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MEMORANDUM FOR: Peer Review Panel  
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FROM: Elizabeth Gaar, Chief, Habitat Conservation Branch  
Steven Landino, National Marine Fisheries Service  
Habitat Conservation Planning Program

SUBJECT: Review of the draft version of an *Ecosystem Approach to Salmonid Conservation, Part II: Planning Elements and Monitoring Strategies for Salmonid Conservation Efforts*, May 1996

Attached is the draft of the second of three documents prepared by ManTech Environmental Technology, Inc., to address development of salmonid conservation efforts for non-federal lands throughout the Pacific Northwest. You may recall that the review of the first and third documents of this three document set occurred last fall. The intent of these three documents was to serve as a basis upon which resource ecologists and private landowners might jointly develop landscape management strategies to conserve salmonids in a way that is consistent with the Endangered Species Act.

This draft document, Part II, lays out the planning elements and monitoring strategies that should form the basis of watershed planning or other conservation programs for anadromous salmonids on non-federal lands. It builds on the technical information provided in Part I, by providing recommendations regarding the important considerations for landowners, salmonid ecologists and managers, in the design of salmonid conservation plans and programs.

You have graciously agreed to review this three document set, and unfortunately, unanticipated delays have prevented us from getting this important part to you sooner. Nevertheless, we appreciate your willingness to take the time to complete the review of this document.

Please be thorough in your review but brief in your comments. As before, we would greatly appreciate it if you could summarize your comments in no more than six pages. We would like to have your comments no later than June 3, 1996. If you have any questions, please feel free to contact me at (360) 753-6054, or Steve Ralph of the U.S. Environmental Protection Agency at (206) 553-4497. Thank you very much.

Attachment



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D-051693

**An Ecosystem Approach to Salmonid  
Conservation  
Volume I: Technical Foundation**

by:

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The Agencies also suggested local and regional experts who have provided materials to be reviewed and synthesized in this document, provided written contributions to sections of the manuscript, and reviewed earlier drafts of the document. This included Robert Beschta and Stanley Gregory, Oregon State University; Patricia McDowell, University of Oregon; and Frank Reckendorf and Vaughn Brown, Soil Conservation Service (retired). In addition, the following were designated as primary contacts for their respective states: Jim Steele, California; Don Zaroban, Idaho; David McAllister, Oregon; and John Mankowski, Washington. These state contacts assisted in obtaining documents for review and periodically assessed the list of materials we were reviewing to insure that we did not overlook significant documents or practices.

Besides the authors, other writers graciously contributed certain sections. Stanley Gregory (3.8), Patricia McDowell (3.1 and 3.5) and Greg Linder (5.1.2). The National Marine Fisheries Service also contributed portions on irrigation (6.7), and harvest (6.9). Cynthia Chapman performed much of the legal research and summarized the State and Federal laws, designed the POPYRUS database, and wrote bibliographic code. Susan Brenard and Frances Beck entered data, maintained the database and the report files, and aptly generated the reports.

Many others, too numerous to mention individually, have met with us to discuss specific issues and suggest documents to review (or others to talk to), have provided us materials to review, or have offered us opportunities to access libraries and document collections.

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**An Ecosystem Approach to Salmonid Conservation**

## 1.0 Executive Summary

### Introduction

There is substantial evidence and growing concern for declines of salmonids in the Pacific Northwest. Anadromous salmonids returning to the Columbia River to spawn have declined from historical highs of 10-16 million wild fish to fewer than 2 million fish, most of which are of hatchery origin. At least 106 salmon stocks have been extirpated, 214 are at high or moderate risk of extinction, and many are being reviewed for listing under the Endangered Species Act, as are several resident species and stocks. Salmon fisheries along coastal regions of Oregon and California have been dramatically curtailed due to dwindling numbers of fish and increasing concern for wild stocks. A number of natural and anthropogenic factors have contributed to these declines: hydropower operations, over exploitation, artificial propagation, climatic and oceanic changes, and destruction and degradation of habitat through land-use and water-use practices. Although the relative impact of these different factors varies among basins and river systems, habitat loss and degradation are considered contributing factors in the decline of most salmonid populations.

This document is intended to provide a comprehensive framework for understanding salmonid conservation principles in an ecosystem context. Aquatic habitats critical to salmonids are the product of processes acting throughout watersheds and particularly within riparian areas along streams and rivers. This document is founded on the premise that salmonid conservation can be achieved only by maintaining and restoring these natural processes. If ecosystems are allowed to function in a natural manner, habitat characteristics favorable to salmonids will result, and fish will be able to reinvade and populate historical habitats, recover from earlier stressors, and persist under natural disturbance regimes. This process-oriented approach is in concert with recent federal and state strategies for management and conservation of forest resources that emphasize watershed and landscape-level functions of ecosystems.

After briefly reviewing evidence of trends for Pacific Northwest salmonids (Section 2), we discuss physical, chemical, and biological processes that affect aquatic ecosystems and the salmonids that inhabit them (Sections 3-4). We then present an overview of habitat requirements of salmonids, including elements that are essential to the general health of aquatic ecosystems, as well as specific habitat requirements at each life stage of salmonids (Section 5). We then discuss how human activities affect watershed and instream processes, focusing on effects of logging, grazing, agriculture (including irrigation withdrawal), mining, and urbanization (Section 6). Effects of dams, species introductions (including hatchery practices), salmon harvest, and other factors are given brief treatment, since these topics were outside the scope of this project. We also review the influence of climatic and oceanic factors on salmonids and how these relate to salmonid conservation (Section 7). Finally, we present an overview of management practices and programs that reduce the detrimental effects of human activities on salmonids (Section 8), followed by a discussion of federal laws and regulations relevant to the conservation of salmonids (Section 9).

The document focuses on anadromous salmonid species, including five Pacific salmon (chinook, coho, chum, pink, and sockeye), trout and char with both resident and anadromous forms (rainbow, cutthroat, and bull trout), and strictly resident species (mountain whitefish). The areal scope was intentionally limited to the regions of the states of California, Idaho, Oregon, and Washington that have supported salmonid populations. For most subject areas, we have relied heavily on comprehensive literature reviews and syntheses already available in the scientific literature. For subject areas where no such summaries were available, we have conducted more extensive literature reviews.

### Physical and Chemical Processes

The physical and chemical characteristics of streams, rivers, lakes and estuaries of the Pacific Northwest are the manifestation of processes operating at many time scales. Tectonic activity and glaciation have continually reshaped the landscape of the Pacific Northwest over millions of years. Alternating glacial and interglacial periods have caused changes in vegetation cover and geomorphic processes. In response to these changes, river channels have shifted from unstable braided channels to relatively stable, meandering channels, achieving present conditions about 6000 to 8000 years ago. Modern coniferous forest communities developed over much of the coastal region within the last 2000 to 5000 years.

Over periods of decades to centuries, large floods, fires, and mass wasting have been dominant influences on river channels. These disturbances can cause abrupt changes in habitat conditions, reconfiguring the stream channel, transporting streambed materials, depositing large quantities of coarse and fine sediments to streams, and altering hydrologic and nutrient cycling processes. These changes may persist for decades or more, influencing the relative suitability of habitats to various salmonids.

At the watershed level, the major processes that affect the features of aquatic ecosystems are hydrology, sediment transport, energy transfer, nutrient cycling/solute transport, and delivery of large woody debris to streams.

Streamflow and stream channel features are largely determined by transport of water, sediment, and wood from the watershed to the stream, especially the timing, duration, intensity, and frequency of extreme events.

Runoff from the watershed affects stream habitats directly by determining the timing and quantity of streamflow, and indirectly by affecting the processes of energy transfer, sediment transfer, and nutrient cycling/solute transport. Runoff is a function of precipitation patterns, evapotranspiration losses, and infiltration rate, which in turn are affected by watershed characteristics, including local climate, topography, soil type, slope, and vegetative cover. Hydrologic regimes of streams in the Pacific Northwest can be divided into three general patterns: rain-dominated systems, which are hydrologically flashy due to frequent rainstorms during the winter (coastal mountains, lowland valleys, and lower elevations of the Cascade and Sierra Nevada Mountains); transient-snow systems, which exhibit both rain and snow during the winter and may experience high flows associated with rain-on-snow events (mid-elevation of the Cascade, northern Sierra Nevada, and Olympic Mountains); and snow-dominated systems, where most precipitation falls as snow during the winter months and is delivered to streams in the spring as snow melts (higher elevations of the Cascade, Sierra Nevada, Olympic, and Rocky Mountains, and mid-elevation areas east of the Cascade/Sierra Crest).

Sediment from upland areas plays a major role in determining the nature and quality of salmonid habitats in streams, rivers, and estuaries. Sediment is generated from surface erosion or mass wasting. Surface erosion occurs when soil particles are detached by wind, rain, overland flow, freeze-thaw, or other disturbance (animals, machinery). Mass wasting (slumps, earthflows, landslides, debris avalanches, and soil creep) result from weathering, freeze-thaw, saturation, groundwater flow, wind stress transferred to soil by trees, earthquakes, and undercutting of streambanks. Bank erosion and bedload movement occur naturally during high flows, but may be exacerbated where riparian vegetation that stabilized banks is removed or when peak flows are increased by human activities. Watershed characteristics affecting sediment transport include climate, topography, geology, soil type and erodibility, vegetative cover, and riparian zone characteristics. West of the Cascades, mass wasting is the major source of sediments in undisturbed systems, whereas east of the Cascades, both surface erosion and mass wasting may be important sources of sediments. Wet, snow-dominated systems generally provide highest sediment yields; rain-dominated systems have intermediate yields; and dry, snow-dominated systems produce the lowest yields.

Stream temperatures influence virtually all aspects of salmonid biology and ecology, affecting the development, physiology, and behavior fish, as well as mediating competitive, predator-prey, and disease-host relationships. Heat energy is transferred to streams and rivers by six processes: short-wave radiation, long-wave radiation, convective mixing with the air, evaporation, conduction with the stream bed, and advective mixing with inflow from ground water or tributaries. The temperature of streams represents a balancing of these factors. During the summer, incoming solar radiation is the dominant source of energy for smaller streams, though groundwater discharge may be locally important. Consequently, riparian vegetation plays a major role in controlling summer stream temperatures; as may topographic features that provide shade. During the winter, solar radiation becomes less important due to lower sun angles, shorter days, and cloudier conditions. Stream characteristics, including width, depth, velocity, and substrate also determine the rate at which heat is gained or lost through radiation, convection, conduction, and evaporation. As streams become larger and less shaded downstream, the influence of both terrestrial vegetation and groundwater inputs diminishes and temperatures tend to equilibrate with mean air temperatures.

Water is the primary agent dissolving and transporting solutes and particulate matter across the landscape, integrating processes of chemical delivery in precipitation, weathering, erosion, chemical exchange, physical adsorption and absorption, and biotic uptake and release. Climate, geology, and biological processes all influence the character and availability of inorganic solutes. The composition and age of parent rock determine the rate of weathering and hence the release of soluble materials. These dissolved materials are transported by surface and groundwater flow to streams. The biota of terrestrial, riparian, and aquatic ecosystems control the sources and cycling of major nutrients and associated organic solutes through processes such as photosynthesis, respiration, food uptake, migration, litter fall, and physical retention. Side channels on floodplains are areas of high nutrient uptake and processing because of low flow velocities and extensive contact with the water column. Riparian vegetation removes a significant proportion of the available phosphorous and nitrogen (60-90%) and thus directly affects stream productivity.

Once in the stream, nutrients are transported downstream until they are taken up and processed by organisms and then released again, collectively termed 'nutrient spiraling.' The average distance over which one complete spiral occurs varies with stream characteristics, including retentive structures that physically trap particulate matter, stream size, water velocity, and the degree of contact between the water column and biological organisms inhabiting the stream bed. Simplification of channel structure increases nutrient spiral length, decreasing retention efficiency. Salmon and lamprey carcasses are also an integral part of nutrient cycling for both aquatic and riparian systems; thus declines in salmonids may cause more fundamental changes in ecosystem productivity than the simple loss of stocks or species.

Riparian and floodplain areas are an active component of the terrestrial-aquatic ecosystem, filtering, retaining, and processing materials in transit from uplands to streams. Riparian vegetation plays a major role in providing shade to streams and overhanging cover used by salmonids. Riparian vegetation stabilizes stream banks by providing root mass to maintain bank integrity, by producing hydraulic roughness to slow flow velocities, and by promoting bank

building by slowing velocities and retaining sediments. Riparian vegetation also provides much of the organic litter required to support biotic activity within the stream, as well as the large woody debris required to create physical structure, develop pool-riffle characteristics, retain gravels and organic litter, provide substrate for aquatic invertebrates, moderate flood disturbances, and provide refugia for organisms during floods. Large woody debris performs important functions in streams, increasing channel complexity, creating hydraulic heterogeneity, and providing cover. Large woody debris also provides critical habitat heterogeneity and cover in lakes, estuaries, and the ocean. In addition to the aquatic functions that riparian areas perform, they provide habitat and create unique microclimates critical to a majority of the wildlife occupying the watershed.

## Biological Processes

The physiology and behavior of organisms, the dynamics and evolution of populations, and the trophic structure of aquatic communities are influenced by the spatial and temporal patterns of water quantity and velocity, temperature, substrate, and dissolved materials. At the organism level, survival of salmonids depends on their ability to feed, grow, undergo smoltification, and reproduce. Habitat characteristics (water depth, velocity, temperature, and chemistry; turbidity; and substrate) influence the quality and amount of food (energy) available, the amount of energy expended for metabolic processes, and hence the amount available for growth and reproduction.

Each phase of the salmonid life cycle (i.e., adult migration, spawning, incubation of embryos and alevins, emergence of fry, and juvenile rearing) requires utilization of and access to distinct habitats. The strong homing ability of salmonids has led to the formation of numerous, relatively isolated stocks, each adapted to the specific environmental conditions found in its natal and rearing habitats. This ability to adapt is reflected in the wide diversity of life histories exhibited by the salmonids of the Pacific Northwest. A major concern is that land use and water use have reduced habitat diversity (through loss or simplification of habitat), which in turn has reduced the life-history diversity exhibited in the salmonid populations. At larger spatial scales, groups of populations or 'metapopulations' interact infrequently through straying or dispersal. Conventional metapopulation models assume that all local populations making up the metapopulation have an equal probability of extinction, and that metapopulations will persist if recolonization rates and exceed extinction rates. An alternative model, the core-satellite model, suggests that extinction probability is not equal among populations, and that certain extinction-resistant populations are important 'seed' sources of recolonizers for habitats made vacant by extinction. Conservation of salmonids thus depends on maintaining: connectivity among habitats to allow reinvasion, sufficient genetic diversity to allow successful recolonization of vacant habitats, and refugia from which dispersal can occur. The concept of Evolutionarily Significant Units presently being used by federal agencies to determine appropriate units of conservation for salmonids is based in part on these metapopulation considerations.

Biotic communities in aquatic ecosystems are influenced by predator-prey, competitive, and disease- or parasite-host relationships within and among species. Current theory suggests that disturbance plays a major role in influencing the outcome of these interactions and, thus, in determining community or assemblage structure. Two models appear to be applicable to stream communities. The 'intermediate disturbance hypothesis' argues that diversity is greatest in systems experience intermediate disturbance, because neither colonizers (favored by frequent disturbance) nor superior competitors (favored by infrequent disturbance) are favored. The 'dynamic equilibrium model' proposes that community structure is a function of growth rates, rates of competitive exclusion, and frequency of population reductions; inferior competitors persist if disturbance occurs often enough to prevent competitive exclusion, but species with long life cycles are lost if disturbance is too frequent. Both of these theories suggest that increases in disturbance frequency caused by human activities are likely to alter community structure.

Food webs in aquatic systems are highly complex, consisting of many species representing several trophic levels, and can be highly modified by environmental changes in the food base (energy or nutrient input); alteration in streamflow, temperature, or substrate; and introductions of non-native organisms. Changes physical habitat characteristics can alter competitive interactions within and among species (e.g., bull trout population declines have in part resulted from their inability to compete with rainbow, brook, and brown trout at warmer stream temperatures). Similarly, changes in temperature or flow regimes may favor species that prey on salmonids, such as northern squawfish. Salmonids are affected by a variety of bacterial, viral, fungal, and microparasitic pathogens. Both the immune system of fishes and the virulence of pathogens are greatly affected by environmental conditions (especially temperature); thus, alteration of temperature, substrate, and flow may increase the incidence of epizootics.

## Salmonid Habitat Requirements

All of the physical, chemical, and biological processes discussed above, operating throughout the watershed and across the landscape, affect the features and characteristics of aquatic habitats from headwater streams and lakes to estuaries and the ocean. Protecting or restoring desirable habitat requires that the natural processes that produce those features and characteristics be maintained or restored. Four general principles should be considered when determining habitat requirements of salmonids:

- Watersheds and streams differ in their flow, temperature, sedimentation, nutrients, physical structure, and biological components.
- Fish populations have adapted—biochemically, physiologically, morphologically, and behaviorally—to the natural environmental fluctuations that they experience and to the biota with which they share the stream, lake, or estuary.
- Specific habitat requirements of salmonids differ among species and life-history types, and change with season, life stage, and the presence of other biota.
- Aquatic ecosystems are changing over evolutionary time.

Consequently, there are no simple definitions of salmonid habitat requirements, and the goal of salmonid conservation should be to maintain habitat elements within the natural range for the particular system.

Five general classes of features or characteristics determine the suitability of aquatic habitats for salmonids, including flow regime, water quality, habitat structure, food (energy) source, and biotic interactions. Flow regimes directly influence the water depth and velocity, and total available habitat space for salmonids and food organisms, as well as performing important functions such as redistributing sediments, flushing gravels, and dispersing vegetation propagules. Water quality requirements include temperatures, dissolved oxygen, nutrient, and low levels of pollutants. Salmonids are cold water species, and temperatures above 25°C are lethal to most species; individual species have specific preference ranges that vary by life-cycle stage. Variation in temperature is required to trigger spawning, support growth, initiate smoltification, and enable other parts of the life cycles of salmonids. Salmonids require well oxygenated water (> 6 mg/l) throughout their life cycle, and any level below saturation can be detrimental. Nutrient levels are variable among streams and must be sufficient to support natural plant and animal assemblages. Important structural attributes of stream includes pools and riffles, substrate, cover (e.g. undercut banks, overhanging vegetation), depth, and hydraulic complexity. The presence of large woody debris enhances channel complexity, creating hydraulic heterogeneity, pools, side channels, back eddies, and other features that are used by salmonids and other aquatic organisms maintaining adequate food sources depends upon maintaining natural inputs of allochthonous material (type, amount, and timing), as well as the physical structures needed to retain these materials. Normal biotic interactions also must be maintained to ensure the health of aquatic ecosystems, including competitive, predator-prey, and disease-parasite relations.

Stream habitat and channel features vary markedly from headwater streams to the estuaries and ocean. Salmonids, particularly anadromous salmonids, utilize the entire range of habitats encountered during completion of their life cycle. The diversity of life histories of the salmonids has developed to accommodate (and fully utilize) the range of habitats encountered, and loss of specific elements of habitat diversity may reduce the diversity exhibited in the salmonid's life histories.

Habitat requirements vary by life stage. During spawning migrations, adult salmon require water of high quality (i.e., cool temperatures or thermal refugia, dissolved oxygen near 100%, and low turbidity); adequate flows and depths to allow passage over barriers between rearing and spawning sites (variable with species); and sufficient holding and resting sites. Spawning areas are selected on the basis of specific requirements of flow, water quality, substrate type, and, for some species, groundwater upwelling. Embryo survival and fry emergence depend upon substrate conditions (gravel size, porosity, permeability, oxygen levels), substrate stability during high flows, and appropriate water temperatures (< 14°C for most species, but < 6°C for bull trout). Habitat requirements for rearing juveniles of anadromous species and adults of resident species also vary with species and size. Microhabitat requirements for holding, feeding, and resting each differ, and these requirements change with season. Migration of juveniles to rearing areas (whether the ocean, lakes, or other stream reaches) requires unobstructed access to rearing and feeding grounds. Physical, chemical (e.g., dissolved oxygen), and thermal conditions may all impede migrations of juvenile fish. Although specific life histories likely have adapted to evolving habitats so that no "optimal factors" can be identified to meet all the needs of all salmonids, a diversity of habitats is required to maintain the diversity of life histories exhibited by the salmonids (that may be needed to accommodate environmental change over evolutionary time) and assure their survival.

## Effects of Human Activities on Watershed Processes and Aquatic and Riparian Habitats

Land-use practices, including forestry, grazing, agriculture, urbanization, and mining alter watershed processes that influence streams, lakes, and estuaries. Logging and grazing affect the greatest percentage of lands in the Pacific Northwest, but effects of agriculture, urbanization, and mining may result in a higher degree of local disturbance. Most of the alterations due to land-use practices result from changes in vegetation and soil characteristics that affect the quantity and routing of water, sediments, nutrients, and other dissolved materials delivered to streams. In addition, application of chemical fertilizers and biocides affect water quality. Activities within the riparian zone can alter shading, transport and supply of sediment, inputs of organic litter and large wood, bank stability, seasonal streamflow regimes, and flood dynamics. Dams, diversions, and road crossings hinder migrations, alter physical and chemical character of streams, and change stream biota. Salmon harvest, introduction of non-native species and hatchery-reared salmonids, and beaver eradication also affect salmonids and their habitats.

### Forestry

Forest practices result in removal and disturbance of natural vegetation, disturbance and compaction of soils, construction of roads, and installation of culverts. Removal of vegetation typically reduces water loss to evapotranspiration, resulting in increased water yield from the watershed. In general, increases in water yield are greater west of the Cascades than they are on the east side. Short-term increases in both peak flows and base flows have been reported, but hydrologic responses vary regionally, and long-term reductions in summer base flows due to logging have been reported from one basin in the Cascade Range. Site disturbance and road construction typically increase sediment delivered to streams through mass wasting and surface erosion, which can elevate the level of fine sediments in spawning gravels. Logging removes canopy, reducing shading and increasing solar radiation reaching the streams, which results in higher maximum stream temperatures and increased diel and seasonal fluctuations. In addition, the loss of riparian vegetation may increase radiative cooling during the winter, enhancing the formation of anchor ice. In other systems, increases in winter stream temperatures have been observed after logging. Temperature increases due to logging depend on the size of the stream, and the type and density of canopy removed. Altered stream temperatures will likely persist until pre-logging shading is reestablished, and may take from less than 10 to more than 40 years. Timber harvest removes biomass, and hence nutrients, but nutrients are more available to streams immediately following harvest, resulting in part from addition of slash to the forest floor, accelerated decomposition of litter, and increased runoff and erosion. This short-term increase diminishes as soils stabilize and revegetation occurs. Where logging occurs in riparian areas, delivery of leaf litter and large woody debris to the stream is reduced, and may significantly alter the nutrient balance and physical character of the stream. Loss of large woody debris, combined with alteration of hydrology and sediment transport, reduces complexity of stream micro- and macrohabitats and causes loss of pools and sinuosity. These alterations may persist from decades to centuries. Changes in habitat conditions may affect fish assemblage structure and diversity (e.g., favoring species that prefer riffles rather than pools), alter the age-structure of salmonid populations, and disrupt the timing of life-history events. Other effects on salmonids include reduced survival of embryos and fry production, decreased growth efficiency, increased susceptibility to disease and predation, lower overwinter survival, blocked migration (e.g., inadequate culverts), and increased angler mortality through improved access to streams.

### Grazing

Grazing results in the removal of natural vegetation, the alteration of plant community composition and modification of soil characteristics, which in turn affect hydrologic and erosional processes. Effects are particularly acute in the riparian zone, where livestock tend to congregate, attracted by water, shade, cooler temperatures, and an abundance of high quality forage. In general, grazed lands have less vegetation and litter cover than ungrazed lands, and in many areas of the West, perennial grasses have been replaced by non-native annual grasses and weedy species. Livestock also affect vegetation and soils through trampling. Trampling soils in arid and semi-arid lands may break up the fragile cryptogamic crust (comprised of symbiotic mosses, algae, and lichens) causing reduced infiltration, increased runoff, and reduced availability of nitrogen for plant growth. In addition, trampling detaches soil particles, accelerating surface erosion in upland areas, and may promote mass wasting along streambanks. Mass wasting also occurs where grazing has eliminated riparian vegetation, and hence the root matrix that helps bind soil together. All of these processes result in increased sediment transport to streams. Animals also redistribute seeds and nutrients across the landscape, especially to riparian zones or other attractors, such as spring seeps or salt blocks. Devegetating riparian zones reduces shading and increases summer stream temperatures, and may also increase the formation of anchor ice in the winter. Grazing also results in changes in channel morphology, through changes in hydrology, sedimentation, and

loss of bank stability. Streams in grazed areas tend to be wider and shallower, and consequently warmer, than in ungrazed reaches. In some instances, streams in grazed areas tend to downcut, effectively disconnecting the stream channel from the floodplain, further altering the hydrology of the stream and changing the plant community occupying the riparian zone from hydric (wetland) to xeric vegetation. Grazing in the riparian zone can reduce recruitment of large woody debris, especially since reestablishment of riparian shrubs and trees rarely occurs if grazing pressure is not reduced. Loss of woody debris reduces retention of gravels, creation and maintenance of pool habitats, and instream cover. General effects of grazing on salmonids include reduced reproductive success due to sedimentation of spawning gravels, alteration of food supplies through changes in primary and secondary production, reduced fish densities, and shifts in fish species and biotic communities.

## Agriculture

Although agriculture is not a dominant land use in the Pacific Northwest (approximately 16% of the total land area), alterations to the land surface are more severe than those caused by forestry or grazing, are generally permanent, and tend to involve repeated disturbance. Replacing natural grasslands, forests, and wetlands with annual crops leaves much area fallow during part of the year, and dramatically changes the function of plants and soil microbes in the tilled areas. Repeated tillage, fertilization, and harvest permanently alters soil character and microorganism function. Infiltration is reduced and runoff is increased. These changes alter seasonal runoff patterns, increasing high flows, lowering water tables, and reducing summer base flows in streams. Channelizing, to reduce local flooding and alter the geometry of cropped lands, also facilitates more rapid routing of water downstream, increasing flooding downstream. Sediment yield from agricultural lands is typically greater than from prairie, forest, or wetland areas. Where riparian shading is lost or summer base flows are reduced, stream temperatures are increased. Nutrients, insecticides, and herbicides are typically elevated in streams draining agricultural areas. Channelization, snag removal, revetments, and removal of riparian vegetation reduce habitat complexity, decrease channel stability, and alter the food base of the stream. Incised and channelized streams in agricultural areas typically support smaller fish and fewer fish species.

## Urbanization

Urbanization has affected only 2% of the land area of the Pacific Northwest, but the impacts to aquatic ecosystems are severe and long-lasting. The land surface, soil, vegetation, and hydrology are all significantly altered in urban areas. Regrading is common, and impervious surfaces greatly increase runoff and reduce the area available for infiltration. Impervious surfaces, roads and gutters, storm drains and drainage ditches in combination quickly shunt precipitation to receiving streams, resulting in an increased magnitude and frequency of peak discharge and reduced summer base flow. Sediment delivery typically increases during construction activities. The total vegetated area is greatly reduced, and replacement vegetation is typically lawns and ornamentals that require water, fertilizers, and pesticides. Riparian corridors are frequently constricted, disabling or altering riparian function. Loss of riparian zones and increased flood flows combine to increase stream temperatures and simplify stream channels. With the loss of riparian zones, the source of large woody debris is diminished, and natural nutrient inputs are replaced with fertilizer (and other chemicals) washed from lawns and streets, and discharge from sewage treatment facilities, causing significant alteration in water quality of receiving streams. The highly altered streams in urban areas generally provide poor habitat for fish.

## Mining

Sand and gravel mining in streams and on adjacent floodplains has a substantial impact on stream channels and hydraulic characteristics. In addition to the immediate morphological changes in stream channels caused by excavation, the channel continues to exhibit instability, accelerated erosion, and altered substrate composition and structure. Downcutting of stream channels frequently follows gravel mining, often exceeding 4-6 meters in depth over periods ranging from months to decades. The downcutting and simplification of stream channels results in increased flood peaks, increased sediment transport, increased temperatures, and decreased base flows. The most direct impacts to salmonids are degradation and simplification of spawning and rearing habitats and increased turbidity. In addition, increased turbidity and decreased substrate stability influence lower trophic levels, upon which salmonids depend for food.

Mineral mining also has substantial impact on aquatic ecosystems. Hydraulic mining (e.g., gold) from stream deposits and hillslopes dramatically altered stream channels, riparian zones, and floodplains. Although hydraulic mining

is uncommon today, previously degraded habitats have not yet recovered and still exhibit excessive sediment transport, downcutting, and instability. Recovery may take generations where, in addition to channel modification, acid drainage, radioactive materials, and metals from mining wastes contaminate streams. Increased sediments, acidification, and chronic pollution from mine wastes significantly degrade aquatic habitats throughout the West.

### Dams and Irrigation Withdrawal

Hydroelectric dams, other impoundments, and irrigation water withdrawals have been a significant factor in the decline of salmonids. These activities alter habitat characteristics and impede migrating adult and juvenile salmonids. Dams have blocked access for adults to 55% of the basin and 33% of the stream miles in the Columbia basin. At dams, injury and mortality to juveniles occurs as a result of passage through turbines, sluiceways, juvenile bypass systems, and adult fish ladders. Dams and reservoirs increase the time it takes juveniles to migrate to the ocean, which increases exposure to predation and contact time for spread of parasites and disease. Below hydroelectric facilities, nitrogen supersaturation may also negatively affect migrating salmon.

Hydrologic effects of dams and irrigation withdrawals included water level fluctuations, altered seasonal and daily flow regimes, reduced water velocities, and reduced discharge volume. Drawdowns and diversions reduce available habitat area and concentrate organisms, increasing predation and transmission of disease and parasites. Impoundments alter natural sediment transport processes, causing deposition of fine sediments in slack water areas, reducing flushing of sediments through moderation of extreme flows, and decreasing recruitment of coarse material (including spawning gravels) downstream of the obstruction. Irrigation return flows tend to have high sediment content and turbidity. Impoundments and irrigation withdrawals also change the thermal regimes of streams. Temperatures may increase in shallow reservoirs and where irrigation return flows have been heated. Below deeper reservoirs that thermally stratify, summer temperatures may be reduced through release of hypolimnetic waters, but fall temperatures tend to increase as heated water stored during the summer is released. Changes in water temperatures affect development and smoltification of salmonids, as well as influence the success of predators (e.g., northern squawfish) and competitors (e.g., American shad), and the virulence of disease organisms. Dissolved oxygen concentrations may be reduced during both summer and winter due to irrigation withdrawals. In summer, high temperatures of irrigation return flows reduces the oxygen-holding capacity of water; in winter, drawdown of irrigation impoundments may facilitate freezing, which diminishes light penetration and photosynthesis, potentially causing fish kills through anoxia.

### Effects of Atmospheric and Ocean Circulation

Marine productivity depends on atmospheric and oceanic circulation and strongly affects abundance of salmonids and other fishes. Surface currents of the northeast Pacific are dominated by the 'West Wind Drift,' which flows west-to-east across the Pacific and bifurcates as it approaches North America into the Alaska Current flowing north and the California Current flowing south. Changes in climatic conditions affect the behavior of the West Wind Drift. In years where a strong Aleutian Low Pressure system develops off the south coast of Alaska (typical of El Niño conditions), a greater percentage of cold, nutrient-rich water is diverted north into the Alaska Current. When the Aleutian Low is weaker (typical of La Niña years), more water from the West Wind Drift is diverted south towards California. These shifts, combined with changes in prevailing wind directions and upwelling patterns, can substantially affect conditions for salmonids entering the ocean. Changes in surface currents and upwelling strength influence temperature, salinity, and nutrients, thereby affecting the abundance of food available to juvenile salmonids, the number and distribution of predators and competitors, and the transport of smolts entering the ocean (along-shore or off-shore). Recent evidence suggests that when ocean conditions are poor for salmonids in the Pacific Northwest, conditions are favorable to Alaskan stocks and vice versa.

Cycles in marine productivity can mask the effects of habitat degradation in freshwater environments or other stressors of salmonid populations. Long-term trends in the ability of freshwater environments to support salmonids may not be evident during periods of favorable oceanic conditions, particularly for populations augmented by hatchery fish. However, as ocean conditions shift towards less favorable conditions (particularly for hatchery fish), increasing pressure from overcapitalized fisheries can dramatically reduce the abundance of wild stocks.

### Programs and Regulations to Protect Aquatic and Riparian Habitats

Virtually all land-use and water-use practices have some effect on aquatic ecosystems. However, there are numerous opportunities, through planning and specific practices, for minimizing these effects or mitigating for past damage. In some locations, dams are being considered for removal to restore habitat and remove barriers (e.g., Elwha River, WA and Savage Rapids, OR). Elsewhere, the impacts of dams are being reduced by assuring instream flows, especially at critical times; screening turbine intakes; improving bypass systems; and attempting to control predation.

Harvest strategies are also being developed to prevent over-exploitation, protect weak stocks, and mitigate the loss of salmon carcasses to the aquatic ecosystems. Hatchery programs are changing their objectives from increasing harvest to conserving endangered species, supplementing weak stocks, and avoiding introductions of competitors. Recognizing the importance of large woody debris and channel complexity to habitat integrity and quality, snagging and channelization is restricted and, where practiced, conducted in a manner to preserve habitat or minimize degradation.

Impacts of forest practices can be reduced through longer rotations; selective harvesting instead of clear-cutting; use of high lead, skyline, and helicopter logging instead of ground-based equipment; use of designated skid trails; minimizing site-preparation practices that compact or scarify soils; retention of riparian buffer zones along streams; designation of no-cut zones in areas prone to mass failures; careful placement and maintenance of roads; and decommissioning and reseeding of roads when logging is completed. These activities function to minimize the percentage of the watershed in a disturbed state, reduce the total area of ground disturbance and soil compaction, minimize surface runoff and sediment loads, and protect and preserve the function of riparian zones.

The effects of range practices can be reduced by resting pastures; decreasing numbers of livestock; controlling livestock distribution through fencing of riparian zones or watering of stock away from riparian areas; controlling forage use; controlling season of use; and determining the kind of livestock best suited for the area. These practices can serve to reduce grazing stress, promote the re-establishment of riparian vegetation (particularly woody shrubs and trees), and keep stock out of riparian zones, although site-specific conditions will determine their relative effectiveness.

Agricultural practices and policies that promote water and soil conservation and that reduce chemical application can all reduce effects on aquatic ecosystems. Examples include switching to crops that do not require irrigation; ditch lining and drip irrigation; screening of irrigation intakes; increasing vegetative cover (e.g., permanent rather than annual crops); conservation tillage; planting grass in water ways (for soil conservation); organic farming; integrated pest management; increasing tax relief for farmers employing conservation practices; and increased penalties for those who do not.

Most of the impacts of gravel mining relate to changes in channel morphology that create channel instability, cause bedload movement, and increase sedimentation. Consequently, these effects can be most effectively reduced by eliminating instream mining or by employing bar scalping instead of below-surface extraction. Effects of mineral mining can be reduced by burying toxic materials below the root zone, rehabilitating the site by creating natural contouring and re-establishing natural vegetation, and controlling mining-generated solids with containment structures.

Urbanization permanently alters many natural watershed processes, and in some cases, little may be done to mitigate effects. Thus, the most effective way to minimize impacts is through careful land-use planning that minimizes development along streams and in natural floodplains. Sewage treatment and programs to foster water conservation, minimize chemical applications, and prevent toxic materials from being dumped into drainage structures can reduce impacts of urbanization to water quality.

Several federal laws, notably the Clean Water Act (CWA), the Endangered Species Act (ESA), the National Environmental Policy Act (NEPA), and the Food Security Act (FSA), are being employed to protect aquatic and riparian habitats to the extent possible. Each of these may be used to provide federal leadership and to make scientific information and funds available to states and private landowners to further the goals of habitat conservation.



## 2.0 Introduction

Physical and biological characteristics of freshwater and estuarine salmonid habitats are the integrated expression of processes that occur in upland areas, within the riparian zone, and in the stream channel. Thus, protection of these salmonid habitats depends on maintaining the natural processes that shape stream ecosystems. This document comprises a technical foundation for understanding salmon conservation principles and developing salmonid conservation plans in ecosystem contexts. We intentionally focus on freshwater habitats but acknowledge that many other factors, including fish harvest, dams, hatchery practices, habitat conditions in near-shore areas, and natural variation in ocean productivity greatly influence the abundance of anadromous salmonids. This document establishes a framework with which the National Marine Fisheries Service, the U.S. Fish and Wildlife Service, and the U.S. Environmental Protection Agency (the Agencies), in cooperation with private landowners, can develop policies and strategies to restore and protect salmonid habitat; it does not provide specific prescriptions for land-use activities. Conclusions are based on our assessment of the scientific literature. Because some topics are well documented in this literature and others are not, certain sections of the document are complete and robust, whereas others seem quite sparse. The Agencies intend to update this document as new information becomes available and to use it for suggesting areas of new research.

### 2.1 Scope

Geographically, the scope of this document is limited to the Pacific Northwest region, including portions of California, Oregon, Washington, and Idaho that have supported salmonid populations. Many of the general concepts and processes examined, however, do apply outside this region. Discussion of specific habitat requirements is restricted to those salmonid species (Table 2.1a) that are endemic to the Pacific Northwest, including the five Pacific salmon (chinook, coho, sockeye, chum, and pink salmon), trout and char, with both resident and anadromous forms (rainbow, cutthroat, and bull trout), and strictly resident species (mountain whitefish). In the remainder of Section 2, we discuss evidence of widespread declines in salmonid abundance that indicate region-wide degradation in habitat quality and ecosystem health. We then identify strategies for restoring salmonid habitats. These strategies emphasize the importance of maintaining watershed processes, providing for the diverse life-history requirements of salmonids, and reestablishing connectivity between salmonid habitats across the landscape. In Sections 3 and 4 we review physical, chemical, and biological processes that occur within watersheds that influence the quality and quantity of available salmonid habitat, and that need to be maintained to ensure the persistence of salmonid stocks. Some physical and chemical processes (Section 3) shape stream habitats over long time periods (e.g., glaciation, volcanism) and others operate in relatively short time scales (e.g., floods, droughts, landslides). Biological processes (Section 4) encompass those occurring at the level of the individual organisms (e.g., physiology, behavior), populations (e.g., life history, adaptation), and communities (e.g., disease, predation, competition). Sections 3 and 4 sufficiently detail ecological processes so that the effects of anthropogenic disturbances on salmonids and their habitat can be understood and evaluated. Section 5 describes habitat requirements specific to each stage of the salmonid life history and general characteristics of healthy aquatic and riparian systems, including physical habitat structure, stream flow, stream temperature, and water quality, as well as the important biological components.

In Section 6, we discuss the effects of human activities on watershed processes and the resulting impacts on salmonids and their habitats. The discussion focuses on effects of forest practices, livestock grazing, agriculture, and urbanization because these activities generally affect salmonids over the greatest areal extent. The effects of mining, dams, and other activities that contribute locally or regionally to declines of salmonids are also reviewed. The impacts of harvesting fish and introducing non-native species are largely outside the scope of this document and are discussed only briefly.

Section 7 briefly reviews general circulation patterns and the dominant physical processes controlling conditions in the Northeast Pacific Ocean. This section discusses how ocean conditions influence abundance and distribution of aquatic organisms, especially salmonids, and the implications for restoration.

Section 8 identifies management systems that are designed to minimize effects of human activities on salmonid habitats, with emphasis on forestry, range, and agricultural practices, as well as urban planning. Section 9 summarizes four federal laws and the amendments that relate to conserving and protecting aquatic habitats and their species. These include the Endangered Species Act (ESA), the Clean Water Act (CWA), the Food Security Act (FSA), and the National Environmental Policy Act (NEPA).

Necessary elements of habitat conservation plans, and an evaluation of selected management approaches and certain laws that apply specifically to habitat conservation planning are identified in Volume II. Selected sources of information useful in preparing habitat conservation plans are identified in Volume III.

## 2.2 Historical Background and Evidence of Habitat Degradation

Many Pacific salmon stocks have been depleted to the point that continued declines will likely result in additional listings under the Endangered Species Act (ESA), and extirpation of some local stocks. Although ample evidence documents historic declines in Pacific salmonids (Ebel et al. 1989), the landmark paper by Nehlsen et al. (1991) alerted both scientists and the public to the extent of salmonid declines. Summarizing the status of Pacific salmon of Washington, Oregon, Idaho, and California, Nehlsen et al. (1991) listed 214 stocks (unique populations) as being at high or moderate risk of extinction or of special concern and at least 106 stocks that have already been extirpated. Since 1985, tribes, professional fishery societies, and conservation organizations have petitioned the National Marine Fisheries Service (NMFS) to list 24 stocks as threatened or endangered. To date four of these have been listed (Sacramento River winter chinook, Snake River sockeye, and Snake River spring/summer and fall chinook). In recent years, commercial and sport ocean harvest in the Pacific Northwest has been sharply curtailed because of dwindling numbers of salmon and concern for wild salmon stocks. In 1994, the commercial troll fishery for coho salmon in Oregon and northern California was shut down entirely, and a similar closure for the Columbia River spring chinook fishery is imminent for 1995.

Fish and fish assemblages serve as sensitive indicators of ecosystem health and biological integrity because they reflect the integrated effects of activities occurring throughout a watershed (Miller et al. 1988; Fausch et al. 1990). Salmonids may be particularly sensitive to environmental deterioration because they depend on clean, cool water and because they require diverse habitats during various stages of their life cycles. Long-term data on the status of salmonid populations may indicate when physical habitat and water quality are changing through time.

Several recent analyses support the conclusion that declines of salmonids and other fishes in the Pacific Northwest are widespread. The 214 at-risk salmon stocks identified by Nehlsen et al. (1991) are distributed throughout Washington, Oregon, California, and Idaho. At least two to three species of fish (including non-salmonids) are extinct or at risk of extinction in most areas of the Pacific Northwest, indicating that species losses are not isolated occurrences (Figure 2.2a) (Frissell 1993b). Nawa (1994) studied population trends for 228 stocks of spring and fall chinook salmon over the period from 1940–1993 and found that 34% were extinct or nearly extinct, 24% were declining, and only 8% were not declining. Other chinook stocks were either hatchery-influenced or had unknown status.

