Murphy and Hall 1981). Rarely does canopy removal cause onsite increases in stream temperatures that are lethal to trout, but the sublethal and downstream cumulative effects can produce physiological and recruitment stress, and substantial deterioration in trout populations (Everest et al. 1985a, Levno and Rothacher 1967, Patton 1973). Elevated stream temperatures can at times block fish migrations.

Increases in water temperatures and in incident light levels within the canopy opening also can increase photosynthesis and productivity by algae in streams (Chamberlin 1982, Duncan and Brusven 1985c, Murphy et al. 1986). This can shift the energetic bases of the affected streams to increasing contributions by autochthonous sources (Everest and Harr 1982). In turn, this can increase the densities of invertebrate food items for fish. The increased water temperatures associated with opening of the canopy can also extend the growing season and increase the growth rate for fish (Holtby 1988).

Various studies have shown that flora and fauna are often more abundant in stream sections with open forest canopies (Everest et al. 1985a). For example in the Cascade Mountains of Oregon, Murphy and Hall (1981) found generally greater biomasses, densities, and species richnesses of both fish and insect predators in clearcut sections where the stream was still exposed to sunlight 5 to 17 years following logging as compared with old-growth (>450-year-old) stream sections. In a related study, Wilzbach et al. (1986) suggested that growth rates of cutthroat trout in logged sections relative to forested sections increased as a consequence of the higher invertebrate productivities and, consequently, higher invertebrate drift rates and foraging successes in the opened stream sections. Johnson et al. (1986) found for southern Alaska streams that summer densities and growths of juvenile steelhead were greatest in clearcut sections relative to forested sections or sections with streamside buffer zones; during the winter, densities decreased by over 90% in the clearcut sections, but increased by 100% and 400% in the forested and buffered sections, respectively. Opening the canopy, however, can produce light or temperature conditions that can be damaging to some species (Everst et al. 1985).

It may be possible to regulate stream temperatures by judiciously designing a streamside harvest and revegetation plan to optimize light and solar heating of streams (Chamberlin 1982). With increased stream primary productivities and stream temperatures, increased fish productivities may result.

Alterations in Water Chemistries

Concentrations of nutrients (e.g., nitrate, calcium, and potassium) in streams generally increase following logging activities (Bowden and Bormann 1986, Chamberlin 1982, Hornbeck et al. 1975, Likens et al. 1977, Schillinger and Stuart 1978, Vitousek and Matson 1984). This results from both decreased uptake of nutrients because of the removal of plants, and increased decay of the remaining dead plant debris. The effects on fish spawning and rearing activities are usually not detectable (Everest et al. 1985a).

Accumulations of fine particulate organic materials accompanying slash can increase the biological oxygen demand (BOD) as this material decomposes, and cause dissolved oxygen levels in the stream or sediment to be depleted below concentrations necessary for survival of incubating eggs (Chamberlin 1982, Everest et al. 1985a). Accumulations of fine debris in spawning gravels associated with debris torrents can lead to decreased dissolved oxygen in the gravels, due both to reduced water permeability through the gravels and to decomposition of the fine organic materials entrained in the gravels (Everest and Meehan 1981a).

Fertilizers, fire retardants, herbicides, and insecticides can be used to protect or enhance forest resources. Salmonids can be affected directly through physiological changes produced by these forest chemicals or indirectly through changes in their habitat conditions. Forest fertilization appears to be used primarily in the Pacific Northwest to lessen nitrogen deficiencies, but even in this region only relatively small portions of all forests are actually fertilized (Norris et al. 1983b). Stay et al. (1978, 1979) investigated the additions of nitrogen compounds to Cascade forests. They found that the fertilization resulted in no prolonged increase in stream nitrogen levels. Breach of buffer strips along streams did, however, result in temporary increases in stream nitrogen concentrations following fertilization. The temporary increases in stream nitrogen were followed by temporary increases in primary productivity by stream periphyton, but this was not accompanied by concurrent increases in stream invertebrate populations that could be conclusively linked to the fertilization. And fish assays showed no mortality associated with the fertilizer applications.

Early fire retardants were borate salts, which were effective and long lasting but were also potent soil sterilants that inhibited revegetation (Norris et al. 1983b). Subsequently, bentonite clay suspensions in water were used, but they proved to have relatively low effectiveness. Most commonly used retardants today contain primarily ammonium phosphate or ammonium sulfate, plus several other minor ingredients (Norris et al. 1983b). Such compounds have known potential toxicities to salmonids, and their direct applications to streams can lead to mortality; little is known, however, about the actual exposure risk to salmonids when these materials are applied outside of streams (Norris et al. 1983b).

The direct chemical effects as well as the behavior and toxicity of commonly used forest insecticides and herbicides have been reviewed by Norris et al. (1983). This review included 2,4-D (PGBE ester), picloram, atrazine, MSMA, forsamine ammonium, glyphosate, dioseb, malathion, carbaryl, azinphos-methyl, carbofuran, and acephate. They concluded that under typical operational use of these pesticides, only dioseb and malathion presented a high potential for impacting fish (see also Woodward 1976).

In total, use of fertilizers, fire retardants, herbicides, and insecticides in forest management generally have little effect on salmonids and their habitats, if direct application to the waters is avoided (Everest et al. 1985a, Norris et al. 1983b).

Impacts Associated with Fire

Forest fires affect salmonid habitat predominately by increasing erosion and deposition rates of sediments; other potential effects of generally less importance include releasing nutrients, increasing water temperatures by removing streamside shade, and increasing summer base flows (Everest and Harr 1982, Tiedemann et al. 1979). The actual nature of the impacts largely depends on the characteristics of the burn, the site's physiography, its soils, climate, and, subsequently, the postburn recovery by the vegetation. Fire does often increase runoff from the burn areas (Albin 1979, Everest and Harr 1982, Helvey 1980, Tiedemann et al. 1979).

The extent that burning impacts soils and, consequently, erosion and salmonid habitat depends on the degree of soil heating, which in turn depends on the (1) types and amounts of fuels present, (2) intensity of the burn, (3) soil moisture content, (4) nature of the soil litter, and (5) and soil composition (Everest and Harr 1982, Tiedemann et al. 1979). Typically, 8% of the heat from a broadcast burn is absorbed by soils, while 92% enters the atmosphere. With water present, soil temperatures will not exceed 100°C until after all soil water evaporates (Everest and Harr 1982). Destruction of soil organic matter, which helps to bind soil particles together, will tend to increase potentials for surface erosion, nutrient loss, and effects to salmonids. Lowest strengths of aggregation and highest potentials for erosion tend to relate directly with increasing quartz contents in the parent materials (Everest and Harr 1982). In contrast, soils derived from parent materials that can form clays, including basalts, andesites, and gabbro-containing materials, tend to support high plant productivities, contain higher contents of organic materials, and have low erosion potentials. With very intense burns, such as occurred at some locations during the 1988 Yellowstone fires, soils can become hydrophobic due to the movement of polar substances from the litter into the soil; formation of hydrophobic soil can slow rates for revegetation and lengthen times of postburn erosion (DeBano and Rice 1973; Savage 1974; R. Swanson, Shoshone National Forest, Cody, Wyoming, personal communication).