Available evidence suggests that there are regional patterns in the status of salmonid stocks. Bisson et al. (1992) report that more Alaskan stocks of chinook, coho, chum, and steelhead increased than decreased from 1968 to 1984 (Figure 2.2b). Conversely, in Washington, the Columbia River Basin, and coastal Oregon and California, declining stocks outnumbered increasing stocks for all four species examined, though the majority of stocks exhibited no significant trend over the 16-year period (Figure 2.2b). Frissell (1993b) examined native fish taxa that are considered extinct, endangered, or threatened in drainage basins of the Pacific Northwest and reports a north-to-south gradient in the degree of endangerment (Figure 2.2a): mean percentages of the native taxa considered to be extinct or at risk of extinction were 13.5% in Washington, 33.0% in Oregon, and 48.0% in California. This pattern is largely influenced by the basin-specific populations of seven widely distributed species of anadromous salmonids rather than locally endemic species such as suckers, pupfishes, minnows, and other fishes. Another status review of the five Pacific salmon and the anadromous steelhead and cutthroat trout indicates a similar latitudinal gradient in the degree of endangerment for most of these species (The Wilderness Society 1993), (Figures 2.2c–k). The general north-to-south gradient in salmonid declines likely reflects several factors: environments in the southern portion of the salmonids' range are more extreme; there has generally been a higher degree of habitat modification in the southern part of the range; and the influence of changing oceanic conditions varies with latitude (see Section 4.4).

In addition to the north-south gradient in species declines, several regions and localized areas have an especially high degree of species endangerment. The risk of extinction is greatest in the upper Columbia—with large hydropower dams and large-scale diversions—as well as in many other undammed coastal and Puget Sound streams. Historically, 10–16 million salmon returned to the Columbia River to spawn; however, recent estimates suggest fewer than 2 million fish have returned to spawn in recent years of which only 20% spawn in the rivers (NPPC 1992b). For example, coho salmon historically were abundant throughout the Columbia Basin and along the coast (Figure 2.2c). Today, coho stocks in the eastern half of their range are extinct, and stocks in the southern two-thirds of their coastal range are considered imperiled (Frissell 1993b). High numbers of threatened and endangered species in the Puget Sound and San Francisco Bay areas suggest that urbanization has contributed to the declines of native taxa.

In addition to the above reports, NMFS is now preparing status reviews of seven eastern Pacific anadromous salmonids over their ranges in the region. These reviews will incorporate information from the publications cited above as well as from state-wide status reviews prepared by the Oregon Department of Fish and Wildlife (in preparation) and the Washington Department of Fisheries and Wildlife (NIFC et al. 1993). These reviews will focus on delineating "evolutionarily significant units" (see Section 4.2.5) pursuant to potential listing as threatened or endangered under the Endangered Species Act, and they may differ somewhat from other reporting efforts.

A number of factors have been implicated in the decline of Pacific salmonids: hydropower operations, overexploitation, disease, predation, artificial propagation, climatic variation, and the destruction and alteration of

habitat. The relative importance of each of these factors in influencing salmonid populations varies across the landscape. However, habitat loss and modification are believed to be the major factors determining the current status of salmonid populations (FEMAT 1993), and habitat degradation is considered a primary or contributing factor in over 90% of listed threatened and endangered populations of Pacific salmon (Nehlsen et al. 1991) and over 75% of fish species extinctions (Miller et al. 1989). Historically, certain land-use practices have contributed to the degradation of aquatic habitats, namely, timber harvesting, livestock grazing, mining, urbanization, road construction, agriculture, and hydroelectric development. Doppelt et al. (1993) summarize several changes resulting from land-use practices that contribute to the degradation of riverine systems: 1) changes in water quality or flow due to irrigation and other withdrawals; 2) modification of channels and riparian ecosystem morphology (damming, making reservoirs, channelizing, draining and filling of wetlands, dredging for navigation); 3) excessive non-point source pollution; 4) deterioration of substrate quality and stability; and 5) degradation of water quality through addition of point-source contaminants. Over time, land-use practices have substantially decreased the physical and biological complexity of ecosystems, thereby diminishing the ability of ecosystems to self repair when perturbed (FEMAT 1993).

Evidence of degrading aquatic habitats is not limited to fishes. Amphibians, which use streams and wetlands as breeding and rearing habitats, are also highly sensitive to environmental (Welsh 1990). Recent field studies in the Pacific Northwest indicate widespread population declines, range reductions, and extinction of amphibians in forest and other ecosystems (Blaustein et al. 1994). Blaustein et al. (1994) identify habitat destruction as the major cause for amphibian losses, but suggest that other factors may be important, including chemical pollution, acid precipitation, increased ultraviolet radiation, introduction of non-native species, pathogens, harvesting by humans, and natural population fluctuations.

### 2.3 Cumulative Effects

The widespread decline of salmonid stocks throughout much of the Pacific Northwest has resulted from the cumulative effects of land-use practices, fish harvest, hatchery practices, and natural fluctuations in environmental conditions. The term "cumulative effects" has been used generally to describe the additive or synergistic effects of land-use practices on ecosystems. A more comprehensive definition of cumulative effects is provided by (Sidle 1989): "changes to the environment caused by the interaction of natural ecosystem processes with the effects of land use, distributed through time and space, or both."

Because of the longitudinal nature of stream ecosystems, the accrual of effects is particularly important along both spatial and temporal dimensions. Activities that take place in headwater streams may influence the suitability of habitats in downstream reaches (e.g., temperature change, sediment input) and affect the response of ecosystem components to additional stresses. Similarly, activities that have occurred in the past may change current habitat conditions through residual effects (e.g., channel morphology altered by splash dams, hydraulic mining); long-term, persistent effects (e.g., reduced woody debris recruitment; channelization and revetment); or latent effects that are triggered by future environmental events (e.g., mass wasting of hill slopes, debris torrents, incision of stream channels).

In the context of conserving and restoring salmonids, the notion of cumulative effects has at least two important implications. First, individual actions that by themselves are relatively benign may be damaging when coupled with other actions that have occurred or may occur in a watershed. Historic and current patterns of land-use practices, as well as other factors, have a significant bearing on how salmonid populations will respond to further anthropogenic disturbances. Traditional management strategies that rely on site-specific analyses without regard for other activities that have occurred or are occurring within a watershed will generally fail to protect salmonid populations against cumulative effects. This premise underlies the recent development of watershed and ecosystem approaches to resource management.

Second, regional declines in salmonid populations are the product of numerous incremental changes in the environment; this fact strongly suggests that recovery of salmonid populations will proceed in a similar fashion—through incremental improvements in habitat conditions. Few activities directed toward improving habitat are likely to have sudden and marked influences on salmonid populations, and in many cases, we may be unable to detect any improvement at all amid the "noise" of natural variation in salmonid production except over long time periods (Hall and Knight 1981). This suggests that we should temper our expectations of how rapidly ecosystem complexity and integrity can be restored. It also means that individual stakeholders can and must play an active role in salmonid habitat restoration even if tangible benefits are slow to come.

Cumulative effects of human activities have resulted in a regional landscape—including the aquatic ecosystems contained therein—that is highly fragmented with few large expanses of land (i.e., whole watersheds) that are relatively intact (Doppelt et al. 1993). Early settlement of the Pacific Northwest was concentrated along low-gradient streams and rivers on relatively gentle terrain that was suitable for farming and ranching. Larger waterways served as primary travel corridors for boats as well as logs that were felled in the riparian zone and floated to sea. Snagging operations removed thousands of logs annually to facilitate this river traffic (Sedell and Luchessa 1981). Similarly, roads and railroads were most readily laid out in valley bottoms adjacent to rivers because of the relative ease

of construction. Diking and brush removal were commonly employed to reduce flooding of lowland areas and to allow farming and construction of houses within the historical floodplain. Streams were channelized to facilitate rapid runoff of stormwaters out of watersheds.

A *de facto* consequence of these (and other) activities and their cumulative effects on salmonid habitats is that many of the most pristine habitats that remain are in headwater streams, where human disturbance has been less severe (Doppelt et al. 1993; Frissell et al. 1993; Henjum et al. 1994). This situation has led to a common misperception among non-scientists and scientists alike that headwater environments are the preferred habitats of salmonids. In reality, headwater streams generally do not contain the wide array of habitats that are necessary or desirable for all life-stages of salmonids or for different fish taxa that have varying habitat requirements (Sheldon 1988). It is generally believed that unconstrained, aggraded floodplain reaches were historically highly productive habitats for some anadromous salmonids (Stanford and Ward 1992). Off-channel areas adjacent to larger rivers have been shown to be important rearing habitats for salmonids during high winter flood events (Tschaplinski and Hartman 1983).

Fragmentation of habitat and the resulting isolation of populations may affect the long-term viability of salmonid stocks (see Section 4.2.4). In addressing fragmentation and connectivity of habitats for the northern spotted owl, Thomas et al. (1990) outline several general principles that are equally applicable to salmonid conservation:

- Large blocks of habitat are preferable to small blocks.
- Patches of habitat that are close together are superior to those that are far apart.
- Contiguous blocks are preferable to fragmented habitats.
- Interconnected patches are better than isolated habitat patches, and corridors linking habitats function better when they resemble the preferred habitat of the target species.

Thus, essential goals of salmonid restoration should be to prevent further fragmentation of aquatic habitats, to improve connectivity between isolated habitat patches, and to protect and restore areas surrounding critical refugia from further degradation so as to allow for the expansion of existing populations.

## 2.4 Strategies for Salmonid Conservation

In the last twenty years, there has been a fundamental shift away from "single-species management" of salmonids toward more holistic watershed and ecosystem approaches that seek to conserve aquatic habitats by protecting processes operating throughout the watershed. The federal agencies responsible for administering public lands have concluded that ecosystem management is essential for arresting further habitat degradation, maintaining habitats that are relatively intact, and aiding in the recovery of at-risk species of fish (FEMAT 1993; FS and BLM 1994). Several recent efforts that incorporate an ecosystem perspective include the Aquatic Conservation Strategy in FEMAT (1993), the Eastside interim protection report (Henjum et al. 1994), and the PACFISH strategy (1994). BLM's strategy for managing wetland and riparian areas recognizes that "entire watershed condition is an important component in assessing whether a riparian-wetland area is functioning properly" (Barrett et al. 1993). EPA's Environmental Monitoring and Assessment Program (EMAP) is also based on the concept that all ecosystems existing in the landscape are integrated components and the condition of one component affects and is affected by the condition of the others. NMFS's current coast-wide status review of coho salmon and steelhead trout as well as the imminent coast-wide reviews of sea-run cutthroat trout and chinook, pink, chum, and sockeye salmon (NMFS 1994), further reflect this new approach.

The FEMAT and PACFISH approaches to aquatic resource conservation as well as other published conservation strategies (Moyle and Sato 1991; Doppelt et al. 1993; Frissell et al. 1993; Henjum et al. 1994) share two common elements. First, each of these strategies recognizes the importance of identifying and protecting those habitats that retain the highest degree of integrity to serve as refugia and centers from which population expansions can occur. Second, they recognize that an effective conservation strategy must emphasize restoring ecological processes and function and must be organized at a watershed (or larger) scale. "Key Watersheds" identified by the FEMAT report, the PACFISH strategy, and the Eastside Assessment illustrate these concepts as applied to federal lands.

We believe that a strategy for conserving salmonids on non-federal lands also must be ecosystem oriented, and must address the diversity of habitats required by different stocks and life-history stages on an integrated-ecosystem approach but one with additional considerations. The historical abundance of many salmonids in the Pacific Northwest was due in part to the diversity of life-history types that evolved to exploit a wide array of available habitats and that allowed temporal and spatial segregation of habitat use. In the diverse, geomorphically and tectonically unstable environments of the Pacific Northwest, well dispersed networks of locally adapted salmonids may be necessary for species persistence (Frissell 1993a). This diversity allows species to adapt to changing environmental conditions. Furthermore, the anadromous life-history pattern exposes the fish to a tremendous diversity of habitats that may include small headwater streams for spawning; larger streams, lakes, or off-channel areas for rearing; still larger streams as migration corridors; and estuaries and oceans for primary growth phases. Resident trout, char, and whitefish also spend

portions of their life histories in streams and lakes of various sizes. The ability (and necessity) to thrive in each habitat is critical to the success of a population.

Conservation of salmonids will require a comprehensive approach that addresses these spatial and temporal needs. Current strategies for managing federal lands begin this process, but because of the spatial distribution of federal lands, protected watersheds presently tend to be concentrated in higher-elevation areas, forested watersheds, and headwater streams. The FEMAT report specifically cites the importance of non-federal lands in an overall riparian conservation strategy, and the Eastside panel further stresses the need to accommodate a wide variety of habitat types through the establishment of Aquatic Diversity Areas. A strategy for non-federal lands should build upon existing conservation plans by reestablishing connectivity between habitats on federal and non-federal lands, and by working towards protection of habitats that are poorly represented in federal plans, particularly the lower-elevation streams and habitats for resident species, including non-game fishes. Both the FEMAT and PACFISH approaches focus on anadromous salmonids. Such a strategy should also provide guidance for managers so that actions at a local scale can be integrated into watershed and regional recovery plans.

Local habitat rehabilitation is essential within this broader context of conserving habitats and biodiversity across broad landscapes. Improved land-use practices and rehabilitation of riparian zones can provide many benefits, including decreased sediment transport to the stream, decreased stream temperatures, increased allochthonous nutrient inputs, increased flood plain interaction, stabilized ground water discharge, and increased inputs of large woody debris (Naiman 1992). As natural processes and conditions are restored, downstream reaches will be improved and connections between habitats reestablished (Salo and Cundy 1987), allowing greater expression of life-history diversity. Thus, private landowners can play a vital role in both improving local conditions and advancing the recovery of salmonids region wide. Furthermore, local actions can enhance other values, including water quality and quantity.

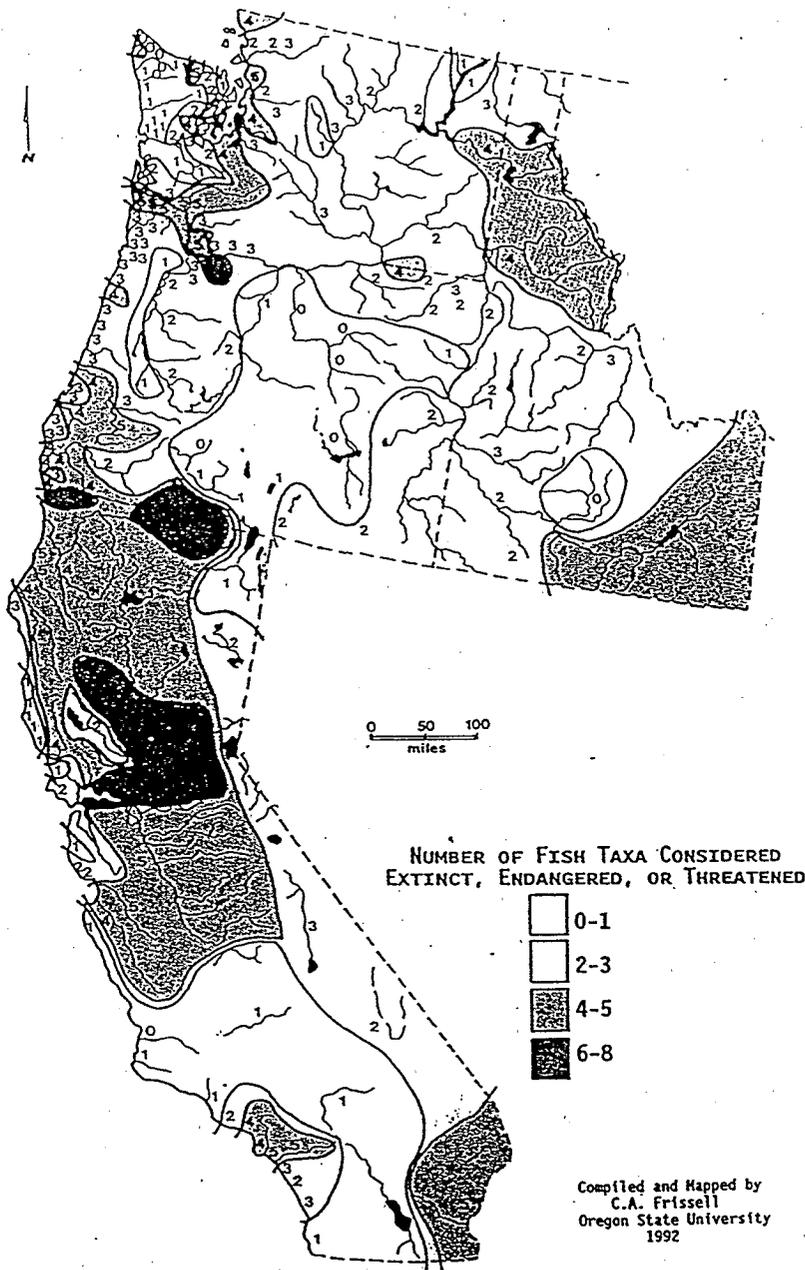
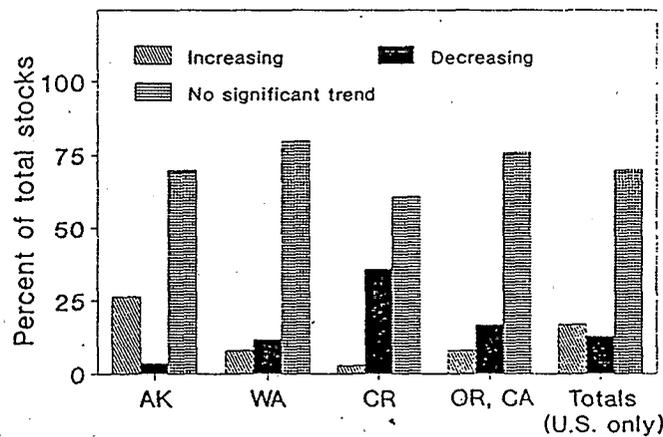


Figure 2.2a

Number of fish taxa considered extinct, endangered, or threatened in the Pacific Northwest and California. From Frissell (1993b). Reprinted by permission of Blackwell Scientific Publications, Inc.



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

Fig. 2.2b

Trends in the abundance of wild stocks of chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), chum salmon (*O. keta*), and steelhead (*O. mykiss*) from river systems along the Pacific coast. UP = percentage of stocks significantly increasing, DN = percentage of stocks significantly decreasing. Data from Konkel and McIntyre (1987), in (Naiman 1992). Reproduced with permission of the publisher.

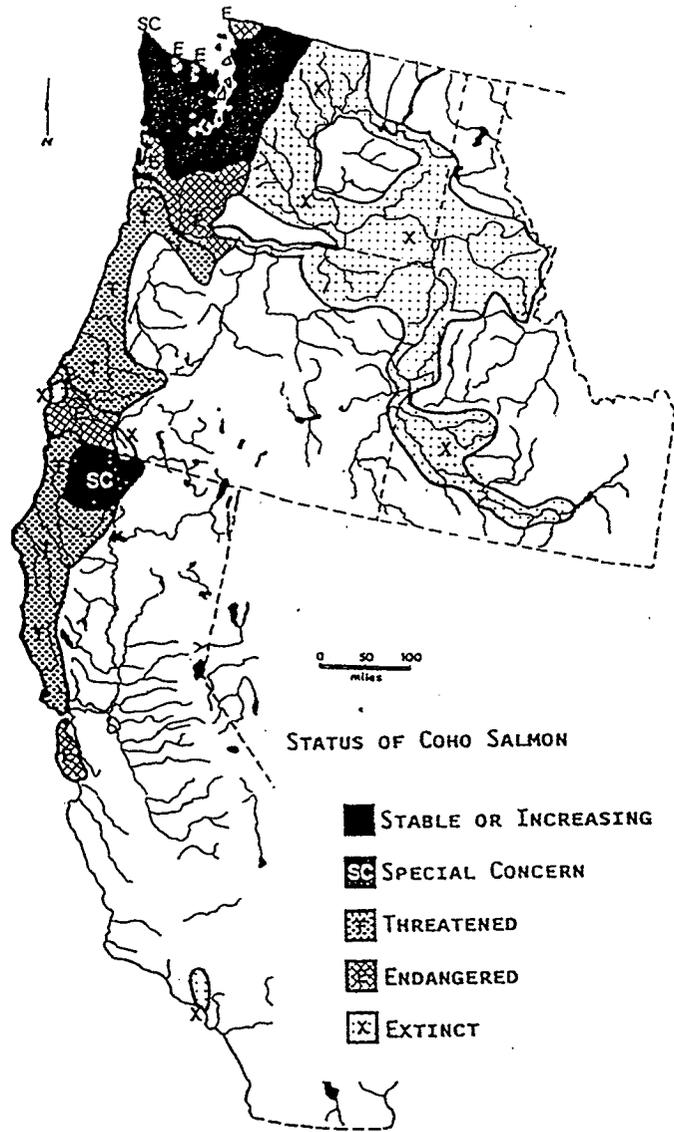


Fig. 2.2c

The historic range and present status of populations of coho salmon, (*Oncorhynchus kisutch*), in the Pacific Northwest and California. From Frissell (1993b). Reprinted by permission of Blackwell Scientific Publications, Inc. Based on data from Nehlsen et al. (1991) and Lee et al. (1980).

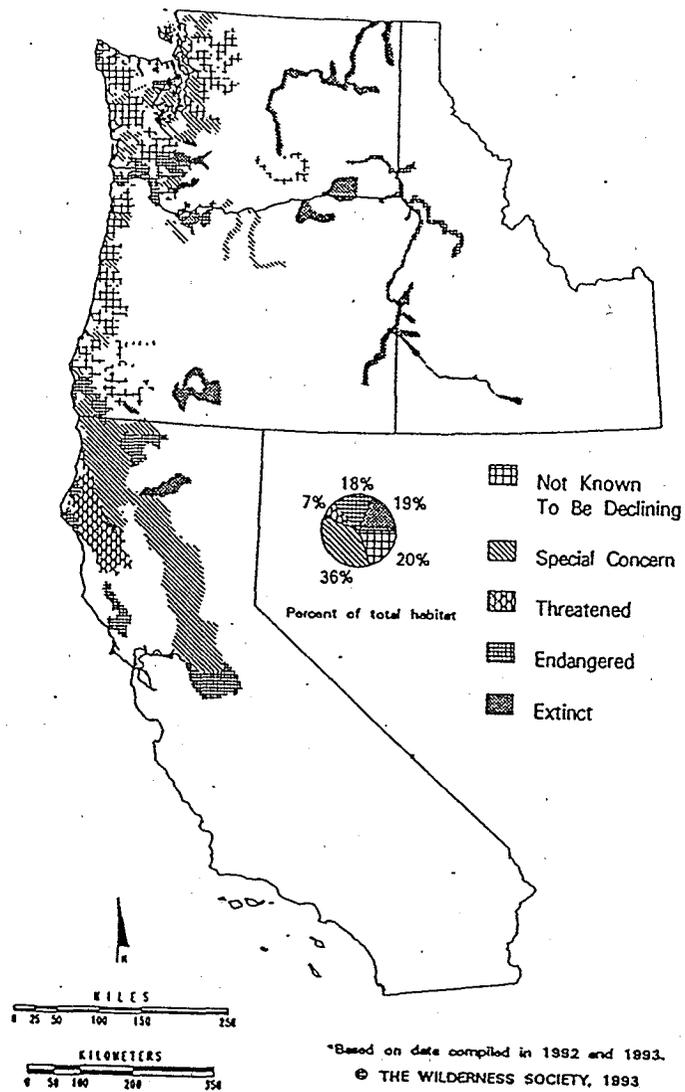


Fig. 2.2d

Status of fall chinook salmon in the Pacific Northwest and California.. From The Wilderness Society (1993). Reproduced with permission of the publisher.

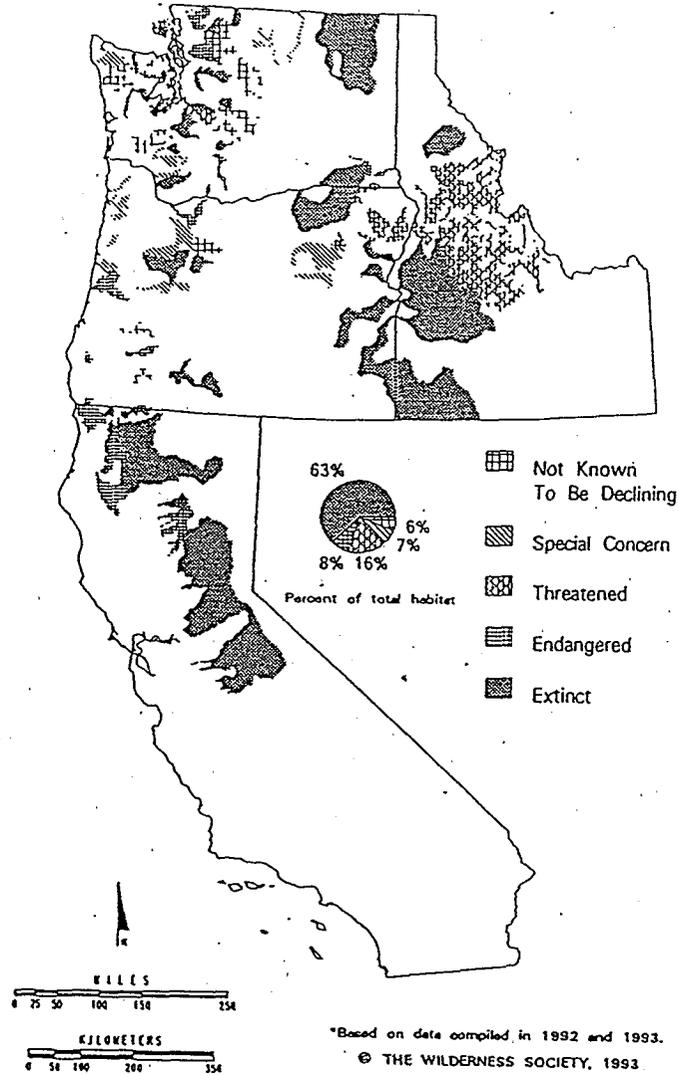


Fig. 2.2e

Status of spring and summer chinook salmon in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

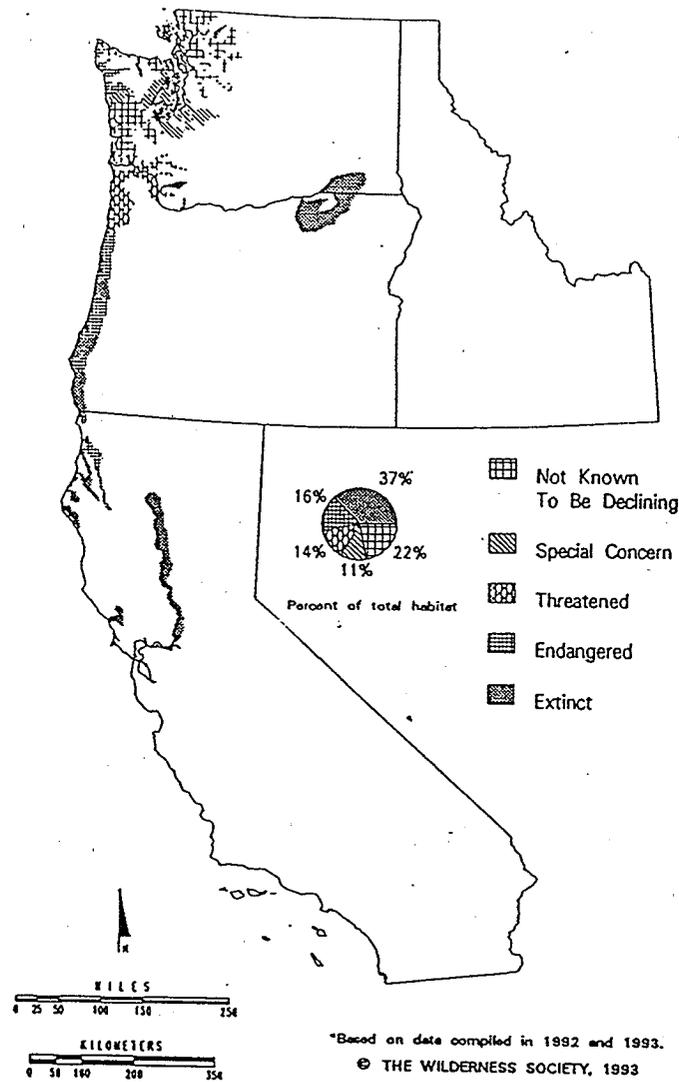


Fig. 2.2f

Status of chum salmon in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

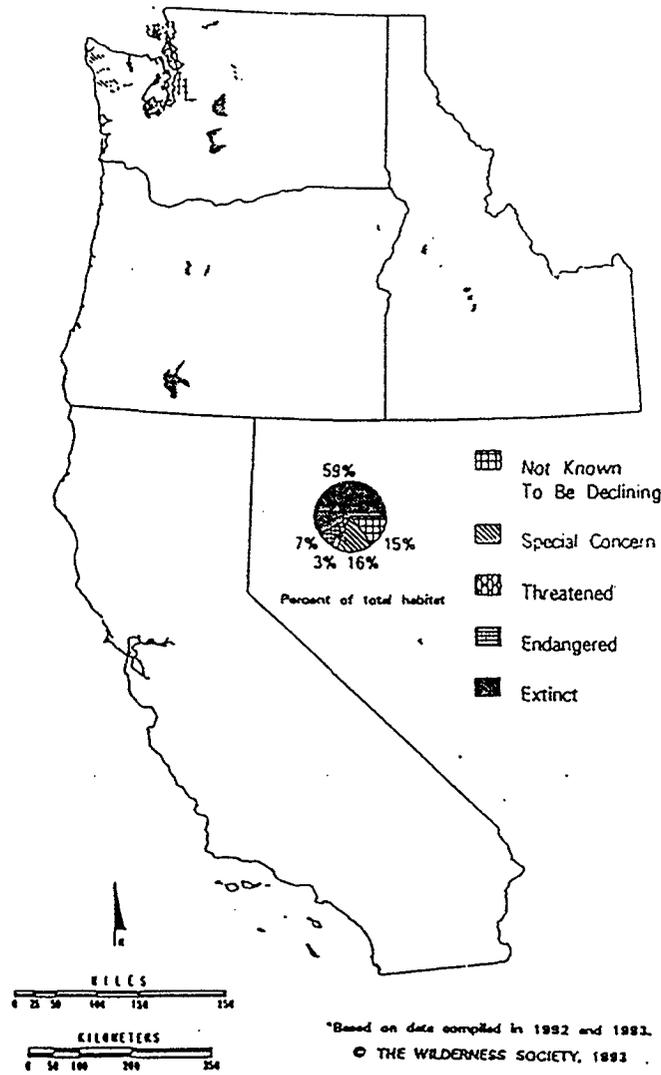


Fig. 2.2g

Status of sockeye salmon in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

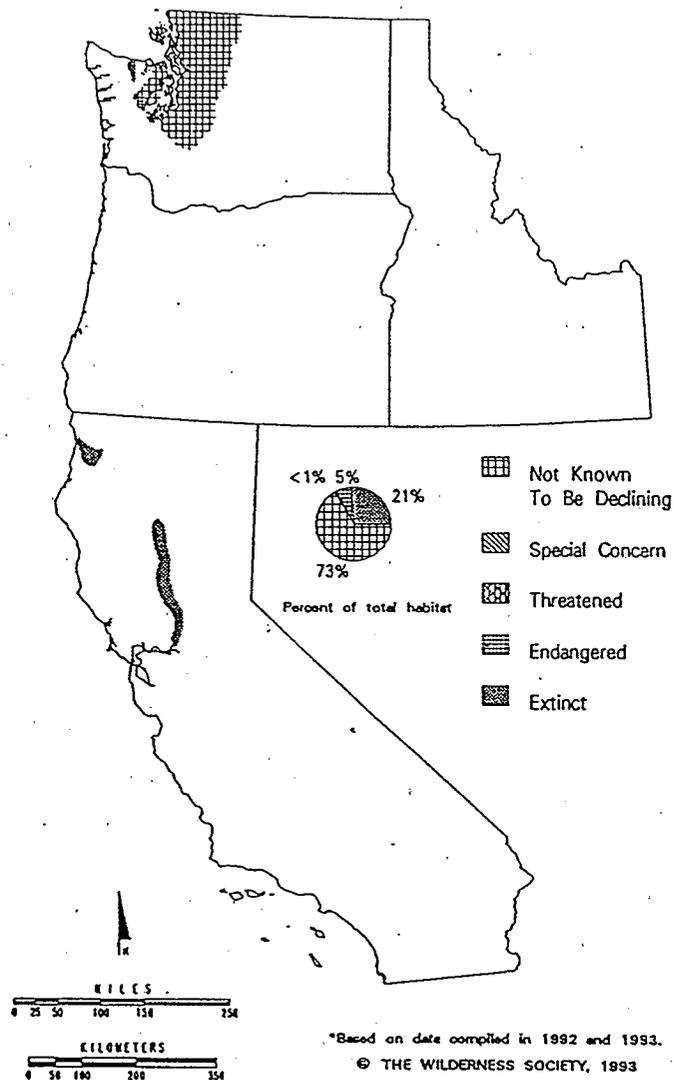


Fig. 2.2h Status of pink salmon in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

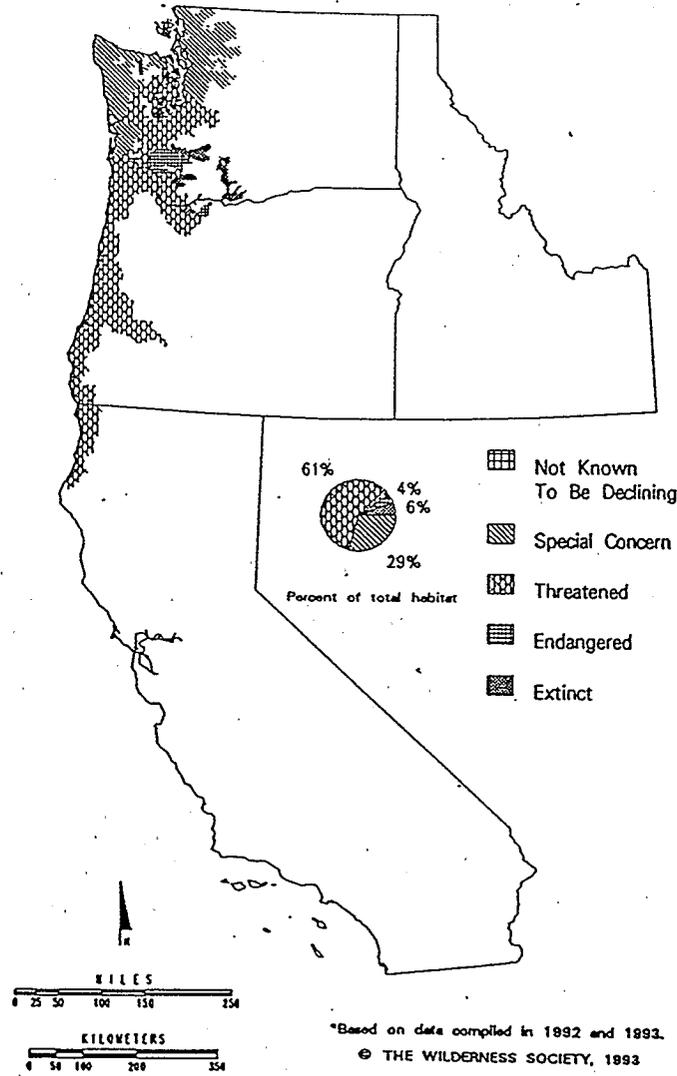


Fig. 2.2i Status of sea-run cutthroat trout in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

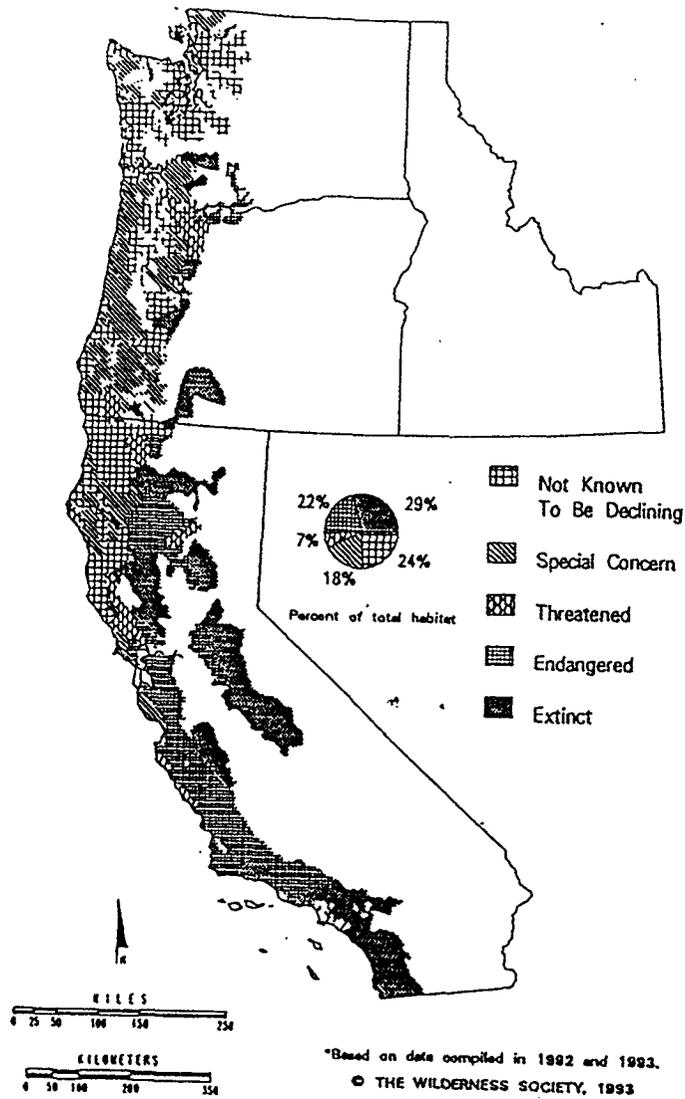


Fig. 2.2j

Status of winter steelhead in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

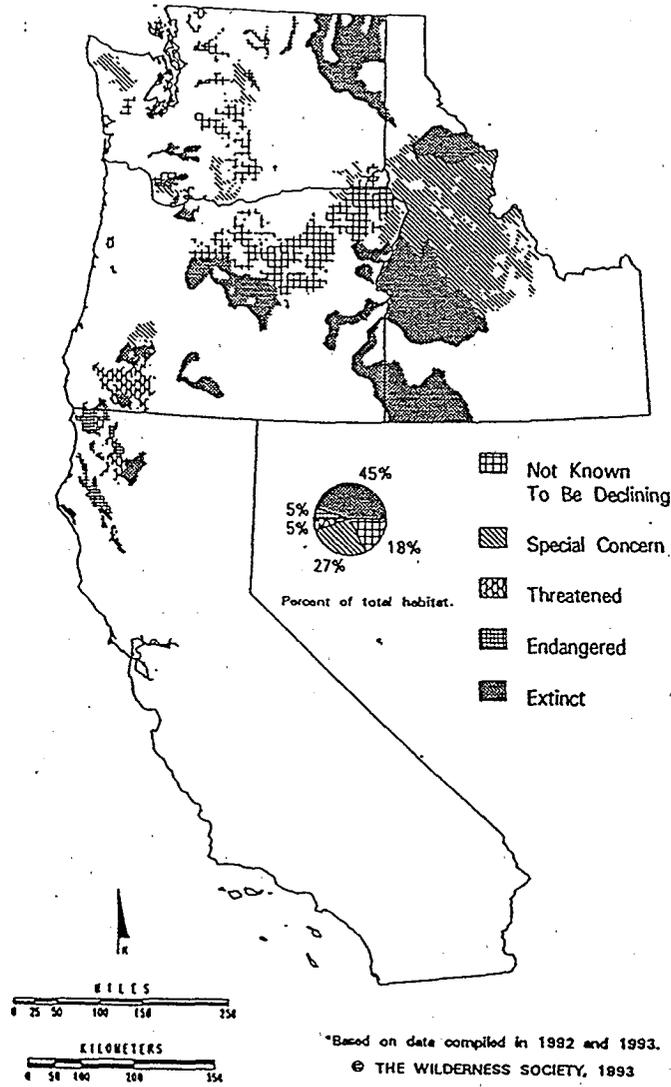


Fig. 2.2k

Status of summer steelhead in the Pacific Northwest and California. From The Wilderness Society (1993). Reproduced with permission of the publisher.

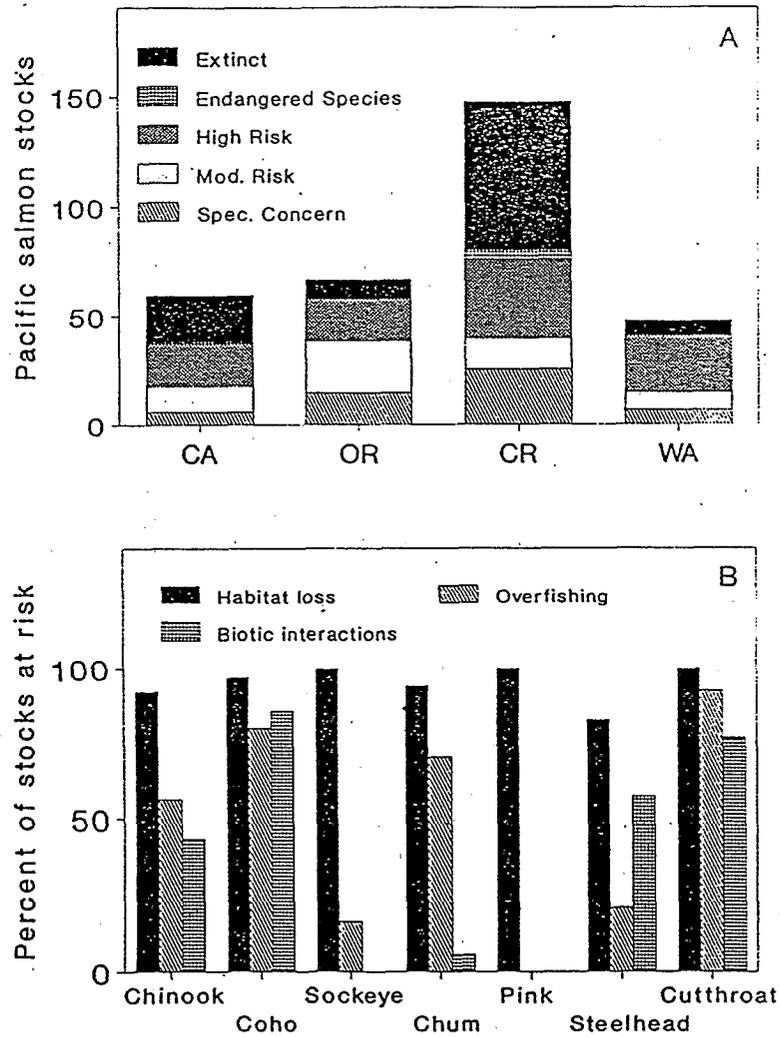


Fig. 2.3.1a. (A) Distribution of Stocks of Anadromous Pacific salmon (*Oncorhynchus*) in different extinction risk categories within various regions of the Pacific coast. (B) The percentage of stocks in which habitat damage, overfishing, and harmful biotic interactions have been implicated in declines of stock abundance. Figure from (Bisson, Quinn et al. 1992). Data from (Nehlsen et al. 1991). Reproduced with permission of the publisher.

Table 2.1a. Common and Scientific Names Salmonids native to the Pacific Northwest

| Common Name        | Scientific Name                                 |
|--------------------|---|
| pink salmon        | <i>Oncorhynchus gorbuscha</i> (Walbaum, 1972)   |
| chum salmon        | <i>Oncorhynchus keta</i> (Walbaum, 1972)        |
| coho salmon        | <i>Oncorhynchus kisutch</i> (Walbaum, 1972)     |
| sockeye salmon     | <i>Oncorhynchus nerka</i> (Walbaum, 1972)       |
| chinook salmon     | <i>Oncorhynchus tshawytscha</i> (Walbaum, 1972) |
| mountain whitefish | <i>Prosopium williamsoni</i> (Girard, 1856)     |
| cutthroat trout    | <i>Oncorhynchus clarki</i> (Richardson, 1836)   |
| rainbow trout      | <i>Oncorhynchus mykiss</i> (Walbaum, 1972)      |
| bull trout         | <i>Salvelinus confluentus</i> (Suckley, 1858)   |
| Dolly Varden       | <i>Salvelinus malma</i> (Walbaum, 1972)         |

### 3.0 Physical and Chemical Processes

The freshwater and estuarine habitats of salmonids are the product of interactions among numerous physical, chemical, and biological processes (Swanston 1991; Marcus et al. 1990) operating over long- and short-term temporal scales, and large and small spatial scales. Over millions of years, tectonic and volcanic activity in the Pacific Northwest has created a region of extreme topographic complexity, characterized by a series of mountain ranges that are oriented along a north-to-south axis and separated from one another by lowlands, and plateaus, or smaller mountain ranges. Significant portions of the Pacific Northwest landscape have been reshaped by glacial advance and recession. These large-scale, long-term geomorphic and climatic processes have created the physical template upon which rivers, and estuarine systems of the Pacific Northwest have formed.

Within a watershed, topographic, geologic, and climatic characteristics control soil development and vegetation cover and influence the transport of water, sediments, wood, and dissolved materials from upland areas to the stream channel. These transport processes occur continuously, but may be greatly accelerated during disturbances such as floods, debris torrents, landslides, and wildfires. The riparian zone acts as a filter that moderates the exchange of materials from terrestrial to aquatic ecosystems. In addition, riparian vegetation directly controls stream environments by providing shade and stabilizing streambanks, and through the input of organic litter and large woody debris.

A useful way to conceptualize how these processes ultimately affect salmonid habitats is in terms of a hierarchy of factors (Frissell et al. 1986; Naiman et al. 1992), where each component exerts influence on other components—usually at the same or lower levels—and all components ultimately influence stream conditions (Figure 3.0). Elements at the top of the hierarchy (e.g. climate and geology) have pervasive effects on other lower-level processes occurring in a basin or watershed (e.g. erosion, streamflow, nutrient cycling, riparian features). Lower-tier processes generally interact with other components at the same and lower levels, but may also influence components at higher levels through feedback loops. The specific characteristics of salmonid habitats are thus the manifestation of highly complex interactions among processes operating over many spatial and temporal scales.

In this section, we provide a broad overview of the dominant physical and chemical processes affecting the landscape and, ultimately, the aquatic ecosystems on which salmonids depend. The relative influence exerted by each specific process varies across the landscape with differences in geomorphology, geology, climate, hydrology, soil, vegetation, and other controlling factors. Consequently, the potential effects of human disturbances on aquatic systems is similarly variable in space. Our objective is to provide sufficient detail of physical and chemical processes so that regional differences in the response of ecosystems to human-caused perturbations can be understood. We begin the discussion with a review of processes that operate over large-temporal and spatial scales and over which humans have minimal influence. Next we review processes that operate at smaller spatial and temporal scales, and that may be substantially altered by land-use activities. Included in this discussion is a review of functional roles of riparian vegetation in influencing salmonid habitats. A summary of the effects of physical and chemical processes on salmonids and their habitats is presented in Section 3.10.

#### 3.1 Tectonism and Volcanism

Tectonic activity operating over millions of years created the rugged montane physiography, high local relief, and steep slopes of the Pacific Northwest—structural features that control the geographic patterns of drainage systems in the region. These processes thus set the stage for other geomorphic processes that shape stream channels.

Direct effects of tectonics on active geomorphic processes are generally limited in spatial extent and relatively infrequent, compared to other processes discussed in this document. The Pacific Northwest is subject to large subduction zone earthquakes at intervals of several hundred years. These large-magnitude earthquakes may cause subsidence in soft alluvial and coastal fills, creating zones of deposition (Atwater 1987; Darienzo and Peterson 1990), and they may also trigger mass movements of soil.

Volcanic activity has been regionally less significant than tectonics and glacial processes, but at local sites it has resulted in catastrophic readjustments of the landscape. Geomorphic impacts depend on the geochemical type of volcanism. Explosive eruptions of silicic volcanoes directly reshape the landscape, blocking and diverting drainage systems by ash flows, filling valleys or channels with mudflows and causing major inputs of sand and silt-sized sediments from tephra (airborne ash). Basaltic volcanic centers may block and divert drainage systems through lava flows and cinder eruptions, and also release limited amounts of tephra. Basaltic volcanic centers active in the Holocene are located in the Cascade Range from southern Washington to California, a few areas of eastern Oregon, and in the eastern Snake River Plain of Idaho. Recently active silicic volcanic centers are limited to the Cascade Range (Sarna-Wojcicki et al. 1983).

Volcanic mud- and ashflows commonly occur from volcanic eruptions in Cascade Range volcanoes. Mudflows can inundate valley floors with deposits less than one meter to tens of meters thick, and have produced the

most widespread geomorphic effects of past eruptions. Mudflows caused by the 1980 eruption of Mount St. Helens inundated valleys and completely buried pre-existing river channels. The channels subsequently reestablished on the mudflow deposits, through alternating episodes of incision, channel widening, and aggradation over a period of at least several years (Meyer and Martinson 1989). During the adjustment period sediment yields were much higher than before the eruption. Geomorphologic adjustments have been prolonged by landslides on slopes that were destabilized by the eruption. Ash flows also move down valleys and bury valley flow (Crandell 1976), while tephra may be deposited many kilometers from the source.

### 3.2 Glaciation

The landscape of the Pacific Northwest has developed under alternating glacial and interglacial periods over the last one million years or longer. Glaciation has affected the region's landscapes through: 1) direct modification of mountain areas and limited lowland areas by glaciers, 2) eustatic sea-level lowering, which has had major effects on coastal rivers and estuaries, 3) glacial-interglacial climatic changes that have influenced the hydrologic regime, and 4) climate-driven changes in vegetation cover that have affected hillslope and stream processes (Table 3.2). In general terms, glacial periods are times of rapid sediment transfer from uplands to lowlands and the ocean by glacial advance and meltwater transport in glaciated areas, and by increased streamflow in unglaciated areas. Interglacial periods tend to be periods of sediment accumulation in upland valleys with limited fluvial transfer out of the uplands (Thorson 1987). Where residence times of sediment accumulations, or recurrence intervals of events, are thousands of years (Dietrich et al. 1982; Kelsey 1982), glacial-interglacial transitions may be the most important periods in landscape formation.

During the last glacial period, about 22,000 to 15,000 years ago, ice sheets and mountain glaciers were developed in many areas of the Pacific Northwest, and sea level was about 100 m below present, exposing large areas of the continental shelf. The Cordilleran ice sheet extended south from British Columbia, covering the Puget Lowland, northern Cascades, Okanogan Valley, and upper Columbia Valley in Washington. South of the ice sheet, mountain ice sheets and glaciers were widely distributed in the mountainous regions of Washington, Oregon, Idaho, and the Siskiyou of Northern California (Crandell 1965; Porter et al. 1982). Climate of the glacial period was much colder than today. Although effective moisture in the Pacific Northwest was less (Thompson et al. 1993), runoff likely was as high or higher than today, due to changed land surface conditions. Down the valley from glaciers and in unglaciated watersheds, frost-weathering and mass-wasting were probably more intense than at present. River systems probably had greater streamflow and transported greater sediment loads. In addition, enormous ice jams periodically developed and broke, resulting in catastrophic floods that formed the coulees of eastern Washington and deposited the deep soils of the Willamette Valley.

In addition to these physical changes, ecological changes resulting from glacial climates may have also influenced geomorphic processes. The pre-glacial ecosystems of these glaciated areas were significantly displaced by glaciation, and species and stocks present today in these ecosystems likely existed in refugia south of and/or at lower elevations than the glaciers. In unglaciated parts of western Oregon and Washington, the vegetation consisted of tundra close to the glaciers and subalpine parkland elsewhere, including the Oregon Coast Range (Worona and Whitlock 1994). In eastern Washington, a sparse periglacial steppe was present (Barnosky et al. 1987; Whitlock 1992; Thompson et al. 1993; Whitlock et al. 1993). In the more sparsely vegetated landscape of the last glacial period and less large organic debris was available to influence streams and valley floors. Present environmental conditions have prevailed in this region for the last 6,000 to 8,000 years. The establishment of current conifer forest communities did not occur until 5000 to 2000 years ago (Whitlock 1992; Worona and Whitlock 1994), and in glaciated watersheds of western Washington, stream channels reached conditions similar to those of the present by about 6,000 to 8,000 years ago (Benda et al. 1994). As forest stand density and height increased with climatic amelioration, woody debris exerted a stronger influence on stream and valley morphology. Some channel incision and narrowing of meander belts probably continued into the late Holocene.

A general model of river channels based on empirical evidence from several parts of the world suggests that channel changes from glacial to interglacial periods follow a specific sequence. Braided channels dominate during glacial periods. During interglacial periods these change to transitional braided meandering channels with mid-channel bars but well-defined thalwegs, and then to large meandering channels adjusted to higher than present discharge. Finally, smaller meandering channels develop during stable conditions typical post-glacial periods (e.g., late Holocene; (Schumm and Brakenridge 1987). In the Pacific Northwest, the late-glacial to early Holocene period was likely characterized by channel incision into thick glacial-period valley fills, formation of terraces, sediment yields higher than present as rivers downcut, and significant changes in channel morphology due to changed hydrologic and sediment regimes (Benda et al. 1994).

In addition to the changes in inland watersheds described above, coastal rivers were directly affected by lowered sea level during glacial periods (McDowell 1987). At the last glacial maximum, global sea level was 100 m or more below the present sea level, and the shore was 10 km or more west of its present location. Coastal streams flowed

across the exposed continental shelf, perhaps in incised valleys. Estuaries were very limited in extent. As global deglaciation began, sea level initially rose very rapidly creating deep coastal estuaries. Beginning 10,000 years ago, sea level rise continued at a decreasing rate, and it has fluctuated close to the present level since 4,000 years ago. Shallow-water conditions in estuaries, including mud and sand flats, have become established only recently (McDowell 1986, 1987).

### 3.3 Wildfires

The historical frequency of fires varies over the landscape as a function of climate and vegetation type. Fires in higher elevation communities, including subalpine fir, western hemlock/red cedar, lodgepole pine, and grand fir tend to recur at an interval of decades to centuries, while low-to-mid elevation juniper, ponderosa pine, Douglas-fir, and white fir forest typically experience fires at intervals of several years to a few decades. Little information is available regarding the historical frequency of fire in grassland, shrubland, and woodland communities east of the Cascade Crest (Agee 1994). Stiff sagebrush and Sandberg's bluegrass plant assemblages may have biomass sufficiently low to prevent large-scale fires, while other communities, including various fescue and bluebunch wheatgrass assemblages may have sufficient biomass to carry fire, but lack sources of ignition (i.e. lightning) during the periods when they are most combustible (Agee 1994). Fire frequency in other sagebrush and woodland communities is poorly documented. West of the Cascades, higher humidity, lush vegetation, and less frequent lightning storms combine to reduce the frequency of wildfire; however, under dry summertime conditions, the effects of wildfire in dense timber stands can be substantial. During the period 1933–1951, four fires in the Coast Range of Oregon, collectively known as the Tillamook fires, burned more than 643,000 acres and had significant and long-lasting effects on forest and riparian communities.

Riparian areas are generally characterized by a higher percentage of deciduous plants than is found in surrounding uplands. In addition, local microclimates tend to be cooler, resulting in moist soils and low fuel moisture. Because of these attributes, riparian areas do not burn or they burn at lower intensity than forests in upland areas (Agee 1994). As such, they may buffer aquatic communities from some of the effects of wildfire. However, in headwater reaches and at higher elevations, stronger winds and greater biomass may facilitate fires of relatively high intensity (Agee 1994). Consequently, it is difficult to generalize about the effects of fires on the riparian zone.

Fires in upland areas and riparian zones can affect aquatic ecosystems by altering vegetation cover, which in turn influences erosion and sediment transport, water infiltration and routing, the quantity of nutrients reaching streams, the amount of shading, and the input of large woody debris into the system (Wissmar et al. 1994). The extent of impacts is generally related to the intensity of the burn. In high intensity fires, soil organic matter that helps hold soils together is consumed, increasing the susceptibility of soils to erosive forces. In addition, volatilization of certain compounds can cause the surface-soil layer to become hydrophobic, thereby reducing infiltration of water and increasing surface runoff (Marcus et al. 1990). The combined effects of vegetation loss and hydrologic changes can alter the frequency of severe debris torrents (Wissmar et al. 1994). Nutrients such as phosphorous, nitrogen, and sulfur may be volatilized into the atmosphere (Everest and Harr 1982), or lost through leaching and soil erosion. The loss of riparian vegetation can increase exposure to solar radiation, causing stream temperatures to warm. Inputs of large woody debris may also change following fire in the riparian zone. In speculating about the effects of the Yellowstone fire of 1988, Minshall (1988) hypothesizes that large woody debris in streams would likely increase immediately following the fire—due to augmentation of existing woody debris with falling branches—and then decrease through time because new growth contributes little to instream woody debris.

Humans have significantly altered natural fire regimes through land-use practices and an extensive and long-term focus on fire suppression. As a result, significant changes in forest vegetation have resulted. East of the Cascades, fire suppression has led to shifts in vegetation from historically open stands of ponderosa pines and western larch to stands with dense understories of Douglas-fir and grand fir (Mutch et al. 1993). Ponderosa pines are well adapted to frequent, low-intensity burns that were characteristic of eastside forests. These fires tended to prevent fire intolerant species from invading. Drought and subsequent insect infestations have killed many understory trees, allowing fuels to accumulate and increasing the probability of high intensity fires (Wissmar et al. 1994). Consequently, ecosystems that once experienced frequent but small wildfire disturbances are now prone to infrequent but much more catastrophic events.

### 3.4 Sediment Transport

Sediment transported from upland areas into stream channels determines the nature and quality of salmonid habitat in streams, rivers, and estuaries. The development and persistence of morphological structures used for spawning, incubation, and rearing depend on the rate at which sediment is delivered and the composition of deposited materials. Sediment delivery rates and composition, in turn, are controlled by climate, topography, geology, vegetation, and hydrology. Local variation in these watershed characteristics ultimately determine the type and quality of habitat found in a given system.

Land-use practices, through alteration of soil structure, vegetation, and hydrology, can significantly alter the delivery of fine and coarse sediments to streams, thereby affecting salmonid habitats. In this section, adapted primarily from Swanston (1991) we discuss surface erosion and mass wasting, the dominant forms of sediment transport, as well as environmental factors that influence these processes. The routing of sediments within the stream channel, and the role of large woody debris in controlling sediment movement are discussed in other sections of this document (see section 3.5 and 3.9.5).

### 3.4.1 Surface Erosion

Surface erosion results from rain and overland runoff. Particulate and aggregate materials are relocated via a two-step process: dislodgement and then downslope transport of detached materials. Detachment is influenced by the size and compaction of particles, and by the protective cover of organic litter and plants. Slope gradient and length, rainfall intensity, and soil infiltration rate determine transport rate (Swanston 1991). Initiation of erosion may be caused by landslides, fire, logging, rain, and drop splash from forest overstory, animal activity, freeze-thaw phenomena, or any other soil surface disturbance. Surface erosion rarely occurs on undisturbed forest lands west of the Cascade crest due to high infiltration rates, though it may occur in areas with steep (> 50 %) slopes (Swanson et al. 1987) and in sparsely vegetated lands east of the Cascades.

Most surface sediments that reach stream channels result from channelized erosion (rilling and gullying) and sheet erosion (Brown 1980; Swanston 1991). Channelized erosion occurs when flows are restricted by landforms, and typically occurs following heavy storms or snowmelt (Beschta et al. 1994). It is considered the most significant form of surface erosion on forest lands (Brown 1980). Although uncommon in undisturbed forested situations, rills may occur when infiltration capacity is reduced (Megahan 1991). In contrast sheet erosion develops from detachment initiated by nonchannelized overland flow, and raindrop splash, or by gravitational and wind movement of dry particles (dry ravel). These processes generally occur on exposed soils and tend to remove soil uniformly over an exposed area. Sheet erosion tends to be of greater significance on low gradient agricultural lands than on forested lands, whereas dry ravel occurs on steep slopes in soils lacking cohesion (Swanson et al. 1987; MacDonald and Ritland 1989).

### 3.4.2 Mass Wasting

Mass wasting, including slumps, earthflows, landslides (or debris avalanches), and soil creep, is often a major component of sediment delivery to streams, particularly in mountainous regions where surface erosion is minor (Swanson and Dymess 1975). Generally episodic in nature, mass wasting can provide large quantities of sediment and organic material to streams. Currently, it is believed that periodic mass wasting (300–500 yr return interval) is necessary to replenish large woody debris and coarse sediments in streams. Under natural conditions, at any given time, a basin would have had reaches with too much or too little coarse sediment. Through time these naturally coarse sediments would have eroded and redeposited. This variation in space and time created areas of naturally excellent and poor salmonid habitat. By altering these spatial and temporal patterns, humans have degraded and simplified stream habitats.

Slumps and earthflows generally develop in deeply weathered soils. These often occur in sedimentary geology (siltstones, sandstones, mudstones) and volcanoclastic rocks. In these soils with primarily clay-sized particles, low soil permeability restricts groundwater movement (Swanston 1991). Slumps are the sliding of soil blocks along a concave surface while earthflows often begin as slumps or a series of slumps. Once initiated, rheological flow of the clay fraction keeps the individual soil blocks moving downslope like a viscous fluid in earthflows. Earthflows tend to be seasonal with most movement occurring after rains have saturated soils. These flows are slow moving, ranging from 2.5–2720 cm/yr (Swanston 1991), and may eventually protrude into the stream channel where they are gradually eroded away. As they erode, residual lag deposits may form which can increase channel gradient downstream through the accumulation zone. These areas, if in otherwise "sediment poor" reaches, and if they contain coarse sediments, may increase the habitat diversity in a morphologically uniform channel and have a long-term beneficial effect on fish habitat.

Soil creep is soil movement that is imperceptible except by measurements taken over long periods of time. Carson and Kirkby (1972) identify causes including systematic reworking of the surface soil layers due to frost heaving, steady application of downward shear stress, and random movements due to organisms or micro-seisms. Continuous creep tends to occur in clay soils, and is absent in coarse-grained soils.

Landslides typically occur in shallow noncohesive soils on steep slopes overlying less permeable bedrock (Beschta et al. 1994). Conditions causing landslides include 1) zones of weakness in soil or bedrock, 2) wind stress transferred to the soil by trees, 3) deformation caused by soil creep, 4) drag caused by seepage pressure, and 5) removal of slope support by undercutting. Landslides—relatively dry soil masses—are distinguished from debris flows, which are typically saturated. When landslides enter stream channels during floods, they become debris flows—large volumes of water containing soil, rock, and frequently, large organic debris. These flows scour the channel and severely modify

fish habitat as they move rapidly downstream. As debris flows move downstream into higher order channels, their effects become less pronounced due to increasing streamflow.

### 3.4.3 Factors Affecting Erosion and Sedimentation Rates

The magnitude, locations and frequency of sediment delivery to active channels is highly dependent upon climate, local topography, soil type, soil saturation, vegetative cover, organic matter, depth and degree of weathering, and degree of upslope disturbance (Swanston 1991; Beschta et al. 1994; Reiter et al. 1994). Rain-dominated watersheds tend to yield more sediment than snow-dominated systems, although inter-basin variability is quite high. Larson & Sidle (1981) examined data from 13 relatively undisturbed watersheds and reported sediment yields of 2.0 to 40.7 tonnes/km<sup>2</sup> for rain-dominated systems. For snow-dominated systems, sediment yield typically ranged from 1.6 to 6.1 tonnes/km<sup>2</sup>/yr; however, two watersheds had substantially higher yields of 39.9 and 117.1 tonnes/km<sup>2</sup>/yr (see Swanston 1991). Within-year variation in sediment production can also be high. Larson and Sidle (1981) reported differences in sediment yield among years of an order of magnitude or more for both rain-dominated and snow-dominated systems.

The timing, frequency, and type of precipitation influences the rate and yield of sediment delivered to stream channels. In rain-dominated ecoregions (see Table 3.6.1a) sedimentation and allochthonous inputs are minimized during summer low-flow periods. Sedimentation and allochthonous inputs increase during the wet months of September to February when soils are saturated and landslide hazards are highest. In snow dominated ecoregions, sedimentation and allochthonous inputs are minimal during the spring, summer and fall, similar to the dry period in rain-dominated regions. However, when rapid snowmelt or a high-intensity storm occurs, high streamflows result, and entire hillslope and channel systems erode, increasing the rate of allochthonous inputs (Swanston 1991).

Topography influences slope steepness, length, elevation, and aspect. Runoff energy is highest on steeper slopes with greater slope length, which increases the volume and velocity of water moving downslope. Failures that occur on lower areas of the hillside nearer streams have a greater potential of reaching the stream.

Parent material and soil types also determine soil texture and erodibility. Erodible materials include soils derived from granite, quartz diorite, granodiorite, Cenozoic non-marine sediments, and schist (Beschta et al. 1994). Diorite and various metamorphic rock derived soils have intermediate erodibility, and non-erodible materials include andesite, basalt, peridotite, serpentinite and pre-Cenozoic and Cenozoic marine sediments. Some important soil properties affecting mass wasting include cohesion, structure, porosity, moisture capacity, drainage, chemical properties, and soil depth, all of which are affected by the relative proportion of clay, silt, and sand in the soil (Swanston et al. 1980). Typically, soils with little cohesion, structure, or porosity, low moisture capacity, and poor drainage are more likely to erode.

Vegetative cover tends to reduce sediment transport by reducing detachment rate and through the binding capacity of root masses (Harvey et al. 1994; Larson and Sidle 1981). Organic matter, utilizing water as the cementing agent, helps to form aggregates which tend to be more resistant to detachment and transport (Dyrness 1967 in (Beschta et al. 1994).

### 3.4.4 Regional Differences

East of the Cascades, soils are most susceptible to surface erosion. However, mass wasting events are important as well. Slumps originate in fine textured soils while debris torrent failures occur in weakly cohesive ash (Harvey et al. 1994) suggest that the high infiltration rates in most soils of the inland Pacific Northwest region make them less susceptible to surface erosion unless slopes are greater than 30%, and unvegetated. Compacted ash and pumice soils on shallow slopes are susceptible to gullying due to their low density and cohesion. In areas characterized by coarse, cohesionless soils and periods of drought, dry creep and sliding of materials from denuded slopes may be an important source of local surface erosion (Swanston 1991). Dry ravel is significant on slopes greater than 40% in pumiceous, cindery, and ashy soils—conditions confined to specific areas in the central Oregon Plateau and eastern portions of the central Washington Cascades.

Mass wasting occurs with high frequency in the western Cascades and coast range (MacDonald and Ritland 1989; Beschta et al. 1994). Wet climatic conditions in the coast range and valleys tend to promote deep soil formation and clays, which are prone to slow continuous failures, including slumps, soil creep and earthflows. At higher elevations near the Cascade crest, shallow cohesionless soils overlying slightly weathered bedrock are susceptible to landslides. Sediment budgets from three sites illustrate these regional differences. The wet, snowmelt-dominated, glaciated, and tectonically active Queen Charlotte Islands of British Columbia have sediment yields an order of magnitude greater than drier, snowmelt dominated, granitic lands of central Idaho, while in the rain-dominated regions of western Oregon and Washington, yields are intermediate to the other two regions (MacDonald and Ritland 1989).

### 3.4.5 Influence of Sediment Transport on Nutrients, and Organic and Chemical Movement into Streams

The mode of sediment transport has an impact on the rate of delivery of nutrients, organic debris, and chemicals to streams. Surface erosion affects productivity through the loss of nutrient-rich organic layers and topsoil, and leaching (Beschta et al. 1994). Due to the uneven distribution of nutrients in soils, minimal losses of organic litter and top soil may result in disproportionately high loss of a nutrient such as nitrogen. If nutrients are limiting, small losses of nutrients may cause substantial losses in productivity. Maintenance of organic material in the soil increases adsorption and decreases erodibility. Mass wasting events such as landslides carry large quantities of large woody debris in forested areas, to the active channel along with large volumes of sediments.

### 3.5 Channel Morphological Features and their Formation

Stream conditions that are important for aquatic habitat can be observed over a range of scales from an entire drainage network to a reach to a channel unit (Gregory et al. 1991). Average values of many stream characteristics, such as width, depth, velocity, and bed material size, vary systematically in a downstream direction. There are, however, important patterns of variation at local scales, such as the reach and the channel unit scales. Reaches are stream and valley segments, typically 1–10 km long, within which gradient, valley width, and channel morphology are relatively homogeneous and distinct from adjacent segments. Reach-scale variation is controlled by geologic factors such as rock type, geologic structure and location of geomorphic features such as terraces, alluvial fans and landslides (Table 3.5a). In many streams, high-gradient reaches with narrow, constrained valley floors are interspersed with lower-gradient, alluvial reaches with wide valley floors (Grant et al. 1994). In such streams, channel width/depth ratios, channel bed morphology, and channel-valley floor relationships will vary significantly from reach to reach. Reach-scale variations influence the location of spawning areas and types of fish that inhabit a stream system (Grant et al. 1994; Montgomery 1994). Consequently, reach-scale variations are an important consideration in watershed planning.

Channel units or habitat units consist of bed morphological features such as pools, glides, riffles, rapids, stepped-pool sequences, cascades, and steps (Table 3.5b). Channel units exert an important influence on local flow hydraulics and bed sediment characteristics (Grant et al. 1990). Channel units therefore provide the local habitat context for aquatic insects, fish, and other animals that inhabit stream channels.

The relative abundance of different channel unit types, such as pools or cascades, varies from reach to reach in response to variation in controls such as bedrock type, reach gradient, mass movement features, sediment size, and position in the channel network. Steep reaches, associated with resistant bedrock types or with coarse substrate deposits that intersect the channel, are dominated by cascades, rapids, and/or steps, with limited pool, glide and riffle area (Grant et al. 1990). Overall these channel units combine to form a step-pool channel morphology in steep reaches. The steep channel units are associated with boulder-sized bed material. Biological processes also play a role by creating steps and pools adjacent to large woody debris accumulations (Beschta and Platts 1986). Although formed of boulder-size material that exceeds the competence of most high flow events (i.e., mean annual floods), stepped-bed channels in steep mountain streams of the Pacific Northwest are not residual features but are in equilibrium with the modern hydrologic regime (Grant et al. 1990). The channel units are reworked by flow events with recurrence intervals of twenty-five to fifty years. In less steep reaches, cascades, rapids and steps are less frequent, bed material is dominated by cobbles and gravels rather than boulders, and the abundance of pools, riffles and glides is higher. Active bedload, transported during frequent high flows (mean annual flood), accumulates in bars that are positive relief features on the channel bed. These bars result in pool-riffle channel morphology that is expressed at moderate to low flows. Pools are located at points of scour during high flow events, and riffles are formed by bar fronts (Lisle 1982; Beschta and Platts 1986; Wohl et al. 1993).

Human modification of flow and sediment regimes can modify the abundance and character of channel units. Human impacts resulting in net aggradation, for example, tend to reduce pool area and depth (Lisle 1982; Beschta and Platts 1986). Human impacts that decrease woody debris input to the channel can have the same effect, as can flood-induced aggradation (Lisle 1982). Human impacts that result in net degradation may also reduce pool area if bedload is depleted and bedrock is exposed in the channel bed.

### 3.6 Hydrology

The flow in streams and rivers represents the integration of the climate, topography, geology, geomorphology, and vegetative characteristics of a watershed. Precipitation may be intercepted by the vegetation and subsequently evaporate, or it may reach the ground either directly or as throughfall. Water reaching the ground either evaporates, infiltrates into the soil, or flows overland until it reaches the stream or an area where infiltration is possible. Water that infiltrates the soil may be taken up by plants and transpired back into the atmosphere, remain in the soil as stored moisture, percolate through the soil into deep aquifers, or enter streams via subsurface flow. Each of these processes affects the amount and timing of streamflow.

Land-use disrupts natural hydrologic processes, altering the amount of evaporation, transpiration, and runoff, the routing of water through the system, and the temporal patterns of streamflow. Regional differences in the hydrologic cycle can affect the response of a watershed to human disturbance. Consequently, an understanding of basic hydrologic processes is critical to understanding how land-use practices influence streamflow and how these effects vary across the landscape. This section provides a brief overview of hydrologic processes that occur in a watershed, with emphasis on those processes that may be substantially modified by human disturbance. A thorough review of hydrologic processes can be found in Swanston (1991).

### 3.6.1 Precipitation

The amount, form, and timing of precipitation differs dramatically across the Pacific Northwest, with the primary controlling factors being latitude, elevation, and proximity to the ocean and mountain ranges (Jackson and Kimerling 1993). Moisture-laden air generated over the Pacific Ocean is uplifted and cooled as it approaches mountainous regions, causing water to condense and fall as precipitation. After the air mass passes over these mountains it warms again, increasing its capacity to hold moisture. Thus, areas on the east slope of mountain ranges receive less rainfall than western slopes of comparable elevation, the so-called "rain shadow" effect. Convection storms (i.e. storms generated by heating and upward expansion of air masses near the earth's surface) may also be a significant source of precipitation during the spring and summer months in mountainous regions and continental climates east of the Cascade and Sierra Nevada crests. These storms tend to be localized events of high intensity and relatively short duration.

Three general precipitation patterns are represented in the Pacific Northwest: rain-dominated systems, transient-snow systems, and snow-dominated systems (Table 3.6.1). Rain-dominated systems include the coastal mountains, lowland valleys, and lower elevations of the Cascade and Sierra Nevada ranges, which are characterized by moderate to high precipitation that falls primarily as rain from late fall to early spring. In some coastal regions and lower elevations of the western Cascades, fog drip from forest canopies may also constitute a significant part of the total precipitation (Oberlander 1956; Azevedo and Morgan 1974; Harr 1982). The transient-snow zone includes mid-elevation areas of the Cascades, northern Sierra Nevada, and Olympic Mountains that also receive most of their precipitation in the late-fall to early spring, as both rain and snow (Swanston 1991). Hydrologically, this transient zone is particularly important during rain-on-snow events. When warm, moist air-masses pass over snowpack, condensation of water on the snow surface occurs, releasing large amounts of latent energy during the phase change of water from vapor to liquid. A small amount of condensation can facilitate the rapid melting of substantial volumes of snow. Snow-dominated systems include those that receive precipitation predominately as snow, including the higher elevations of the Cascade, Sierra Nevada, Olympic, and Rocky Mountain (and associated) ranges, as well as mid-elevation interior basins of the Columbia and Snake rivers. In the mountainous regions west of the Cascade and Sierra crests, precipitation is highly seasonal with most falling from fall through spring. East of the Cascades and Sierra Nevada, the strong seasonal signature of precipitation diminishes and precipitation is spread more evenly throughout the year, particularly where spring and summer convective storms contribute substantially to the total annual precipitation. In high elevation areas of eastern Washington, the Cascades, and the Rocky Mountains, rime and hoar-frost formation may also contribute significantly to the overall water balance of a watershed (Berndt and Fowler 1969; Gary 1972; Hindman et al. 1983).

### 3.6.2 Evapotranspiration

Evapotranspiration losses include those water losses from interception by the canopy and subsequent evaporation, evaporation of water that reaches the soil, and water that enters the soil and is subsequently taken up by plants and transpired back into the atmosphere. The amount of water lost through these processes depends on vegetation type, season, and the nature of the precipitation event, including the intensity, duration, and form of the precipitation, as well as climatological conditions during the event (e.g. temperature, humidity, wind speed).

*Interception Losses* - Dense coniferous canopies have greater interception storage capacity than those of sparse coniferous forests, deciduous forests, shrublands, or grasslands (Wisler and Brater 1959; Zinke 1967; Rothacher 1963) reported interception and evaporation losses of nearly 100% during low-intensity rainfall events (<0.13 cm) compared with losses of only 5–12% during high-intensity events (>5 cm) in an old-growth Douglas-fir forest in western Oregon. Annual interception losses for woodland-chaparral vegetation in central California ranged from 5–8%, with seasonal losses of 4% during the winter and 14% during the spring and summer when vegetation was in full foliage (Hamilton and Rowe 1949).

Interception by coniferous canopies during snowfall can also be substantial. Snow may be temporarily stored in the canopy and then delivered to the snowpack during the storm as branches become heavily laden or following the storm by melt or wind action. Satterlund and Haupt (1970) found that 80% of the snow held in the canopy of a forest in

Idaho subsequently reached the ground. Only 5% of the total snowfall was lost to interception and subsequent evaporation.

**Evaporation Losses** - Evaporation directly from the soil or vegetation depends on vapor pressure gradients between the air and the wetted soil or leaf surface, solar radiation, and wind. Vapor pressure gradient in the air is a function of both temperature and humidity. The temperature required for evaporation increases with increasing humidity. Under dense forest canopy, evaporation from wetted soils occurs slowly because of the high degree of shading, low temperatures, relatively high humidity, and low wind speeds typically found in these environments. More open forests allow for greater radiation and higher wind speeds that help remove water vapor from the air-soil interface, maintaining a higher vapor pressure gradient. For soil surfaces exposed to direct solar radiation, evaporation may dry soil more rapidly than transpiration because of high surface temperatures and low humidity (Satterlund and Adams 1992).

**Transpiration Losses** - Transpiration is the passage of water vapor from living plant tissues into the atmosphere through pores or "stomates." Transpiration rates vary based on a number of plant characteristics, including leaf surface area, stomatal characteristics, and depth of roots; they also are affected by whether the plants are annual or perennial, and deciduous or coniferous. Coniferous forests generally have the highest leaf surface area and thus the greatest potential for transpiration losses, followed in descending order by deciduous trees, shrubs, grasslands, and desert shrubs. Trees and shrubs with deeper roots can extract moisture from greater depths than grasses and forbes. Coniferous trees in xeric conditions east of the Cascades and Sierra Nevada may have large tap roots that penetrate deep into the soil, allowing moisture to be extracted even during dry periods.

Transpiration rates also depend on climatic conditions including temperature, humidity, and wind speed. In general, transpiration rates increase with increasing temperature; however, stomates will close in response to excessively high or low temperatures, increasing resistance to moisture loss. High humidity reduces the vapor pressure gradient between the plant leaf and the atmosphere, thereby reducing transpiration losses. Winds transport evaporated water vapor away from the leaf surface, thereby maintaining a higher vapor pressure gradient and increasing transpiration.

Soil conditions also influence how much water is available for transpiration. Loam soils tend to have higher water storage capacity than sandy soils. Similarly, deep soils hold more water than shallow soils. As soil moisture is depleted, the resistance to further uptake by plants increases, and water is supplied to plants at a slower rate (Satterlund and Adams 1992). Insufficient moisture causes closure of leaf stomates, which reduces transpiration losses. Consequently, when soils are moist, transpiration approaches maximum values; when soils are dry, transpiration is substantially less.

**Total Evapotranspiration** - Estimates of total evapotranspiration losses (interception+evaporation+transpiration) for a number of vegetation communities in the Pacific Northwest indicate that total losses are generally highest for coniferous forest types and slightly lower for chaparral and woodland communities (Table 3.6.2); however, losses from chaparral, woodland, and semi-arid communities represent a greater percentage of annual total precipitation. This is significant in eastside systems, in part because a substantial amount of precipitation occurs during spring and fall periods when temperatures are warm and evaporation and transpiration rates are high. In contrast, precipitation in the Coast Range and western Cascades generally falls during winter, when transpiration losses are relatively low due to low solar radiation, high humidity, and cool temperatures. These differences between hydrologic processes in eastside versus westside systems are important in determining the potential effects of land-use practices and are discussed in greater detail in Section 6.1.

### 3.6.3 Infiltration, Subsurface Flow, and Overland Flow

The amount of water that infiltrates into the soil depends upon the physical structure of the soil and antecedent moisture conditions. Sandy and gravelly soils derived from colluvium, alluvium, glacial tills, or soils that are rich in organic matter tend to be highly porous and allow rapid infiltration (Swanston 1991). Soils derived from finer-grained materials, including marine and lacustrine materials, or from weathered siltstones, sandstones, and volcanic rocks are less permeable and have lower infiltration capacities. During a given storm event, infiltration capacity decreases through time as soil pores are filled with water (Bedient and Huber 1948). If rainfall intensity (or snowmelt) exceeds infiltration capacities, overland flow occurs. Consequently, the likelihood of overland flow increases with storm intensity and duration.

In forested watersheds, most precipitation that reaches the forest floor, infiltrates into the soil (Satterlund and Adams 1992). Surface soils in old-growth forests areas typically have high organic content and porosity. Consequently, infiltration capacities are high and overland flow is uncommon except in areas where soil structure has been modified through human activity or natural disturbance. The majority of water that falls on a forested landscape thereby enters

streams via downslope subsurface flow. As a result, there is usually a lag between the time of peak rainfall and maximum streamflow (Swanston 1991)

In arid and semi-arid systems, as well as deforested lands, vegetation and organic litter are less abundant, and the routing of water once it reaches the soil differs. In areas where the soil surface is exposed, the impact of raindrops can detach and mobilize fine sediments (splash erosion), which settle into soil interstices, creating an impervious surface layer (Wisler and Brater 1959; Heady and Child 1994a). As a result of this "rain compaction", a significant proportion of rainfall or snowmelt runs off overland to the stream. Thus, in contrast to forested watersheds, precipitation events in arid and semi-arid systems cause rapid increases in streamflow. This may be particularly evident when soils are further compacted through land-use activities.

### 3.6.4 Stream Hydrology

*Regional Patterns* - Differences in precipitation patterns, evapotranspiration rates, and infiltration processes lead to marked regional differences in hydrologic regimes of streams. In addition, the size of the drainage basin significantly influences the characteristics of streamflow at a particular point downstream. As a general rule, small headwater streams are more hydrologically dynamic than larger streams because runoff occurs more rapidly over steeper areas and because high intensity events are more common in small areas. In the discussion below, we generalize about hydrologic patterns in lower order streams.

In the Coast Range and western Cascades, as well as the Puget Lowlands and Willamette Valley, frequent and heavy precipitation from November to March leads to a highly variable stream hydrograph, with multiple peaks that closely correspond to precipitation (Swanston 1991). In the early part of the rainy season, soil moisture is typically low and much of the water that falls as rain replenishes depleted soil moisture. In addition, evapotranspiration rates decrease during the winter as temperatures drop. Consequently, precipitation events of similar intensity will result in higher peak flows in the winter, when soils are more fully saturated and transpiration demands are low, than in the fall. Streamflows are lowest during the summer when precipitation is low, evapotranspiration demands are high, and soil moisture is depleted.

In the transient-snow zone of the mid-elevations of the Cascades and northern Sierra Nevada, soils become saturated as rainfall increases in the fall. During the winter, a combination of rain and snow events occur. During rainfall events, water tends to run off quickly to the stream channel because soil moisture is high and evapotranspiration is low. Consequently, increases in streamflow tend to coincide with rainfall. Precipitation that falls as snow is stored above ground for varying lengths of time, but generally melts within a few weeks of falling (Swanston 1991). Thus increases in streamflow from melting snow will occur days, or even weeks after the peak snowfall. Some of the more notable high flow events occur when substantial snowfall is followed by high intensity rains. These "rain-on-snow" events can release large volumes of water over short time periods.

In snow-dominated systems, including the high Cascades, Sierra Nevada, Blue Mountain and Northern Rocky mountain ranges, moisture from precipitation is stored in snowpack through much of the winter and released when temperatures warm in the late spring. Stream hydrographs are thus characterized by low winter flows followed by rapid increases during the spring snowmelt period. As snowpack diminishes, streamflow recedes and summer flows during the dry summer months are typically low, although minor peaks may result from intense convection storms. In the fall, rainstorms of moderate intensity can cause additional peaks in flow (Swanston 1991). Runoff from these events occurs most rapidly in high-elevation areas where soils are shallow and composed of fast-draining colluvial deposits and where transpiration demands are low because of sparse vegetation.

Arid and semi-arid regions east of the Cascades and Sierra Nevada tend to have high numbers of large ephemeral and intermittent stream channels. In part, this is because the timing of precipitation can coincide with periods of relatively high solar radiation in the spring, summer, and fall, unlike west of the Cascades where most precipitation falls during cold, cloudy periods. Much of the precipitation that falls in the warmer months is either rapidly evaporated from the ground or forest canopy or transpired by vegetation. In high intensity events, sudden increases in streamflow can occur where soils are relatively impervious and water is routed rapidly to the stream channel. Those streams that flow year round are generally fed by snowmelt from higher elevations or by ground-water discharge from aquifers recharged during periods of high precipitation.

*Floods* - Large, infrequent floods play an important role in shaping stream channels through the erosion transport and deposition of bed materials. Floods with recurrence intervals of 100 years or more can result in major channel changes, and several decades may be required to re-establish an equilibrium approaching pre-flood conditions. Some features produced by large floods may last longer than the recurrence interval of the event (Anderson and Culver 1977), implying that large floods may be responsible for specific aspects of valley floor formation rather than simply acting as disturbance events.

In December 1964, a rain-on-snow storm produced floods with a recurrence interval exceeding 100 years over much of northern California and Oregon. Studies conducted after this event provide information on the

geomorphic effects of large floods and the time needed to achieve a new dynamic equilibrium following such an event. The storm caused numerous debris slides and debris avalanches on slopes, and the resulting flood caused channel erosion and destruction of streamside vegetation (Lisle 1982; Sarna-Wojcicki et al. 1983). Hillslope and valley-bottom erosion put large amounts of sediment into the channel of the Van Duzen River, equal to seventeen times the mean annual sediment input into the channel system (Kelsey 1980). The result was a prolonged period of channel aggradation (five to fifteen years), followed by a period of degradation that was not complete in some reaches after twenty years. Erosion and the increased sediment load changed channel morphology, increasing channel width and decreasing channel depth, pool depth, and roughness (Lisle 1982).

The effectiveness of large floods to shape channel morphology may vary depending on stream size and position in the drainage network, as well as land cover (or recent changes in land cover such as logging). In steep mountain streams, only large, infrequent floods significantly modify valley floor landforms. In lower gradient alluvial reaches, smaller, more frequent events and ongoing processes modify the valley floor (Grant et al. 1994).