Burns can cause release of chemicals, including nitrogen, phosphorus, sulfur, and other nutrients, when soil temperatures exceed their temperatures of volatility (Everest and Harr 1982). Other nutrients can be lost following fire via leaching and soil erosion. Such nutrient losses can retard recolonization by terrestrial vegetation and prolong sediment losses to streams (Tiedemann et al. 1979).

Stream temperatures can increase directly as a result of the burn and indirectly as a result of opening the canopy cover. Early observations on increased stream temperatures due to direct heating by fires in Yellowstone National Park found the fires did not impact resident salmonid populations (Albin 1979). Subsequently, the more intense fires in and around this park in 1988 were found to produce direct mortality of resident salmonids; the heat from some of these fires was great enough to split rocks on stream bottoms (R. Swanson, personal communication).

The extent of postburn temperature increase related to canopy removal depends on latitude, elevation, cloudiness, and the relative importance of groundwater contributions (Everest and Harr 1982). Some of the temperature increase may be mitigated by increased stream discharges following the burns, but the degree of this increase depends on the extent of the vegetation destroyed and, consequently, the decrease in transpiration.

In general, forest burns appear to have little impact on stream nutrient concentrations. While relatively slight increases in stream concentrations for some nutrients (e.g., calcium, magnesium, potassium, chloride, sulfate, phosphate, and total organic carbon) have been associated with runoff from burn areas, no harmful effects on water quality have been observed for the burning of slash, prescribed burns, or for wildfires in Yellowstone, the Pacific Northwest, or the Atlantic and Gulf Southeast (Albin 1979, Everest and Harr 1982, Richter et al. 1982, Tiedemann et al. 1979). Similarly, changes in water qualities associated with forest fires also have not been found to adversely affect benthic invertebrate populations (Tiedemann et al. 1979). Nutrient changes associated with forest fires will likely have little impact on salmonid habitats. Only when logging residues were burned directly in the streams were significant increases in concentration of nutrients and toxic metals observed (Everest and Harr 1982).

In conclusion, existing evidence suggests that except for potentially serious effects by increased sediment impacts, most forest fires will minimally impact streams and salmonid habitats. The potential severity of any fire-related impact will increase, however, as the severity of the fire increases; total destruction of the surrounding forests and riparian zones can produce dramatic and long-lasting negative impacts on aquatic habitats. Unfortunately, little documentation exists on the consequences of large fires on either short-term or long-term changes in water qualities or salmonid habitat quality.

Impacts of Grazing

Haugen (1985) summarized the position of the American Fisheries Society: "Livestock grazing is one of the multiple uses of the riparian area rangelands; however, years of improper grazing is one of the major reasons why so much of North America's public and private riparian area rangelands are in poor condition." He also stated the society's request "that land managers develop best management practices for managing riparian areas, and update these practices as new research and management information becomes available."

Livestock grazing produces a variety of changes in the riparian zone and stream channel that may be detrimental to salmonids. Platts and Raleigh (1984) presented the following categories:

1. Increased stream temperature due to loss of overhanging vegetation
2. Increased sedimentation from bank and upland erosion
3. Increased coliform bacteria counts (though Gary et al. (1983) and Johnson et al. (1978) noted this was unlikely to harm salmonid populations)
4. Increased channel width due to hoof-induced bank sloughing and consequent erosion
5. Channel alteration
6. Plant community alteration and/or vegetation loss
7. Loss of riparian vegetation due to channel degradation and lowering of the water table
8. Stream channel trenching or braiding and consequent shift to more xeric species in the riparian zone.

Blackburn (1984) suggested that the removal of vegetation by grazing could lead to increased raindrop impact, decreased soil organic matter, increased surface crusts, decreased infiltration rates, and/or increased erosion. Other impacts are reviewed by Kauffman and Krueger (1984), Meehan and Platts (1978), and Skovlin (1984).

When comparing heavily grazed streams with lightly grazed or ungrazed streams, the former were generally shallower, wider, and had less overhanging vegetation (Hubert et al. 1985; Platts and Nelson 1985a, 1985b; Stuber 1985). Trout populations often increased in response to reduced or no grazing (Prichard and Upham 1986, Stuber 1985). Platts and Rinne (1985) cited 16 studies demonstrating benefits to the riparian zone from eliminating grazing; trout populations had also increased in 12 of these studies. However, Platts and Nelson (1985a, 1985b) reported that the trout population did not increase in a grazing exclosure despite improvements in the riparian zone, and speculated this was due to water quality degradation caused by grazing upstream from the exclosure. And trout populations may not respond as rapidly to reduced grazing as riparian vegetation (Hubert et al. 1985, Platts and Nelson 1985b).

Successful management of salmonid fisheries on grazed ranges remains a difficult problem. Platts (1981a) and Skovlin (1984) concluded that no existing range management strategies were compatible with improving or protecting riparian vegetation. Complete exclusion of grazing by fencing riparian zones is expensive and controversial (Platts and Wagstaff 1984). But Platts and Rinne (1985) contended that riparian zones recover relatively quickly, and that undercut banks, important for use by salmonids as cover, may form. Additionally, Stabler (1983, 1985) reported that some formerly intermittent streams became perennial within fenced exclosures. He hypothesized this was due to vegetative regrowth, channel aggradation, and reduced soil compaction that led to increased infiltration rates and storage capacity.

Recent research indicates that some grazing strategies may be compatible with protecting riparian areas. Platts (1986a, 1986b) suggested that creating a separate riparian pasture or grazing during winter reduced livestock impacts. Bank damage by cattle and soil moisture have been linked (Marlow and Pogacnik 1985), thus grazing when soil moisture is less than 10% may prevent damage. Rest-rotation grazing or sheep grazing may be less deleterious than other livestock use (Platts 1986a). Nonetheless, rest-rotation grazing can overuse riparian zones if not aggressively managed (Platts and Nelson 1985a), and the benefits of sheep grazing are lost if the animals are allowed to concentrate in the riparian zone (Platts 1981a). In some circumstances grazing might be beneficial to salmonids; to reduce the sediment yield of highly incised streams with sheer banks, Siekert et al. (1985) suggested that summer or fall grazing could round these channels, and that such alteration would reduce sediment transport to streams containing salmonids.