Floods also deposit sediments onto the surrounding floodplain, transport and rearrange large woody debris within the channel, clean and scour gravels in streams, recharge floodplain aquifers, and disperse propagules of riparian vegetation.

**Droughts** - Below-average precipitation and runoff can have significant effects on streams and watersheds. The recent drought in the Pacific Northwest has focused much attention on the health of forest ecosystems east of the Cascade Crest (Quigley 1992). Substantial die-off of forest vegetation has resulted from the synergistic effects of fire suppression and forest practices, which have led to changes in species composition of terrestrial vegetation (see Section 3.3). Drought conditions have weakened trees, making them more vulnerable to infestation by insects or disease. The influence of drought on watershed processes is not well documented; however, it is likely that droughts affect the input of nutrients, allochthonous materials, and large woody debris to stream channels. Within the stream channel, low flows can constrict the available habitat and allow water temperatures to warm, stressing fish or creating thermal barriers that block migration. A potential benefit of drought is that it provides the opportunity for establishment of riparian vegetation within the active stream channel, which in turn can stabilize channel features, dissipate hydraulic energy, and collect sediment when flows rise again (Blau 1995).

Tree-ring records from eastside forests indicate that a number of significant droughts lasting from 5 to 20 years have occurred during the past 300 years (Agee 1994). The timing of droughts may be asynchronous within subregions of the Pacific Northwest. For example, droughts in the Columbia Basin are not always coincident with droughts west of the Cascade Range or in northern California (Graumlich 1987).

### 3.7 Thermal Energy Transfer

Because most aquatic organisms are ectothermic, water temperature plays an important role in regulating biological and ecological processes in aquatic systems. Temperature directly and indirectly affects physiology, development, and behavior of salmonids, as well as mediates competitive interactions, predator-prey relationships, and the incidence of parasitism and disease (see section 4). Land-use practices can significantly change seasonal and diel temperature regimes in streams, primarily through the alteration of forest and riparian canopy but also through irrigation, impoundments, heated industrial effluents, and thermal power plants. In this section, we review the dominant energy transfer processes that are responsible for the heating and cooling of streams, rivers, and lakes to provide the basis for evaluating the effects of land-use practices on salmonid habitat. The role of riparian vegetation in controlling these processes is emphasized.

#### 3.7.1 Heat Exchange in Streams

Heat energy is transferred to and from streams and rivers by six processes: short-wave radiation (primarily incoming solar), long-wave radiation, convective mixing with the air, evaporation, conduction with the stream bed, and advective mixing with inflow from groundwater or tributary streams (Beschta et al. 1987; Sullivan et al. 1990). These processes occur in all streams, but the importance of each process on stream temperatures varies with location and season (Sullivan et al. 1990).

Direct solar radiation is generally the dominant source of energy input to streams and rivers. The amount of solar radiation that reaches and is absorbed by streams and rivers is influenced by season, latitude, topography, orientation of the watershed, local climate, and riparian vegetation. Season and latitude together determine the amount of daylight and the solar angle, both of which affect the amount of energy absorbed by streams (Brown 1980). In mountain or canyon regions, topography may provide substantial shade to streams, particularly at times of the year when the sun is low in the sky and in north-facing drainages. Local climate, and particularly cloud cover, significantly influences how much solar radiation reaches the stream channel. The amount and type of riparian vegetation play dominant roles in regulating incoming solar radiation in smaller streams (Brown 1980; Beschta et al. 1987; Caldwell et

al. 1991). The percentage of total solar radiation that reaches the stream surfaces in forested reaches may vary from less than 16% under dense coniferous canopies found in old-growth stands of the Coast Range and western Cascades (Summers 1983) to 28% in old-growth forests east of the Cascades (Anderson et al. 1992, 1993). In alpine, arid, and semi-arid ecosystems, the degree of shading may be less. Deciduous vegetation can provide significant shading during the spring and summer months, but it has minimal effect after leaf drop in the fall. The influence of riparian vegetation on radiation inputs diminishes in a downstream direction. As streams become larger and wider, riparian vegetation shades a progressively smaller proportion of the water surface (Beschta et al. 1987).

Long-wave radiation back into the atmosphere plays a relatively minor role in the overall energy budget of a stream. Long-wave radiation loss is determined primarily by the temperature differential between water and air, with greater exchange occurring when the difference between the air and water temperatures is greatest. Riparian vegetation influences long-wave radiation through its effect on microclimate within the riparian zone. Temperatures in the riparian zone tend to be cooler during the day and warmer at night than those above the forest canopy, which moderates long-wave radiative gains and losses.

Convective and evaporative heat transfer are controlled by temperature and vapor-pressure gradients, respectively, at the air-water interface (Beschta et al. 1987). Greater convective exchange occurs when the temperature differential between air and water is highest. Similarly, evaporative losses are highest at low humidity. Wind facilitates both convective and evaporative losses by displacing air near the air-water interface as it approaches thermal equilibrium with the water, and as it becomes more saturated through evaporation. Riparian vegetation modifies convective and evaporative heat exchange losses by creating a microclimate of relatively high humidity, moderate temperatures, and low wind speed compared with surrounding uplands. These conditions tend to reduce both convective and evaporative energy exchange by minimizing temperature and vapor-pressure gradients.

Conductive transfer of heat generally represents a minor component of a stream heat budget. The amount of heat transferred depends on the nature of the substrate, with bedrock substrates being more efficient in conducting heat than gravel beds (Beschta et al. 1987). Brown (1980) estimates that heat flow into bedrock stream beds may be as high as 15–20% of the incident heat. Heat that is transferred to the streambed during the daylight hours serves to heat streams during periods of darkness, thus dampening diel fluctuations. In shallow, clear streams, without shade from riparian vegetation, solar energy may penetrate through the water column and heat the substrate directly.

The role of advection depends on the volume of groundwater or tributary inputs relative to the total stream discharge; consequently, the importance of advection tends to diminish in a downstream direction. Nevertheless, even when groundwater inputs are small, they may provide thermal heterogeneity within streams that is biologically important (see section 5). In addition, certain regions east of the Cascade Range (e.g. the Deschutes Basin) are underlain with porous basaltic formations that absorb large amounts of water during periods of high runoff and release it later in the year. These groundwater inputs can significantly moderate streamflow and temperature regimes.

As subsurface flow moves laterally and downward towards stream beds, water temperatures equilibrate with those in the subsurface soil layers (Beschta et al. 1987), thus the temperature of water that enters streams from groundwater flow depends on ambient conditions in the soil environment. Surface soil temperatures follow seasonal air temperature patterns with a time lag that increases with increasing depth (Meisner 1990). Seasonal fluctuations are greatest at the surface and decrease with depth down to the "neutral zone," generally about 16–18m below the surface, where temperatures remain constant throughout the year (Meisner 1990). If the groundwater flow originates below the neutral zone, then groundwater temperatures will remain constant; if it originates above the neutral zone, then groundwater temperatures will exhibit seasonal variation (Meisner 1990). In snow-dominated systems, melting snow infiltrates into the soil at temperatures approaching 0°C (Beschta et al. 1987).

### 3.7.2 Stream Temperature Regulation

All of the above processes interact to produce the temperature regimes observed in streams and rivers; however, the relative importance of each process differs among locations. In small- to intermediate-sized streams of forested regions, incoming solar radiation represents the dominant form of energy input to streams, with convection, conduction, evaporation, and advection playing relatively minor roles (Brown 1980; Beschta et al. 1987; Sullivan et al. 1990). Groundwater inputs may be important in small streams where they constitute a large percentage of the overall discharge, particularly during periods of the year when flows are low. Downstream, where flow increases, the effects of riparian shading and advective mixing generally diminish, and the importance of evaporative heat loss increases.

Channel characteristics may also significantly affect heat exchange processes. The amount of heat that is gained or lost and the rate at which exchange takes place depend on the surface area of the stream or river. Wide, shallow streams exhibit greater radiative, convective, and evaporative exchange and, consequently, heat and cool more rapidly than deep, narrow streams. Similarly, the rate of energy exchange is affected by seasonal changes in stream discharge, which alter surface-to-volume ratios and determine the relative importance of groundwater inputs. In most streams in the Pacific Northwest, groundwater inputs are critical to cool streams during the warm summer months. Regional differences in stream temperatures result from differences in climatic factors (e.g., humidity, air temperature).

Streams in the Coast Range and western Cascades are moderated by the maritime climate and undergo smaller seasonal temperature fluctuations than those in the continental climates east of the Cascades. Elevation also influences stream temperatures, primarily because of elevational gradients in air temperatures that lead to greater convectional heating (Beschta et al. 1994). Finally, high turbidity in streams and rivers substantially increases the absorption of high-energy, shortwave radiation (Wetzel 1983), and thus can affect stream heating.

### 3.7.3 Lakes and Reservoirs

Lakes and reservoirs are heated primarily by incoming solar radiation, although some heat is transferred by convection, conduction (in shallow waters), and evaporation (Wetzel 1983). In clear water, over one-half of the incoming solar radiation is absorbed in the upper two meters of water, and more may be absorbed in waters with high turbidity. In temperate lakes, incoming solar radiation exceeds outgoing long-wave radiation during the summertime, and water at the surface is gradually warmed. Because warm water is less dense than cold water, it tends to remain near the surface and is resistant to mixing by the wind. As a result, thermal stratification can occur, with a warm and relatively well-mixed "epilimnion" overlaying a cooler "hypolimnion." Between these two layers is a transition zone, or "metalimnion," where temperatures rapidly decrease with increasing depth. During the fall as solar radiation decreases, temperatures in the surface layers cool and the mixing of epilimnetic and hypolimnetic waters occurs as they reach comparable temperature and density. During the winter, lake waters tend to remain mixed except where temperatures are sufficiently cold to freeze over. Because water reaches maximum density at 4°C, lakes that freeze are colder near the surface and warmer near the bottom (inverse stratification). When lakes become ice-free in the spring, density is relatively uniform and mixing of the water column will occur again provided there is sufficient wind at the surface.

The above pattern is characteristic of deeper lakes in the Pacific Northwest. In shallower lakes and ponds, lakes may turn over many times each year, whenever high wind conditions occur. Such systems are usually poor habitat for salmonids because they warm throughout the water column.

Thermal structure plays an important role in determining the distribution and production of aquatic organisms within a lake or reservoir. Stratification of lakes may restrict the habitats of fishes and other aquatic organisms. In addition, the circulation (turnover) of water during the spring and fall mixing periods brings nutrient-rich waters to the surface and stimulates production of phytoplankton and zooplankton. Release of either epilimnetic or hypolimnetic water from stratified reservoirs can markedly influence downstream temperature regimes in ways that may have adverse consequences for fish.

## 3.8 Nutrient Cycling/Solute Transport

Water is the major agent for the flux of dissolved and particulate matter across the landscape, integrating processes of chemical delivery in precipitation, geologic weathering, erosion, chemical exchange, physical adsorption and absorption, transport and retention in surface waters, and biotic uptake and release. Concentrations of nutrients or suspended material at any point within a landscape or catchment result from many abiotic and biotic processes.

Geology, climate, and biological processes across a landscape determine patterns of nutrient cycling. The primary determinant of the chemistry of most surface waters is the composition and age of the parent geology. The major rock types—igneous, sedimentary, and metamorphic—have characteristic compositions of major cations and anions, as well as minor chemical constituents that serve as nutrients for biota (e.g., nitrogen, phosphorus). The high temperatures and pressure under which igneous and metamorphic rocks are formed alters the chemical composition by volatilizing elements and compounds that are released as gases to the atmosphere (e.g., nitrogen, inorganic carbon). Sedimentary rocks contain minerals that have been weathered from other sources previously, and they may contain greater amounts of biologically derived material because of the less harsh conditions of their formation. Geochemistry of the parent material governs rates of dissolution or weathering and, thus, influences concentrations of dissolved chemicals in surface waters.

Climate strongly influences general surface water chemistry and nutrient concentrations through two major processes—direct input of chemicals through precipitation and influence on hydrology (Gibbs 1970). The atmosphere is a major source of elements and compounds. Weather patterns affect the available source areas for water and chemicals in the atmosphere and subsequent precipitation that falls on land. Both natural and anthropogenic sources may create distinctive chemical signatures in precipitation. Climate also determines the general hydrologic regime and establishes physical conditions that influence evaporative losses of water. Arid areas typically exhibit high concentrations of dissolved ions because of the high rates of evaporation and subsequent concentration of chemicals in solution. The hydrologic regime is a function of climate and geographic features of the landscape, and it is a major determinant of weathering rates, dilution, and timing of nutrient transport. Patterns in runoff may be mirrored by differences in surface water chemistry. The flashy flow-regimes of rain-dominated and rain on snow dominated systems create a similar episodic pattern in nutrient transport, while the more steady flow regimes of snow-dominated systems produce more predictable nutrient transport patterns.

The biota of terrestrial, riparian and aquatic ecosystems strongly influence the cycling of major nutrients and associated chemical parameters (Likens et al. 1977; Meyer et al. 1988); through such processes as photosynthesis, respiration, food consumption, migration, litter fall, and physical retention. Surface waters are exposed to various sources of inputs, sites of biological uptake, and surfaces for physical exchange (Gregory et al. 1991). Stream substrates serve as sites for colonization and attachment by aquatic organisms ranging from microbes to vertebrates. Many aquatic organisms have distinct substrate relationships; therefore, the composition of the stream bed can directly influence nutrient cycling. Organic substrates, such as leaves and wood, create important sources for microbial colonization and subsequent nutrient cycling (Aumen et al. 1985a, b; Meyer 1989). These organic substrates also serve as sources of dissolved organic carbon for microbial activity or transport into the water column (Dahm 1981). Woody debris in particular plays a critical role as a food resource, substrate, site of physical exchange, site for biological uptake, and roughness element that reduces water velocity and increases retention (Harmon et al. 1986). Land use practices typically alter the organic substrates of stream channels, and thereby influence water quality.

Vegetated floodplains along streams and rivers, as well as mudflats and vegetation beds in estuaries, create a mosaic of geomorphic surfaces and riparian plant communities (Fonda 1974; Gregory et al. 1991; Bayley and Li 1992). Floodplains influence the delivery and transport of material by 1) delivering stored material during high flows, 2) retaining material in transport from the main channel, 3) providing a matrix of sediment for subsurface flow, and 4) reducing velocities of water and increasing the potential for retention. Elimination of floodplains greatly reduces the assimilative and storage capacity of a stream system and is one of the major forms of anthropogenic alteration of nutrient cycling in lotic ecosystems (Smith et al. 1987; Junk et al. 1989; Sparks et al. 1990). Side channels on floodplains and in estuaries are habitats with extensive contact with the water column and lower velocities than the main channel, thus these lateral habitats typically exhibit high rates of nutrient uptake and biological productivity (Cooper 1990).

Streamside forests, estuarine vegetation beds (tidal marshes), and other plant communities create a filter through which nutrients in solution must pass before entering surface waters (Pionke et al. 1988; Gregory et al. 1991). Retention of nutrients in groundwater is a critical component of nutrient cycling within a basin (Simmons et al. 1992). Commonly, these vegetative corridors remove 60–90% of the nitrogen and phosphorus in transport (Lowrance et al. 1983; Peterjohn and Correll 1984; Lowrance 1992). Modification of riparian forest structure can substantially change long-term patterns of nutrient cycling within a catchment (Pinay et al. 1992).

One of the most overlooked components of a stream and its valley is the hyporheic zone, the area of subsurface flow beneath the surface of the stream bed (Stanford and Ward 1988; Bencala 1993). Recent research indicates the hyporheic zone plays important roles in nutrient cycling, temperature modification, dissolved oxygen microbial processes, meiofaunal communities and refugia for a wide range of organisms (Pinay and Decamps 1988; Stanford and Ward 1988; Triska et al. 1990; Valett et al. 1990; Hendricks and White 1991). In many streams, as much as 30–60% of the flow occurs in the hyporheic zone and may exceed these levels in porous bed materials or during low flow conditions. The majority of nutrient uptake in streams may occur in the hyporheic zone in desert, forest, or grassland ecoregions (Duff and Triska 1990).

### 3.8.1 Major Chemical Species and Dissolved Nutrients

Surface waters contain a complex array of major chemical species, biologically important nutrients, and numerous trace elements and compounds. The major dissolved constituents include cations and anions that are required by living organisms but are so abundant that they rarely limit biological production. In addition, surface waters contain essential nutrients that may be present in such low concentrations that they limit rates of production of plants, microbes, or consumers. The major nutrients or macronutrients are nitrogen, phosphorus, and carbon. Micronutrients are generally required in such low amounts that their availability is rarely limiting, but studies over the last several decades have demonstrated that the productivity of some systems may be limited by micronutrients and many processes are commonly limited by the availability of these chemicals. This review only covers the macronutrients.

The major cations in surface waters include the divalent cations of calcium and magnesium and the monovalent cations of sodium and potassium. In general, the order of dominance in surface waters of the world is  $\text{Ca}^{++} > \text{Mg}^{++} > \text{Na}^+ > \text{K}^+$ , but local geology can alter their relative abundance (Gibbs 1970). These elements play critical roles in all biological systems as well as influence the reactivity and abundance of other elements. The exchange of these cations, either physically or through biological absorption, can alter the availability of hydrogen ions and thereby alter pH, which strongly influences biota and fundamental ecological processes.

The major anions in surface waters consist of the divalent anions of carbonate and sulfate and the monovalent anions of bicarbonate and chloride (Gibbs 1970). The order of dominance of anions in temperate waters is:  $\text{HCO}_3^- > \text{CO}_3^{--} > \text{SO}_4^{--} > \text{Cl}^-$ . Inorganic carbon and sulfate are biologically important in all ecosystems, and the inorganic carbon species largely determine the buffering capacity and thus the pH conditions of the world's surface waters.

**Nitrogen** - Nitrogen exists in solution as both inorganic forms—nitrogen gas ( $N_2$ ), nitrate ( $NO_3^-$ ), nitrite ( $NO_2^-$ ), ammonia ( $NH_3$ ), or ammonium ( $NH_4^+$ )—and organic forms (organic N). In many areas of the Pacific Northwest, surface waters commonly have extremely low concentrations of dissolved nitrogen because of the underlying volcanic parent geology, which was created under intense temperature and pressure (Thut and Haydu 1971; Sollins and McCorison 1981; Norris et al. 1991).

Biological processes largely mediate the different forms of nitrogen (Gosz 1981). Nitrogen fixation converts  $N_2$  into  $NH_3$  under anaerobic conditions or in specialized cells, and organisms subsequently use the ammonia to form amino acids and proteins. Organic nitrogen is metabolized to ammonium as a waste product or microbial decomposition converts organic N to ammonium through the process of ammonification. Certain microorganisms are capable of oxidizing ammonia to nitrite or nitrate. Plants and heterotrophic microorganisms can then reduce nitrate to form ammonia and subsequently proteins and amino acids. Under anoxic conditions, certain microorganisms can reduce  $NO_3^-$  to  $N_2$ . These transformations create intricately linked cycles of nitrogen, and under nitrogen limited conditions, these links are tightly coupled. As a result, certain forms—such as ammonia or nitrate—are rarely present in high concentrations because they are so rapidly incorporated into other nitrogenous molecules or modified.

Riparian areas play major roles in nitrogen cycling by providing year-round anaerobic conditions (Green and Kauffman 1989; Mulholland 1992). Rates of denitrification (and nitrogen fixation) are enhanced in the anaerobic conditions and the high moisture and organic substrates that denitrifying bacteria require (Myrold and Tiedje 1985; Ambus and Lowrance 1991; Groffman et al. 1991). Rates of denitrification in riparian soils in the Cascade Mountains of Oregon are four to six times higher than in upslope forests, and alder-dominated reaches exhibit the highest observed rates (Gregory et al. 1991). Alder is a common streamside plant and is also a nitrogen fixer; thus alder-dominated riparian areas are potential sources of nitrogen in stream water (Tarrant and Trappe 1971). As noted above, elevated rates of denitrification may negate the contribution of alders, but it is possible for extremely high concentrations of nitrate ( $> 5 \text{ mg } NO_3^-/l$ ) to occur where litter inputs are high and water velocities are low (Taylor and Adamus 1986). These conditions have been observed primarily in the Coast Range where alder may extend from stream's edge to the ridgeline.

Catchments generally process nitrogen efficiently because it is such an important biological requirement. A small western basin retained approximately 99% of the nitrate that entered in precipitation (Rhodes et al. 1985). Loss of nitrogen from terrestrial ecosystems is mediated by uptake in the aquatic ecosystems, particularly in nitrogen-limited ecosystems, such as the basalt-dominated Pacific Northwest (Triska et al. 1982, 1984). Studies of nitrogen uptake in streams of the Cascade Mountains indicate that approximately 90% of the nitrate or ammonium introduced into stream water is assimilated within 500–2000 m, depending on the size of the stream (Lamberti and Gregory 1989).

**Phosphorus** - Phosphorus in surface waters is largely derived from mineral sources. Inorganic phosphorus includes many compounds incorporating the phosphate ( $PO_4^{3-}$ ) ion. Concentrations of inorganic phosphorus are low in many geologic areas and as a result, phosphorus commonly is a limiting nutrient for primary production and microbial processes (Wetzel 1983). In much of the Pacific Northwest, however, the basaltic parent geologic material contains abundant and relatively easily weathered forms of inorganic phosphorus; thus, concentrations of phosphorus in streams and rivers commonly exceed  $10 \mu\text{g } PO_4\text{-}P/l$  (Fredriksen et al. 1975; Salminen and Beschta 1991; Bakke 1993).

### 3.8.2 Nutrient Spiraling and Retention

Nutrient cycling is often viewed as a closed system in which chemicals pass through various states and reservoirs within the ecosystem of interest. Stream ecosystems present an interesting contrast to this perspective because of their unidirectional flow from headwaters to large rivers to the ocean. The Nutrient Spiraling Concept was developed to more accurately represent the spatially dependent cycling of nutrients and the processing of organic matter in lotic ecosystems (Newbold et al. 1982; Elwood et al. 1983).

The longitudinal nature of streams and rivers strongly influences patterns of nutrient uptake. In the Nutrient Spiraling Concept, one complete cycle of a nutrient depends upon the average distance a nutrient atom moves in the water compartment (i.e., the uptake length), the average distance a nutrient atom moves in the particulate compartment, and the average distance a nutrient atom moves in the consumer compartment. The Nutrient Spiraling Concept provides a useful conceptual framework for investigating the dynamics of dissolved and particulate material in streams and rivers (Mulholland 1992). Alteration of riparian areas, stream channels, and biotic assemblages can be viewed in terms of changes in flux and uptake, the two major components of spiraling length. Efficiency of nutrient use can be quantified in terms that are relevant to the cycling of nutrients along a river valley or drainage network.

Downstream transport of dissolved or particulate material is a complex function of physical trapping, chemical exchange, and biological uptake (Minshall et al. 1983; Speaker et al. 1984). Retention of material in streams is not necessarily uniform along a reach of stream. Physical discontinuities, such as debris dams, boulders, pools, and sloughs, alter retention patterns. The ionic strength or salinity of surface water tends to increase from headwaters to large rivers, reflecting the accumulation of weathering products and material produced by terrestrial and aquatic

ecosystems (Dahm et al. 1981). Spiraling length increases and retention efficiency decreases as streams become larger because of the decreased friction, increased average velocity, and lower probability of being trapped by bed material. This pattern is moderated in braided channels and at high flows as streams flow out of their banks and are slowed by the roughness of adjacent forests and floodplains (Welcomme 1988; Junk et al. 1989; Sparks et al. 1990).

Different environments may alter retention patterns for dissolved and particulate matter. Areas of intense biological activity increase biotic uptake and alter patterns of retention. Simplification of stream ecosystems will tend to make longitudinal patterns of retention more uniform and less efficient, thus lowering biological productivity.

Retention of material represents a fundamental ecological feature that integrates the supply and use of nutrients and food resources. In the Pacific Northwest, major inputs of organic matter brought into freshwater ecosystems from the Pacific Ocean were historically represented by salmon carcasses (Bilby and Bisson 1992), and lampreys. It has long been recognized that the abundance of salmon carcasses is correlated with the productivity of sockeye lakes in Alaska for the subsequent year class (Donaldson 1967). Recent studies have demonstrated that as much as 30% of the nitrogen for higher trophic levels in streams in the Pacific Northwest may be derived from marine ecosystems (Bilby et al. In press). In addition, the retention of carcasses in streams has been linked to channel complexity and abundance of woody debris (Cederholm and Peterson 1985). Declines in anadromous fishes in the Pacific Northwest (Nehlsen et al. 1991) may signal more fundamental changes in stream ecosystem productivity than the simple loss of stocks or species.

Disturbances can accelerate or slow the loss of nutrients and the efficiency with which terrestrial and aquatic ecosystems use them (Vitousek and Melillo 1979; Beschta 1990). Generally, such disturbances disrupt nutrient cycling over the short-term (i.e., less than a decade) (Resh et al. 1988), but as ecosystems recover, they more efficiently cycle available nutrients. Many disturbances also increase habitat complexity (Swanson et al. 1982), thereby increasing the efficiency of retention after an initial recovery period (Bilby 1981; Aumen et al. 1990). The frequent disturbances associated with stream ecosystems make them one of the most dynamic ecosystems with respect to nutrient cycling and biotic community organization (Minshall et al. 1985; Minshall 1988). Changes in community organization and process rates in response to changes in long-term nutrient availability may not be fully exhibited for years (Stottlemyer 1987; Power et al. 1988; Peterson; Deegan et al. 1994).

### 3.9 Roles of Riparian Vegetation

Riparian zones constitute the interface between terrestrial and aquatic ecosystems (Swanson et al. 1982; Gregory et al. 1991), performing a number of vital functions that effect the quality of salmonid habitats, as well as providing habitat for a variety of terrestrial plants and animals. While processes occurring throughout a watershed can influence aquatic habitats, the most direct linkage between terrestrial and aquatic ecosystems occurs in the riparian area adjacent to the stream channel. Consequently, the health of aquatic systems is inextricably tied to the integrity of the riparian zone (Gregory et al. 1991; Naiman et al. 1992).

Riparian vegetation provides numerous functions including shading, stabilizing streambanks, controlling sediments, contributing large woody debris and organic litter, and regulating the flux and composition of nutrients (FEMAT 1993; O'Laughlin and Belt 1994; Cederholm 1994). Recognition of the importance of riparian-aquatic interactions has made the establishment of riparian buffers a central element of forest practices rules and watershed restoration efforts. Several recent publications have advocated a functional approach to riparian management, attempting to identify "zones of influence" for critical riparian processes (FEMAT 1993; O'Laughlin and Belt 1994). These approaches recognize that the influence of riparian vegetation on stream ecosystems generally diminishes with increasing distance from the stream channel. In this section, we review the principal functions of riparian vegetation and summarize the available literature regarding zones of riparian influence. Some functions are discussed in greater detail in previous sections.

#### 3.9.1 Shade

In small headwater streams, riparian vegetation moderates the amount of solar radiation that reaches the stream channel, thereby dampening seasonal and diel fluctuations in stream temperature (Beschta et al. 1987) and controlling primary productivity. The effectiveness of riparian vegetation in providing shade to the stream channel depends on local topography, channel orientation and width, forest composition, and stand age and density (Beschta et al. 1987; FEMAT 1993). Naiman et al. (1992) reported that the amount of solar radiation reaching the stream channel is approximately 1–3% of the total incoming radiation for forested small streams and 10–25% in forested mid-order streams (Naiman et al. 1992). In winter, streamside vegetation provides insulation from radiative and convective heat losses (see section 3.6.1), which helps reduce the frequency of anchor ice formation (Murphy and Meehan 1991). Thus riparian vegetation tends to moderate stream temperatures year round. The numerous biological and ecological consequences of elevated stream temperatures on salmonids include effects on physiology, growth and development, life history patterns, competitive and predator-prey interactions, and disease (see Section 4.3).

The FEMAT (1993) report presents a generalized curve relating cumulative effectiveness of the riparian canopy in providing shade relative to distance from the stream channel for westside forests (Figure 3.9.1a). They propose that, as a general rule, close to 100% shading can be maintained by buffer zone widths equal to one site-potential tree height (i.e., the potential height of a mature tree at the particular location). In the Oregon Coast Range and western Cascades, buffer zones of 100 feet or more can provide as much shade as intact old-growth forests (Brazier and Brown 1973; Steinblums et al. 1984). Similar assessments for eastside forests, as well as arid and semi-arid shrublands have not been published, and may be substantially different.

### 3.9.2 Bank Stabilization

Riparian vegetation increases streambank stability and resistance to erosion via two mechanisms. First, roots from woody and herbaceous vegetation bind soil particles together, helping to maintain bank integrity during erosive high streamflow events (Swanson et al. 1982). Diverse assemblages of woody and herbaceous plants may be more effective in maintaining bank stability than assemblages dominated by a single species; woody roots provide strength and a coarse root network, while fine roots fill in to bind smaller particles (Elmore 1992). The root matrix promotes the formation of undercut banks, an important habitat characteristic for many salmonids (Murphy and Meehan 1991). Second, stems and branches moderate current velocity by increasing hydraulic roughness. East of the Cascades, grasses, sedges, and rushes tend to lie down during high flows, dissipating energy and protecting banks from erosion (Elmore 1992).

Riparian vegetation may also facilitate bank-building during high flow events by slowing stream velocities, which in turn helps to filter sediments and debris from suspension. This combing action helps to stabilize and rebuild streambanks, allowing the existing channel to narrow and deepen, and increasing the effectiveness of riparian vegetation in providing bank stability and shade (Elmore 1992). During overbank flows, water is slowed and fine silts are deposited in the floodplain, increasing future productivity of the riparian zone.

Vegetation immediately adjacent to the stream channel is most important in maintaining bank stability. The FEMAT (1993) report suggests that the role of roots in maintaining streambank stability is negligible at distances of greater than 0.5 tree heights from the stream channel (Figure 3.9.1a). In wide valleys where stream channels are braided, meandering, or highly mobile, the zone of influence of root structure may be substantially greater.

### 3.9.3 Sediment Control

The regulation of sediment flow is a major function of the riparian zone. Riparian vegetation and downed wood in the riparian zone can reduce the amount of sediment delivered from upland areas to the stream channel in several ways. By providing physical barriers, standing or downed vegetation can trap sediments moving overland during rainfall events. Riparian zones, however, are less effective in regulating channelized erosion. Most erosion occurs in channelized flows that may travel thousands of feet (O'Laughlin 1991; O'Laughlin and Belt 1994). Thus riparian vegetation may have little influence on mass soil movements derived from outside of the riparian zone. Riparian vegetation may also influence sediment inputs by reducing the likelihood of mass failures along the stream channel, through the stabilizing action of roots, and by buffering the stream from mass wasting that initiates in upland areas, although riparian vegetation may have little effect during large, deep-seated landslides (Swanson et al. 1982).

The zone of riparian influence for sediment regulation is difficult to define because of the different ways sediment may enter the stream channel. The FEMAT (1993) review of the literature suggests that riparian zones greater than 200 feet (i.e., about one site-potential tree height) from the edge of the floodplain are probably adequate to remove most sediment from overland flow. However, O'Laughlin and Belt (1994) suggest sediment control cannot be achieved through riparian zones alone, since channel erosion and mass wasting are significant sources of sedimentation in forested streams. For these events the zone of influence may extend several hundred meters from the floodplain (FEMAT 1993), depending on the soil type, slope steepness, and other factors that influence the susceptibility of hillslopes to mass wasting or channelized erosion.

### 3.9.4 Organic Litter

Riparian vegetation contributes significant quantities of organic litter to low- and mid-order streams, which constitutes an important food resource for aquatic communities (Naiman et al. 1992). The quality, quantity, and timing of litter delivered to the stream channel is dependent on the vegetation type (i.e., coniferous versus deciduous), stream orientation, side slope angle, stream width, and the amount of stream meander (Cummins et al. 1994). In conifer-dominated riparian zones, 40–50% of the organic litter consists of low quality cones and wood, which may take several years to decades to be processed. In contrast, high quality material from deciduous forests may decay within a year. Although conifers have the greater standing biomass, shrub- and herb-dominated riparian assemblages provide significant input in many streams (Gregory et al. 1991). Over 80% of the deciduous inputs, primarily leaves, are

delivered during a 6–8 week period in the fall (Naiman 1992), while coniferous inputs are delivered throughout the year (Cummins et al. 1994).

The extent of the riparian zone of influence for organic litter inputs is dependent on geomorphology and stream size. Upland forests beyond the riparian zone can contribute litter to small streams in steep basins through direct leaf-fall and overland transport of material by water. In 3rd to 5th order streams, streams are more influenced by vegetation in the immediate riparian zone. Large lowland streams tend to have complex floodplain channels with minimal upland interactions. However, the lateral movement of unconstrained alluvial channels effectively increases the potential riparian zone of influence. In westslope forested systems, most organic material that reaches the stream originates within 0.5 tree heights from the stream channel (Figure 3.9.1a) (FEMAT 1993). Vegetation type may also influence the riparian zone of influence since deciduous leaves may be carried greater distances by the wind than coniferous litter.

### 3.9.5 Large Woody Debris

Large woody debris (LWD) in stream channels provides critical structure to stream channels, although full recognition of the importance of large wood in stream ecosystems has only come in the last 20 years (Swanson et al. 1976; Swanson and Lienkaemper 1978). For more than 100 years, large wood was removed from stream channels to facilitate boat traffic and the floating of logs downstream. In addition, during the 1940's and 1950's, biologists viewed large wood as an impediment to fish migration and routinely cleared woody debris from stream channels (Sedell and Luchessa 1981). Consequently, the many roles of large wood in streams, from small headwaters to large river systems, have been greatly diminished over time.

Large wood enters the stream channel through two different pathways: the steady toppling of trees as they die or are undercut by streamflow, and catastrophic inputs associated with windstorms, mass failures, and debris torrents (Bisson et al. 1987; Cummins et al. 1994). Once in the stream channel, large woody debris influences coarse sediment storage; increases habitat diversity and complexity, gravel retention for spawning habitat, and flow heterogeneity; provides long term nutrient storage and substrate for aquatic invertebrates; moderates flow disturbances; increases retention of allochthonous inputs, water, and nutrients; and provides refugia for aquatic organisms during high and low flow events (Bisson et al. 1987). The ability of large wood to perform these functions depends in part on the size and type of wood. In general, the larger the size of the debris, the greater its stability in the stream channel, since higher flows are needed to displace larger pieces. In addition, coniferous logs are more resistant to decay than deciduous logs and hence exhibit greater longevity in the stream channel (Cummins et al. 1994).

Although LWD performs essential functions in all streams, the relative importance of each of the processes listed above varies with stream size. In small, steep headwater streams (1st and 2nd order), large volumes of stable LWD tend to dominate hydraulic processes. Generally, woody debris is large enough to span the entire channel, resulting in a stepped longitudinal profile that facilitates the formation of plunge pools downstream of obstructions. This stepped profile increases the frequency and volume of pools, decreases the effective streambed gradient, and increases the retention of organic material and nutrients within the system, thus facilitating biological processing (Bisson et al. 1987). Woody debris within the channel increases velocity heterogeneity and habitat complexity by physically obstructing the stream flow, creating small pools and short riffles (Swanston 1991). Diverted currents create pools (plunge, lateral, backwater) and riffles, flush sediments, and scour streambanks to create undercut banks (Cummins et al. 1994). In sediment-poor systems, LWD retains gravels that are essential for spawning salmonids. Larger debris dams store fine sediment and organic materials, reducing their rate of transport downstream. In addition debris dams protect the downstream reaches from rapid changes in sediment loading, which may degrade spawning gravels, fill pools, and reduce invertebrate populations.

In mid-order streams, large woody debris functions primarily to increase channel complexity and flow heterogeneity by: 1) anchoring the position of pools along the thalweg, 2) creating backwaters along the stream margin, 3) causing lateral migration of the channel, and 4) increasing depth variability (Maser et al. 1988). Large wood deposits tend to occur along margins, or in mid-channel where physical obstructions collect wood during storms Bisson et al., (1987) and Bilby and Ward (1989) examined streams in western Washington and found a number of differences in the roles of large wood in relation to stream size. Average diameter, length, and volume of pieces of wood were generally greater in mid-order streams than in low-order streams. Large wood was important in pool formation in mid-sized streams; however, these were more likely to be debris scour pools than plunge pools. In addition, the ability of wood to accumulate sediment diminished as streams became larger. Distributions of organisms associated with woody debris, including various salmonids, changed relative to the changes in woody debris distribution along the stream channel. Other important functions of large wood in mid-order streams include the retention of salmon carcasses and organic detritus, which provide nutrients to the flora and fauna within the stream and in the adjacent riparian area (Bilby et al. In press).

The role of large woody debris in high order streams is generally less well documented; however, historical records indicate that large debris jams once played a major role in floodplain and channel development on major rivers,

such as the Willamette River in Oregon and other systems in the Puget Lowlands of Washington (Sedell and Luchessa 1981). In these high-order streams, large woody debris increased channel complexity by creating side channels, backwaters, and ponds, as well as refugia for aquatic organisms during winter storm events. During high flows, sediments were deposited on the floodplains and in riparian zones, increasing the productivity of these soils. Extensive snag removal and channelization over the last 100 years have diminished these roles of wood in larger river systems. Today, solitary pieces of woody debris are generally not large enough to span the active channel or substantially modify flows (Mase et al. 1988), although woody debris along the outside bends of river banks provides habitat in an otherwise simplified habitat zone (Swanston 1991). Wood snags that remain in the main channel are utilized by insects and fish, particularly in larger river systems with unstable sand substrate (Marzolf 1978; Benke et al. 1984).

Defining the zone of influence for input of large woody debris is difficult because methods of delivery differ. The greatest contribution of large wood to streams comes from trees within one tree height of the stream channel (Figure 3.9.5a) (FEMAT 1993). However, the likelihood that a falling tree will enter the stream channel depends on tree height, distance from the stream channel, and the nature of the terrain. On level terrain, the direction that a tree will fall is essentially random. On steep terrain, however, there is generally a higher probability that the tree will fall downslope into the stream channel. For episodic inputs of large woody debris via mass wasting and debris torrents, defining the zone of influence becomes more difficult. The likelihood of wood entering the stream will vary with conditions that control the frequency of mass wasting, including slope, soil type, and hydrology. Assessing appropriate zones of influence for these events is probably beyond our current level of scientific understanding. Cummins et al. (1994 Draft) and Reeves et al. (In Press) report that 0-order channels generate most landslides containing trees and coarse sediments.

### 3.9.6 Nutrients

Riparian zones control the flow of nutrients to the stream and are therefore important regulators of stream production. Subsurface flow from upland areas carries nutrients and dissolved organic matter to the riparian zone, where these material are taken up by vegetation for plant growth or chemically altered (Naiman et al. 1992). Lowrance et al. (1984) found that even narrow riparian zones along streams in agricultural lands significantly affected stream chemistry. Riparian forests modify the chemical composition and availability of carbon and phosphorus, and promote soil denitrification through changes in the position of oxic-anoxic zones. (Pinay et al. 1990 in (Naiman et al. 1992)). During overbank flows, nutrients from floodwaters may be absorbed by riparian vegetation, reducing the total nutrient load in the stream (Cummins et al. 1994). Dissolved organic matter inputs can occur from numerous sources besides groundwater. These include leachate from entrained litter and large woody debris in the channel, algal, invertebrate, and fish excretions; and floodplain capture at the time of inundation (Gregory et al. 1991).

We found no published attempts to define zones of influence for nutrient cycling. Most likely, this reflects the difficulty in tracing the movement of nutrients, particularly with those elements such as nitrogen for which the number of alternative pathways is great. As discussed in Section 3.5, conditions throughout the watershed influence stream chemistry, thus the zone of influence extends to the top of the watershed, even though it may be years before nutrients ultimately find their way to the stream. However, the zone of most intense interaction is within the floodplain and hyporheic zones, where subtle changes in oxygen levels can dramatically affect nutrient composition and bioavailability.

### 3.9.7 Microclimate

Although not well documented (O'Laughlin and Belt 1994), streamside vegetation can have a significant influence on local microclimates near the stream channel (FEMAT 1993). Chen (1991) reported that soil and air temperatures, relative wind speed, humidity, soil moisture, and solar radiation all changed with increasing distance from clear-cut edges in upslope forests of the western Cascades. The FEMAT(1993) concluded that loss of upland forests may therefore influence conditions within the riparian zone. Thus they concluded that buffers around riparian zones may need to be wider in order to maintain riparian microclimates, compared with buffers needed to maintain other riparian functions (Figure 3.9.7a).

### 3.9.8 Wildlife Habitat

Although riparian areas generally constitute only a small percentage of the total land area, they are extremely important habitats for wildlife. The attractiveness of riparian zones to wildlife likely reflects two attributes: the presence of water, which is essential to all life and generally scarce in the West, and the fact that riparian areas tend to support more diverse plant assemblages than surrounding upland areas. The latter characteristic derives from the dynamic nature of riparian zones, which typically leads to a mosaic of plant assemblages in different stages of ecological succession (Kauffman 1988). Brown (1985) reported that 87% of wildlife species in western Oregon and Washington utilize wetlands or riparian areas during some or all of their life cycle (FEMAT 1993). Thomas et al. (1979) found that

82% of all terrestrial vertebrates in the Great Basin of southeastern Oregon are either directly dependent on riparian zones or utilize riparian habitats more than any other habitat. Dependence of a majority of species on riparian zones has been demonstrated for all major vertebrate classes. Bury (1988) reported that 8 of 11 species of amphibians and 5 of 6 species of reptiles in Oregon either reside or breed in aquatic or riparian habitats. In northern California, approximately 50% of both reptiles and amphibians prefer riparian or aquatic habitats (Raedeke et al. 1988). Raedeke (1988) reviewed the published literature and found that 67% of native large mammals in the Pacific Northwest are either dependent on riparian areas, or are more abundant in riparian areas than surrounding uplands. Similar preferences for riparian habitat by small mammals, and especially bats, have also been documented (Cross 1988). Beschta et al. (1994) reported 55 species of birds in Oregon (approximately 46% of the total for which data were available) that are dependent or exhibit preferences for riparian habitats. For eastside ecosystems, the dependence of birds and other species on riparian zones is likely higher than for westside systems, where water is more abundant.

### 3.10 Implications for Salmonids

The above discussion highlights the highly complex array of physical and chemical processes that occur across the landscape, in the riparian zone adjacent to streams and rivers, and within the stream channel. Large-scale geomorphic and climatic processes have together shaped the landscape of the Pacific Northwest, exerting dominant control over channel gradient and configuration. Although these processes operate at scales of thousands to millions of years (Table 3.10), they are nevertheless important considerations in the development of salmonid conservation strategies. The current distribution of salmonids and other fish species in the Pacific Northwest is a direct consequence of tectonic activity dating back tens of millions of years, which has both isolated and reconnected drainage basins through vertical lift and shifted subplates (McPhail and Lindsey 1986; Minckley et al. 1986). Similarly, climatic shifts and glaciation have alternately eliminated and stimulated reinvasion of fishes over significant portions of the Pacific Northwest landscape, as well as redistributed species into lower elevations or more southerly areas. Furthermore, the isolation of individual populations by geomorphic and glacial processes over time has allowed the evolution of unique stocks and species. Evolutionarily significant units (Waples 1991b) reflect the historical legacy wrought by geologic and climatic conditions over the millennia, as well as adaptation to local environmental conditions that have prevailed since the last glaciation. Finally, long-term geomorphic and climatic process together with hydrologic processes and vegetation cover, determine the rate at which nutrients, sediments, organic material, and water are transported from upslope areas into the stream channel. Consequently, the geomorphic and climatic setting determines the normal background rates of these processes, regulates the frequency and magnitude of natural episodic disturbances that reset and replenish streams, and govern the responses of specific watersheds to human perturbations.

Nested within this geomorphic and climatic context are a number of physical and chemical processes that further modify the landscape and that directly influence stream channel characteristics and water chemistry. These processes, including surface erosion, landslides, floods, debris torrents, ice flows, droughts, beaver activity, and wildfire, operate at ecological times scales—generally from days to decades or centuries—regulating the input of sediment, nutrients, and organic material to the stream (Table 3.10). The riparian zone acts as the interface between terrestrial and aquatic ecosystems, moderating the effects of upslope processes, as well as providing other critical functions (e.g., shading, bank stabilization, nutrient transformation, allochthonous inputs). Together, these processes determine the macrohabitat characteristics, including general channel morphology and pool-riffle sequences, and microhabitat characteristics, such as depth, velocity, cover, temperature, and substrate.

The processes that influence salmonid habitats may be either cyclical in their occurrence (e.g., seasonal temperature, streamflow, and leaf-fall patterns), or episodic in nature (e.g., wildfires, landslides, floods, debris torrents). It is critical to recognize that these cycles or disturbances are fundamental and vital parts of ecosystem function, even though they may be temporarily disruptive of aquatic ecosystems. Studies of geomorphology and paleoecology indicate that disturbance is continual, sometimes across large areas, and often unpredictable. In eastside ecosystems the changes are most often associated with climatic changes that render vegetation more susceptible to disturbances such as fire and disease (Johnson et al. 1994). Eastside forests have evolved with periodic disturbances and when they do not receive them they become increasingly unstable (Henjum et al. 1994; Johnson et al. 1994). If drought or fire do not alter those forests, then disease or insects will. Naturally occurring mass soil movements and erosion introduce large woody debris, rock, gravel, and fine materials into stream channels, substantially modifying conditions for salmonids. Floods and debris torrents are dominant disturbances affecting westside stream systems (Swanston 1991), and may significantly alter channel morphology, scouring channels and creating debris jams and coarse sediment deposits that eventually produce important spawning and rearing areas for salmonids.

Salmonids have evolved not only to the general conditions that are typical of a watershed, but to the specific disturbance regimes found in that watershed. Human activities potentially modify disturbance regimes in three distinct ways: by increasing the frequency of disturbance events, by altering the magnitude of these events, and by affecting the response of the stream channel to disturbance events through modification of instream characteristics. Sediment delivery, for example, is essential to the development and maintenance of spawning gravels for salmonids. However,

alteration in sediment composition, delivery rates, or fate can be damaging to salmonids, resulting in the degradation of spawning gravels and rearing habitats. Similarly, floods and droughts are important determinants of fish community structure; however, increases in the frequency of these events may result in population declines, shifts in community structure, and decreases in biodiversity. The effects of human-caused alterations on salmonids and their habitats are discussed in greater detail in Section 6 of this document.

Table 3.2.a Past Controls and Effects on Landscape Development in the Pacific Northwest

| Period                          | Controls   | Probable geomorphic and ecological effects  |
|---------------------------------|--|---|
| Glacial                         | <p>IN GLACIATED AND PERIGLACIAL AREAS:<br/>                     advance of Cordilleran ice sheet;<br/>                     development of mountain ice sheets and alpine glaciers;<br/>                     very cold climate with reduced precipitation</p> <p>IN UNGLACIATED AREAS:<br/>                     lowered sea level;<br/>                     cold climate with reduced precipitation</p> | <p>glacial erosion and deposition and formation of outwash trains in valleys;<br/>                     periglacial churning and mass movement, intensified mechanical weathering;<br/>                     glacial meltwater discharge;<br/>                     displacement of interglacial ecological communities<br/>                     vegetation cover absent or greatly reduced</p> <p>displacement and shrinkage of estuary areas;<br/>                     reduced vegetation cover;<br/>                     mechanical weathering, mass movement and slope erosion rates greater than modern;<br/>                     increased streamflow and fluvial sediment transport;<br/>                     accumulation of coarse valley fills;<br/>                     reduced organic inputs to streams</p> |
| Late Glacial and early Holocene | <p>retreat and downwasting of glaciers;<br/>                     rapid sea-level rise;<br/>                     warming;<br/>                     effective moisture greater than modern ca. 14,000 to 11,000 yrs ago, then less than modern until ca. 7,000 yrs ago</p>   | <p>glacial deposition and exposure of glaciated land surfaces;<br/>                     landward displacement of estuaries, increase in estuary depth and area;<br/>                     mass movement and slope erosion rates decreasing but still greater than modern;<br/>                     streamflows probably greater than modern;<br/>                     stabilization and then incision of valley fills;<br/>                     increasing vegetation cover and changes in community composition;<br/>                     increased organic inputs to streams, but still less than modern;<br/>                     minor fluctuations in alpine glaciers</p>   |
| Middle Holocene to modern       | <p>sea-level stable;<br/>                     climate approaching modern conditions, with short-term fluctuations</p>  | <p>estuaries filling and shallows developing;<br/>                     slope stabilization and decrease in mass movement rates;<br/>                     decreased mechanical and increased chemical weathering;<br/>                     streamflows near modern, with short-term fluctuations;<br/>                     continued but slowed incision of valley fills;<br/>                     development of modern ecological communities;<br/>                     high rates of organic inputs to streams;<br/>                     minor fluctuations in alpine glaciers</p>  |

Table 3.5a Reach-scale Variations in channel-forming features

| Gradient        | Typical controls  | Valley morphology                          | Channel morphology              |
|-----------------|---|--|---------------------------------|
| steep           | resistant bedrock;<br>landslide control of<br>local base level or<br>coarse sediment<br>input | narrow, contrained                         | cascade;<br>rapid;<br>step-pool |
| moderate to low | weak bedrock;<br>structural basins;<br>fine sediment input                                    | medium,<br>unconstrained, or<br>wide flats | pool-riffle;<br>plane-bed       |

Table 3.5b Types of Channel Units (Habitat Units)\*

| Type        | Morphology  | Hydraulic characteristics   | Ecological function   |
|-------------|---|---|---|
| Pool        | deepest, lowest gradient unit; depth varies within unit; may have asymmetrical cross-section; may accumulate fine bed material  | slow, tranquil, subcritical flow without hydraulic jumps during low flow; scour, turbulence and energy dissipation during high flow | fish rearing  |
| glide (run) | intermediate, uniform depth; symmetrical cross-section; gravel or cobble-bedded   | tranquil subcritical flow generally without hydraulic jumps;  |   |
| riffle      | shallow depth; gravel or cobble bedded  | tranquil, generally subcritical flow with small hydraulic jumps over boulders or cobbles  | insect production; salmon and trout spawning; steelhead rearing; may be winter cover for salmon and trout; aeration |
| rapid       | shallow depth; often have transverse ribs of emergent boulders and pocket pools; common emergent boulders   | 15 - 50% of area in supercritical flow (jumps, standing waves) at low flow  | aeration  |
| cascade     | shallow depth; steeper overall than rapid; consists of a series of short steps over boulders or bedrock ledges; common emergent boulders                              | >50% of area in supercritical flow at low flow  | aeration; may be migration barrier  |
| step -      | isolated small falls, 1 - 2 m high and less than one channel width in length; over boulders, bedrock or large woody debris; common emergent boulders, bedrock or wood | shallow depth; steepest unit  | aeration; may be migration barrier  |

\* Modified from Grant et al. (1990); USFS (1993) and Beschta and Platts (1986).

Table 3.6.1a Precipitation Patterns for Ecoregions in the Range of the Anadromous Pacific Salmonids

| Ecoregion                            | Avg. Annual Precipitation cm (Inches) | Dominant Form                         | Season  |
|--------------------------------------|---------------------------------------|---------------------------------------|---|
| Coast Range                          | 140-318 (55-125)                      | rain                                  | mid fall - early spring                                 |
| Puget Lowlands                       | 89-127 (35-50)                        | rain                                  | mid fall - early spring                                 |
| Willamette Valley                    | 89-114 (35-45)                        | rain                                  | mid fall - early spring                                 |
| Central CA Valley                    | 38-64 (15-25)                         | rain                                  | winter  |
| Southern & Central CA Plains & Hills | 51-102 (20-40)                        | rain                                  | winter  |
| Cascades                             | 127-254 (50-100)                      | rain (low elev.)<br>snow (high elev.) | mid fall - early spring                                 |
| Sierra Nevada                        | 46-216 (18-85)                        | rain (low elev.)<br>snow (high elev.) | mid fall - early spring                                 |
| Eastern Cascades Slopes & Foothills  | 30-64 (12-25)                         | snow                                  | mid fall - early spring                                 |
| Columbia Basin                       | 23-64 (9-25)                          | rain/snow                             | fairly uniform fall - spring                            |
| Blue Mountains                       | 25-102 (10-40)                        | snow                                  | late fall - early spring; >10% summer convective storms |
| Snake River Basin/ High Desert       | 20-64 (8-25)                          | rain/snow                             | fairly uniform with slight peaks in fall and spring     |
| Northern Rockies                     | 51-152 (20-60)                        | snow                                  | fall - spring   |

\*most precipitation  
Data from Omernik and Gallant (1986).

Table 3.6.2a Estimated Precipitation and Evapotranspiration for Western Vegetation Communities

|  | Precipitation |          | Evapotranspiration |          |
|--|---------------|----------|--------------------|----------|
|  | (cm)          | (Inches) | (cm)               | (Inches) |
| <b>Forest</b>                          |               |          |                    |          |
| Lodgepole pine                         | 51-114        | 20 - 45  | 48                 | 19       |
| Engelmann spruce-fir                   | 51-114        | 20 - 45  | 38                 | 15       |
| White pine-larch-fir                   | 64-152        | 25 - 60  | 56                 | 22       |
| Mixed conifer                          | 38-178        | 15 - 70  | 56                 | 22       |
| True fir                               | 51-254        | 20 - 100 | 61                 | 24       |
| Aspen                                  | 51-114        | 20 - 45  | 58                 | 23       |
| Pacific Douglas-fir<br>hemlock-redwood | 51-254        | 20 - 100 | 76                 | 30       |
| Interior ponderosa pine                | 51-76         | 20 - 30  | 43                 | 17       |
| Interior Douglas-fir                   | 51-89         | 20 - 35  | 53                 | 21       |
| <b>Chaparral and Woodland</b>          |               |          |                    |          |
| Southern California<br>chaparral       | 25-102        | 10 - 40  | 51                 | 20       |
| California woodland-<br>grass          | 25-102        | 10 - 40  | 46                 | 18       |
| Arizona chaparral                      | 25-51         | 10 - 20  | 43                 | 17       |
| Pinyon-juniper                         | 25-51         | 10 - 20  | 38                 | 15       |
| Semi-arid grass and shrub              | 13-51         | 5 - 20   | 28                 | 11       |
| Alpine                                 | 64-203        | 25 - 80  | 51                 | 20       |

Table 3.10a Approximate Ranges of Recurrence of Landscape and Channel-forming Processes and the Effects of these Events on Stream Habitats

| Event                 | Range of Recurrence (years) | Channel Changes   | Habitat Effects   |
|-----------------------|-----------------------------|---|---|
| Tectonics             | 1,000–1,000,000             | Creation of new drainages; major channel changes including stream capture due to regional upwarping and faulting  | Subsidence in alluvial and coastal fills creating zones of deposition with increased fines; Steep erosive channels caused by upwarping leads to coarser sediments.  |
| Climatic change       | 1,000–100,000               | Major changes in channel direction; major changes in channel grade and configuration; valley broadening or downcutting; alteration of flow regime   | Changes in type and distribution of spawning gravels; changes in frequency and timing of disturbance events; shifts in species composition and diversity  |
| Volcanism             | 1,000–100,000               | Local blocking and diversion of channel by mudflows and tephra; valley filling and widening; major changes in channel grade and configuration;  | Changes in type and distribution of spawning gravels. Major inputs of sand and silt from tephra   |
| Slumps and earthflows | 100–1,000                   | Low-level, long term contributions of sediment and large woody debris to stream channels; partial blockage of channel; local baselevel constriction below point of entry; shifts in channel configuration | Siltation of spawning gravels; scour of channel below point of entry; accumulation of gravels behind obstructions; partial blockage of fish passage; local flooding and disturbance of side-channel rearing areas   |
| Wildfire              | 1–500                       | Increased sediment delivery to channels; increased large woody debris in channels; loss of riparian vegetation cover; decreased litterfall; increased channel flows; increased nutrient levels in streams | Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and overwintering habitat; decreased availability of fine woody debris; increased availability of food organisms |
| Windthrow             | 10–100                      | Increased sediment delivery to channels; decreased litterfall; increased large woody debris in channel; loss of riparian cover  | Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and overwintering habitat; decreased fine organic debris   |
| Insects and disease   | 10–100                      | Increased sediment delivery to channels; loss of riparian vegetation cover; increased large woody debris in channels; decreased litterfall  | Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and overwintering habitat  |

Table 3.10a Approximate Ranges of Recurrence of Landscape and Channel-forming Processes and the Effects of these Events on Stream Habitats

| Event   | Range of Recurrence (years) | Channel Changes   | Habitat Effects   |
|---|-----------------------------|---|---|
| Activities of beavers   | 5.0–100                     | Channel damming; obstruction and redirection of channel flow; flooding of banks and side channels; ponding of streamflow; siltation of gravels behind dams  | Improved rearing and overwintering habitat; increased water volumes during low flows; slack-water and back-water refuge areas during floods; refuge from reduced habitat quality in adjoining areas; limitation on fish migration; elevated water temperatures; local reductions in dissolved oxygen                                |
| Debris avalanches and debris torrents   | 5.0–100                     | Large, short-term increases in sediment and large woody debris contributions to channel; channel scour; large-scale movement and redistribution of bed-load gravels and large woody debris; damming and obstruction of channels; accelerated channel bank erosion and undercutting; alteration of channel shape by flow obstruction; flooding | Changes in pool to riffle ratio; shifting of spawning gravels; siltation of spawning gravels; disturbance of side-channel rearing areas; blockage of fish access; filling and scouring of pools and riffles; formation of new rearing and overwintering habitat   |
| Major storms; floods; rain-on-snow events   | 1.0–10                      | Increased movement of sediment and woody debris to channels; flood flows; local channel scour; movement and redistribution of coarse sediments; flushing of fine sediments; movement and redistribution of large woody debris   | Changes in pool to riffle ratio; shifting of spawning gravels; increased large woody debris jams; siltation of spawning gravels; disturbance of side-channel rearing areas; increased rearing and overwintering habitat; local blockage of fish access; filling and scouring of pools and riffles                                   |
| Seasonal precipitation and discharge; moderate storms; freezing and ice formation | 0.1–1.0                     | Increased flow to bank-full width; moderate channel erosion; high base-flow erosion; increased mobility of in-channel sediment and debris; local damming and flooding; sediment transport by anchor ice; gouging of channel bed; reduced winter flows   | Changes in pool to riffle ratio; siltation of spawning gravels; increased channel area; increased access to spawning sites; flooding of side-channel areas; amelioration of temperatures at high flows; decreased temperatures during freezing; dewatering of gravels during freezing; gravel disturbance by gouging and anchor ice |
| Daily to weekly precipitation and discharge                                       | 0.01–0.1                    | Channel width and depth; movement and deposition of fine woody debris; fine sediment transport and deposition   | Minor siltation of spawning gravels; minor variation in spawning and rearing habitat; increased temperature during summer low flows   |

Modified from Swanston (1991).

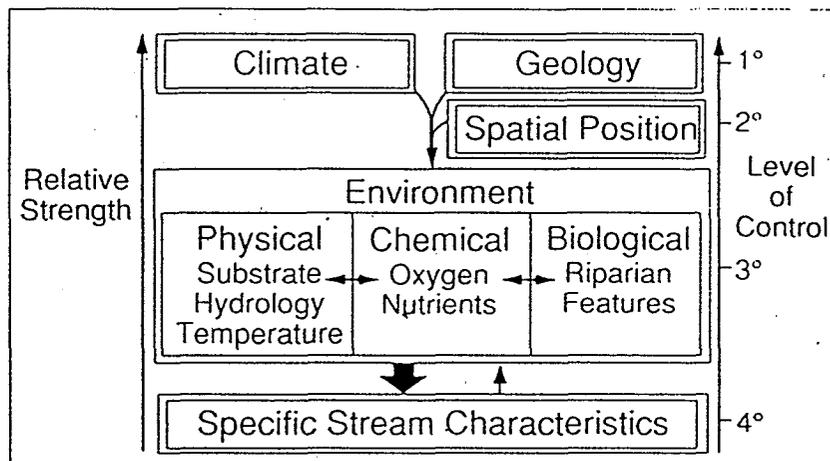


Figure 3.0a Relative strength of factors influencing stream characteristics, and principal feedback loops between components. From Naiman et al. (1992). Reproduced with permission from the publisher.

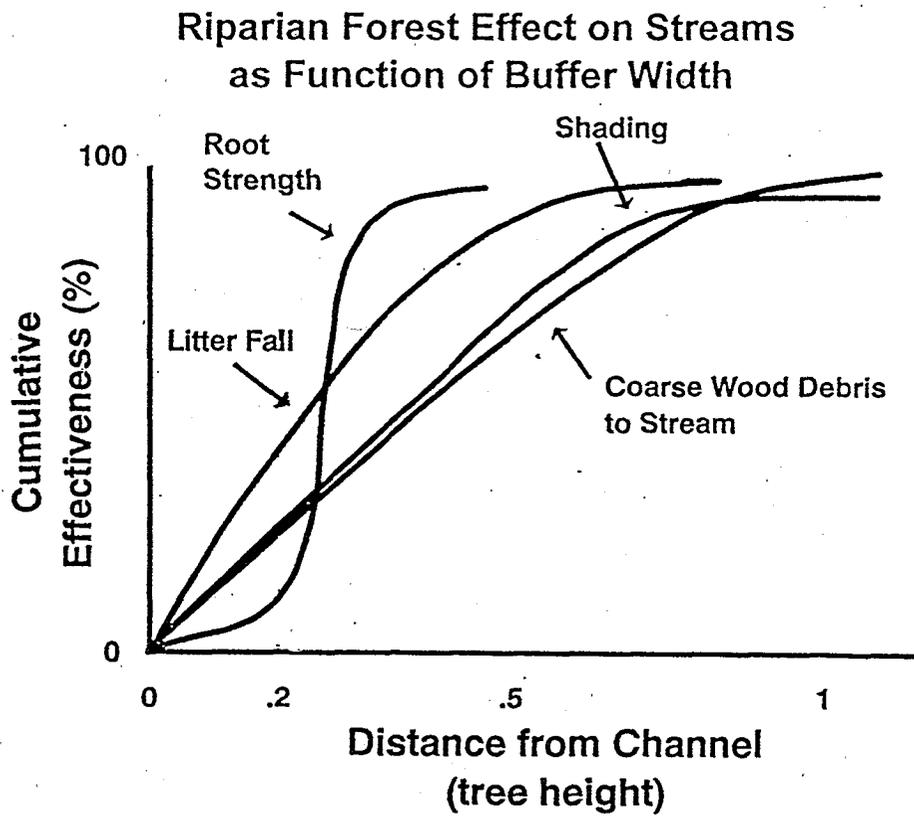


Figure 3.9.1a Riparian forest effect on streams as function of buffer width. From FEMAT (1993).

### Riparian Buffer Effects on Microclimate

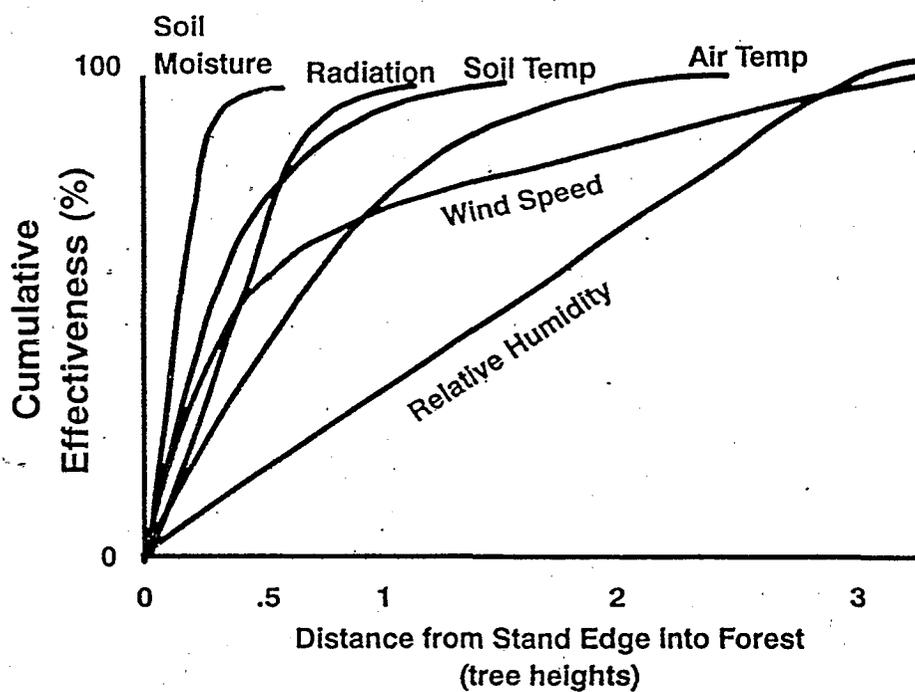


Figure 3.9.7a Riparian buffer effects on microclimate. From FEMAT (1993).

## 4.0 Biological Processes and Concepts

The physical and chemical environment of aquatic ecosystems forms the template upon which biological systems at all levels are organized (Southwood 1977; Poff and Ward 1990). The spatial and temporal patterns of water quantity and velocity, temperature, substrate, and dissolved materials influence the physiology and behavior of organisms, the dynamics and evolution of populations and metapopulations, and the trophic structure and diversity of aquatic communities. Modification of physical and chemical attributes of water bodies through land-use and water-use practices, and direct alteration of specific biological components of these systems, can result in changes to individual organisms, populations, and communities. In this section, we briefly review some fundamental biological processes that occur in aquatic ecosystems, focusing on those processes that are likely to be affected by modifications to physical and chemical habitat characteristics.

### 4.1 Organism Level

The survival of fish in the wild depends on their ability to carry out basic biological and physiological functions including feeding, growth, respiration, smoltification, migration, and reproduction. The fate of populations and the outcome of higher-level biological interactions—competition, predation, and disease—ultimately depends on the performance of individuals in obtaining food, defending space, maintaining physiological health, or otherwise coping with their ecological circumstances. Characteristics of the physical and chemical environment of fish, particularly water temperature, regulate the rate at which these processes occur. A detailed discussion of the complex interactions between fish and their environments is beyond the scope of this document. However, a brief review of the fundamental biological and physiological processes is essential to understanding how habitat modifications may affect salmonids.

#### 4.1.1 Bioenergetics and Growth

A useful way of viewing the effects of environmental factors on individual fish is through a simple energy budget. Food energy that is ingested by fish (I) has several potential fates. It is either expended during metabolic processes (M), deposited as new body tissue (G), or excreted as waste products (E) (Jobling 1993). Thus the energy balance can be expressed as:

$$I = M + G + E$$

Environmental conditions influence all aspects of a fish's energy budget. Temperature, in particular, has pervasive effects on bioenergetic pathways. In general, appetite, standard and active metabolic rates, and food conversion efficiency (i.e., the proportion of food energy absorbed by the fish) increase with increasing temperature up to the physiological optimum for the species and then decline at higher temperatures (Brett 1971; Jobling 1993). Digestion rate continues to increase even at high temperatures. Water velocity in streams dictates active metabolic rates for fish holding within the current. Dissolved oxygen concentrations may affect food consumption and metabolic rates (Warren 1971), as do various chemical pollutants. All these environmental factors interact to determine the amount of energy expended on metabolic processes, and hence determine the energy left over for growth and reproduction.

#### 4.1.2 Food and Feeding

Juvenile salmonids are generally opportunistic in their feeding habits while in fresh water, consuming primarily drifting aquatic or terrestrial invertebrates in streams, and macroinvertebrates and zooplankton in lakes and estuaries. Bull trout especially, as well as rainbow and cutthroat trout, may feed on other fishes and amphibians during their adult stages, particularly in systems where they attain large sizes. A summary of specific dietary items for anadromous and resident salmonids can be found in Meehan and Bjornn (1991); a more detailed examination of dietary habits of Pacific salmon can be found in Groot and Margolis (1991).

Environmental conditions can influence the amount and type of prey available to salmonids, as well as the energetic costs of obtaining that food. Many salmonids seek out areas of slow water velocity immediately adjacent to faster waters, presumably because these areas provide greater food per unit of energy expended in maintaining position (Smith and Li 1983; Fausch 1984). Velocity heterogeneity, therefore, creates microhabitats that are energetically favorable. Turbidity in streams may reduce light penetration and decrease the reactive distance of salmonids to prey, as well as limit benthic algal production. Nutrients may affect total food availability by controlling primary production. Sedimentation may alter substrate composition, thereby affecting the total abundance and composition of prey.

#### 4.1.3 Respiration

Most of the energy used by salmonids to swim, locate food, grow, and reproduce is provided through metabolic processes that require oxygen. Because water contains only about 3.3% of the amount of oxygen contained in air, the efficient extraction of oxygen is critical to survival. Fish, and salmonids in particular, have evolved elaborate gill structures that facilitate the uptake of oxygen for delivery to other parts of the body. Environmental conditions can have a significant influence on the oxygen demands of fish, the amount of oxygen present in water, and the ability of fish to take up that oxygen. In general, the oxygen demands increase with increasing temperature, although oxygen consumption may decrease as temperatures approach lethal levels, particularly at high levels of activity (Brett 1971). In contrast, dissolved oxygen levels in saturated water are inversely proportional to temperature, with water at 5°C holding approximately 30% more oxygen than water at 20°C. Oxygen demand is also influenced by water velocity, which determines the swimming speed required of salmonids to maintain their position in the current. High levels of suspended solids in water may influence respiration by abrading or clogging gill surfaces (Warren 1971). Similarly, pollutants can cause mucous secretions to coat gill surfaces, inhibiting the exchange of oxygen. Excessive amounts of easily decomposable organic material in water increases microbial oxygen demand and thereby decreases dissolved oxygen concentration. On the other hand, gas supersaturation from dam spills and intense algal photosynthesis can create gas bubbles in fish gills and tissue, resulting in decreased respiratory efficiency, disease, or death. All of these factors can influence the ability of fish to satisfy their oxygen demands.

#### 4.1.4 Reproduction and Embryological Development

One measure of the success of an organism is its ability to reproduce. Energy reserves of salmonids must be sufficient to allow for gamete production after other metabolic costs are incurred. Anadromous salmonids have particularly high energy requirements, in that they must have sufficient reserves to undergo lengthy migrations and negotiate barriers in order to reach the ocean and then return to their spawning tributaries. Modifications of temperature, water quality, streamflow and physical structure all may affect how much energy can be devoted to reproductive output. The development of embryos and alevins in the gravel is affected by several environmental factors. Water temperature greatly influences times to hatching and emergence for Pacific salmonids. Development time decreases in an asymptotic fashion with increasing incubation temperatures, with the rate of change in development time relative to temperature increase being greatest at the low end of the tolerable temperature range (Beacham and Murray 1990). Consequently, small increases in temperature at the low end of the range can substantially alter the time of hatching and emergence of salmonids. Early emergence due to warming of water temperatures may increase exposure of fry to high flow events, and alter the natural synchrony between emergence and predator or prey cycles. Scrivener (1988) found that chum salmon in Carnation Creek emerged and migrated to sea 4 to 6 weeks earlier after logging compared with pre-logging years in response to water temperature increases. In a companion study, Holtby (1988) reported that coho salmon emerged up to 6 weeks early in response to logging. Temperatures may influence the size of emerging fry. For example, coho salmon reared at 4°C were larger than those reared at warmer temperatures (Beacham and Murray 1990). In contrast, pink salmon fry tended to be larger when reared at 8°C than when reared at 4°C.

Dissolved oxygen concentrations in redds also influence the survival and development rate of embryos and alevins, as well as the size of emerging fry (Warren 1971). Streamflow may regulate the flow of water through redds and hence the levels of dissolved oxygen. Environmental changes, such as siltation or altered flow regimes, that reduce the flow of water can thus adversely affect embryo and alevin development and survival. Silt concentration in gravels may also impede emergence of fry. Phillips et al. (1966) found that emergence of coho fry decreased as the percentage of fine sediments in the gravel increased, presumably because of reduced oxygen content and increased difficulty of fry in reaching the surface. Increased frequency of high scouring flows or debris torrents, which are associated with disturbed catchments (Swanston 1991) may further affect egg and alevin survival.

#### 4.1.5 Smoltification

The transition from fresh to salt water marks a critical phase in the life history of anadromous salmonids. Emigration to the ocean is preceded by rapid physiological, morphological, and behavioral transformations that preadapt fish for life in salt water and initiate their downstream movement (Folmar and Dickhoff 1980; Groot 1982; Wedemeyer, Saunders et al. 1980). Once in the ocean, newly arrived smolts must acclimate to a markedly different set of ecological circumstances, including new food resources, new predators, and a substantially different physical environment. Much of the total ocean mortality incurred by salmon smolts is believed to occur during this period of early ocean life (Manzer and Shepard 1962; Matthews and Buckley 1976; Walters et al. 1978; Fisher and Pearcy 1988; Pearcy 1992). Consequently the timing of ocean entry is likely adaptive to maximize survival and/or growth (Riddell and Leggett 1981; Murphy et al. 1988; Beacham and Murray 1990; Miller and Brannon 1981).

Because development and growth are highly influenced by water temperatures, modifications to thermal regimes can potentially alter the time of smoltification (reviewed in Wedemeyer et al. (1980) and Hoar (1988). Similarly, temperature and streamflow patterns may be important cues for releasing migratory behavior (Hoar 1988). Consequently, alterations in normal hydrologic and thermal patterns may trigger movement into the ocean at times that are less favorable for growth and survival. The parr-smolt transformation may also be affected by exposure to contaminants, including heavy metals, which alter enzymatic systems involved in osmoregulation and may inhibit migratory behavior (Wedemeyer et al. 1980). Structural alterations that hinder salmonids during the smolt transformation include loss of large woody debris and habitat complexity in streams and estuaries, which reduces cover and food supplies during this critical period.

## 4.2 Population Level

### 4.2.1 Generalized Life Cycle

The life cycle of anadromous salmonids consists of several distinct phases, at least three of which involve significant shifts in habitat. Adult salmon migrate from the ocean into their natal stream to spawn. Females construct a "redd" in the stream gravel into which eggs are deposited, fertilized by males, and subsequently covered with gravel. All adult salmon die after spawning, usually within a few weeks. Females will typically spend one to three weeks guarding the redd site before dying, whereas males may seek out and spawn other females. The fertilized embryos develop for a period of one to several months, depending on temperature and dissolved oxygen availability, before hatching occurs. The emergent "alevins" remain in the gravel, nourished by a yolk sac, for another few weeks to a month or more. Once yolk-sac absorption is complete, the fry emerge from the gravel and begin actively feeding on drifting material. The period of freshwater rearing lasts from a few days to several years, depending on the species, after which juveniles undergo smoltification. Smolts migrate to the ocean, where the majority of growth occurs, before returning to spawn as adults, completing the cycle (Figure 4.2.1a).

The life cycles of the anadromous trout and char differ from those of salmon in that some adults may survive after spawning, migrate back to the ocean, and return to spawn a second or third time. Resident salmonids, including kokanee salmon (i.e. landlocked sockeye salmon), bull trout, cutthroat trout, rainbow trout, and mountain whitefish, do not have an oceanic phase, but commonly undergo substantial migrations to and from rearing areas in lakes or larger rivers. With the exception of kokanee salmon, which die after spawning, the resident forms usually spawn multiple times over their lifetimes.

### 4.2.2 Life History

#### 4.2.2.1 Life History Patterns

Although all anadromous salmonids share the general life cycle discussed above, substantial differences exist in the period of time that the different species spend in freshwater and marine environments (Table 4.2.2.1a), and the types of habitat they use for spawning and rearing. In addition, a high degree of variation in life histories can exist within each species. Extensive reviews of the life histories and general habitat preferences of the Pacific salmon, trout, and char can be found in Groot and Margolis (1991) and Meehan and Bjornn (1991), from which much of the information below was taken. Pink and chum salmon typically spawn in coastal streams not far from tidewater—chum occasionally within the tidal zone—and have the shortest freshwater phase, entering the ocean soon after they emerge from the spawning gravels (Table 4.2.2.1b). Almost without exception, pink salmon mature at 2 years of age, at which time they return to freshwater to spawn. Chum salmon are more variable, spending from 2 to 5 years in the ocean before returning to their natal area to spawn. Coho salmon generally spawn in small, low-gradient streams or stream reaches in both coastal and interior systems. Juveniles typically spend from 1 to 3 years in freshwater; however, in the southern portion of their range (including Washington, Oregon, and California) most fish migrate to sea after just one year. Adults return to spawn after approximately 18 months at sea, although "jack" males may return after only six months in the ocean (Sandercock 1991). The life histories of sockeye and chinook salmon are more variable. Sockeye salmon most often spawn in the inlet or outlet streams of lakes. Shortly after emergence, sockeye fry migrate into these lakes, where they reside for 1 to 3 years. Juveniles then migrate to the ocean, where they spend 2 to 3 years. Chinook salmon generally spawn in small to medium-sized rivers, but may also spawn in large river systems such as the mainstem Columbia. Chinook salmon display two dominant life history types, an ocean type that is typical of fall-run stocks and a stream type that is characteristic of spring-run fish. Those exhibiting the ocean-type life history typically spend only a few months in freshwater before migrating to sea. Stream-type fish spend 1 to 2 years in freshwater. Both ocean- and stream-type fish can spend anywhere from 2 to 5 years in the ocean, although jacks may spend less than a year at sea before returning to spawn. Within any given population, multiple life-history patterns may be observed. Based on time of freshwater and estuarine residence, Reimers (1973) identified five distinct life-history patterns for fall chinook salmon in the Sixes River, Oregon.

The anadromous trout and char, including steelhead and cutthroat trout, and Dolly Varden, exhibit considerable life-history variation as well. Steelhead trout tend to spawn in small streams and favor relatively high-gradient reaches. Freshwater residence can last from 1 to 4 years, while ocean residence ranges from a few months ("half-pound") males to 4 years. Although most adult steelhead die after spawning, up to 30% may live to return to the ocean and spawn again in subsequent years, particularly in coastal streams where the spawning migrations are fairly short (Meehan and Bjornn 1991). Consequently, the number of potential life-history types is large. Anadromous cutthroat trout most commonly spawn in small headwater streams and spend 2 to 4 years in freshwater before migrating to the ocean during the spring, where they generally remain until the next fall. As with steelhead trout, some adults may live after spawning, migrate back to the ocean, and return a second or third time. Dolly Varden spawn in coastal streams and exhibit complex life-history patterns. Juveniles typically rear in higher-velocity habitats for several years (Meehan and Bjornn 1991). After smoltification, Dolly Varden enter the ocean, but may repeatedly return to freshwater habitats during the winter months to rear in lakes, sometimes away from their natal areas. Thus, it is difficult to generalize about the periods of freshwater and marine residence for Dolly Varden.

Resident trout, char, and whitefish spend their entire lives in freshwater; however, life-history patterns may still be quite diverse. Varley and Gresswell (1988) identified four principal life history patterns for Yellowstone Lake cutthroat trout: fluvial populations that remain in their natal streams throughout their lives, fluvial-adfluvial populations that reside in larger rivers but spawn in small tributaries, lacustrine-adfluvial populations that reside in lakes and spawn in tributary streams, and allucustrine populations that reside in lakes and migrate down outlet streams to spawn. Rainbow trout may spawn in streams or lake inlets or outlets and rear in streams or lakes (Behnke 1992). Bull trout reside in a variety of freshwater habitats including small streams, large rivers, and lakes or reservoirs (Meehan and Bjornn 1991). Some populations spend their entire lives in cold headwater streams. In other populations, juveniles spend from 2 to 4 years in their natal stream before migrating into lakes or reservoirs, where they reside for another 2 to 4 years before returning to their natal stream to spawn. Mountain whitefish spawn in streams and rivers and reside there throughout their lives, although substantial migrations from larger rivers into smaller spawning tributaries are common.

#### 4.2.2.2 *Implication of Life-History Diversity for Salmonid Conservation*

The remarkable diversity of life histories exhibited by Pacific Northwest salmonids reflects adaptation to a wide array of habitats. As a group, the salmonids inhabit streams ranging from mountain headwaters to large lowland rivers, in regions varying from arid and semi-arid shrublands to temperate rainforests. Reproduction may occur in streams, lakes, or intertidal sloughs; rearing of juveniles occurs in streams and lakes for some species, and in estuaries and oceans for others. In any particular habitat, spatial and temporal differences in micro- and macrohabitat utilization permit the coexistence of ecologically similar species (Everest et al. 1985). Within species, life-history diversity allows salmonids to fully utilize available freshwater, estuarine, and ocean environments. Species that occupy several habitat types, or that have multiple freshwater and marine residence times, effectively spread ecological risk (*sensu* Den Boer 1968) such that the impacts of environmental fluctuation on populations are distributed through time and space. Consequently, species are likely to differ in their response to human-caused perturbation. The diversity among species and by life stage indicates that most accessible freshwater habitats are used year round if environmental conditions are suitable (Table 4.2.2.2).

Life history diversity should be considered in the development of salmonid conservation strategies and local enhancement measures (Carl and Healey 1984; Lichatowich et al. 1995). The historically high abundance of salmonids in the Pacific Northwest was due in part to the diversity of life-history patterns exhibited by the various species. Habitat simplification through land-use and water-use practices has effectively simplified this diversity in life-history organization. In the Columbia River, for example, salmonids of various species and life stages were present in the mainstem year round. Because of alteration of temperatures and flow regimes, the temporal usage of the mainstem and major tributaries is now far more restricted. Historical records indicate that the Yakima River once supported both ocean-type and stream-type chinook salmon, the ocean type being found in warmer, low-elevation sites. Today, because irrigation withdrawals have reduced flows and increased temperatures, the population consists only of stream-type fish isolated in cooler headwater streams (Spence 1995). Restoration of such stocks to a harvestable level will require restoration of habitat conditions suitable for ocean-type chinook salmon, which will restore the former life-history diversity. Differences in life histories also affect the response of salmonids to harvest. Salmon that spend several years at sea before maturing are more vulnerable to troll fisheries than those that spend only a year at sea.

#### 4.2.3 Stock Concept and Local Adaptation

Among the most remarkable characteristics of anadromous salmonid species is their tendency to return to their natal stream to spawn during a particular season, often after ocean migrations of a thousand miles or more. Although the strong homing tendency of salmonids is most conspicuous in anadromous species, it may be common in resident populations as well. Lake-dwelling populations of cutthroat and bull trout that spawn in tributaries have also been

shown to return to their natal stream to spawn with low rates of straying (Pratt 1992; Gresswell et al. 1995), and it is likely that stream-dwelling residents also display some fidelity to their natal area. As a consequence of homing, salmonid species typically comprise numerous local populations or "stocks" that are to varying degrees reproductively isolated from other such populations. Ricker (1972) defined a stock as "the fish spawning in a particular lake or stream (or portion of it) at a particular season [that] to a substantial degree do not interbreed with any group spawning in a different place, or in the same place at a different season."

The homing and the resultant reproductive isolation of stocks provide a mechanism by which local populations become uniquely adapted to the specific suite of environmental conditions encountered during their life histories. Ricker, in his classic 1972 paper that formalized this concept, catalogued dozens of examples of local variation in morphological, behavioral, and life-history traits, and provided evidence that many of these traits are to some degree heritable. For a trait to be considered adaptive, it must not only be differentially expressed, but it must confer some advantage to the individuals exhibiting that trait. More formally, Taylor (1991) defined local adaptation as "a process that increases the frequency of traits within a population that enhance the survival or reproductive success of individuals expressing such traits." He identified many examples of variation in morphological, behavioral, developmental, biochemical, physiological, and life-history traits in the family Salmonidae that are both heritable and believed to be adapted to local conditions. Results from his extensive review are summarized in Table 4.2.3.

Despite the fact that the stock concept is generally credited to Ricker, the implications of stock formation and local adaptation in salmonid conservation have long been recognized. Rich (1939) proposed that conservation of a species that is made up of numerous isolated, self-perpetuating units depends on conserving each constituent part. While Rich argued that local adaptation was not necessary for stocks to be the appropriate unit of management, the recognition that stocks do differ in heritable traits and that these differences are a consequence of differential selection serves to strengthen the argument for conserving individual salmonid stocks. The loss of local stocks changes the genetic composition and reduces the genetic variability of the species as a whole (Nehlsen et al. 1991), reducing its ability to respond to environmental change.

From Table 4.2.3, it is evident that many traits of salmonids are adaptations to environmental conditions that may be significantly altered by human activities. In the wake of rapid and extensive anthropogenic change, traits that were once adaptive may be rendered maladaptive. For example, the timing of spawning, emergence, and smoltification of salmonids are clearly linked to stream temperature regimes, as are development rates of eggs and juveniles. Warming of stream temperatures through loss of riparian canopy, releases from reservoirs, or irrigation practices can advance development or alter the timing of life-history events and potentially disrupt natural synchronies in biological cycles that have evolved over thousands of years. Alteration of temperatures may also affect embryo and alevin survival, as well as enzyme activity, in populations that are specifically adapted to warm or cool environments. Thus, small changes in temperature may prove ecologically damaging even though such changes would produce no evidence of acute or chronic physiological stress. Other characteristics, including body morphology, agonistic and rheotactic behavior, and the timing of smolt and adult migrations, are tied to streamflow. Changes in the timing or magnitude of flows due to hydroelectric operations, agricultural diversions, or disruption of hydrologic processes from forest and range practices may effect these characteristics of fish. In this context, the ability of species-specific (versus stock-specific) criteria for water quality, instream flows, and other habitat attributes to adequately protect individual salmonid stocks may need to be reevaluated.

#### 4.2.4 Metapopulation Dynamics

The stock concept focuses on the reproductive isolation and subsequent adaptation of local populations to the particular environments that they inhabit. Metapopulation theory is concerned with the behavior of groups of populations, or "metapopulations," that interact via individuals moving among populations through the processes of dispersal or straying (Hanski and Gilpin 1991). The term metapopulation dynamics thus describes the long-term behavior of a metapopulation over time.