Impacts of Water Development Projects on Streams

Reservoirs and diversion dams disrupt the natural continuum of streams by converting previous flowing water environments into standing water environments (Ward and Sanford 1983). Dams can block migration routes for resident stream organisms, alter flow regimes, and change temperature and nutrient patterns in downstream flows.

Over the impounded stretch of stream, many of those species most adapted to life in flowing waters are displaced by other organisms better adapted to life in standing waters. Concurrently, management problems change from managing stream species to managing reservoir species. Further consideration of these problems is beyond the scope of this review, but they have been previously discussed in depth (American Fisheries Society 1967, Hall 1971, Hall and Van Den Avyle 1986, Marcus 1987, Petts 1984).

Alterations in Flows Below Reservoirs

Natural daily and seasonal streamflow patterns can be dramatically altered downstream of reservoirs. With most reservoirs built for water supply and flood control, maximum spring streamflows are stored and minimum summer flows are augmented. In comparison with natural flow regimes, maximum flows downstream from such reservoirs are often markedly reduced, while summer minimum flows can be greatly enhanced. With diversion dams, downstream flows can be depleted during any season. Downstream of hydroelectric reservoirs, both the frequency and magnitude of daily flow fluctuations can be magnified greatly above the natural regimes. The abnormal fluctuations in daily and seasonal flow patterns created below reservoirs can lead to low-flow dewatering of spawning beds, and both low-flow and high-flow induced spawning interference, incubation mortality, and rearing mortality in resident fish (Fraley et al. 1986, Jacobs et al. 1987, Nelson 1986). Similar relationships also occur in invertebrate populations (Gislason 1985).

Downstream dewatering and desiccation are undoubtedly the worst of the possible adverse impacts on the stream and riparian habitats resulting from stream impoundment. For example, one study of the potential cumulative effects from microhydroelectric facilities on the Swan River drainage in northwestern Montana indicated that the associated dewatering could eliminate 23%, 19%, and 6% of the high quality rearing habitat for cutthroat, bull, and brook trout, respectively (Leathe and Graham 1983). Also, 20% of the known redds for bull trout would lie in dewatered reaches, which could significantly reduce this fishery.

Downstream Streambed Dynamics and Flushing Flows

In general, masses of sediment released downstream during construction of dams may be increased by more than 50% over historical sediment loadings (Petts 1984). The Montana microhydroelectric study further indicated that a significant negative correlation existed between juvenile bull trout densities and the amount of fine sediment in the streambed. Therefore, the authors projected that increased sediment delivered to the streams during construction of these facilities could significantly reduce this fishery (Leathe and Graham 1983).

Reservoirs and diversion dams disperse and disrupt flows and the kinetic energy patterns downstream. With this dispersal of energy, virtually all sediment carried by stream inflows settle to the reservoir bottom, causing aggradation (lifting of the river valley) upstream of the dams (Heede 1980, Newbury 1985, Petts 1984, Simons 1979). Consequently, waters discharged from reservoirs tend to be nearly devoid of suspended inorganic particles (Neel 1963, Petts 1984).

Alterations in downstream flows also redistribute the historical downstream supply of kinetic energy. In turn, this redistribution can substantially alter the morphology of the downstream channel as it accommodates the new energy regime in the regulated stream. In effect, a new "dynamic equilibrium" is created between the stream channel and the regulated flows (Heede 1980, Petts 1984). In most cases, however, a new equilibrium cannot be reached over the relatively short economic life (ca. 60 years) of most North American water projects, and instabilities in the downstream environments persist. Where instabilities cannot be controlled, major shifts to new streambed geometries occur.

Various instream forces contribute to changing the geometries of downstream channels. "Sediment-hungry" waters discharged from reservoirs rapidly entrain sediment to achieve equilibrium loads, degrading streambeds and eroding river banks (Heede 1980, Simons 1979). If uncontrolled, the degradation and erosion can endanger downstream structures, including bridges and roads. The extent of streambed degradation, however, is limited by the extent of bed armoring, which is a layer of cobble or rubble too large to be moved by the existing hydraulic conditions and which, in turn, protects finer particles that accumulate below the armor layer (Simons 1979). With this protection, potentially greater sediment masses erode from streambanks, decreasing bank stability and often destroying riparian vegetation. Following destruction of the streambanks, downstream sediment loads can further alter the downstream geometry of the stream (Petts 1984).

Beyond these problems, stream regulation can lead to clogging of the bed gravels with fine sediment. In unregulated streams, natural peak flows during seasonal or storm-related runoff events mix the upper streambed layers and flush accumulated fine sediment from the deeper layers. But in regulated streams where natural peak flushing flows are greatly reduced, fine sediment can accumulate in the deeper layers, clogging the free flow of water (R.W. Nelson et al. 1987; Reiser et al. 1985, 1987). This can adversely impact the intragravel habitat important to the survival of benthic insects, incubating eggs, and rearing larvae. Considering the biology of regulated streams, flushing flow releases from reservoirs are needed whenever sediment accumulations begin to adversely affect aquatic habitats and interfere with life functions of the resident organisms.

Numerous methods are available to measure accumulations and changes in accumulations of fine sediment within streambeds (Reiser et al. 1985, 1987; Wesche et al. 1977, 1985b, 1987c). Reiser et al. (1985, 1987) evaluated their usefulness. While many are good, no method or group of methods presently stand out as the best approach. Similarly, at least 15 potentially useful methods exist to determine magnitude, timing, and duration of flows necessary to flush deep sediment from streambeds, but again no method is clearly superior. One of the simplest approaches is the Tennant (1975) method, which is the most common method used in the western United States. Under this method an adequate flushing flow is defined as 200% of the average annual flow. However, no recommended duration for an adequate flushing flow is provided by this technique; many state resource agencies recommend a period of 14 days.

An alternative approach has been applied in Wyoming by Wesche et al. (1977). Using information from the literature plus their own field data and observations, they recommended that bankfull flows needed to be maintained for 3 days to obtain adequate flushing of sediment. Subsequent to their recommendations, natural precipitation produced a series of 3 flushing flows that met or exceeded the magnitude and duration prescribed. Analysis of the sediment content in the streambed prior to and following these events indicated that the recommended flushing flows were "somewhat successful" in removing the deposited sediment (Wesche et al. 1985b, 1987c).

While flushing flows can require high flow rates to remove sediment from deep deposits in some streambeds, these high flow rates sometimes can be very stressful and damaging to biota resident in the stream channel. Also, not all streams develop clogged beds as a result of stream regulation. For example, if large sediment sources, particularly those with large deposits of silt and clays, are lacking within the watershed, flushing flows may be unnecessary.

Reiser et al. (1985) concluded that determining potential needs for flushing flows required consideration of six points:

1. Physical location of the water development in relation to major sediment sources
2. Topography and geology of the area
3. Susceptibility of the drainage to catastrophic events
4. Sensitivity of target fish species and their life history stages to sediment depositional effects
5. Extent of human-induced activities within the drainage
6. Operational characteristics of the project.

Further, Reiser et al. (1985, 1987) noted that after needs for flushing flows are established, the problem becomes one of selecting the best time for implementation. Here, consideration is given to the species of fish present in the system; life histories of these species; the natural, historical runoff periods that the species have adapted to; and the potential flow available for flushing after the project has been completed. Ideally, the best time for implementing flushing flows is when greatest potential benefits to the biological community can be derived. Since no single, standard approach to determine appropriate flushing flows exists, the most reliable method for establishing required flushing-flow rates is to observe the effects of various test flow releases in the stream of interest (Reiser et al. 1985, 1987). Results from research currently underway by Wesche et al. (1983) should help improve our understanding of appropriate methods to determine flushing flow necessary for maintaining stream channels.

Reservoir Effects on Downstream Temperature Patterns

Amplitudes of both daily and seasonal natural temperature regimes also can be significantly altered downstream, depending primarily on the depth of the reservoir outlet. In fact, no daily thermal cycles exist for waters in the initial stream reaches below reservoirs having deep-water outlets. Daily fluctuations, however, do tend to increase with distance downstream below the dams, especially as tributary inflows contribute greater influences to the downstream flows (Ward 1974).

Because natural lakes and reservoirs that have surface water outlets release warmed water relatively soon after heating, temperatures in downstream waters can often be considerably warmer than upstream waters during much of the spring, summer, and early fall (Martin and Arneson 1978, Wright 1967). But in reservoirs having deep-water outlets, heated surface waters accumulate in the upper layers during most of the spring and summer, while cooler, deeper waters are discharged (Martin and Arneson 1978, Wright 1967). Then, during most of the fall and winter, cool influent waters tend to be stored in the upper layers of the reservoir, while warmer, deeper layers are discharged. Consequently, compared with temperatures in tributary inflows, water downstream of deep-release reservoirs tend to be cooler during the spring and summer and warmer during the fall and winter. This warming of downstream waters during the winter can alter or even prevent downstream ice formation (Neel 1963).

Studies of benthic macroinvertebrates below reservoirs indicate that the greater seasonal consistency of temperature reduces the numbers of resident species by providing competitive advantages to one or a few species; seasonal colder summer waters can slow larval development and prevent maturation and life cycle completion for some invertebrates; and seasonal warmer temperatures can speed the morphogenic development of some invertebrates, causing early (winter) emergence of some species (Ward 1974, 1976). Presumably, similar physiological and developmental relationships also hold for fish downstream of deep-release reservoirs. Certainly, warm summer temperatures can exclude former resident fish species below some reservoirs (Young and Maughan 1980).

Reservoir Effects on Downstream Nutrient Patterns

Another potentially major impact created downstream of reservoirs is the change of natural nutrient input patterns. Construction and filling of reservoirs flood terrestrial environments, leading to leaching of chemicals from flooded soils and from rotting forest debris. Decaying forest materials consume dissolved oxygen and elevate carbon dioxide, nutrients, and dissolved organic materials. As a result, heavy algae growths can be supported, undesirable levels of color and odorous substances may be produced, and conditions that enhance aquatic productivity or that can be toxic to aquatic life may result (Canter 1985, Hendricks and Silvey 1977, Sylvester 1965). Reservoirs, even at high elevations, also can permit the growth of nitrogen-fixing algae (Marcus 1987).

All of these substances become available for discharge from reservoirs, particularly from those with deep-water releases, since these materials tend to concentrate in the deep waters of reservoirs and lakes (Martin and Arneson 1978, Wright 1967). Consequently, productivities downstream of reservoirs can be enhanced over upstream rates, and this may lead to productivity enhancements in downstream fish populations (Marcus 1980, McConnell and Sigler 1959, Neel 1963, Pfitzer 1954). In the most extreme cases, the downstream discharges of chemicals can produce nuisance growths of aquatic weeds and toxic conditions to fish (Baxter 1977, Petts 1984).

Mitigating Downstream Impacts from Reservoirs

Various methods are available to at least partly mitigate the effects of blocked migration routes on stream fishes. R. W. Nelson et al. (1978) discussed various trap and haul systems, fishways, conduits and culverts, and turbine bypasses potentially useful to permit fish to pass beyond the dam structure. Also, Nettles and Gloss (1987) present an example of using radio-tagged salmon smolt to evaluate the effectiveness of a bypass structure and various orientations of trash racks at a small-scale hydroelectric site. They found that the bypass structure and angled trash racks significantly reduced entrainment through the penstock and turbine.

Methods are available to minimize the diversity of potentially adverse impacts and to manage fisheries below reservoirs (Marcus 1987, R. W. Nelson et al. 1978, Petts 1984). These include alternative scheduling of water releases to optimize needed flows, discharging water from alternative reservoir depths to manipulate water temperatures and water qualities, and physical enhancements of downstream environments.

Impacts of Mining and Mining Reclamation on Forest Water Quality

Mining-related activities can degrade water qualities and destroy physical habitats of salmonids. Often as a consequence of either metal or coal mining operations, dissolved concentrations of heavy metals and acids are increased in the adjoining surface waters. The resulting concentrations can be toxic to fish and other aquatic organisms and serve as instream barriers to migration (Canton and Ward 1977, Duff 1972, Gray and Ward 1977, Herricks 1977, Lewis 1977). Metals most frequently associated with mining-related toxicity to aquatic organisms in the central Rocky Mountains include zinc, copper, cadmium, and lead (Davies and Woodling 1980, Roline 1982). These metals can also bioaccumulate in organisms and present potential hazards to those consuming the organisms (Parkhurst et al. 1983, Roline 1982).

In addition to the direct toxic effects from these materials, metals generally precipitate from solution as solution pH increases and coat bottom materials with hydrous metal oxides. Frequently, this occurs downstream from a mining-contaminated stream after its confluence with an uncontaminated stream. These depositions can decrease abundances of periphyton and benthic macroinvertebrates and decrease potentials for spawning success downstream from the confluence (McKnight and Feder 1984, Roline 1982).