Central to metapopulation theory is the idea that local populations within a metapopulation periodically go extinct as a result of natural disturbances, leaving vacant habitat patches that may subsequently be recolonized by individuals from other populations (Hanski and Gilpin 1991). Metapopulation persistence requires that, among local populations, the recolonization rate must exceed the extinction rate (Hanski 1991).

While discussion of metapopulation dynamics of anadromous and resident salmonids is largely absent from the literature (see Li et al. In Press); a number of principles from metapopulation theory relate to salmonid conservation. Evidence from other taxonomic groups suggests that the probability of a local extinction increases with decreasing population size, decreasing size of habitat patches, and increasing isolation from other conspecific populations (reviewed in Hanski 1991; Sjögren 1991). The risk of extinction is also believed to be greater for populations that undergo large natural fluctuations in abundance (Harrison 1991). Recolonization rates are similarly influenced by population size and distance between habitat patches. Reestablishment of populations depends on sufficient numbers of individuals invading that habitat, which in turn depends on dispersal rates, the population size of

source populations, the proximity and size of nearby habitat patches, and the availability of suitable migration corridors between patches. Hanski (1991); Hansson (1991); and Harrison (1991) have argued that metapopulation persistence may be dependent on the existence of a few extinction-resistant populations serving as sources of recolonization for other extinction-prone populations.

Salmonid metapopulations exhibit many characteristics that would appear to make them vulnerable to extinction. Nehlsen et al. (1991) identified 101 stocks of anadromous salmonids that have had escapements under 200 within the last 1 to 5 years. These stocks are at increased risk of extinction from stochastic genetic, demographic, or environmental events. For example, the 1976–1977 drought in California elevated water temperatures and resulted in precipitous declines of winter Chinook salmon in the Sacramento River, leading to their listing under the Endangered Species Act (Nehlsen et al. 1991). Many extant salmonid stocks have been eliminated from lower-elevation stream reaches and persist only as remnant populations confined to smaller harsher headwater streams that have been less affected by habitat alterations. Low-order streams in steep headwaters tend to be hydrologically and geomorphically more unstable than low-gradient, high-order streams. Thus, salmonids are being restricted to habitats where the likelihood of extinction due to random environmental events is greatest. Increased fragmentation of aquatic habitats and isolation of salmon populations reduces the chances that straying individuals from other populations can help restore depleted stocks. Snake River sockeye salmon provide a good example of an isolated population that is unlikely to be rescued by strays from other populations, since stocks in the region are separated by hundreds of river kilometers and straying rates are low. Lastly, salmonids have historically experienced wide interannual variation in numbers as a consequence of variation in both freshwater and marine conditions. Numbers of coho salmon returning to streams in Oregon, Washington, and California can vary by an order of magnitude or more in different years (Hall and Knight 1981). Similar variability in escapement of pink and sockeye salmon has also been documented (Burgner 1991; Heard 1991). The probability of extinction because of fluctuating numbers combined with random environmental events may be particularly high for those species such as pink and coho salmon that have comparatively rigid life histories. In these species, the loss of a particular year-class may have longer-lasting effects than in populations with greater diversity in the age of spawning adults.

A final aspect of metapopulation theory that is relevant to salmonid conservation relates to temporal difference in the dynamics of the local populations that constitute the metapopulation. Hanski (1991) proposed that metapopulation persistence should be greatest where local populations fluctuate independently of each other, i.e., asynchronously, and lowest where local populations fluctuate synchronously in response to regional environmental conditions. The widespread declines in salmon populations throughout the Pacific Northwest suggest that fluctuations in these populations are synchronous, therefore, the risk of metapopulation extinction is relatively high.

#### 4.2.5 Evolutionarily Significant Units

Under the Endangered Species Act or ESA (as amended in 1978), a "species" is defined to include:

"any subspecies of fish or wildlife or plants, or any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature."

For anadromous Pacific salmon and trout, most stocks are, to varying degrees, reproductively isolated—and hence potentially distinct population segments—but the ESA provides no direction for determining what constitutes a distinct population segment. To address this concern, the National Marine Fisheries Service (NMFS) has adopted a definition of "species" that is based on the concept of "Evolutionarily Significant Units" or "ESUs" (Waples 1991b). A population is considered an ESU if it meets two criteria: 1) it is substantially reproductively isolated from other conspecific population units, and 2) it represents an important component in the evolutionary legacy of the species (Waples 1991b). For the first criterion to be met, isolation of the population need not be complete, but it must be sufficient to allow accrual of differences in specific traits among populations. Nor is isolation by itself sufficient for a population to be considered distinct (Waples 1991b). A population may meet the second criterion if it contributes to the overall genetic diversity of the species. In addition, because ecological diversity may foster local adaptations, stocks occupying distinct or unusual habitats or that are otherwise ecologically distinct may also be ESUs (Waples 1991b).

The intent of the ESU framework is to conserve the genetic diversity of species and the ecosystems that species inhabit, two fundamental goals of the ESA (Waples 1991b). The genetic variability within a stock or population represents both the legacy of past evolutionary events and the ability of the population to respond to future environmental changes. The loss of individual stocks or the alteration of the genetic composition of stocks through hatchery introductions can fundamentally alter the ability of the species cope with local environmental conditions, to respond to environmental change, and hence to persist over the long-term.

Waples (1991b) advocates a two-step approach for determining whether or not a population represents a distinct unit. The first step is to evaluate the degree of reproductive isolation of the population. With salmonids, and particularly anadromous forms, reproductive isolation is rarely complete because of straying and is more a matter of

degree. Waples (1991b) recommends several approaches for assessing the degree of reproductive isolation including: 1) use of tags to estimate straying rates, 2) intentional genetic marking of populations, 3) use of genetic indices to estimate levels of gene flow, 4) observation of recolonization rates, and 5) identification of physical or geographic features likely to act as barriers to migration. The second step is to evaluate whether or not the population exhibits evidence of substantial ecological or genetic diversity. Factors to consider include: 1) genetic traits, including unique alleles, different allelic frequencies, total genetic diversity; 2) phenotypic traits, including morphological or meristic characters, occurrence of parasites, and disease or parasite resistance; 3) life-history traits, such as time, age, or size at spawning, fecundity, migration patterns, and timing of emergence and outmigration; and 4) habitat characteristics, including temperature, rainfall, streamflow, water chemistry, or biological attributes of the particular system (Waples 1991b).

As Waples (1991b) notes, interpretation data for reproductive isolation is not always straightforward. For example, assessments of straying rates may be confounded by behavior of migratory adults (e.g., temporary entry of fish into non-natal streams). Measures of gene flow may require assumptions of selective neutrality for the alleles used. Assessment of allelic frequencies or presence of unique alleles may be influenced by sampling design, including number of samples and time of sampling. Similarly, interpreting ecological or genetic diversity data may be equally difficult. Variation in phenotypic and life-history characteristics may be attributable to both genetic and environmental factors; thus phenotypic or life-history variation alone is insufficient for determining population distinctness. The ability to distinguish distinct and unusual habitats is limited by both science and differences of opinion as to what are important habitat characteristics.

Identification of evolutionarily important biological units for protection is made more difficult by the fact that a significant number of salmonid stocks have already been lost, and as a result, our understanding of metapopulation structure and function is incomplete. Li et al. (in press) note that few high-quality habitats remain and that many of these lie at the extremes of species' ranges. They argue that conservation strategies should differ depending on metapopulation structure. For example, Levins' (1969) model of metapopulations assumes that populations within each metapopulation each carry equal "evolutionary weight," whereas the "core-satellite" model of Harrison (1991) proposes that dispersal from larger "core" populations results in smaller satellite populations. With the first model, the conservation strategy might be to protect as many populations as possible in order to protect diversity. In the core-satellite system, emphasis should be placed on protecting core populations, since failure to do so would result in marginal populations of narrow specialists occupying the extremes of the species' range (Li et al., in press). Waples (1991b) similarly argues that threatened and endangered status should be considered for metapopulations as well as more discrete population units.

Finally, an assumption of the ESU concept is that not all populations need to be protected in order to preserve the genetic integrity of the species (Waples 1991b). Local populations that are not reproductively isolated or that are isolated but fail to exhibit any important and distinctive genetic or life history traits do not qualify for protection under the ESA. In practice, such populations are typically not genetically differentiable from hatchery populations. Where such populations are lost, their ecological function in the aquatic community will also be lost and other organisms may be affected over the evolutionary short term (see Sections 4.3 and 5.4). However, over longer evolutionary time scales the ESU conservation strategy will result in available habitats repopulated by native fish from either within the local ESU or from neighboring ESUs. This should result in fish populations locally adapted or more able to survive and reproduce in the wild, thereby fulfilling their role in the ecosystem.

### 4.3 Community Level

The biotic communities of aquatic systems are highly complex entities. Within communities, assemblages and species have varying levels of interaction with one another. Direct interactions may occur in the form of predator-prey, competitor, and disease- or parasite-host relationships. In addition, many indirect interactions may also occur between species. For example, predation of one species upon another may enhance the ability of a third species to persist in the community by releasing it from predatory or competitive constraints. These interactions continually change in response to shifting environmental and biotic conditions. Human activities that modify either the environment, the frequency and intensity of disturbance, or species composition can shift the competitive balance between species, alter predatory interactions, and change disease susceptibility, all of which may result in community reorganization.

The role of disturbance in regulating stream community organization has been a principal focus of aquatic ecology in the past decade. In a recent review, Resh et al. (1988) identify three theories (equilibrium, intermediate disturbance hypothesis, dynamic equilibrium model) that reflect our present understanding of disturbance theory as it relates to stream community structure. The equilibrium theory proposes that environments are more or less constant and that community organization is determined by biotic interactions, including competition, mutualism, and trophic interactions. The intermediate disturbance hypothesis assumes a competitive hierarchy of species. In the absence of disturbance, superior competitors eliminate inferior ones, whereas in systems with frequent or severe disturbances,

resident competitors are eliminated and colonizing species dominate. In systems with intermediate disturbance regimes, species richness is maximized; colonizers exploit disturbed areas and are thus able to coexist with superior competitors. The dynamic equilibrium model proposes that community structure is a function of growth rates, rates of competitive exclusion, and frequency of population reductions. Inferior competitors persist in the community if disturbances occur often enough to eliminate competitive exclusion; however, if disturbances are too frequent, species with long life cycles are eliminated. Species diversity is determined by the influence of the environment on the net outcome of species interactions. Both the intermediate disturbance hypothesis and dynamic equilibrium model emphasize the role of the environment in regulating stream communities, and Resh et al. (1988) conclude that these hypotheses are more generally applicable to stream ecosystems than the equilibrium model. All three models may be applicable depending on spatial and temporal scales.

#### 4.3.1 Food Webs

The food energy available to fish and other organisms in aquatic ecosystems comes from two sources: aquatic plants (macrophytes, benthic algae, and phytoplankton) that convert solar energy into biomass, and organic litter that falls into the stream and provides the energy base for fungi and bacteria (OWRRI 1994). The relative importance of these energy sources changes with the size and morphology of a river, estuary, or lake system (see Section 4.4.2) and the availability of nutrients in the catchment. Herbivorous aquatic invertebrates consume algae and other aquatic plants, whereas detritivorous invertebrates consume decaying organic matter. Many invertebrates select food on the basis of size, rather than source, while others are generalized feeders. Predatory invertebrates may add an additional trophic level to the food web. Collectively, these invertebrates form an important food base for many juvenile anadromous salmonids and adult trout, although some species may feed on other fishes and terrestrial insects that fall into the stream. Fishes, in turn, are consumed by a host of terrestrial and aquatic predators, including other fishes, birds, mammals, reptiles, and amphibians. The resulting food webs can be highly complex, consisting of many species representing several trophic levels.

Food webs may be highly modified by environmental changes, including alterations of the food base; changes in streamflow, temperature, and substrate; and the introduction of non-native organisms. Alterations of individual components of a food web can propagate throughout the system, leading to community wide adjustments in food web composition. For example, impoundments on the Columbia River have shifted the food base from coarse detrital material derived from wetland emergent vegetation and fine material derived from periphyton to a phytoplankton-derived micro-detritus food base, creating numerous adjustments throughout food web (Simenstad et al. 1990; Palmisano et al. 1993b). In the estuary, amphipods and isopods—the preferred food items of salmonid smolts (Dawley et al. 1984)—have now been replaced by suspension feeding epibenthos (Simenstad et al. 1990), which are a primary food source for juvenile American shad. An increasingly favorable environment for shad, coupled with relatively low predation rates, has allowed the population to increase dramatically over the last few decades from less than 200,000 to approximately 4 million (Palmisano et al. 1993b). Elimination of woody riparian vegetation from rangeland streams has shifted the food base from coarse, terrestrially derived material to periphyton. The latter is most efficiently consumed by shell cased macroinvertebrates that are unsuitable prey for juvenile salmonids (Tait et al. 1994).

Changes in water temperatures may change the composition of algal communities (Bush et al. 1974); disrupt the development and life-history patterns of benthic macroinvertebrates (Nebeker 1971; Lehmkuhl 1972) and zooplankton (Hutchinson 1967); and decrease the abundance of certain benthic invertebrates, especially species that are stenotherms (Hynes 1970).

Introductions of non-native fish, either as game fish or forage for other fish, have led to food web alterations in most river systems of the Pacific Northwest. In California and Oregon, introduced fishes constitute 35 and 29% of the total species, respectively. The impact of these fish on native species is poorly known, but they are potential predators and competitors of both the juvenile and adult salmonids. The mainstem Columbia river is host to numerous non-native fish (Figure 4.3.1), many of them piscivorous, that have acclimated to the lentic habitat of the reservoirs and now dominate many of the trophic pathways. Several mechanisms have been identified that allow introduced fish to succeed in displacing native species, including competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization (Moyle et al. 1986). Similarly, introductions of invertebrates can modify food webs. The introduction of opossum shrimp to Flathead Lake, Montana, resulted in the disappearance of two cladoceran species, which in turn had negative effects on the kokanee salmon that were intended to benefit from the introduction (Spencer et al. 1991).

#### 4.3.2 Competition

Competition among organisms occurs when two or more organismic units (i.e., individuals or species) use the same resources, and when availability of those resources is limited (Pianka 1978). Two types of competition are generally recognized: interference competition, where one organism directly prevents another from using a resource

through aggressive behavior, and exploitation competition, where one species affects another by utilizing a resource more efficiently (Moyle et al. 1986). Although competition is difficult to demonstrate (Fausch 1988), salmonids likely compete for food and space resources both within species (intraspecific) and between species (interspecific). Within species, stream-dwelling salmonids frequently form dominance hierarchies, with dominant individuals defending holding positions against subordinate fish through agonistic encounters. Evidence suggests that dominant individuals occupy the most energetically profitable holding positions, which in turn leads to higher growth rates (Fausch 1988). Similar interactions occur between salmonid species; however, in assemblages that have co-evolved, species with similar ecological requirements may segregate according to both micro- and macrohabitats at various life stages.

Changes in physical (e.g., temperature, streamflow, habitat structure) and biological (e.g., food availability, species composition) characteristics of streams and lakes can alter competitive interactions within and among species, potentially resulting in a restructuring of fish communities. In a laboratory study, Reeves et al. (1987) found that stream temperature affected interspecific interactions between juvenile steelhead trout and redbside shiner (*Richardsonius balteatus*), with trout competing more effectively at cold temperatures through interference, and shiner competing more successfully at warm temperatures through both exploitation and interference. Cunjak and Green (1986) found that interactions between brook trout and rainbow trout are also influenced by water temperature, rainbow trout being superior competitors at 16°C and brook trout at 9°C. Ratliff (1992) suggests that the decline of bull trout populations in Oregon may in part reflect the inferior ability of bull trout to compete with rainbow, brook, and brown trout at warmer temperatures.

Changes in streamflow in the Columbia River system have resulted in increased plankton production, which have apparently increased the success of American shad. Palmisano et al. (1993a; 1993b) conclude that increased numbers of shad, which also feed on benthic invertebrates, may be competing with juvenile salmonids. Cunjak and Green (1984) reported that brook trout tended to dominate social interactions with rainbow trout when in pool habitats, but not in faster waters. Larson et al. (1995) suggest that the dynamics of brook trout and rainbow trout interactions in a southern Appalachian stream may be affected by both temperature and flow conditions. During years of low discharge, rainbow trout encroached on upstream habitats of brook trout, possibly because warmer temperatures favored rainbow trout. During periods of higher discharge, encroachment was reversed, presumably because brook trout are more well-adapted to the steep stair-stepped channel morphology. In general, decreases in streamflow decrease available habitat, and may thereby intensify inter- and intraspecific competition for suitable rearing, feeding, spawning, and refuge habitats.

The introduction of non-native species increases the potential for competition in aquatic systems. In natural fish assemblages, salmonids have presumably adapted to other native species with similar ecological requirements through resource partitioning or segregation in time or space. However, with the introduction of non-native species, there has been no opportunity for natural selection to ameliorate competition (Fausch 1988). Several studies have documented influences of introduced species on native salmonids. In a British Columbia lake, cutthroat trout were found to shift from midwater areas when allopatric to littoral zones when sympatric with rainbow trout (Nilsson and Northcote 1981). Dambacher et al. (1992) found that introduced brook trout outcompeted bull trout in Sun Creek, Oregon, in areas of co-occurrence. Intraspecific interactions may also become more intense with the introduction of hatchery fish. Nickelson et al. (1986) concluded that competition between larger hatchery coho salmon and wild juveniles resulted in 44% replacement of the wild fish.

#### 4.3.3 Predation

Adult and juvenile salmonids have evolved strategies to coexist with numerous natural predators including a variety of fish, birds, and mammals. Native fish piscivores include sculpin (*Cottus* sp.), bull trout, rainbow trout, cutthroat trout, northern squawfish (*Ptychochilus oregonensis*), and possibly white sturgeon (*Acipenser transmontanus*). These fish prey on juvenile salmonids during instream rearing and during smolt migrations. Northern squawfish are considered important predators of outmigrant salmon and steelhead trout particularly in slackwater habitats (Poe et al. 1991). Bird predators of smolts and juveniles (Kaczynski and Palmisano 1993) include ring-billed gulls (*Larus delawarensis*), common mergansers (*Mergus merganser*), herons (*Ardea* spp.), and kingfishers (*Megaceryle alcyon*). Kingfishers were found to have increased feeding efficiency in slower moving waters. Pinnipeds, including harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and Stellar sea lions (*Eumetopia jubatus*) are the primary marine mammals preying on salmonids, although Pacific striped dolphin (*Lagenorhynchus obliquidens*) and killer whale (*Orcinus orca*) may also prey on adult salmonids. Seal and sea lion predation is primarily in saltwater and estuarine environments though they are known to travel well into the freshwater environment after migrating fish. All of these predators are opportunists, searching out locations where juveniles and adults are most vulnerable.

Habitat alterations can affect predation rates by: reducing cover, which increases vulnerability to capture by predators; altering flow regime and water velocity, which may favor certain piscivorous fishes; modifying temperature,

which affects the metabolism of piscivorous fish and the ability of fish to elude predators; and by obstructing passage, which may delay migrations and thereby increase exposure to predators. In the Columbia Basin altered flow regimes have contributed to the increased success of northern squawfish, walleye, and smallmouth bass, which tend to avoid areas of high water velocity (Faler, Miller et al. 1988). Laboratory experiments with squawfish have shown that maximum consumption of salmonid smolts increased from 0.5 smolts per day at 8.3 °C to 7 smolts per day at 21.7 °C (Vigg and Burley 1991), indicating that temperature increases may indirectly cause greater predation on juvenile salmonids (Palmisano et al. 1993b). The high incidence of predation by sea lions at such places as Ballard Locks in Washington is in part attributable to the unnatural congregations of fish as they attempt to pass through the locks.

#### 4.3.4 Disease and Parasitism

Salmonid fishes are affected by a variety of bacterial, viral, fungal, and microparasitic pathogens. In the Pacific Northwest, numerous diseases may result from pathogens that occur naturally in the wild or that may be transmitted to wild fish via infected hatchery fish. Among these are bacterial diseases, including bacterial kidney disease (BKD), columnaris, furunculosis, redmouth disease, and coldwater disease; virally induced diseases, including infectious hepatopoietic necrosis (IHN) and erythrocytic inclusion body syndrome (EIBS); protozoan-caused diseases, including ceratomyxosis and dermocystidium; and fungal infections, such as saprolegnia (SRSRT 1994). Brief descriptions of the most prevalent pathogens and the associated diseases are shown in Table 4.3.4.

Water temperature greatly influences the immune system of fishes, the number and virulence of pathogens, and in the case of microparasites, the occurrence of infective life stages in natural and aquacultural environments. Consequently, changes in water temperatures caused by forest and range practices, dams, and irrigation can alter the susceptibility of salmonids to infection by these pathogens. Most work on fish pathogens has concerned fish in culture situations and the incidence of disease and its role of fish population dynamics and in structuring fish assemblages in natural waters is poorly understood (Austin and Austin 1993). Nevertheless, laboratory studies indicate that water temperature has a direct effect on the of infection rate of most pathogens and the mortality rate of infected salmonids. With most pathogens, the susceptibility of salmonids to infection tends to increase with increasing water temperatures, although mortality from coldwater disease is greater when temperatures are lower (Holt et al. 1993). A summary of the general relationship between temperature and important pathogens in Pacific Northwest environments is shown in Table 4.3.4.

Several recent epizootics indicate that temperature may play a significant role in mediating disease in natural populations. Prespawning mortality in fall chinook salmon was highly correlated with mean maximum stream temperatures in the Rogue River (Oregon) during August and September, with mortality rates increasing abruptly at temperatures greater than 20 °C (ODFW 1992). *Flexibacter columnaris* was commonly found in dead and dying fish and was presumed to be the primary agent causing mortality. Release of heated reservoir water during the late summer and early fall has been implicated in outbreaks of *Dermocystidium salmonis* in anadromous fish in the lower Elwha River, Washington (USDI et al. 1994).

While epizootics provide the most dramatic examples of the potential for pathogens to affect salmonid populations, sublethal chronic infections can impair the ability of fish to perform in the wild and thereby contribute secondarily to mortality or reduced reproductive success. Fish weakened by disease are more sensitive to other environmental stresses. Furthermore, infected fish may become more vulnerable to predation (Hoffman and Bauer 1971), or less able to compete with other species. For example, Reeves et al. (1987) found that the interspecific interactions between juvenile steelhead trout and redbreast shiner (*Richardsonius balteatus*) were affected by water temperature. They speculated that these differences were in part because most juvenile steelhead were infected with *F. columnaris* at high temperatures, whereas shiners showed a higher incidence of infection at lower temperatures.

The susceptibility of salmonids to disease may be affected by other stressors, including dissolved oxygen, chemical pollution, and population density. Temperature may interact synergistically with these factors, causing disease to appear in organisms that might be resistant in the absence of other forms of stress. Susceptibility also varies among salmonid species and life stages. For example, older chinook have been shown to be more resistant to *F. columnaris* than younger fish (Becker and Fujihara 1978).

#### 4.4 Connectivity Among Processes

The biotic communities found in streams and rivers reflect physical and chemical gradients that occur both across the landscape, and along a stream from the headwaters to the ocean. In the preceding sections, we have reviewed fundamental biological processes that occur at the level of organisms, populations and communities, and the relationship between these processes and habitat characteristics that are affected by human activities. In this section, we discuss two concepts, the river continuum concept and the ecoregion concept, that address spatial relationships between these physico-chemical and biological processes. The river continuum concept (Vannote et al. 1980), focuses on interrelationships between physical and biological processes along streams from their headwaters to the ocean. The

ecoregion concept, relates regional patterns in physical and chemical gradients to the biological communities contained therein.

#### 4.4.1 River Continuum Concept

The river continuum concept (Vannote et al. 1980) proposes that the physico-chemical variables (e.g. light, nutrients, organic materials) within a river system change in a systematic way as a stream flows from headwaters to larger river systems to the ocean, and that the biological communities found along this gradient change accordingly (Figure 4.4.1a). In forested headwater reaches, energy inputs are dominated by coarse allochthonous materials, particularly leaf litter from riparian vegetation. As streams increase in size, canopy cover becomes less complete and more light reaches the stream; consequently, the contribution of instream primary production from algae and macrophytes increases relative to energy derived from allochthonous materials. In still-larger systems, fine particulate material transported from upstream areas forms the dominant energy source, particularly where depth and turbidity limit algal growth. These gradients lead to corresponding changes in the biological communities that use these changing energy sources. Invertebrate communities shift from those dominated by shredders and collectors in small streams, to collectors and grazers in mid-order streams, to mostly collectors in large rivers. Fish assemblages shift from invertivores in headwater reaches, to piscivores and invertivores in mid-order reaches, and include some planktivores in larger rivers.

Although the River Continuum Concept was developed in forested biomes, it can also be applied more generally. Meehan (1991) suggests that meadows and deserts, which lack shading and have reduced allochthonous inputs, obtain most of their energy from autochthonous sources, in contrast to woodland streams which have stronger terrestrial influences and therefore greater quantities of coarse particulate detritus. They conclude that desert streams are more similar to the downstream reaches of forested streams. Minshall et al. (1985) illustrate this conceptually by proposing a sliding scale to indicate that streams enter the continuum at different points. Similarly, primary production by algae may be high in headwater streams of alpine systems, where riparian inputs are comparatively low. Consequently these systems may have a different sequence in the biological communities along the continuum.

#### 4.4.2 Ecoregions

Ecological processes that influence salmonids and other aquatic species in the Pacific Northwest vary greatly across the landscape because of the high diversity of climate, topography, geology, vegetation, and soils. Ecoregion classifications represent attempts to identify areas of relative homogeneity in ecological systems or in the relationships between organisms and their environments (Omernik and Gallant 1986). Several federal agencies, including the Environmental Protection Agency (EPA), (Omernik and Gallant 1986; Omernik 1987), the U.S. Forest Service (Bailey 1976), and the Soil Conservation Service (Norris et al. 1991), have developed or are in the process of developing ecoregion classifications in order to address spatial issues in the management of natural resources. Landscapes, water bodies, and the biota that they support are expected to be similar within an ecoregion and to differ between ecoregions. We believe some form of ecoregion classification will be essential to defining the natural range of physical, chemical, and biological characteristics of salmonid habitats across the landscape.

The various ecoregion delineation processes differ. Omernik and his colleagues synthesized a number of factors (climate, geology, topography, soil, vegetation, land cover) to assess patterns at multiple spatial scales. Bailey considered many of the same factors, but used only one at any single scale of resolution. For example, his first divisions were by climatic patterns and his last were by vegetation. The Soil Conservation Service, as might be expected, focused on soil and agricultural land uses. Omernik's approach is favored by many state water quality agencies because of its ability to assess patterns at multiple scales and its adaptability, and it has been recommended by other scientific organizations (Science Advisory Board 1991; NRC 1992).

Although there are serious limitations to the application of Omernik's ecoregions at the site or small catchment scales, they are useful for stratifying the regional variability of the Pacific Northwest (Table 4.4.2) into relatively homogenous units. In addition, ecoregions offer a framework for aggregating and extrapolating data collected at the local level. A regional perspective is also essential for managing widely distributed resources, such as Pacific salmonids, because of the natural variability among sites and the human tendency to focus on local issues while losing sight of regional ones. In addition, subregions can be developed in a hierarchical manner to facilitate more precise landscape classification at local scales (Clarke et al. 1991; Thiele and Clarke In press). Direct applications of ecoregion concepts to aquatic ecosystems have demonstrated the utility of this approach. Whittier et al., (1988) showed that fish assemblages in rivers and small streams exhibited patterns concordant with Omernik's ecoregions in Oregon. In evaluating a number of different data sets from basin to state scales, Hughes et al. (1994) found that: ecoregions that differed markedly supported dissimilar fish assemblages, similar ecoregions supported more similar fish assemblages, and within-region variation was less than among-region variation.

#### 4.5 Summary

In the preceding sections, we have discussed biological processes at three levels of biological organization: organisms, populations, and communities. Grouping processes into these discrete categories serves to simplify thinking about the effects of environmental perturbations on salmonids and their ecosystems, but it should be reiterated that salmonids are simultaneously affected by processes occurring at all levels of biological organization. Physiological stresses influence the ability of salmonids to acquire food and defend space from competitors, to escape or avoid predators, and to fend off infectious diseases and parasites, all of which affect community structure. Populations have evolved specific mechanisms for coping with environmental conditions in their natal and rearing streams. These adaptations include morphological, biochemical, physiological, behavioral, and developmental traits that allow fish to survive and thrive with the specific physical, chemical, or biological constraints imposed by the environment, and that ensure specific activities (e.g., timing of migration and emergence) coincide with favorable environmental and ecological conditions. Adaptation is also evident in life-history strategies (e.g., fecundity and straying rates) that accommodate natural disturbance regimes and allow populations to persist over evolutionary time. Unlike the biological diversity of fishes in the Mississippi Basin, which centers on species diversity, the fish diversity in the Pacific Northwest centers on stock and life-history diversity. The evolution of a wide variety of life-history strategies has allowed salmonids to invade and thrive in the diverse habitats of the Pacific Northwest. The linkage between biological communities and the physical and chemical characteristics of streams are illustrated through the River Continuum and Ecoregion concepts, which offer means for assessing patterns in aquatic community structure across the landscape, and for predicting the response of aquatic ecosystems to anthropogenic disturbance. These concepts are essential in developing site-specific and region-specific salmonid conservation strategies and goals.

Table 4.2.2.1a. Life Histories of Pacific Salmonids (fw = freshwater)

| Species                          | Spawning Migration        | Spawning Period             | Spawning Area           | Life History   | Most Common Age at Maturity <sup>†</sup>         |
|----------------------------------|---------------------------|-----------------------------|-------------------------|--|--|
| <u>Anadromous Salmon</u>         |                           |                             |                         |  |  |
| Chum Salmon                      | Summer to Winter          | Summer to Winter            | Usually near Tidewater  | Fry go directly to sea. Adults age 2-5 yrs.  | 4 <sub>1</sub>                                   |
| Pink Salmon                      | Late summer to early Fall | Late Summer to early Fall   | Usually near Tidewater  | Fry go directly to sea. Adults all return at 2 yrs.  | 2 <sub>1</sub>                                   |
| Sockeye Salmon                   | Spring to Fall            | Late Summer to Fall         | Tributaries of Lakes    | 1-3 yrs in lake<br>2-3 yrs in ocean  | 4 <sub>2</sub><br>5 <sub>3</sub>                 |
| Coho Salmon                      | Summer to Fall            | Fall to early Winter        | Small Headwater Streams | 1-3 yrs fw<br>6 mos Jack<br>18 mos adult } Ocean   | 3 <sub>2</sub>                                   |
| Chinook Salmon                   | Spring to Fall            | Summer to early Winter      | Large Rivers            | 3 mos-2 yrs fw<br>2-7 yrs total age  | 4 <sub>1</sub> (Ocean)<br>5 <sub>2</sub> (River) |
| <u>Anadromous Trout and Char</u> |                           |                             |                         |  |  |
| Rainbow (Steelhead trout)        | Summer to Winter          | Late Winter to Spring       | Small Headwater Streams | Summer-run fish return to fw in summer and spawn in early spring<br>Winter-run fish enter fw in fall to winter and spawn in spring, All may survive to spawn again | 1-4 yrs fw<br>1-4 yrs in ocean                   |
| Cutthroat (Searun)               | Fall to Winter            | Late Winter to early Spring | Small Headwater Streams | Fw 2-4 yrs, migrate to ocean for spring and summer. Adults 2 yrs and older   | Mature sometime after 2nd year                   |

Table 4.2.2.1a. Life Histories of Pacific Salmonids (fw = freshwater)

| Species                 | Spawning Migration  | Spawning Period        | Spawning Area                                | Life History  | Most Common Age at Maturity <sup>†</sup> |
|-------------------------|---------------------|------------------------|--|---|--|
| Dolly Varden            | Late Summer to Fall | Fall                   | Main Channels on Rivers                      | 2-4 yrs in fw<br>2-4 yrs saltwater  | mature at 5-6 years, die at 6-7 years    |
| <u>Resident Species</u> |                     |                        |  |   |  |
| Kokanee salmon          | Late Summer to Fall | Late Summer to Fall    | Tributaries of Lakes                         | Fw only, Dies after spawning  | 3-4 yrs                                  |
| Rainbow trout           | Spring              | Spring                 | Small Headwater Streams                      | Fw only   | 2-3 yrs                                  |
| Cutthroat trout         | Spring              | Spring to Early Summer | Small Headwater Streams                      | Adfluvial, lacustrine-adfluvial, varied   | 25-50 cm, size dependent                 |
| Bull trout              | Fall                | Fall                   | Large streams with ground water infiltration | Juveniles migrate from tributaries to lakes or larger stream - 2 yrs; highly variable               | 4-9 yrs                                  |
| Mountain whitefish      | Fall                | Fall                   | midsized streams                             | Forms occur in streams and lakes. Mass spawning over gravels in streams or lakes. No redd is built. | 3-4 yrs                                  |

Data from Groot and Margolis (1991); Meehan and Bjornn (1991); Pratt (1992); and Moyle (1976).

<sup>†</sup> Gilbert-Rich Age Designation

Table 4.2.2.1b. Variation in Life Histories of Salmonids

| Species/<br>race                | Life<br>History* | Reproduces in: |         | Rears in: |         |           |       |
|---------------------------------|------------------|----------------|---------|-----------|---------|-----------|-------|
|                                 |                  | Lakes          | Streams | Lakes     | Streams | Estuaries | Ocean |
| Pink<br>Salmon                  | Anadromous       |                | X       |           | X       | X         | X     |
|                                 | Anadromous       |                | X       |           |         |           | X     |
| Chum<br>Salmon                  | Anadromous       |                | X       |           | X       | X         | X     |
|                                 | Anadromous       |                | X       |           | X       |           | X     |
|                                 | Anadromous       |                | X       |           |         |           | X     |
| Coho<br>Salmon                  | Anadromous       |                | X       |           | X       | X         | X     |
|                                 | Anadromous       |                | X       |           | X       |           | X     |
| Sockeye<br>Salmon               | Anadromous       |                | X       | X         |         |           | X     |
|                                 | Anadromous       | X              |         | X         |         |           | X     |
| Sockeye<br>Salmon<br>(kokanee)  | Resident         |                | X       | X         |         |           |       |
| Chinook<br>Salmon<br>(spring)   | Anadromous       |                | X       |           | X       | X         | X     |
|                                 | Anadromous       |                | X       |           | X       |           | X     |
| Chinook<br>Salmon (fall)        | Anadromous       |                | X       |           | X       | X         | X     |
|                                 | Anadromous       |                | X       |           |         |           |       |
| Pygmy<br>whitefish              | Resident         | X              |         | X         |         |           |       |
| Mountain<br>whitefish           | Resident         |                | X       |           | X       |           |       |
| Golden<br>trout                 | Resident         |                | X       | X         | X       |           |       |
|                                 | Resident         |                | X       |           |         |           |       |
| Cutthroat<br>trout              | Resident         |                | X       | X         | X       |           |       |
|                                 | Resident         |                | X       |           |         |           |       |
| Cutthroat<br>trout<br>(searun)  | Anadromous       |                | X       |           | X       | X         | X     |
|                                 | Anadromous       |                | X       |           | X       |           | X     |
| Rainbow<br>trout                | Resident         |                | X       | X         | X       |           |       |
|                                 | Resident         |                | X       |           |         |           |       |
| Rainbow<br>trout<br>(steelhead) | Anadromous       |                | X       |           | X       |           | X     |
| Brown trout                     | Resident         |                | X       |           | X       |           |       |
|                                 | Resident         |                | X       | X         |         |           |       |
| Bull trout                      | Resident         |                | X       |           | X       |           |       |
|                                 | Resident         |                | X       | X         |         |           |       |
| Brook trout                     | Resident         | X              |         | X         |         |           |       |

Table 4.2.2.1b. Variation in Life Histories of Salmonids

| Species/<br>race   | Life<br>History* | Reproduces in: |         | Rears in: |         |           |       |
|--------------------|------------------|----------------|---------|-----------|---------|-----------|-------|
|                    |                  | Lakes          | Streams | Lakes     | Streams | Estuaries | Ocean |
|                    | Resident         |                | X       |           | X       |           |       |
|                    | Resident         |                | X       | X         |         |           |       |
| Dolly Varden       | Anadromous       |                | X       |           | X       | X         | X     |
|                    | Anadromous       |                | X       |           | X       |           | X     |
|                    | Anadromous       |                | X       | X         |         |           | X     |
| Lake trout         | Resident         | X              |         | X         |         |           |       |
| Arctic<br>grayling | Resident         |                | X       |           | X       |           |       |
|                    | Resident         |                | X       | X         |         |           |       |

\*Some species have several races with different life history patterns. From Everest et al. (1985).

Table 4.2.2.2a Seasonal Occurrence of Adult and Juvenile (Embryos in Gravel and Young) Anadromous Salmonids in Freshwaters of Western Oregon and Washington\*

| Species                 | Life-stage | Months |      |      |      |     |      |      |      |       |      |      |      |
|-------------------------|------------|--------|------|------|------|-----|------|------|------|-------|------|------|------|
|                         |            | Jan.   | Feb. | Mar. | Apr. | May | June | July | Aug. | Sept. | Oct. | Nov. | Dec. |
| Pink salmon             | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Chum salmon             | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Coho salmon             | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Sockeye salmon          | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Spring chinook          | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Fall chinook salmon     | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Sea run cutthroat trout | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Winter steelhead trout  | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Summer steelhead trout  | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |
| Dolly Varden            | Adult      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Young      |        |      |      |      |     |      |      |      |       |      |      |      |
|                         | Eggs       |        |      |      |      |     |      |      |      |       |      |      |      |

\*From (Everest, F. H., Armantrout et al. 1985).