Instream sediment loads also tend to increase as a consequence of mining-related physical disturbances, including haul road activities, runoff from mine tailing piles, and washout of settling ponds (American Fisheries Society 1982). Additional sediment problems and the total, if only temporary, destruction of instream and riparian habitat also can result from mining of stream and lake bottoms (Harvey 1986, McLeay et al. 1987, Thomas 1985).

Acid- and metal-containing waters from mine sites can be treated through a variety of proven technologies to remove toxic potentials, allowing recolonization by fish and other aquatic organisms (e.g., Herricks and Cairns

1977, Reiser et al. 1982, Todd et al. 1982). Also, several viable methods are available for restoring mined streams (Gore 1985). The AFS Western Division Riparian Habitat Committee (American Fisheries Society 1982) has presented a selection of best management practices that are useful not only to protect and manage riparian communities along western streams associated with mining, but also to minimize associated instream impacts. These include adherence to NEPA (National Environmental Protection Act) regulations, thoughtful development of mine operation plans, creation of streamside buffer zones, construction of settling ponds, appropriate channelization and culvert installation procedures, definition of instream flows needs, providing for overburden segregation, control and treatment of mine wastewaters, correct mine site revegetation practices, and minimizing impacts from suction dredging.

Impact of Recreation, Fishing, and Fish Stocking

Impacts of Recreation

Generally, nonconsumptive recreation has not been directly linked with reductions in salmonid populations, but it has been coupled with water quality changes. Often, campgrounds or water-based recreation do not significantly lower water quality (Aukerman and Springer 1976, Gary 1982, Gary and Adams 1985). Nonetheless, Aukerman and Springer (1976) suggested that campgrounds accessible to automobiles did increase bacterial water pollution more than did backpack campgrounds. And Johnson and Carothers (1982) reported that recreationists may concentrate use in riparian zones and thus increase soil compaction and vegetation damage. Potter et al. (1984) noted that ski areas may contribute significant amounts of sediment to streams, and could desynchronize snowmelt and lead to unusually high peak flows. Snow making at ski areas also can reduce streamflows. Due to poor sewage disposal, homes located in forested watersheds may elevate concentrations of coliform bacteria and nutrients in streams (Ponce and Gary 1979, Potter et al. 1984).

Effects of Fishing and Fishing Regulations

Under some circumstances, anglers possess the ability to alter the size, structure, and distribution of fish populations. Bjornn et al. (1977b) stated that cutthroat trout in certain northern Idaho streams could be fished to extinction, if it was permitted. Alternatively, regulations restricting bait fishing and the harvest of fish under 30.5 cm did not increase the abundance of brown and rainbow trout in a northern Colorado stream (Klein 1974). Yet Wesche et al. (1987b) found stream accessibility to anglers was significantly related to the standing stocks of brown trout in southeastern Wyoming streams. Therefore, trout must be susceptible and accessible to anglers if regulations are to affect trout populations.

Restrictive regulations have produced dramatic changes in trout populations. The mean length of Yellowstone cutthroat trout in Yellowstone Lake has increased approximately 20 mm since the implementation of a two-fish, 330-mm maximum-size limit (Gresswell 1982). After 5 years of a three-fish, 330-mm minimum-size limit, cutthroat trout abundance increased fourfold in the St. Joe River in Idaho (Johnson and Bjornn 1978). Over the same interval, but under no-kill restrictions, the cutthroat trout population in Kelly Creek expanded nearly 1,200% (Bjornn et al. 1977b). Anderson and Nehring (1984) noted that a no-kill section of the South Platte River contained a larger biomass of trout and a larger proportion of trout greater than 30 cm long than did a section with an eight-fish-per-day limit. Thus fish populations can respond to restrictive regulations in some circumstances.

Not all trout species are equally susceptible to angling. In a small Idaho stream after 32 hours of fishing, anglers removed 31% of the cutthroat trout but only 7% of the brook trout (MacPhee 1966). Furthermore, fish greater than 15 cm were more vulnerable than smaller trout. Nyman (1970) noted that brown trout were less likely to be caught by anglers than were brook trout. Similarly, anglers more readily caught rainbow trout than brown trout, but both brown and rainbow trout greater than 30 cm were more susceptible to angling than were smaller fish (Anderson and Nehring 1984). Data from Favro et al. (1986) indicated that anglers more frequently captured both brown and rainbow trout greater than 33 cm than did electrofishing. Young (1986) noted that westslope cutthroat trout greater than 30 cm in length were more than twice as vulnerable to angling than were smaller trout in a western Montana river. Based on the available data, vulnerability to angling appears to be greatest for cutthroat trout, then brook trout, then rainbow trout, with brown trout seeming to be the least vulnerable. Also, susceptibility to capture by anglers apparently increases with trout size.

Impacts of Fish Stocking

Stocking consists of introducing nonnative or hatchery-produced fish to streams and lakes. Stocking has created a number of salmonid fisheries in streams historically lacking trout, e.g., the North Platte River drainage in Colorado and Wyoming (W.A. Hubert, Wyoming Cooperative Fish and Wildlife Research Unit, Laramie, personal communication). Additionally, the recovery of the federally threatened greenback cutthroat trout has relied heavily on the introduction of hatchery-reared fish (U.S. FWS 1983). Nonetheless, the potential for interaction between hatchery-raised and wild fishes should be evaluated prior to stocking (Li and Moyle 1981).

Trout stocking may have a significant impact on naturally reproducing trout populations. Vincent (1987) reported that 4 years after the cessation of stocking catchable-size rainbow trout in the Madison River in Montana, the fall populations of wild brown and rainbow trout had increased by 160% and 870%, respectively, despite an estimated 40% increase in angling pressure over those 4 years. He speculated that stocking produced increased stress via disruption of stable dominance hierarchies and thus led to greater mortality in wild fish. Bohlin (1977) noted that "resident" brown trout tended to repel "introduced" fish, but perhaps this behavior could be overridden if densities of introduced fish were high. Pollard and Bjornn (1973) noted that introduced rainbow trout took over feeding sites previously occupied by juvenile steelhead.

Introductions of nonnative salmonids can also lead to hybridization. Hybridization with rainbow trout may pose the greatest threat to the preservation of various subspecies of cutthroat trout (Leary et al. 1984). And hybrids between native bull trout and introduced brook trout have been found in one Montana stream (Leary et al. 1983). Furthermore, the genetic content of hatchery trout and wild trout of the same species often differ radically; Allendorf and Phelps (1980) found a 57% reduction in polymorphic loci, a 29% reduction in the average number of alleles per locus, and a 21% reduction in the average heterozygosity in hatchery stocks of westslope cutthroat trout compared with wild stocks. Ryman and Stahl (1980) noted similar changes in hatchery populations of brown trout.

Hybrids between different species (and between different stocks of a single species) tend to possess greater developmental instability due to the loss of coadapted gene complexes, and this has been associated with reduced fitness in several interspecific hybrids of salmonids (Leary et al. 1985). Thus, if hatchery fish breed with wild fish, their offspring may suffer greater mortality. Fraser (1981) reported that survival of wild brook trout was greater than that of a hybrid of two wild stocks. In comparisons of wild, hatchery, and wild-hatchery crosses of steelhead trout, wild fish had significantly greater survival in streams than did hatchery fish or wild-hatchery fish (Reisenbichler and McIntyre 1977). Interestingly, hatchery fish survived significantly better in a hatchery pond than did the other two groups.

Urbanization Impacts on Water Quality, Aquatic Habitats, and Fisheries

A variety of studies have shown that urbanization can affect streams through physical changes, including removal of riparian vegetation and channelization, and chemical changes, including point and nonpoint pollution sources. Overall, urbanization tends to accentuate peaking flows of stormwater runoff; elevate suspended and bed loads for sediment; plus increase dissolved concentrations of petroleum products, heavy metals, other toxicants, and nutrients, particularly phosphorus and nitrogen (cf., Davies 1986, Dillon and Kirchner 1974, Schillinger and Stuart 1978, Shapiro and Pfannkuch 1973).

A series of studies comparing the physical, chemical, and biotic conditions of Kelsey Creek in Bellevue, Washington, with conditions in a nearby nonurban reference stream provides perhaps the most complete analysis of the influences by urbanization on the trophic structure and salmonid fishery in a stream (Pedersen and Perkins 1986, Scott et al. 1986, Sloane-Richey et al. 1981). Urbanization changed the stream's hydrologic regime and its channel morphology, which caused increased runoff scouring of attached algae and allochthonous materials, increased suspended sediment, decreased streambed stability, and increased light and nutrient availability (Sloane-Richey et al. 1981). Removal of riparian vegetation reduced inputs of allochthonous debris. Also, the deposits of fine silts had relatively lower carbohydrate contents. Thus, not only was the temporal availability of particulate organic material limited in the system, but its value as food was also limited. Its quality was, however, sufficient to support invertebrate growth (Richey 1982).

Comparing invertebrate populations colonizing artificial substrates in these streams suggested there was no significant difference in their population densities, but the biomasses sometimes were 4 times greater in the urban stream (Pedersen and Perkins 1986, Richey 1982). The dominant taxa in the urbanized stream were burrowers tolerant of unstable substrates. Kelsey Creek also lacked benthic taxa adapted to utilize large particulate organic material, while such taxa were abundant in the reference stream. [This is consistent with the observation by Sloane-Richey et al. (1981) that large organic debris was largely absent from Kelsey Creek.] Amphipods, one of the most common invertebrates in Kelsey Creek, often were flushed from the creek during runoff events.

The abundance of invertebrates in Kelsey Creek was adequate, however, to support populations of coho salmon and cutthroat trout; total biomass of fish was not markedly different between these streams (Scott et al. 1986). Growth rates for salmonids in the urban stream, in fact, exceeded those in the reference stream, and its cutthroat reached the smolt stage and migrated downstream 1–2 years earlier than occurred in other less productive streams in the region. This increase in growth rate for salmonids in the urban stream appeared to be related to warmer stream temperatures caused by removal of riparian vegetation, and the sometimes greater absolute densities of benthic invertebrates in the urban stream.

Neither increased storm runoff peaks nor increased pollutant load pulses appeared to increase early downstream migration or displacement of salmonids from the urbanized stream, though some evidence suggested that other fish taxa may have been affected by such events. Streambed scouring and instability did appear to markedly reduce spawning habitats available for coho and sculpin, causing egg mortality in these species. Most salmon in Kelsey Creek appear to hatch in a tributary stream that drains a relatively undeveloped section of the watershed. Cutthroat trout appeared to be much more successful in finding satisfactory spawning habitat in Kelsey Creek.

In total, fish diversity in urbanized Kelsey Creek was less than in the reference stream: proportions and biomasses of cutthroat trout were greater, while those for coho salmon, longnose dace (*Rhinichthys cataractae*), largescale suckers (*Catostomus macrocheilus*), and sculpin (*Cottus* spp.) were less. Age-0 cutthroat were found to predominate in Kelsey Creek and other urbanized streams in the drainages of the Puget Sound. Overall for tributaries to Lake Washington, proportions of cutthroat trout directly correlated with proportions of impervious area (e.g., asphalt or concrete) within the watersheds, while the fish diversity inversely correlated with impervious areas (Scott et al. 1986). Thus, cutthroat trout appear to be somewhat less sensitive to disturbance relative to other salmonid and nonsalmonid taxa in these streams.

Cumulative Impacts

Impacts to aquatic systems and fisheries rarely occur in isolation, but rather often originate from multiple sources, produce multiple effects, and extend to subsequent downstream reaches. Regulations promulgated by the Council of Environmental Quality under the National Environmental Policy Act of 1970 require that environmental impact statements contain evaluations of cumulative effects, which are defined (40 C.F.R. 1508.7) as

> . . . the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time.

Coats (1987) and Coats and Miller (1981) present an 18-compartment conceptual model showing how timber harvest can relate to hydrological and erosional processes involved in cumulative watershed effects. Salo and Cederholm (1980) suggest five processes that can act in a cumulative manner on aquatic organisms in streams:

1. Increases in suspended or deposited sediment
2. Changes in diurnal or seasonal temperatures producing patterns outside the natural extremes
3. Changes in the physical habitat produced by excessive increases or decreases in loads of organic materials
4. Detrimental changes in other water quality parameters
5. Detrimental changes in water quantity and flow regimes.

To deal with the general lack of specific data on effects, cumulative watershed effects (CWE) evaluations are often completed through development of models using equivalency tables based on different forest management activities and effects threshold indices (e.g., Klock 1985). Then, through these tables, various management activities can be evaluated and rescheduled, if necessary, to "prevent" cumulative effects from occurring. Such rescheduling may include postponing certain percentages of scheduled timber harvests. However, there has been little evaluation on whether such CWE modeling approaches are achieving their goals (Ice 1987).