Table 4.3.2a. Examples of Local Variation in Traits of Salmonid Fishes and Their Presumed Adaptive Advantage (Data from a review by Taylor (1991). (E = embryo/alevin, F = fry, J = juvenile, S = smolt, A = adult)

| Trait                               | Adaptive Advantage   | Species  | Source  |
|-------------------------------------|--|--|---|
| <b>MORPHOLOGY AND MERISTICS</b>     |  |  |   |
| Streamlined body and larger fins    | Streamlined body and larger fin size adapted for higher water velocities.                                | Coho Salmon (J)<br>Atlantic Salmon (J)<br>Pink Salmon (A)<br><br>Chum Salmon (A) | Taylor & McPhail 1985<br>Riddell & Leggett 1981<br>Beacham 1985, Beacham, et al. 1988b<br>Beacham 1984, Beacham & Murray 1987, Beacham et al. 1988a |
| Jaw size and shape                  | Adapted to local differences in food particle size.  | Arctic Char  | Barbour 1984<br>Skulason, et al. 1984   |
| Gill raker number and length        | Adapted to local differences in food particle size.  | Lake Whitefish   | Ihssen et al. 1981<br>Lindsay 1981  |
| <b>BEHAVIOR</b>                     |  |  |   |
| Direction of fry migration          | Emerging fry migrate in direction of rearing lakes.  | Sockeye Salmon (F)<br><br>Rainbow Trout (F)                                      | Brannon 1972, Raleigh 1971<br><br>Raleigh 1971, Kelsø et al. 1981   |
| Compass orientation of emerging fry | Local differences in orientation facilitate migration to feeding areas.                                  | Sockeye Salmon (F)   | Quinn 1982, 1985  |
| Rheotactic behavior                 | Adapted to local differences in optimal timing of downstream migration.                                  | Chinook Salmon (S)   | Taylor 1990a  |
| Homing accuracy                     | Differences in straying rates potentially reflect differences in environmental stability.                | Pink Salmon (A)  | Bums 1976   |
| Migratory behavior                  | Adapted to local differences in predation pressure, local feeding areas, and hydrologic characteristics. | Brown trout (A)  | Svardson & Fagerstron 1982  |

Table 4.3.2a. Examples of Local Variation in Traits of Salmonid Fishes and Their Presumed Adaptive Advantage (Data from a review by Taylor (1991). (E = embryo/alevin, F = fry, J = juvenile, S = smolt, A = adult)

| Trait                        | Adaptive Advantage   | Species   | Source   |
|------------------------------|--|---|--|
| <b>BEHAVIOR (Continued)</b>  |  |   |  |
| Migration timing             | Differences between fall and spring races reflect seasonal variation in accessibility to spawning streams.   | Chinook Salmon (A)  | Belding & Kitson 1934<br>Smith 1969  |
| Spawning timing              | Differences in spawning timing reflect temperature differences in streams.   | Pink Salmon (A)<br>Chinook Salmon (A)<br>Sockeye Salmon (A) | Sheridan 1962a, Royce 1962<br>Burgin, et al. 1985<br>Brannon 1987            |
| Agonistic behavior           | Reduced level of agonistic behavior in "ocean type" juveniles compared with "stream types" that establish territories.   | Chinook Salmon (J)  | Taylor 1988, 1990a   |
|                              | Lower levels of agonistic behavior in populations with high predation; displays may increase risk.   | Coho Salmon (J)   | Rosenav & McPhail 1987   |
| Agonistic behavior           | Lower levels of agonistic behavior for fish in lakes or other slowmoving habitats. Higher levels in streams dwelling fish, where territorial defense is advantageous.          | Coho Salmon (J)   | Grant & Noakes 1988<br>Swain & Holthy 1989                                   |
| <b>DEVELOPMENTAL BIOLOGY</b> |  |   |  |
| Embryo/Alevin survival       | Survival of embryos/alevins from populations native to cold water environments greater at low temperatures than for populations from warm water environments (and vice versa). | Chum Salmon (E)<br>Sockeye Salmon (E)<br>Pink Salmon (E)    | Tallman 1986<br>Beacham & Murray 1989<br>Beacham 1988, Beacham & Murray 1988 |

Table 4.3.2a. Examples of Local Variation in Traits of Salmonid Fishes and Their Presumed Adaptive Advantage (Data from a review by Taylor (1991). (E = embryo/alevin, F = fry, J = juvenile, S = smolt, A = adult)

| Trait                                       | Adaptive Advantage   | Species                        | Source                      |
|---|--|--------------------------------|-----------------------------|
| <b>DEVELOPMENTAL BIOLOGY</b><br>(Continued) |  |                                |                             |
| Developmental rate                          | Faster development in late spawning stocks may facilitate synchronous emergence with fry of early spawners. Synchrony adaptive for predator swamping or narrow window of favorable oceanic conditions. | Chum Salmon (E)                | Tallman 1986                |
| <b>BIOCHEMICAL TRAITS</b>                   |  |                                |                             |
| Lactate dehydrogenase                       | Temperature-dependent selection of certain allozymes that are more active at colder or warmer temperatures. Allozymes dominant in northern populations are more active in cold water.                  | Sockeye Salmon                 | Kirpichnikov & Ivanova 1977 |
| Esterase-2 locus                            |  | Pink Salmon                    | Kirpichnikov & Ivanova 1977 |
| Isocitrate dehydrogenase                    |  | Arctic Char                    | Nyman & Shaw 1971           |
| Malic enzyme-2 locus                        |  | Steelhead Trout                | Redding & Schreck           |
| Lactate dehydrogenase 5                     |  | Atlantic Salmon                | Verspoor & Jordan 1989      |
|   |  | Brown Trout                    | Henry & Ferguson 1985       |
| <b>PHYSIOLOGICAL TRAITS</b>                 |  |                                |                             |
| Swim bladder function                       | Greater swim bladder gas retention ability in fish inhabiting deeper lakes.  | Lake Trout                     | Inssen & Tait 1974          |
| Swimming ability                            | Superior prolonged swimming ability in stocks with long freshwater migrations.   | Steelhead Trout<br>Coho Salmon | Taylor & McPhail 1985       |
| Temperature tolerance                       | Resistance of fish naturalized to warm water environments and to high temperatures.  | Rainbow Trout                  | Morrissy 1973               |

Table 4.3.2a. Examples of Local Variation in Traits of Salmonid Fishes and Their Presumed Adaptive Advantage (Data from a review by Taylor (1991). (E = embryo/alevin, F = fry, J = juvenile, S = smolt, A = adult))

| Trait                                  | Adaptive Advantage   | Species  | Source   |
|--|--|--|--|
| Time to smolting                       | More rapid development adapted to streams with short growing seasons.  | Atlantic Salmon (S)  | Jensen & Johnsen 1986  |
| <b>DISEASE RESISTANCE</b>              |  |  |  |
| Resistance to <i>Ceratomyxa shasta</i> | Populations that have coevolved with <i>C. Shasta</i> have greater resistance than those that have not.                    | Chinook Salmon<br>Coho Salmon<br>Steelhead Trout   | Zinn et al. 1977<br>Hemmingsten et al. 1986<br>Buchanan et al. 1983  |
| <b>DISEASE RESISTANCE (CONTINUED)</b>  |  |  |  |
| Resistance to <i>Ceratomyxa shasta</i> | Populations that have coevolved with <i>C. Shasta</i> have greater resistance than those that have not.                    | Chinook Salmon<br>Coho Salmon<br>Steelhead Trout   | Zinn et al. 1977<br>Hemmingsten et al. 1986<br>Buchanan et al. 1983  |
| <b>LIFE HISTORY TRAITS</b>             |  |  |  |
| Large size                             | Larger size of adults adaptive in populations undertaking difficult migrations or experiencing high flows during spawning. | Chinook Salmon<br>Sockeye Salmon<br>Chum Salmon<br>Brown Trout<br>Atlantic Salmon<br>Pink Salmon | Taylor 1990b<br>Rogers 1987<br>Beacham & Murray 1987<br>L'Abée-Lund, et al. 1989<br>Power 1986<br>Rogers 1987, Beacham & Murray 1988 |
| Small Size                             | Adaptation to streams with low summer flows.   | Brown Trout (J)  | Borgstrom & Heggenes 1988<br>Titus & Mosegaard 1989  |
| Precocious males/parr maturation       | Increased incidence of precocious males or parr maturation may be adaptive in populations with long, difficult migrations. | Chinook Salmon<br>Atlantic Salmon  | Myers et al. 1986<br>Taylor 1989   |

Table 4.3.2a. Examples of Local Variation in Traits of Salmonid Fishes and Their Presumed Adaptive Advantage (Data from a review by Taylor (1991). (E = embryo/alevin, F = fry, J = juvenile, S = smolt, A = adult)

| Trait     | Adaptive Advantage   | Species            | Source   |
|-----------|--|--------------------|--|
| Egg size  | Late spawners tend to produce smaller faster-developing eggs than early spawners; facilitates synchronous emergence. | Chum Salmon        | Beacham & Murray 1987<br>Flemming & Cross 1990 |
| Fecundity | High fecundity favored in populations that experience high pre-reproductive mortality.                               | Chinook Salmon (A) | Healey & Heard 1984                            |

Table 4.3.4a Pathogens of Salmonids Found in Pacific Northwest Streams.

| Pathogen                         | Disease                       | Comments   |
|----------------------------------|-------------------------------|--|
| <u>Bacteria</u>                  |                               |  |
| <i>Aeromonas salmonicida</i>     | Furunculosis                  | Obligate pathogen of fish; Low mortality at temps < 6.7°C; Increasing mortality at 9.4°C; At 20.5°C, 93-100% mortality for all species. (Groberg et al. 1978)  |
| <i>Aeromonas hydrophila</i>      |                               | Stress facilitated infection; Mortality is associated with elevated water temperatures (>9.4°C), presence of pollutants (particularly nitrate at ≥6 mg/l), (Austin and Austin 1993)  |
| <i>Flexibacter columnaris</i>    | Columnaris                    | Low mortality at temperatures < 15°C; Increasing mortality at 20°C for all species. Virulence at low temperatures depends on specific "shair?"; Naturally occurring bacteria present at low levels in resident fish (suckers, carp and whitefish; Stress increases fish susceptibility (high density increases potential for contact (Inglis et al. 1993). |
| <i>Flexibacter psychrophilia</i> | Coldwater disease             | Appears in spring when temperatures are between 4-10°C; 30-50% mortality for infected alevins; Quickest mortality at 15°C; Mean time to death increases with temperatures from 15-23°C; Mode of transmission unknown; Resident salmonids are probable carriers; Possible vertical transmission. (Inglis et al. 1993)                                       |
| <i>Renibacterium salmoniarum</i> | Bacterial kidney disease, BKD | Obligate pathogen of fish. Disease progresses more rapidly at higher temperatures (15-20.5°C), but mortality may be highest at moderate temperatures (12°C); Transmission is both horizontal and vertical (intra-ovum); Crowding and diet stress can increase susceptibility. (Inglis et al. 1993, Fryer and Lannan 1993)                                  |
| <i>Yersinia ruckerii</i>         | Redmouth disease              | Mortality may be low in chronic infections or become much high with stress from poor water conditions (elevated temperatures, ammonia, metabolic waste, copper); Transmission through water, via baitfish, introduced fish, bird feces, fish farms. (Inglis et al. 1993)   |

Table 4.3.4a Pathogens of Salmonids Found in Pacific Northwest Streams.

| Pathogen                               | Disease          | Comments   |
|--|------------------|--|
| <u>Protozoans</u>                      |                  |  |
| <i>Ceratomyxa shasta</i>               | Ceratomyxosis    | Endemic to many river systems of Northwest. Temperature dependent; increasing mortality for fish exposed at temperatures $\geq 10^{\circ}\text{C}$ . High mortality for nonadapted (no genetic resistance) species and stocks. (Ratliff 1983).   |
| <i>Dermocystidium salmonis</i>         | -                | Pathogen of salmonids in Pacific Northwest. Horizontal transmission through water; Uptake is via gills. Epizootics appear to be temperature dependent. (Olson et al. 1991)   |
| <i>Ichthyobodo/Costia</i> sp           | -                | Ectoparasite affects osmoregulation. Juvenile salmonid mortality high (63-70% in 48 hr tests) upon introduction to marine waters (Urawa 1993)  |
| <i>Myxobolus cerebralis</i>            | Whirling disease | Salmonid infection by mature triactinomyxon via ingestion or through gills. Horizontal transmission via intermediate host is tubifex worm from soft mud habitats. Lethal to salmonids (Rich Holt, pers. comm. 1995).   |
| <u>Viruses</u>                         |                  |  |
| Infectious Hepatopoetic Necrosis Virus | IHNV             | Endemic to most areas. Mortality high for young fry. Most mortality occurs at temperatures of $12^{\circ}\text{C}$ or less; Some outbreaks at $15^{\circ}\text{C}$ ; At temperatures over $10^{\circ}\text{C}$ , disease produces less mortality but leads to more carriers of disease. (Wolf 1988). |
| Erythrocytic Inclusion Body            | EIB              | Potential vertical transmission and known horizontal transmission. Greatest mortality of salmonids found at $8\text{-}10^{\circ}\text{C}$ . (Takahashi et al 1992, Leek 1987)  |
| <u>Fungi</u>                           |                  |  |
| <i>Saprolegnia</i>                     | -                | Ubiquitous in water. Transmitted horizontally or from substratum to fish. Elevated temperatures increase growth rate. If untreated, progressive and terminal.  |

Table 4.4.2a Predominant Characteristics of Ecoregions in the Pacific Northwest

| Ecoregion  | Land Surface Form  | Potential Natural Vegetation  | Land Use   | Soils*   |
|--|--|---|--|--|
| Coast Range                                      | Low to high mountains  | Spruce/cedar/hemlock, cedar/hemlock/ Douglas-fir, redwood   | Forest and woodland mostly ungrazed  | Udic soils of high rainfall areas  |
| Puget Lowland                                    | Tablelands with moderate relief, plains with hills or mountains, or open hills                                 | Cedar/hemlock/ Douglas-fir  | Mosaic including forest, woodland, pasture, and cropland                       | Alfisols, Inceptisols, Mollisols, Spodosols, and Vertisols of valleys          |
| Willamette Valley                                | Plains with hills, or open hills   | Cedar/hemlock/ Douglas-fir, mosaic of Oregon oakwoods and cedar/hemlock/ Douglas-fir                                    | Emphasis on cropland with some interspersions of pasture, woodland, and forest | Xeric Mollisols, Vertisols, and Alfisols of interior valleys                   |
| Cascades   | High mountains   | Silver fir/Douglas-fir, fir/hemlock, western spruce/fir, Douglas-fir, cedar/hemlock/ Douglas-fir, spruce/cedar/ hemlock | Forest and woodland mostly ungrazed  | Udic soils of high rainfall mountains  |
| Sierra Nevada                                    | High mountains   | Mixed Conifer forest (fir, pine, Douglas-fir), red fir, lodgepole pine/subalpine forest (hemlock)                       | Forest and woodland grazed   | Xeric soils of moderate rainfall areas   |
| Southern and Central California Plains and Hills | Irregular plains, tablelands of moderate to considerable relief, low mountains                                 | California oakwoods, chaparral (manzanita, ceanothus), California steppe (needlegrass)                                  | Open woodland grazed   | Light-colored soils of subhumid regions  |
| Central California Valley                        | Flat plains  | California steppe (needlegrass), tule marshes (bulrush, cattails)   | Irrigated agriculture, cropland with grazing land                              | Recent Alluvial soils, light-colored soils of the wet and dry subhumid regions |
| Eastern Cascades Slopes and Foothills            | Varied: Tablelands with moderate to high relief, plains with low mountains; open low mountains, high mountains | Western ponderosa pine  | Forest and woodland grazed   | Xeric soils of moderate rainfall areas   |
| Northern Rockies                                 | High mountains   | Cedar/hemlock/pine, western spruce/fir, grand fir/Douglas-fir, Douglas-fir  | Forest and woodland mostly ungrazed  | Eastern interior mountain soils with acidic rock types, Inceptisols            |

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Table 4.4.2a Predominant Characteristics of Ecoregions in the Pacific Northwest

| Ecoregion                     | Land Surface Form   | Potential Natural Vegetation  | Land Use  | Soils*  |
|-------------------------------|---|---|---|---|
| Columbia Basin                | Varied: Irregular plains, tablelands with moderate to high relief, open hills (excludes extremes) | Wheatgrass/bluegrass, fescue/wheatgrass, sagebrush steppe (sagebrush, wheatgrass) | Mostly cropland, cropland with grazing land         | Xerolls, Channeled Scablands                                |
| Blue Mountains                | Low to high open mountains  | Grand fir/Douglas-fir, western ponderosa pine, western spruce/fir, Douglas-fir    | Forest and woodland grazed                          | Soils of eastern interior mountains, Mollisols, Inceptisols |
| Snake River Basin/High Desert | Tablelands with moderate to high relief, plains with hills or low mountains                       | Sagebrush steppe (sagebrush, wheatgrass), saltbush/greasewood                     | Desert shrubland grazed, some irrigated agriculture | Aridisols, aridic Mollisols                                 |

\*Soils are presented in this table as they appear from mapped units of resource soil maps. The table may differ somewhat from the text descriptions, as certain soil map units integrate soils of limited and therefore non-representative extent. Data from Omernik (1987).

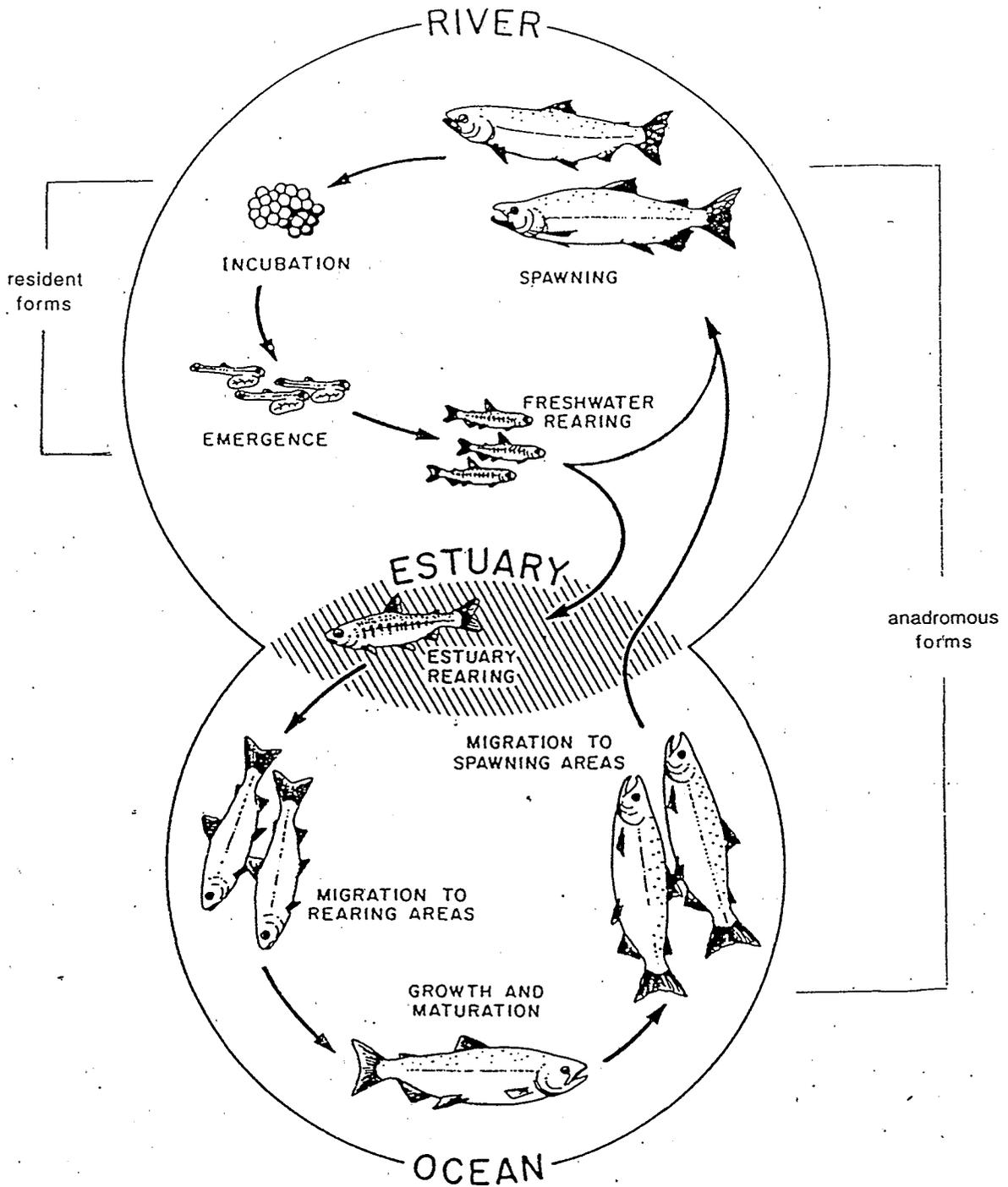


Figure 4.2.1a Generalized salmonid life cycle, showing freshwater and ocean components. Modified from Hankin et al. (1993).

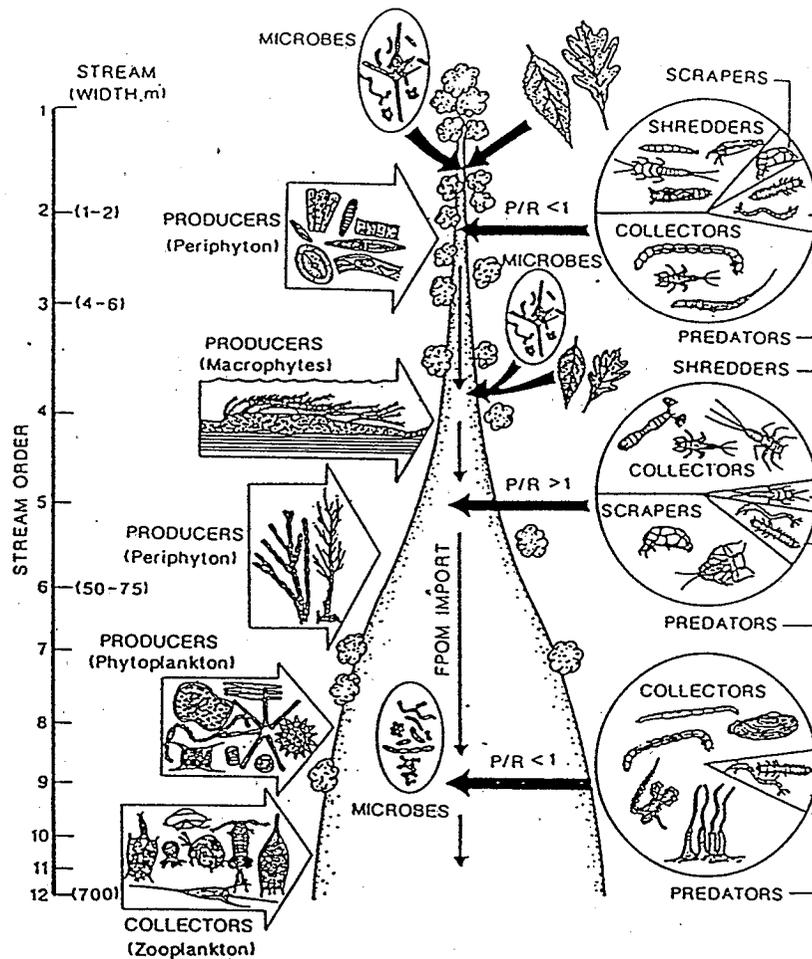


Figure 4.4.1a Trends in energy sources, ratios of autotrophic production to heterotrophic respiration, and functional groups along a river continuum.

## 5.0 Habitat Requirements of Salmonids

Karr et al. (1991) define biological integrity as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region." They further state that a biological system can be considered "ecologically healthy" when "its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed." Specific attributes of streams and lakes, such as streamflow, water temperature, substrate, cover, and dissolved materials—all the elements that typically are associated with the term habitat—are the result of physical, chemical, and biological processes operating throughout a watershed and across the landscape (see Sections 3 and 4). Protecting and restoring desirable habitat attributes of streams and lakes for salmonids requires that the natural processes that produce these characteristics be maintained or restored. In other words, if the processes are protected, desirable aquatic habitat characteristics will develop; if the processes are altered, the integrity of the aquatic ecosystem, and the ecosystems' ability to support salmonids, are diminished. These concepts are directly reflected in the Aquatic Conservation Strategy Objectives detailed by FEMAT (1993).

Four important considerations in assessing the habitat requirements of salmonids are that: 1) all watersheds and streams are different to some degree in terms of their temperature regimes, flow regimes, sedimentation rates, nutrient fluxes, physical structure, and biological components; 2) the fish populations that inhabit a particular body of water have adapted—biochemically, physiologically, morphologically, and behaviorally—to the natural environmental fluctuations that they experience and to the biota with which they share the stream, lake, or estuary; 3) the specific habitat requirements of salmonids differ among species and life-history types, and change with season, life stage, and the presence of other biota; and 4) aquatic ecosystems are changing over evolutionary time. From these general principles, it is obvious that there are no simple definitions of desirable habitat characteristics of salmonids. Defining acceptable or natural ranges of variability for specific habitat attributes is not only difficult, but can also be misleading as well. For example, the same total sediment yield in two different watersheds may affect salmonid habitats differently, depending on geology, topography, hydrology, stream size, and the abundance of large woody debris. Similarly, Behnke (1992) has suggested that stocks of trout native to warmer streams may exhibit greater tolerance to high temperature extremes than stocks inhabiting naturally cooler waters, thus, simply defining the range of temperatures at which a species has been observed does not ensure that stocks will be "safe" or healthy as long as temperatures remain in that range. The FEMAT (1993) report concluded that current scientific information is inadequate to allow definition of specific habitat requirements of salmonids throughout their life histories. Karr et al. (1991) identified five classes of environmental factors that affect the biotic integrity of aquatic ecosystems—food (energy) source, water quality, habitat structure, flow regimes, and biotic interactions—as well as ecological changes that may occur in response to human-induced alterations (Figure 5.0a). Although this model was intended to address all aquatic biota, the elements provide a useful framework for discussing salmonid habitat requirements. In Section 5.1, we use the model of Karr et al. (1991) to outline general habitat requirements of salmonids, focusing on processes and characteristics that must be maintained in order to ensure the ecological health of aquatic ecosystems. An extended discussion of water quality concerns is presented in this section because most water quality standards are intended to protect ecological function rather than specific-species or life stages of fish. In Section 5.2, we review specific habitat requirements of the Pacific salmonids at each life stage: adult migration, spawning and incubation, rearing, and juvenile migration.

### 5.1 General Habitat Requirements

Everest et al. (1985) note that although each species of anadromous salmonid differs somewhat in its specific habitat requirements, all share some common habitat needs. A slightly modified list of these characteristics that includes requirements for resident species is as follows:

- cool, flowing waters
- free migratory access to and from spawning and rearing areas
- clean gravel substrate for reproduction
- water of low sediment content during the growing season (for visual feeding)
- high dissolved oxygen concentrations in rearing and incubation habitats
- sufficient cover
- invertebrate organisms for food

#### 5.1.1 Food (Energy) Source

As discussed in Section 4.1, salmonids require sufficient energy to meet their basic metabolic needs, to grow, and to reproduce. Maintaining the integrity of aquatic ecosystems depends on maintaining the natural spatial and temporal patterns and amount of primary production. In streams where energy inputs are dominated by allochthonous

materials detrital particles are generally larger than in streams where autochthonous production dominates. In addition, in streams with an intact riparian canopy, the timing and type of material delivered to the channel differs between coniferous and deciduous forest. Together, these factors determine the abundance and species composition of aquatic invertebrates, which are the principal food source for most salmonids. Removal of riparian vegetation in smaller streams changes the dominant energy inputs from allochthonous to autochthonous sources. The conversion of riparian vegetation from conifer-dominated communities to deciduous-dominated communities, or from shrub-dominated to grass-dominated communities, alters the type of food energy available to the system, the temporal patterns of allochthonous inputs, and the invertebrate communities that feed on those resources. Although not all of these changes are necessarily detrimental to salmonids, they represent fundamental changes to ecosystem function.

Characteristics of the physical and chemical environment, including temperature, streamflow, turbidity, nutrient availability, and physical structure all influence the composition and abundance of invertebrate communities within streams, lakes, and estuaries, as well as the ability of salmonids to obtain these food resources. Thus physical and chemical processes must be maintained to ensure that food resources remain within the natural range of abundance for the particular site.

### 5.1.2 Water Quality

Water temperature, turbidity, dissolved gases (e.g., nitrogen and oxygen), nutrients, heavy metals, inorganic and organic chemicals, and pH all influence water quality and ability of surface waters to sustain fish populations. With the exception of organic and inorganic chemicals of anthropogenic origin, each of these factors is naturally occurring and exhibits daily or seasonal fluctuations in concentration or magnitude. If the magnitude or concentration of any of these factors exceeds the natural range for a specific location and time of year, biological processes are altered or impaired.

#### 5.1.2.1 Temperature

Perhaps no other environmental factor has a more pervasive influence on salmonids and other aquatic biota than temperature. The vast majority of aquatic organisms are poikilothermic—their body temperatures and hence their metabolic demands are determined by temperature (see Section 4.1.1). Consequently, virtually all biological and ecological processes are affected by ambient water temperature. Many of the effects of temperature on these processes have been discussed elsewhere in this document. Below is a brief list of some of the more important physiological and ecological processes affected by temperature, with reference to sections of this document where more detailed discussions may be found:

- Decomposition rate of organic materials
- Metabolism of aquatic organism, including fishes (Section 4.1.1)
- Food requirements, appetite, and digestion rates of fishes (Section 4.1.2)
- Developmental rates of embryos and alevins (Section 4.1.4)
- Growth rates of fish (4.1.1)
- Timing of life-history events, including adult migrations, fry emergence, and smoltification (Sections 4.1.4, 4.1.5)
- Competitor and predator-prey interactions (4.3.2, 4.3.3)
- Disease-host and parasite-host relationships (4.3.4)

From this list, it is evident that protection and restoration of salmonid habitats requires that temperatures in streams and lakes remains within the natural range for the particular site and season.

Most of the literature on salmonid temperature requirements refers to "preferred", "optimal", or "tolerable" temperatures or temperature ranges (Everest et al. 1985; Bell 1986; Bjornn and Reiser 1991). Preferred or optimal temperatures are generally derived in laboratory studies of behavior (e.g. temperature selection) or performance (e.g. growth, survival, metabolic scope). In general, the term "preferred temperature" is used to describe the temperature to which, given unlimited acclimation time, a fish will ultimately gravitate towards (Fry 1947). The "optimum temperature" is taken to mean the temperature at which a fish can best perform a specific activity. The "tolerable temperature" range includes temperatures at which fish can survive indefinitely. Although studies of temperature preferences, optima, and tolerances are useful in establishing general physiological requirements, they do not address the ecological requirements of salmonids or local adaptation to specific thermal regimes. For this reason, water quality criteria that are designed to prevent temperatures from exceeding physiologically stressful levels alone are unlikely to prevent more subtle ecological changes.

### 5.1.2.2 Turbidity and Suspended Solids

Turbidity in streams is caused by phytoplankton and by inorganic and organic materials that become suspended during high flow conditions. Inorganic and organic solids enter the aquatic environment in surface runoff, or as particles derived from erosion associated with natural (e.g., slumping of unstable banks, storm runoff, volcanoes) or anthropogenic activities (e.g., forestry, grazing, mining, and agricultural practices) (Leidy 1980; Stumm and Morgan 1981; Dickson et al; Maki et al. 1987; Adriano 1992; Hem 1992).

Turbidity and suspended solids in surface waters can effect algae and phytoplankton by reducing light transmission and causing physical damage through abrasion and scouring (Chandler 1942; Chapman 1963; Bullard 1965; Cairns et al. 1972). A number of studies have indicated that turbidity is a major factor to phytoplankton abundance (Buck 1956; Cordone and Pennoyer 1960; Herbert et al. 1961; Benson and Cowell 1967; Sherk et al. 1976). In addition, diminished light penetration and streambed stability can lead to reductions in algal productivity (Samsel 1973), and changes in plant species composition. Samsel (1973) found that a reduction of transparency of about 50 percent caused a threefold reduction in algal productivity in a Virginia impoundment. Chapman (1963) noted that moving sediment may grind or dislodge algae. Shifting of deposited sand (0.008–0.015 in.) prohibited establishment of periphyton along an English riverbed (Nuttall 1972).

Siltation reduces the diversity of invertebrates and aquatic insects, by reducing interstices in the substrate. When fine sediment is deposited on gravel, species diversity and densities drop significantly (Cordone and Pennoyer 1960; Herbert et al. 1961; Bullard 1965; Reed and Elliott 1972; Nuttall and Bielby 1973; Bjornn et al. 1974; Cedarholm et al. 1978). Deposited sediment may reduce accessibility to microhabitats by embedding the edges of cobbles (Brusven and Prather 1974), and it may also entomb benthic organisms which then die of oxygen depletion (Ellis 1931). Suspended sediments also limit insect populations and benthic organisms (Tarzwell 1938; Rees 1959; Branson and Batch 1971). In a 10-year stream survey Roback (1962) found numbers of caddis fly larvae genera decreased from 16–7 at sediment concentrations in excess of 500 ppm. Addition of more than 80 ppm of inert solids to the normal suspended particle concentration of 40 ppm caused a 60 percent reduction in population of riffle macroinvertebrates (Gammon 1970). Estuarine copepods ingested fewer food organisms as silt concentration increased (Sherk et al. 1976).

Siltation and turbidity adversely affect fish at every stage of their life cycle (Iwamoto et al. 1978). In general, deposited sediments have a greater impact on fish than do suspended sediments, with spawning and incubation habitats most directly affected (see Section 5.2.2.3). Particulate materials physically abrade and mechanically disrupt respiratory structures (e.g., fish gills) or surfaces (e.g., respiratory epithelia of benthic macroinvertebrates) in aquatic vertebrates and invertebrates (Rand and Petrocelli 1985). Sediment covers intergravel crevices which fish use for shelter, thereby decreasing the carrying capacity of streams for young salmon and trout (Cordone and Kelley 1961; Bjornn et al. 1974). Fish vacate pools in summer after heavy accumulation of sediments (Gammon 1970). Finally, turbidity affects light penetration, which in turn affects the reactive distance of juvenile and adult salmonids for food capture (see Section 5.2.2.4).

Although salmonids typically prefer water with low turbidity and suspended sediment content, low levels of turbidity may have beneficial effects. Particulates and dissolved chemical solids, including materials harmful to salmonids, may adsorb to the surfaces of colloidal materials, which in turn can reduce their bioavailability. Thus adverse effects potentially associated with exposures to inorganic and organic chemicals may be diminished, and biological processes associated with adsorption of dissolved organic solids (e.g., microbial transformation) may enhance the biodegradation and detoxification of organic chemicals in the water (Rand and Petrocelli 1985; Dickson et al. 1987; Adriano 1992; Hem 1992). While adsorption associated with colloids may attenuate adverse biological effects associated with some chemicals, toxicity of other dissolved chemical solids may increase because of interactions with colloidal materials in the water column. The exposure of fish to heavy metals may increase or the solubilization of heavy metals from otherwise insoluble metal compounds may increase in the presence of suspended solids having a high colloidal content (Leidy 1980; Rand and Petrocelli 1985; Brown and Sadler 1989).

### 5.1.2.3 Dissolved Oxygen and Nitrogen

In general, all salmonids require high levels of dissolved oxygen. Reduced levels of oxygen can affect the growth and development of embryos and alevins, the growth of fry, and the swimming ability of adult and juvenile migrants. In most natural situations, DO levels are sufficient to allow normal function, but concentrations in small streams may be reduced by large amounts of organic debris, nutrient enrichment from sewage treatment plants and agricultural runoff, and excessively high temperatures. Bjornn and Reiser (1991) reviewed a number of papers and concluded that while thresholds for survival are generally low (3.3 mg/l), growth and food conversion efficiency are affected at DO levels of 5 mg/L, and that DO levels of 8-9 mg/L or more are needed to ensure that normal physiological functions of salmonids are not impaired. The EPA's water quality criteria for dissolved oxygen are 9.5 mg/L for a 7 day mean, and 8.0 mg/L for a one day minimum (EPA 1986). Supersaturation of oxygen gas may occur associated with

spills from dams or highly turbulent waters. The EPA standards for maximum levels of oxygen is 110 % of normal saturation. A more detailed discussion of specific oxygen requirements at each life stage is presented in Section 5.2.

Numerous studies of nitrogen supersaturation indicate that dissolved nitrogen generally affects fishes when saturation exceeds 110–130%, with the threshold level depending on water depth (Rucker and Tuttle 1948; Harvey and Cooper 1962; Fickeison et al. 1973; Blahm 1974; Jensen 1974; Meekin and Allen 1974; Meekin and Turner 1974; Rucker and Kangas 1974; Blahm et al. 1975; Dawley et al., 1975; Weitkamp 1975; Bentley and Dawley 1976; Bouck et al. 1976; Nebeker and Brett 1976). Gas bubble disease (GBD) and mortality are the primary detrimental effects associated with dissolved nitrogen concentrations at threshold levels (Parametrix 1975). The detrimental effects of nitrogen supersaturation vary according to the length of exposure (Blahm et al. 1975). Chinook, coho, steelhead, rainbow trout, whitefish and largemouth bass were exposed to nitrogen levels of 130% for 8 of every 24 hours. Mortality did not exceed 50% fish were placed in nitrogen-saturated water (i.e., 100 %) of the remaining 16 hours. However, when fish were continuously exposed to supersaturated levels of nitrogen (130 %), mortality rates exceeded 50% during the first day. Various species of juvenile salmonids may compensate for total nitrogen saturation levels up to 125% by remaining in deeper water (Parametrix 1975). Hydrostatic pressure increases with depth, so in deeper water nitrogen remains in solution in the blood of fish, inhibiting GBD.

#### 5.1.2.4 Nutrients

Nutrient levels should remain within the natural range for the area and season, and sustain the normal level of primary production. Various inorganic constituents of surface water are nutrients that are required for biological processes reviewed in Section 3.8; and phosphorus are clearly the most important nutrients affecting productivity of aquatic systems. Natural sources of nitrogen and phosphorus in natural ecosystems are discussed at length in section 3.8.1. Inputs to surface- and groundwaters can be affected by vegetation changes associated with land-use activities, as well as through direct enrichment from sewage effluents, run-off from agricultural lands, and industrial water.

Nitrogen generally occurs in natural waters as nitrite or nitrate anions, as cationic forms like ammonium, and as intermediate oxidation states like those that occur in biological materials (e.g., decomposing organic solutes). In surface waters or in groundwaters that are impacted through human use, cyanide from industrial sources and various other complex effluents (including agricultural runoff) may also be sources of nitrogen. Nitrite-nitrogen is short-lived in natural environments and, consequently, rarely exists in concentrations that would be toxic to salmonids. Acute toxicity values for salmonids range from 100 to 900 ppb as  $\text{NO}_2\text{-N}$  (48- or 96-h  $\text{LC}_{50}$ ); chronic effects are poorly understood, in part, because nitrite does not persist in surface waters under natural conditions.

Nitrate is formed by the complete oxidation of ammonia through the nitrification process, and can be found in relatively high concentrations in surface waters. Unlike ammonia and nitrite, nitrate does not form un-ionized species in aqueous solutions and is considered essentially nontoxic for aquatic vertebrates and invertebrates (e.g., acute  $\text{LC}_{50}$  greater than 1300 ppm for salmonids). However, much lower concentrations of nitrate may lead to adverse effects associated with eutrophication and the development of oxygen depleted waters (Leidy 1980; Rand and Petrocelli 1985).

Ammonia frequently acts as a toxicant in surface waters subject to high inputs of nitrogen, especially through anthropogenic activities (e.g., agricultural runoff, sewage effluents). For salmonids, ammonia is acutely toxic at concentrations as low as 80 ppb, but the initiation of ammonia toxicosis is highly variable, primarily as a function of pH. Physiological responses to ammonia exposure are frequently exacerbated by low dissolved oxygen concentrations; for salmonids, acute toxicity is increased two-fold when dissolved oxygen is decreased from 80 % to 30 % saturation. In the laboratory, chronic effects of ammonia have been documented as low as 2 ppb, but little work has been completed to identify the effects of long-term exposures under field settings (Rand and Petrocelli 1985; Reader and Dempsey 1989).

In contrast to nitrogen, phosphorus does not leach as readily from soil. In natural waters phosphorus occurs in very low concentrations, most often in tenths of a milligram per liter (or less). Orthophosphate and its intermediates most frequently occur in surface waters, and are routinely measured as "total phosphorus" in water quality monitoring activities. Phosphorus most frequently occurs in surface waters as phosphates, which are generally considered nontoxic to aquatic vertebrates and invertebrates (Stumm and Morgan 1981; EPA 1986).

Nitrogen and phosphorus are the principal causes of nutrient enrichment of surface waters. Aquatic vegetation (rooted submerged and floating vascular plants) and algae are dependent to varying degrees on dissolved nitrogen and phosphorus for their nutrient supply. Growth of benthic algae and phytoplankton is particularly sensitive to the ratio of nitrogen to phosphorus. Enrichment leads to high rates of biomass production (e.g., algal blooms) that are undesirable for other aquatic biota, especially when subsequent decomposition creates high biochemical oxygen demand and oxygen depletion results. While the enhanced growth rates of aquatic vegetation can reach maximal conditions under nitrogen and phosphorus enrichment, phosphorus frequently acts as the limiting factor in aquatic habitats and will tend to control production rates (Leidy 1980; Stumm and Morgan 1981; Hem 1992).

### 5.1.2.5 Biocides

Agricultural chemicals are potentially widespread in the environment, and surface waters and groundwaters may be affected by chemical use that accompanies changes in land-use practices. Various classes of chemicals are currently used in the agricultural industry, including herbicides, insecticides, fungicides, nematicides, defoliant, rodenticides, and growth regulators. These are primarily organic chemicals, but inorganic chemicals, such as mineral salts and nutrients, may also be used as fertilizers and may directly impact receiving waters. Similarly, complex chemical mixtures in the form of municipal sludges and solid wastes may impact water resources through runoff or infiltration to groundwater (Leidy 1980; Rand and Petrocelli 1985).

Agricultural chemicals are regulated to decrease the likelihood that they are released to surface waters and groundwaters. However, water quality criteria have been established for agrichemicals as indicated for selected examples in Table 5.1.2.5a. There are several properties of agricultural chemicals that influence their fate and effects in the environment. For surface waters and groundwaters, a chemical's adsorptivity, stability, solubility, and toxicity will determine the extent to which that chemical will migrate and adversely impact a water resource. Among the thousands of agricultural chemicals available for users (farmers, orchardists, and home gardeners), these properties will vary significantly. Depending upon the chemical's physicochemical properties, the potential contamination of water resources may be complex. For example, a chemical's water solubility will influence whether it occurs in solution or adsorbed to sediments or colloids held in suspension (Rand and Petrocelli 1985; Dickson et al. 1987). In addition to being toxic to fish and invertebrates that fish eat, agricultural chemicals may indirectly affect non-target species through habitat alteration (e.g., changes in plant community structure as a result of targeting weedy species), and such changes may occur even under the best management practices (Leidy 1980).

### 5.1.2.6 Heavy Metals

Metal concentrations in surface water vary regionally and reflect the geochemical composition of the underlying parent material, and the soils characteristic of the watershed. Most frequently, metals occur in trace quantities as a result of soil leaching and geochemical processes that occur in the underlying bedrock. The concentration of metals in surface waters may be increased by anthropogenic activities such as mining and related industrial practices (e.g., electroplating and metals refining), (Leidy 1980; Stumm and Morgan 1981; Rand and Petrocelli 1985).

Many metals are toxic to fish at very low concentrations; however, metals are necessary trace nutrients and the distinction between metal concentrations that are nutritionally beneficial and those associated with metal toxicosis may be relatively slight, especially when other water quality conditions influence the bioavailability of the metals. For example, metals that are nutritional requirements must be absorbed by the organism. Metals may occur in solution and may be available for uptake directly from the water, or they may be adsorbed to colloidal particles in the water column. The extent to which metals are adsorbed and then intentionally or coincidentally ingested may influence the onset of metal toxicosis in aquatic biota, especially when the interaction between the metals in solution and metals adsorbed to colloids of various forms (e.g., relatively simple organic ligands versus complex organic structures like the humic acids) is influenced by other water quality conditions such as pH. Table 5.1.2.6a lists regulatory criteria for selected metals and metalloids that are frequently considered toxicants of concern, especially in surface waters impacted by human use. The physical features of the surrounding habitat (e.g., land-use in riparian areas) may influence a chemical's toxicity, and seasonal variations in bioavailability of contaminants (e.g., changing redox potentials of sediments and availability of metals) must also be considered (Stumm and Morgan 1981; Dickson et al. 1987; Adriano 1992).

### 5.1.2.7 pH

Acidic surface waters may occur naturally as a result of dissolution of parent materials in bedrock and overlying soils, biological decomposition (especially processes yielding organic acids such as fulvic and humic acids), or through geothermal activity or catastrophic events related to volcanic activity. More frequently however, surface water acidity results from anthropogenic activities related to land-use (e.g., mining) or resource-use (e.g., combustion of fossil fuels) with the subsequent deposition of materials capable of generating, directly or indirectly, and releasing hydrogen ions to the environment (Leidy 1980; Rand and Petrocelli 1985). The influence of hydrogen ions on aquatic organisms is influenced by watershed characteristics, including the buffering capacity of soils, as well as by concentrations of dissolved materials in surface waters (Rand and Petrocelli 1985; Brown and Sadler 1989).

In general, fish may be adversely affected by surface water with pH 5.6 or less; however, the threshold for adverse effects is species-specific and water quality-dependent (e.g., buffering capacity). Hence, no one single pH value can be regarded as a threshold for anticipating population-level responses to acidic surface waters. Respiratory problems are frequently observed in experimental fish exposed to low pH. Mucous clogging, increased ventilation, coughing and hypoxia are commonly recorded in acid-exposed fish. Aluminum and other metals exacerbate the

physiological response to increased hydrogen ion. Low pH alters the specific form of metals in soils, increasing both their mobility and their bioavailability to aquatic organism. In addition, low pH acts synergistically with heavy metals in surface waters to yield adverse biological effects (Stumm and Morgan 1981; Rand and Petrocelli 1985; Brown and Sadler 1989).

### 5.1.3 Habitat Structure

The physical structure of streams, rivers, and estuaries plays a significant role in determining the suitability of aquatic habitats to salmonids, as well as other organisms upon which salmonids depend for food. These structural elements are created through interactions between natural geomorphic features, sediments that are delivered to the stream channel, and riparian vegetation, which provides bank stability and inputs of large woody debris. Structural attributes of streams vary naturally among regions and along the longitudinal dimension of streams in response to differences in topography, geology, geomorphic features, hydrologic regime, sediment load, and riparian vegetation (see Sections 3.5 and 3.9.5). These spatial differences and gradients give rise to the variety of macro- and microhabitat attributes that are used by salmonids at various stages of their life histories. Macrohabitat features include pools, glides, and riffles. The relative frequency of these habitat types change with size of the stream, the degree of channel constriction, and the presence of large woody debris. Microhabitat attributes include characteristics such as substrate type, cover, depth, and hydraulic complexity.

Because of the great diversity in the physical attributes of western streams and in the requirements of various salmonids, and because few undisturbed watersheds remain to serve as reference points, it is difficult to quantify natural ranges of physical habitat features in streams, rivers, and estuaries. For example, historically, mid-order streams west of the Cascade crest had 16–38 pools per km (25–60 per mi) (Figure 5.1.3a) (FEMAT 1993). Pool frequencies in 10 human-influenced tributaries of the upper Grande Ronde River ranged from 3.8–26.2 per km (6–42 per mi) in 1941 and 1.4–7.4 per km (2–12 per mi) in 1990 (McIntosh et al. 1994). In the Yakima Basin, an unmanaged watershed (Rattlesnake Creek) averaged 1.6 pools per km (2.5 per mi) in 1935–36 and 3.9 pools per km (6 per mi) in 1987–92 (McIntosh et al. 1994); similar pool frequencies were also reported for the Chewack River in the Methow River Basin. In low gradient streams on the Olympic Peninsula, Washington, pools constituted 81.1% of the stream surface area (160 pools > 10 m<sup>2</sup>/mile) (Grette 1985). In southeast Alaska, pools accounted for 39–67% of the surface area depending upon bank full width (Murphy et al. 1984 in Peterson et al. 1992). This high degree of variation illustrates the importance on local physical features, stream size, and riparian influence on stream habitat characteristics.