Many such CWE models incorporate considerable potential error (Grant 1987, Harr 1987). First, there is substantial doubt that actual effects thresholds exist. Second, all models that attempt to minimize impacts by rescheduling include the assumption that the principal downstream effects are caused by peak flow increases; however, recent results indicate that increased sediment delivery and transport of large woody debris are more important than peak flows (Grant 1986, 1987). Third, such models make no allowance for basin differences in hydrologic responses to forest practices and geomorphic responses to hydrological changes. Finally, current approaches to mitigate impacts do not include monitoring to determine whether the methods are working. Overall, many such models include the erroneous belief that "*simplicity can be willed on the forest hydrological system*. This belief encourages the implementation of simplistic guidelines, the adoption of arbitrary thresholds of concern, and the search for all-encompassing methodologies to predict consequences of forest activities on water resources" (Harr 1987:137).

Grant (1987) emphasizes that cumulative effects must be analyzed individually. Each case will involve a wide range of hydrologic and erosional phenomena, complexly linked by a cascading series of causes and effects (Grant et al. 1984). For example, Grant (1987) suggests that potential effects related to peak flows, one of several classes of potential cumulative watersheds effects, can be decomposed and addressed through a series of five questions:

1. How do harvest activities affect peak flows?
2. How are increased peak flows routed to and through the stream network?
3. How do increased peak flows in channels affect frequency of movement of bed and bank materials?
4. How do changes in frequency or timing of movement of bank and bed materials affect channel structure and stability?
5. How do changes in channel structure or stability affect aquatic communities?

To address such questions requires integrating existing knowledge, future research results, and specific data from the field.

APPROACHES FOR MANAGING AND EVALUATING SALMONID HABITAT

Methods for Monitoring and Evaluating Salmonid Habitat Quality

Once the variables that constitute salmonid habitat have been defined, the next step is to evaluate the quality of the habitat based on some measure of the variables. Following this, the variables can be monitored through time to detect changes in habitat quality. Unfortunately, researchers have failed to consistently define, evaluate, and sample the aforementioned variables, making comparisons of different studies difficult (Armantrout 1982). Various researchers have attempted to develop a field-validated, statistically sound, unified system of habitat assessment (Platts et al. 1983b; USDA Forest

Service 1988). Note, however, that the definition of habitat quality is difficult and equivocal, and that a single definition of quality does not apply to all salmonid species or their habitats (Binns and Eiserman 1979, Bowlby and Roff 1986, Kozel 1987). Consequently, we chose to list a portion of the currently available techniques (table 3).

Techniques for Restoration, Rehabilitation, and Enhancement

A frequently employed approach to manipulating the physical habitat in streams is by altering the actual appearance of the stream channel by placing artificial structures within the stream. In use, these structures can improve fisheries habitat by reducing bank erosion, increasing habitat diversity through creating new series of riffles and pools, providing cover, or improving substrates suitable for spawning. For example, installing small rock dams and deflectors in a small mountain stream in Quebec not only increased the biomass of brook trout, but also increased biomasses of aquatic invertebrates and increased usage of the stream by mink and raccoons (Burgess 1985).

Table 3.—Some techniques for directly or indirectly assessing salmonid habitats.

Sample design and analysis	Green 1979 Seber 1982 Armour et al. 1983 Platts et al. 1983b Snedecor and Cochran 1980
Stream habitat surveys	Armour and Platts 1983 Cuplin 1974 Duff and Cooper 1976 Greentree and Aldrich 1976 Heller and Baker 1974 Oswood and Barber 1982 Pfankuch 1975 Platts et al. 1983b Platts et al. 1987 Rinne 1985 Seehorn 1970 USDA Forest Service 1970 USDA Forest Service 1988
Impact assessment	Green and Vascotto 1978 Hall et al. 1978 Hurlburt 1984 Stewart-Oaten et al. 1986
Fish population estimates	Hankin 1984 Hankin 1986 Hankin and Reeves 1988 Platts et al. 1983b Rinne 1985 Slaney and Martin 1987 Van Deventer and Platts 1983
Invertebrate assessment	Jackson and Resh 1988 Winget 1985
Habitat variable definition	Binns and Eiserman 1979 Bisson et al. 1982 Bovee and Cochnauer 1977 Bowlby and Roff 1986 Everest et al. 1982 Milhous et al. 1981 Platts et al. 1983b Platts et al. 1987 Rosgen 1985 Wesche 1980

Wesche (1985) presented a comprehensive discussion on the relationship of fish with components of their habitat, including stream velocity, substrate, depth, invertebrate drift, spawning areas, and protective cover. In reviewing the use of artificial structures, he concluded that the most commonly used within-channel structures are current deflectors, low profile overpour dams and weirs, bank cover, and boulder placement. Table 4 lists the commonly used options available for enhancing stream habitats. Both Wesche (1985) and R.W. Nelson et al. (1978) provide guidance on appropriate application, construction, and installation techniques for the structures. However, Reeves and Roelofs (1982) noted that the changes in fish populations caused by such enhancement are rarely evaluated. Thus funds for evaluation of structures should be considered a part of habitat improvement projects.

Developing and Using Models to Estimate Potentials for Stream Fisheries

Maximum habitat available for stream fisheries depends ultimately on the volume of water flowing down the stream during the specified period. Consequently, much research has focused on relationships among stream fisheries, habitats, and flows. This research has been extensively reviewed by Wesche and Rechard (1980), Loar and Sale (1981), Terrell (1984), Armour et al. (1984), EA Engineering, Science and Technology, Inc. (EA ES&T 1986), and Fausch et al. (1988). Included in the review by EA ES&T (1986) were 54 instream flow and habitat quality methods, while 99 models were reviewed by Fausch et al. (1988). Rather than again reviewing the available models, this section focuses on considerations important when developing, evaluating, and using these models.

Most instream flow and habitat investigations endeavor to develop techniques and models through which standing crops and/or other measures of biological productivity, generally pertaining to fish, can be described or predicted using a set of habitat variables. Underlying all of the resulting models is the premise that for flow, as well as for other environmental measures, there are definable limits, beyond which conditions become unsuitable for fisheries. Somewhere between these upper and lower extremes, optimal conditions exist that grade predictably to the unacceptable conditions. Ultimately, after the appropriate relationships are derived, it is hoped that a few well-chosen, easily obtained measurements made for a stream can be entered into a model to predict the stream's potential carrying capacity and/or standing stock of fish.

Present models, which have been developed using both qualitative and quantitative approaches, include as few as 1 to as many as 21 input variables (EA ES&T 1986). Some models use variables that are transformed from or

Table 4.—Trout habitat enhancement structures and practices with references describing their application. Information on unreferenced items can be found in R. W. Nelson et al. (1978) and Wesche (1985).

- Current deflectors
  - Rock-boulder deflectors
  - Gabion deflectors—House and Boehne 1986
  - Double-wing deflectors—Seehorn 1985
  - Underpass deflectors
  - Half log deflectors—Stuber 1984
  - Boulder placement—House and Boehne 1985
  - Trash catchers
- Low-profile check dams
  - Rock-boulder dams—Reeves and Roelofs 1982
  - Single and multiple log dams—Rinne 1981
  - Plank or board dams
  - Gabion check dams—Binns 1982b
  - Beaver introduction
- Bank cover treatments
  - Log overhangs
  - Artificial overhangs—Brusven et al. 1986
  - Tree revetments—Binns 1986
  - Bank revegetation—McCluskey et al. 1983
  - Riprap—Knudsen and Dilley 1987
  - Erosion-control matting
  - Streambank fencing
  - Grazing control
  - Buffer strips—Barton et al. 1985
  - Adding large organic debris—Boehne and Wolfe 1986
- Sediment removal
  - Settling basins—Hansen 1973
  - Scour and storage—Klassen and Northcote 1986
  - Riffle sitter—Hall and Baker 1982

derived (recombined) using previously measured variables. Overall, the variables include details on basin morphology, channel morphology, flow rates, habitat structure, species present, and other physical and chemical measures.

Assorted real and potential problems are associated with all existing habitat models (EA ES&T 1986, Fausch et al. 1988). Probably most damaging in the long term is the lack of standard methods for use in measuring habitat variables (Armantrout 1982, Fausch et al. 1988, Scarnecchia and Bergersen 1987, Wesche 1983). Because methods used to collect data are reflected in any resulting models developed, single data sets cannot be accurately explored using otherwise similar models that were developed using alternative sampling methods.

Various problems associated with many of these models have statistical bases (Fausch et al. 1988). First, presentations for most models lack information necessary to critically evaluate how the model was statistically (or otherwise) selected or how the model may perform in general applications.

Second, many of these models are based on small sample sizes. This can limit the potential applicability of the model to only conditions within the restricted range of the data used in its development. If models are used to extrapolate outside these limits, the resulting predictions can be biased and unreliable. This restriction must be strongly considered when applying models to habitat conditions other than those used to develop the model. Emphasis of this detail is often neglected by model developers when promoting general application of their own models, and ignored by investigators when applying and criticizing models developed by other investigators (see Scarnecchia and Bergersen [1987] for an exception).

Third, rarely have errors associated with measuring habitat variables been evaluated during model development (Fausch et al. 1988). If measurements upon which models are based are biased, the model will yield similarly biased predictions. Most models have not been tested with data that was not used in developing the models; thus we know little of their overall realism, precision, or generality (Levins 1966).

Fourth, use of derived variables, including indices, ratios, proportions, and percentages, creates a potential tangle of statistical problems (Green 1979, Sokal and Rohlf 1981). These can include (1) greatly increasing the variance of the derived variable relative to the original variables; (2) potentially yielding biased estimates of the true means for derived variables; (3) subsequently, potentially producing unusable, nonnormal, and possibly intractable distributions for derived variables; and (4) a frequent tendency to obscure rather that clarify intervariable relationships. Despite these problems, we sometimes have no choice other than to use derived variables.

Various potentially unreasonable assumptions about habitat relationships are also implicit in many of these models (Annear and Conder 1984, EA ES&T 1986, Fausch et al. 1988, Mathur et al. 1985). These include the assumptions (1) that fish primarily respond to average water velocities at some defined depth below the surface (e.g., 0.6X); (2) that stream depths, velocities, and substrates are not related to each other (i.e., an important underlying assumption in regression analysis, through which most habitat models are developed, is that the independent variables are uncorrelated); and (3) that large amounts of suboptimal habitat are equivalent to small amounts of optimal habitat. Yet, studies show that fish respond more to flow differences in the microhabitats; that many habitat variables, including stream depths, velocities, and substrates, are often highly correlated; and that suboptimal habitats can often be uninhabitable (Fausch 1984, Fausch et al. 1988, Grossman and Freeman 1987, Moyle and Baltz 1985, Shirvell and Dungey 1983).

Finally, most models often include the assumption that measured densities or biomasses of fish in streams are at carrying capacities for the habitat and that these carrying capacities are defined by the physical and chemical conditions measured in the habitat. This assumption precludes such effects as predation (including fishing), competition, or some other unmeasured environmental variable as having any potential influences on the populations (Conder and Annear 1987). For example, both Grossman et al. (1982) and R. L. Nelson et al. (1987) suggest that unpredicted environmental disturbances can be the most important regulator of fish community structure.

Of the available models, the Habitat Quality Index (HQI), developed by Binns (1979, 1982a) and Binns and Eiserman (1979) in Wyoming, often provides reasonable predictions of standing crops for trout in unregulated, coldwater streams in the central Rocky Mountains (Annear and Conder 1983, 1984; Conder and Annear 1986; EA ES&T 1986). But this model, too, has the problem of including many nonindependent variables (i.e., multicollinearity). And, it includes many subjective variables that can be difficult to score (see Scarnecchia and Bergersen 1987).

Other models have recently been developed relating salmonid population to habitat variables in the central Rocky Mountains (e.g., Chisholm and Hubert 1986, Lanka et al. 1987, Scarnecchia and Bergersen 1987, Wesche et al. 1987b, Winkle 1988). These models all show good applicability to the specific range of habitat conditions that they address. Yet, these models still are limited by the range of habitat data included in model development, inclusion of nonindependent variables without an evaluation of the potential problem caused by the inherent multicollinearity, and inclusion of sometime subjective and/or derived variables. The potential severity of these problems in these and other such models awaits evaluation. As the quest for the "best" salmonid-habitat model continues, we will likely find in the end that the ultimate solution will be a series of highly specific models developed to address specific habitat related problems for specific types of habitats.

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This report includes a general review and analysis of the literature summarizing the available information relevant to salmonid-habitat relationships, particularly as it pertains to the central Rocky Mountains. Also included is a comprehensive indexed bibliography.

Keywords: See Index to Bibliography

U.S. Department of Agriculture
Forest Service

Rocky Mountain Forest and Range Experiment Station

Rocky
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The Rocky Mountain Station is one of eight regional experiment stations, plus the Forest Products Laboratory and the Washington Office Staff, that make up the Forest Service research organization.

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Research programs at the Rocky Mountain Station are coordinated with area universities and with other institutions. Many studies are conducted on a cooperative basis to accelerate solutions to problems involving range, water, wildlife and fish habitat, human and community development, timber, recreation, protection, and multiresource evaluation.

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