Despite the inherent differences in streams, it is clear that habitat complexity is an important feature of aquatic systems. In streams of the Pacific Northwest, large woody debris creates both macro- and microhabitat complexity that is essential to salmonids and other aquatic organisms. Large wood creates habitat heterogeneity by forming pools, back eddies, and side channels, and creating channel sinuosity and hydraulic complexity. Large wood also functions to retain coarse sediments (e.g., spawning gravels) and organic matter, in addition to providing substrate for numerous aquatic invertebrates. McIntosh et al. (1994) reported that changes in substrate composition towards smaller fractions coincided with reduced frequency of large woody debris in streams of the upper Grande Ronde River. Consequently, large woody debris plays a significant part in controlling other structural elements of streams.

Large woody debris provides an important component to estuarine habitats of coastal rivers (Maser et al. 1988). Woody debris increases habitat complexity in areas where the bottom consists mainly of fine sediments. Numerous invertebrates rapidly process the wood, liberating nutrients for some organisms while others use the wood as refugia. In salt marshes, large woody debris traps sediments to increase the extent of the marsh. As exceptionally high tides displace the logs, depressions left in the sediments increase habitat diversity that is important to juvenile fishes. In areas that are predominantly mud bottomed, large wood further serves as a repository for herring egg spawn. Estuaries are identified as critical rearing habitats for some anadromous salmonids (Table 4.2.2.1b).

The functional roles of large woody debris in streams, and how these change from headwater reaches to estuaries, are reviewed in greater detail in Section 3.9.5. The functions of large woody debris relative to specific life stages of salmonids are discussed in Section 5.2.

### 5.1.4 Flow Regime

Flow regimes in streams and rivers determine the amount of water available to salmonids and other aquatic organisms, the types of micro- and macrohabitats that are available to salmonids (see Section 5.2), and the seasonal patterns of disturbance to aquatic communities. High flow events redistribute sediments in streams, flushing fine sediments from spawning gravels and allowing recruitment of gravels to downstream reaches. In addition, extreme flow events are essential in the development and maintenance of healthy floodplain systems through deposition of sediments, recharge of groundwater aquifers, dispersal of vegetation propagules, recruiting large woody debris into streams, and transporting wood downstream. In alluviated reaches, high flows may create new side channels and flood off-channel areas that are important rearing habitats for salmonids. Low flow may also be important for the establishment of

riparian vegetation on gravel bars and along stream banks (Section 3.6). Thus, dynamic flows are needed to perform essential functions that are important in the long-term persistence of salmonid populations, even though over shorter time scales high or low flow events may temporarily reduce salmonid numbers.

The specific flow requirements of salmonids vary with species, life history stage, and time of year (see Section 5.2). Local salmonid populations have evolved behavioral and physical characteristics that allow them to survive the flow regimes encountered during each phase of their development. Protection of salmonid habitats requires streamflows to fluctuate within the natural range of flows for the given location and season.

### 5.1.5 Biotic Interactions

Protecting and restoring the biological integrity in surface waters also depends on maintaining natural biological interactions among species. These interactions may be affected directly by the introduction of nonnative species and stocks (see Sections 4.3 and 6.10), overexploitation (Section 6.9) or indirectly through modification of physical and chemical characteristics of streams, lakes, and estuaries (reviewed in Section 4.3). Human-induced impacts on biological interactions include changes in primary and secondary production, disruption in timing of life history events or seasonal rhythms, increased frequency of disease or parasitism, and changes in the outcome of predator-prey and competitive interactions. Together these perturbations lead to changes in food webs and trophic structure of aquatic systems.

## 5.2 Habitat Requirements by Life Stage

Salmonids, particularly anadromous salmonids, utilize a wide range of habitats making them highly vulnerable to altered and degraded habitats. To persist, each species or stock must be able to survive within the entire range of habitats encountered during its life. Moreover, anadromous fish have developed life histories that are complex and species-specific, using freshwater streams, lakes, or intertidal sloughs for reproduction; streams and lakes for juvenile rearing in some species; and estuarine and ocean rearing for juveniles of other species (Table 4.2.2.1b). Differences in spatial and temporal use of specific habitats exist for each species, yet the diversity among species and by life stage suggests that most freshwater habitats are utilized year round (Table 4.2.2.2a). Juvenile-to-adulthood rearing generally occurs in the ocean, but there is considerable variation (Table 4.2.2.1b), even within each species. Much of the available information has been identified in reviews by (Bell 1973; Everest et al. 1985; Bjornn and Reiser 1991). These are the primary sources for this section unless otherwise noted.

Most of the quantitative descriptions of salmonid habitat requirements presented in this section consist of microhabitat observations of salmonids observed in nature or results from laboratory studies that measure the performance of salmonids (often hatchery fish) under controlled conditions. Microhabitat measurements are frequently made during a single season (usually summer, when sampling is most convenient) and the resulting data are often reported in the literature without accompanying data on habitat availability. Habitat utilization constitutes a "preference" only when the particular range of depths, velocities, or cover type is used at a frequency greater than its general availability in the environment. In addition, microhabitat measurements at holding positions of salmon and trout do not always capture the range of velocities or depths needed for feeding, which are commonly higher. For these reasons, care must be taken when interpreting microhabitat data published in the literature. Similarly, optimal conditions for development, growth, and survival as determined in the laboratory do not always correspond to the most favorable conditions in natural environments (See Section 5.1).

### 5.2.1 Adult Migrations

The migrations of anadromous salmonids from river mouths to their natal streams vary in length from a few hundred meters (e.g., chum salmon spawning in intertidal zone) to well over a thousand kilometers. Resident fish may make substantial migrations as well between lakes and streams or between sections of a stream (Everest et al. 1985). Anadromous salmonids need holding or resting sites, and suitable flow and water quality during upstream migrations. In addition to these requirements, resident salmonids may feed during their migrations, and thus have more diverse requirements.

#### 5.2.1.1 Physical Structure (*pools, large woody debris, gravels*)

Large woody debris, boulders, and other obstructions diversify the flow and provide resting stations while fish move upstream during spawning migrations. Residents utilize structure to pause out of the main current while waiting for prey to drift by in adjacent, faster waters. At redd sites, adequate areas of stable, appropriately sized gravel containing minimal fines are required for successful spawning (see Section 5.2.2.2). In shallower reaches, riparian vegetation provides cover to form predators. Large woody debris and other structures provide flow stability, which facilitates temperature stratification and the development of coldwater thermal refugia (see Section 5.2.1.3)

### 5.2.1.2 *Flows and Depth*

Streamflows during the spawning migration must be sufficient to allow passage over physical barriers including falls, cascades, and debris jams, and as a result the migrations of many stocks occur coincident with high flows. Coho salmon frequently wait near stream mouths until a freshet occurs before moving upstream (Sandercock 1991). Holtby et al. (1984) found that entry of coho salmon into Carnation Creek was continuous during years of high flow, but pulsed when freshets were infrequent. Spring and summer chinook stocks migrate during periods of high flows that allow them to reach spawning tributaries in headwater reaches, while fall run stocks, which typically spawn in lower reaches, may enter streams during periods of relatively low flow (Healey 1991).

Minimum depths that will allow passage of salmonids range from 12 cm for trout to 18 cm for the smaller anadromous species (i.e. pink, chum, steelhead, sockeye, and coho salmon), to 24 cm for large chinook salmon (Bjornn and Reiser 1991); however, substantially greater depths may be needed to negotiate larger barriers. Reiser and Peacock (1985) report that maximum leaping ability varies from 0.8 m for brown trout to 3.4 meters for steelhead. Pool depths must exceed barrier height by approximately 25% to allow fish to reach the swimming velocities necessary to leap to these heights (Stuart 1962). The ability to pass a barrier is also influenced by pool configuration. Water plunging over a steep fall forms a standing wave that may allow salmonids to leap higher (Bjornn and Reiser 1991). Less severe inclines (e.g., cascades) may be more difficult to pass if pool depths are inadequate and velocities are high.

### 5.2.1.3 *Water Quality (temperature, dissolved oxygen, turbidity)*

Temperature - Most adult salmonids typically migrate at temperatures less than 14°C; however, spring and summer chinook salmon migrate during periods when temperatures are substantially warmer (Table 5.2.1.3a). Excessively high or low temperatures may result in delays in migration (Major and Mighell 1966; Hallock et al. 1970; Monan et al. 1975). Adult steelhead that move from the ocean into river systems in the summer and fall may overwinter in larger rivers, delaying entry into smaller spawning tributaries until they are free of ice in the spring (Bjornn and Reiser 1991). Similarly, spring-spawning resident salmonids, including cutthroat and rainbow trout, may hold at the mouths of spawning streams until temperatures warm up to the preferred temperature range. In addition to delaying migration, excessively high temperatures during migration may cause outbreaks of disease (see Section 4.3.4).

Cold-water refugia may also be important to adult salmon as they migrate upstream. Adult summer-run steelhead in the Middle Fork Eel River of California were observed in thermally stratified pools, but they were absent or infrequent in non-stratified pools of similar depth (Nielsen et al. 1994). Cold-water pockets in stratified pools ranged from 4.1 to 8.2 °C cooler than ambient stream temperatures. Spring chinook salmon have also been observed to hold in cold-water pools for several months prior to spawning in the Yakima River of eastern Washington, moving as much as 60 kilometers from holding pools to spawning sites (NRC 1992). The authors suggest that this behavioral thermoregulation lowers metabolic rates and thereby conserves energy for gamete production, mate selection, redd construction, spawning, and redd guarding.

Streamflow, channel morphology, and the presence of large woody debris may play significant roles in mediating the formation and persistence of cold-water refugia (Nielsen 1994; Bilby 1984). In some streams and rivers, maintaining cold-water refugia may depend on gravel bars or other structures that isolate incoming tributaries or seep areas from main stem waters which thereby inhibit the mixing of waters (Nielsen et al. 1994). In larger systems, thermally stratified pools need not be associated with cold-water inputs provided that deep scour pools exist and flows are sufficiently low to prevent turbulent mixing. Consequently, in larger systems management practices that reduce large woody debris, increase instability of stream channels, or modify stream flows may negatively eliminate cold-water refugia.

Dissolved Oxygen - The high energy expenditures of sustained upstream swimming by salmonids requires adequate concentrations of dissolved oxygen (DO). Davis et al. (1963) found adult and juvenile swimming performance impaired when DO dropped below 100% saturation levels for water temperatures between 10-20 °C. DO concentrations below 6.5-7.0 mg/L greatly impaired performance at all temperatures studied. Migrating adults exhibited an avoidance response to DO levels below 4.5 mg/L (Hallock et al. 1970). Migration resumed when DO levels increased to 5 mg/L.

Turbidity - High suspended sediment concentrations may delay or divert spawning runs, and in some instances cause avoidance by spawning salmon (Smith 1939; Servizi et al. 1969; Mortensen et al. 1976). Salmonids were found to hold in a stream where the suspended sediment load reached 4,000 mg/L. Though high sediment loads may delay migration, homing ability does not seem to be adversely affected. Cowlitz River chinook salmon returned to the hatchery seemingly unaffected by the sediments derived from the eruption of Mount St. Helens, WA, though in the highly impacted Toutle River tributary of the Cowlitz, coho did stray to nearby streams for the first two years following the eruption (Quinn and Fresh 1984).

## 5.2.2 Spawning and Incubation

Although spawning and incubation utilizes the same habitat, adults and embryos have slightly different habitat needs. Adult site selection is determined by substrate composition, cover, and water quality and quantity. Embryo survival in and fry emergence from, an intergravel environment depends upon intergravel and extragravel physical, hydraulic and chemical variables including substrate size, channel gradient and configuration, water depth and velocity, DO, water temperature, biochemical oxygen demand in the gravel, and permeability and porosity of the gravel in the redd (Bjornn and Reiser 1991).

### 5.2.2.1 Physical Structure

All salmonids require sufficient gravels of necessary size with a minimum of fines for spawning. Usable gravel size is generally proportional to adult size, i.e. larger individuals spawn in larger substrate (Marcus et al. 1990). Bjornn and Reiser (1991) reviewed the available literature and found that anadromous salmon typically use gravels in the 1.3–10.2 cm size range, whereas steelhead and resident trouts may use smaller substrates (0.6–10.2 cm). The depth that salmonids deposit eggs is also a function of size (Everest et al. 1985), and may be critical to incubation success. Nawa and Frissell (1993) found that gravel beds may be both scoured and filled during the same flood event potentially leaving little net change in bed surface elevation. Eggs deposited within the zone of scour and fill would be washed downstream. Bedload and bank stability arising from LWD, and intact upslope, floodplain, and/or riparian zones minimize this risk. Large woody debris diversifies flows, reducing stream energy directed towards some portions of the stream (Naiman et al. 1992). This creates pockets of relatively stable gravels better protected from the scouring effects of high water events.

### 5.2.2.2 Flow and Depth

The number of spawning salmon and trout that can be accommodated in a given stream depends on the availability of suitable habitats for redd construction, egg deposition, and incubation (Bjornn and Reiser 1991). Two characteristics of spawning habitats that are directly tied to streamflow are water depth and current velocity. Salmonids typically deposit eggs within a range of depths and velocities that minimize the risk of desiccation as water level recedes and that ensure exchange of water between the surface and substrate interstices is adequate to maintain high oxygen levels and remove metabolic wastes from the redd. In general, the amount of habitat suitable for spawning increases with increasing streamflow; however, excessively high flows can cause scouring of the substrate, resulting in mortality to developing embryos and alevins (Hooper 1973).

Bjornn and Reiser (1991) have recently reviewed studies quantifying specific water depths and velocities at sites used by salmonids for spawning in rivers and streams. Results from their review have been supplemented with data from four other reviews on spawning sites for anadromous salmonids in Table 5.2.2.2a (Healey 1991; Heard 1991; Salo 1991; Sandercock 1991). In general, depth and velocity of water at spawning sites is related to the size of spawners, with larger species spawning at greater depths and faster water velocities than smaller species. There is also substantial variation among rivers, probably reflecting differences in habitat availability. Most species typically spawn at depths greater than 15 cm, with the exception of kokanee and smaller trout (Table 5.2.2.2a) which spawn in shallower waters. Location of redd sites based on water depths and velocities may also vary depending on spawner density. For example, pink salmon tend to spawn in shallower waters when conditions are crowded or streamflows are low (Heard 1991). Several species of salmonids may seek out areas of upwelling for spawning; these include sockeye, chum, coho salmon, and bull trout (Burgner 1991; Salo 1991; Sandercock 1991; Pratt 1992). Upwelling increases circulation of water through redds, eliminating wastes and preventing sediments from filling in spawning gravel interstices. Thus infiltration that recharges groundwater, which eventually discharges in subsurface springs and seeps, must be maintained.

### 5.2.2.3 Water Quality (temperature, dissolved oxygen, turbidity)

**Temperature** - Salmonids have been observed to spawn at temperatures ranging from 1–20°C (Bjornn and Reiser 1991), but most spawning occurs at temperatures between 4 and 14°C (Table 5.2.1.3a). Resident trouts, including rainbow and cutthroat trout, may spawn at temperatures up to 20.0°C and 17.2°C, respectively, while coho salmon, steelhead trout, bull trout, Dolly Varden, and mountain whitefish tend to prefer lower temperatures. The wide range of spawning temperatures utilized by most salmonid species strongly suggests that local adaptation has allowed salmonids to persist in a variety of thermal environments, and that attempting to identify species-specific preferences may be misleading relative to the ecological requirements of a specific stock.

Among the salmonids, the preferred incubation temperatures have been best documented for the anadromous species. Bell (1986) suggested preferred temperature ranges of 4.4–13.3 °C for pink, chum, coho and sockeye salmon, and 5.0–14.4 °C for chinook salmon (Table 5.2.1.3a). More recent laboratory studies have demonstrated that coho and sockeye salmon embryos tend to be less sensitive to cold temperatures and more sensitive to warm temperatures than

pink, chum, or chinook salmon (Beacham and Murray 1990); Murray and McPhail 1988). Coho and sockeye salmon embryos incubated at 1.0°C had survival rates higher than 50%; chum and chinook salmon embryos exhibited 50% mortality at temperatures below 2.5 and 3.0°C, respectively; and even and odd-year pink salmon exhibited 50% mortality at 3.5 and 4.5°C, respectively (Beacham and Murray 1990). Conversely, 50% mortality occurred at temperatures above 13.5°C for coho salmon embryos, compared with 15–15.5°C for pink and sockeye salmon, and 16°C for chum and chinook salmon. The alevin stage is generally less temperature sensitive than the embryonic stages, with lower low-temperature thresholds, and higher high-temperature thresholds (Beacham and Murray 1990). Salmonid embryos and alevins can tolerate short periods during which temperatures are below or above incipient lethal levels (Bjornn and Reiser 1991).

Seymour (1956) carried out comprehensive studies on temperature effects on the development of chinook salmon from the egg to fingerling stage. Environmental temperature was correlated with the number of vertebrae, egg mortality, the number of abnormal fry, and the duration of the hatching period. For eggs reared at temperatures between 4.4 and 14.4°C minimal effects occurred, but defects and mortality increased at both higher and lower temperatures. Combs (1965) identified lower (4.4–5.8°C) and upper (12.7–14.2°C) temperature thresholds for normal development of sockeye eggs.

Dissolved Oxygen - Embryos and alevins generally need high levels of oxygen to survive (Shirazi and Seim 1981), although sensitivity varies with stage of development (Bjornn and Reiser 1991). Field studies have demonstrated positive correlations between DO and survival for steelhead trout (Coble 1961) and coho salmon (Phillips and Campbell 1961). Phillips and Campbell (1961) suggest that DO levels must average greater than 8.0 mg/l for embryos and alevins to survive well.

In addition to being directly lethal, low levels of dissolved oxygen can have sublethal effects of salmonids as well. The rate of embryological development, the time to hatching, and size of emerging fry are all affected by low levels of dissolved oxygen. Garside (1966) found that the rate of embryonic development was increasingly retarded by progressively lower levels of dissolved oxygen (DO), resulting in delayed hatching. Doudoroff and Warren (1965) reported that the rates of embryonic and larval development, the size of larvae at the times of hatching, and of completion of yolk absorption were all dependent on oxygen concentration. Silver et al. (1963) and Shumway et al. (1964) observed that steelhead, coho, and chinook reared in water with low or intermediate oxygen concentration were smaller in size and had a longer incubation period than those raised at high DO. Similarly, Brannon (1965) found a positive relationship between DO and the size of sockeye salmon alevins at time of hatching. Alderdice et al. (1958) found that very low oxygen levels at early egg incubation stages produced severe morphometric abnormalities in chum salmon, in addition to delaying hatching. Low DO levels stimulated eggs in an advanced stage of development to hatch prematurely, causing mortality.

Bjornn and Reiser (1991), summarizing four different studies, concluded that critical dissolved oxygen levels needed to meet respiratory demands vary with state of development. Early embryological states (pre-eyed) require the lowest levels of oxygen, while embryos nearing hatching have the highest DO requirements.

Turbidity - Salmonids require gravels that have low concentrations of fine sediments and organic material for successful spawning and incubation. Bedload or suspended organic and inorganic materials that settle out over spawning redds can affect the intragravel environment of salmonid embryos in several ways. Inorganic sediments, as discussed above, may clog substrate interstices and thereby diminish intragravel flows. In addition fine sediments may act as a physical barrier to fry emergence (Cooper 1956, 1959, 1965; Wickett 1958; McNeil and Ahnell 1964; Koski 1972). Eggs deposited in fine gravel or gravel with a high percentage of fines have lower survival to emergence (Harrison 1923; Hobbs 1937; Shapovalov and Berrian 1940; Shaw and Maga 1943; Koski 1966). McHenry et al. (1994) found that excessive fines (< 0.85 mm and > 13%) resulted in intragravel mortality for coho and steelhead embryos due to oxygen stress. Organic materials that enter the substrate interstices use up oxygen as they decompose (Bjornn and Reiser 1991), further reducing DO concentrations. In addition, salmon and trout avoid areas with high percentages of sand, silt, and clay (Burner 1951; Stuart 1953).

### 5.2.3 Rearing Habitat - Juveniles and Adult Residents

The abundance of juveniles and resident adult salmonids is influenced by the quantity and quality of suitable habitat, food availability, and interactions with other species, including predators and competitors (Bjornn and Reiser 1991). Complex physical structure provides the necessary diversity of habitats to minimize negative conspecific and intraspecific interactions. However, there is no set of "optimal factors" for all species at all life stages. Bjornn and Reiser (1991) suggest that at any given time, certain environmental parameters may be better suited for some individuals, populations or species, while other parameters may not be as favorable, yet must be kept in a suitable range for organism persistence.

### 5.2.3.1 Physical Structure

A variety of lentic and lotic habitats are potentially available to juvenile salmonids due to species-specific differences in ecological specialization. Cover is necessary to the survival of most species, and undercut banks, large woody debris, overhanging vegetation, cobble or boulder areas, and pools provide numerous refugia. Sockeye juveniles rear in lakes for up to three years. Chinook rear in mainstem rivers and estuaries. Coho prefer pool habitats in summer and move into side channels, sloughs, or beaver ponds for winter (Meehan and Bjornn 1991). Backwaters and numerous side-channels that developed along unconstrained reaches in alluvial floodplains were historically important rearing habitats for salmonid juveniles (Sedell and Luchessa 1981; Sedell and Froggatt 1984).

Species-specific differences of habitat use can be illustrated with coho, cutthroat, and steelhead. Large woody debris functions to create a range of hydraulic gradients that increase microhabitat complexity, which allows multiple species to coexist as a community. Two western Washington streams—Beaver Creek and Thrash Creek—differ in pool/riffle ratios because of dissimilar LWD loading. Beaver Creek, a stream with high levels of large woody debris, has a complex microhabitat predominated by pools. Thrash Creek is a debris-poor stream with a high percentage of riffle area. The pool habitat is favored by juvenile coho and cutthroat during the summer, while juvenile steelhead prefer riffles and glides (Everest et al. 1985). As a result, juvenile coho and cutthroat inhabit Beaver Creek and juvenile steelhead inhabit Thrash Creek.

Comparison of microhabitat requirements among species is made difficult by the fact that microhabitat selection is influenced by numerous factors including life stage, time of year, food availability, and presence of other salmonids. Selected data from microhabitat studies are shown in Table 5.2.2.2a; however, these values should be viewed with caution since they were collected from various locations and under different environmental conditions.

### 5.2.3.2 Flow and Depth

The amount of physical space available to juvenile and adult salmonids rearing in streams and the quality of that habitat is directly related to stream discharge (Everest et al. 1985). Within stream environments, salmonids select specific microhabitats where water depth and velocity fall within a specific range, or where certain hydraulic properties occur (Table 5.2.3.2a). These depth and velocity preferences change both with season and life stage. Consequently, streamflows must be adequate to both satisfy minimum requirements for survival during periods of stress (e.g., low flow), as well as to provide specific microhabitat characteristics that are favorable to salmonid populations throughout their period of freshwater residence.

For many salmonids, smaller-sized fish tend to select shallower, slower moving waters than larger individuals (Chapman and Bjornn 1969; Everest and Chapman 1972; Moyle and Baltz 1985). Newly emerged fry may be vulnerable to downstream displacement by flow and typically select velocities lower than 10 cm/s (reviewed in Bjornn and Reiser 1991). During summer months, salmonids often select holding positions at moderate velocities but immediately adjacent to faster waters (Chapman and Bjornn 1969; Jenkins 1969; Everest and Chapman 1972). These positions are believed to confer the greatest energetic advantage to the fish. The amount of food delivered to a particular location is proportional to water velocity (Wankowski and Thorpe 1979; Smith and Li, 1983). Consequently, fish that hold in water adjacent to faster feeding lanes can maximize food intake while minimizing energy expenditures associated with maintaining position in the current (Smith and Li, 1983; Fausch 1984).

During winter months, metabolic demands and, thus, food requirements decrease as temperatures drop. Swimming ability also decreases with decreasing temperature (Brett 1971; Dickson and Kramer 1971; Griffiths and Alderdice 1972), and fish may be less able to maintain positions in fast waters for extended periods of time. As a result, salmonids tend to select slower water velocities, move to off-channel habitats, or seek refuge in substrate interstices when temperatures drop below a certain threshold temperature (Bustard and Narver 1975; Tschaplinski and Hartman 1983; Campbell and Neuner 1985; Johnson and Kucera 1985; Sheppard and Johnson 1985). Larger resident trout may abandon feeding sites in riffles and runs and move to slow moving pool habitats if substrate refugia are unavailable.

For resident salmonids and juveniles of anadromous species that spend a year or more in freshwater, streamflows during the summer low-flow period must be adequate to prevent streams becoming excessively warm or drying up altogether. Under drought conditions, flows may be insufficient to maintain continuous flows and fish may be restricted to isolated pools. Such conditions can result in increased competition for food, reduced dissolved oxygen levels, increased physiological stress, and vulnerability to predators.

### 5.2.3.3 Water Quality, (temperature, dissolved oxygen, turbidity)

Temperature - Juvenile and resident salmonids are variable in their temperature requirements, though most species are at risk when temperatures exceed 23–25 °C (Bjornn and Reiser 1991). Upper and lower lethal temperatures, as well as the "preferred" temperature ranges of several western salmonids is shown in Table 5.2.3.3a. These values provide a general range of tolerable temperatures; however, the ability of fish to tolerate temperature extremes depends on their

recent thermal history. Fish acclimated to low temperatures, for example, have lower temperature thresholds than those acclimated to warmer temperatures.

Temperatures exceeding the upper incipient lethal level may be tolerated for brief periods of time, particularly during diel fluctuations, or may be avoided by seeking coldwater refugia provided by seeps or springs. Bull trout (not shown in table) appear particularly sensitive to warm waters. Temperatures higher than 14°C may act as a thermal barrier to migration of bull trout (OWRRI 1994). McPhail and Murray (1979) found that bull trout grew most rapidly at temperatures of 4°C, about 10°C colder than optimal growth temperatures for most species of *Oncorhynchus*. Lower lethal temperatures are near 0°C for most species of salmonids.

Many salmonid-bearing streams in the Pacific Northwest, particularly those in the southern, eastern, and low elevation portions of the range, experience summer maximum temperatures that approach or exceed upper lethal levels for salmonids. Cold-water refugia in the form of springs, seeps, cold tributaries, and thermally stratified pools may allow populations to persist in streams these streams that would otherwise be inhospitable. Nielsen et al. (1994) found that juvenile steelhead moved into thermally stratified pools when mainstem temperatures were between 23–28°C in a coastal northern California stream. Similarly, Li et al. (1991) reported that resident rainbow trout in an eastern Oregon stream selected natural and artificially created cold-water seep habitats when main-channel temperatures exceeded 24°C, but showed no preference for coldwater areas when temperatures in the main channel dropped below 20°C.

**Dissolved Oxygen** - Salmonids are strong, active swimmers and require highly oxygenated waters. Maximum sustained swimming performance dropped off for coho and chinook salmon when DO concentrations decreased much below air-saturation levels (8–9 mg/L at 20°C) (Davis et al. 1963; Dahlberg et al. 1968). Alabaster et al. (1979) concluded growth rate and food conversion efficiency were probably limited if DO concentrations fell below 5 mg/L for Atlantic salmon, while Davis (1975) found that salmonids would suffer no impairment if DO concentrations remained near 8 mg/L (76–93 % saturation), and determined that DO deprivation would begin at approximately 6 mg/L (57–72 % saturation). High water temperatures which decrease oxygen solubility, would further increase the stress on fish caused by low DO concentrations. A recent literature review resulted in criteria for salmonids presented in Table 5.2.3.3b (Oregon DEQ 1994). The dissolved oxygen criteria proposed for Idaho, Oregon, and Washington are more protective of salmonids than EPA's minima.

**Turbidity** - Turbidity is elevated in all streams for short durations during storm and snowmelt events. Juveniles and adults appear to be little affected (Sorenson et al. 1977) by these transitory episodes though Bisson and Bilby reported coho avoidance of water exceeding 70 NTU's (nephelometric turbidity units), levels that may occur in some watersheds with high erosion potential. In a laboratory setting, coho and steelhead juveniles had reduced growth rates and a greater number were found to emigrate from turbid streams (25–50 NTU) than clear streams (Sigler et al. 1984). Lloyd et al. (1987) found that juvenile salmonids avoided chronically turbid streams including glacially influenced streams and those disturbed by human activities. Turbidity also influences foraging behavior of juvenile salmonids by reducing the distance from which they can locate drifting prey.

## 5.2.4 Juvenile Migration

Depending upon the species or population of juvenile salmonids, some fish migrate to the sea, others to lakes, and still others remain in a relatively small reach of stream for their entire lives (Everest et al. 1985; Bjornn and Reiser 1991). Juvenile fish require unobstructed access to upstream or downstream reaches for migration or dispersal to feeding grounds.

### 5.2.4.1 Physical Structure

Physical structure may be utilized by migrating juveniles as refugia during resting periods, as sources of food, or as cover from predators. Juveniles migrating to lakes such as sockeye or adfluvial resident populations, may be traveling upstream or downstream and require structure for food and refugia for streamflow and predators.

### 5.2.4.2 Flow and Depth

Streamflow is important in facilitating downstream movement of salmonid smolts. Smolt migration is believed to be regulated by "priming" factors, such as photoperiod and temperature, that alter the disposition of the fish in anticipation of downstream migration, and "releasing" factors, including changes in temperature or streamflow, that trigger movement once a state of "readiness" is obtained (Groot 1982). Dorn (1989) found that increases in streamflow triggered downstream movement of coho salmon in a western Washington stream. Similarly, Spence (1995) also found short-term increases in streamflow to be an important stimulus for smolt migration in four populations of coho salmon. Thus the normal range of streamflows may be required to stimulate downstream migration.

Streamflow is also important in determining the rate at which smolts move downstream. Bjornn and Reiser (1991) state that the time required to travel from the Salmon River in Idaho to the Dalles Dam increased by as much as 30 days during low flow periods following the construction of six dams on the Columbia-Snake system (Fried et al. 1978).

#### 5.2.4.3 *Water Quality (temperature, dissolved gasses, turbidity)*

Temperature - Temperature affects migration timing of smolts in two fundamental ways: by influencing the rate of growth and physiological development and by affecting the responsiveness of fish to other environmental stimuli (Groot 1982). Consequently, alteration of thermal regimes through land-use practices and dam operations can influence the timing of migration. Holtby (1988) found that coho salmon smolts emigrated approximately 8 days earlier after logging. In addition, the age-class distribution was shifted from populations evenly split between one- and two-year old smolts to populations dominated by one-year old fish. Such shifts in age structure may be detrimental in that overall population survival may be enhanced when risk is spread over two years instead of one.

The specific temperature requirements of juvenile anadromous salmonids during their seaward migration are not well documented. Sockeye smolts have been reported to migrate at temperatures ranging from 2–10°C (reviewed in Burgner 1991). Coho salmon have been observed to migrate at temperatures as low as 2.5°C and as high as 13.3°C (Sandercock 1991); however, most fish migrate before temperatures reach 11–12°C. Ocean-type chinook typically migrate during March and April at temperatures between 4.5°C and 15.5°C (Healey 1991), whereas stream-type chinook smolts tend to migrate 1 to 2 months later when conditions are substantially warmer. Once temperatures exceed a threshold level in the spring, salmon smolts will revert to a pre-smolt physiology and remain within the stream. No summary information for pink and chum salmon were readily available.

Dissolved Gasses - Supersaturation of dissolved gasses (particularly nitrogen) has been found to cause gas bubble disease in upstream and downstream migrating salmonids (Ebel and Raymond 1976). Steelhead appear to be more susceptible than salmon to the disease since salmon have been found to better sense and avoid highly supersaturated waters (Stevens et al. 1980). However, all salmonids are susceptible.

Turbidity - Turbid waters have been mentioned as affecting migration but little documentation is available in the literature. Thomas (1975) found fry migration increased as turbidity increased. Lloyd et al. (1987) found that turbid streams were avoided by juveniles except when the fish must pass through them along migration routes. There is also some evidence that diel migrations of salmonids is influenced by turbidity. Many salmonids tend to migrate during the evening hours Burgner (1991), presumably to avoid predation; however, in streams with higher turbidity, migrations may be evenly dispersed during both the day and night.

Table 5.1.2.5a Water quality criteria for selected herbicides, pesticides, and fungicides in freshwaters (USEPA 1986)—PAP 1329?.

| Chemical                    | Maximum Acceptable Levels<br>(ppb) | Period of Exposure                           |
|-----------------------------|------------------------------------|--|
| Aldrin                      | 3.00e+00                           | instantaneous                                |
| Dieldrin                    | 2.50e+00<br>1.90e-03               | instantaneous<br>24 hr avg                   |
| Chlorophenoxy<br>Herbicides |                                    |  |
| 2,4-D                       | 1.00e+02                           | instantaneous*                               |
| 2,4,5-TP                    | 1.00e+01                           | instantaneous*                               |
| Chloropyrifos               | 8.30e-02                           | 1 hr avg, not more than<br>1 time per 3 yrs. |
| DDT and Metabolites         |                                    |  |
| DDT                         | 1.10e+00<br>1.00e-03               | instantaneous<br>24 hr avg                   |
| TDE                         | 6.00e-01                           | instantaneous                                |
| DDE                         | 1.05e+03                           | instantaneous                                |
| Endosulfan                  | 2.20e-01<br>5.60e-02               | instantaneous<br>24 hr avg                   |
| Endrin                      | 1.80e-01<br>2.30e-02               | instantaneous<br>24 hr avg                   |
| Guthion                     | 1.00e-02                           | instantaneous                                |
| Heptachlor                  | 5.20e-01<br>3.80e-03               | instantaneous<br>24 hr avg                   |
| Malathion                   | 1.00e-01                           | instantaneous                                |
| Methoxychlor                | 3.00e-02                           | instantaneous                                |
| Mirex                       | 1.00e-03                           | instantaneous                                |
| Parathion                   | 4.00e-02                           | instantaneous                                |
| Pentachlorophenol           | **                                 |  |
| Toxaphene                   | 1.60e+00<br>1.30e-02               | instantaneous<br>24 hr avg                   |

\* human-health based criteria

\*\* criteria based on pH- See current Water Quality Criteria

Table 5.1.2.6a Metals and Metalloids, Found in Surface Waters, Their Sources, and Factors Affecting Their Toxicity to Fish.

| Metal     | Essential Nutrient | Anthropogenic source  | Factors influencing toxicity           |
|-----------|--------------------|-----------------------|--|
| Arsenic   | no                 | agricultural          | -                                      |
| Boron     | yes                | agricultural          | -                                      |
| Cadmium   | no                 | mining/industrial     | water hardness, PH,                    |
| Chromium  | no                 | industrial            | PH                                     |
| Copper    | yes                | mining/industrial     | PH, valence, temperature, other metals |
| Lead      | no                 | mining/industrial     | turbidity, PH, water hardness          |
| Manganese | yes                |                       |  |
| Mercury   | no                 | industrial            | DOC, microbial activity                |
| Nickel    | no                 | mining/industrial     | water hardness                         |
| Selenium  | yes                | irrigated agriculture | -                                      |
| Silver    | no                 | mining                | water hardness                         |
| Zinc      | yes                | mining/industrial     | hardness, PH, temperature, valence     |

Table 5.2.1.3a Tolerable and Preferred Temperature Ranges (°C) for Adult Migrations, Spawning, and Incubation of Embryos for Native Salmonids in the Pacific Northwest. Modified after Bjornn and Reiser (1991).

| Species | (Life Stage)                           |                            |                              |
|---------|--|----------------------------|------------------------------|
|         | Spawning Migration (minimum - maximum) | Spawning (preferred range) | Incubation (preferred range) |
|         | 5                                      | 17                         | 8                            |

Table 5.2.2.2a Water Depths and Velocities Used by Anadromous and Resident Salmonids for Spawning

| Species                                   | Preferred Depth <sup>†</sup><br>(cm) | Preferred Velocity <sup>‡</sup><br>(cm) | Source                     |
|---|--------------------------------------|---|----------------------------|
| Chinook salmon <sup>††</sup>              | 0.15–0.43                            | 0.37–0.69                               | Bovee (1978)               |
| Chinook salmon <sup>††</sup> (Skagit R.)  | 0.52–1.28                            | 0.55–1.13                               | Graybill et al. (1979)     |
| Spring Chinook salmon                     | ≥24                                  | 30–91                                   | Thompson 1972              |
| Summer Chinook salmon <sup>†</sup>        | ≥30                                  | 32–109                                  | Reiser and White 1981a     |
| Summer Chinook salmon                     | 5–700                                | 10–189                                  | Healey 1991                |
| Fall Chinook salmon                       | 0.24                                 | 0.30–0.91                               | Thompson (1972).           |
| Fall Chinook (Columbia R.) <sup>†</sup>   | 1.22–1.98                            | 0.84–1.14                               | Chambers et al. (1955).    |
| Fall Chinook salmon                       | ≥24                                  | 30–91                                   | Thompson 1972              |
| Spring Chinook salmon <sup>††</sup>       | 0.18–0.38                            | 0.24–0.61                               | Bovee (1978)               |
| Spring Chinook salmon                     | 0.24                                 | 0.30–0.91                               | Thompson (1972).           |
| Spring Chinook (Cowlitz R.)               | 0.30–1.07                            | 0.30–0.44                               | Chambers et al. (1955).    |
| Chum salmon                               | ≥18                                  | 46–101                                  | Smith 1973                 |
| Chum salmon                               | 13–110                               | 0–168 <sup>§</sup>                      | Salo 1991                  |
| Coho salmon                               | ≥18                                  | 30–91                                   | Thompson 1972              |
| Coho salmon                               | 4–33                                 | 30–75                                   | Sandercocock 1991          |
| Coho salmon <sup>††</sup>                 | 0.12–0.35                            | 0.25–0.61                               | Bovee (1978)               |
| Coho salmon                               | 0.18                                 | 0.30–0.91                               | Thompson (1972).           |
| Coho salmon                               | 0.20–0.25                            | 0.25–0.70                               | Li et al. (1979)           |
| Pink salmon                               | ≥15                                  | 21–101                                  | Collings 1974              |
| Pink salmon                               | 10–100                               | 30–104                                  | Heard 1991                 |
| Sockeye salmon                            | ≥15                                  | 21–101 <sup>†</sup>                     | †                          |
| Sockeye salmon                            | 15–300                               | nd                                      | Buigner 1991               |
| Sockeye salmon <sup>††</sup>              | 0.17–0.49                            | 0.34–0.58                               | Bovee (1978)               |
| Sockeye salmon <sup>††</sup>              | 0.15–0.55                            | 0.28–0.79                               | Stober and Graybill (1974) |
| Sockeye salmon                            | 0.30–0.46                            | 0.53–0.55                               | Clay (1961)                |
| Kokanee salmon                            | ≥6                                   | 15–73                                   | Smith 1973                 |
| Kokanee salmon <sup>††</sup>              | 0.06–0.23                            | 0.12–0.41                               | Bovee (1978)               |
| Steelhead trout                           | ≥24                                  | 40–91                                   | Smith 1973                 |
| Steelhead trout <sup>††</sup>             | 0.18                                 | 0.30–0.91                               | Stober and Graybill (1974) |
| Steelhead trout <sup>¶</sup>              | 0.12–0.70                            | 0.37–1.09                               | Hunter (1973)              |
| Steelhead trout <sup>††</sup> (Skagit R.) | 0.27–0.88                            | 0.46–0.91                               | Graybill et al. (1979)     |
| Winter Steelhead trout <sup>††</sup>      | 0.24–0.45                            | 0.43–0.87                               | Bovee (1978)               |
| Rainbow trout                             | ≥18                                  | 48–91                                   | Smith 1973                 |
| Rainbow trout                             | 0.15–0.43                            | 0.27–0.79                               | Chambers et al. (1955).    |
| Rainbow trout                             | 0.21–0.30                            | 0.30                                    | Li et al. (1979)           |
| Cutthroat trout                           | ≥6                                   | 11–72                                   | Hunter 1973                |
| Cutthroat trout <sup>†</sup>              | 0.17–0.30                            | 0.15–0.46                               | Chambers et al. (1955)     |
| Mountain whitefish <sup>††</sup>          | ≥0.22                                | 0.30–0.66                               | Bovee (1978)               |
| Mountain whitefish                        | 6.10–12.20                           | ≥0.15                                   | Li et al. (1979)           |

<sup>†</sup> Range of depths shown in parentheses

<sup>‡</sup> Range of velocities shown in parentheses

<sup>§</sup> Zero velocity likely at redds with upwelling

<sup>†</sup> Estimated by Bjornn and Reiser 1991 from criteria for other species

<sup>††</sup> Utilizes 80% probability range

<sup>††</sup> Utilizes 50% probability range

nd no data

Table 5.1.2.6a Metals and Metalloids, Found in Surface Waters, Their Sources, and Factors Affecting Their Toxicity to Fish.

| Metal     | Essential Nutrient | Anthropogenic source  | Factors influencing toxicity           |
|-----------|--------------------|-----------------------|--|
| Arsenic   | no                 | agricultural          | -                                      |
| Boron     | yes                | agricultural          | -                                      |
| Cadmium   | no                 | mining/industrial     | water hardness, PH,                    |
| Chromium  | no                 | industrial            | PH                                     |
| Copper    | yes                | mining/industrial     | PH, valence, temperature, other metals |
| Lead      | no                 | mining/industrial     | turbidity, PH, water hardness          |
| Manganese | yes                |                       |  |
| Mercury   | no                 | industrial            | DOC, microbial activity                |
| Nickel    | no                 | mining/industrial     | water hardness                         |
| Selenium  | yes                | irrigated agriculture | -                                      |
| Silver    | no                 | mining                | water hardness                         |
| Zinc      | yes                | mining/industrial     | hardness, PH, temperature, valence     |

Table 5.2.1.3a Tolerable and Preferred Temperature Ranges (°C) for Adult Migrations, Spawning, and Incubation of Embryos for Native Salmonids in the Pacific Northwest. Modified after Bjornn and Reiser (1991).

| Species            | (Life Stage)                           |                            |                              |
|--------------------|--|----------------------------|------------------------------|
|                    | Spawning Migration (minimum - maximum) | Spawning (preferred range) | Incubation (preferred range) |
| <b>Anadromous</b>  |  |                            |                              |
| Pink Salmon        | 7.2 - 15.6*                            | 7.2 - 12.8*                | 4.4 - 13.3*                  |
| Chum Salmon        | 8.3 - 15.6*                            | 7.2 - 12.8*                | 4.4 - 13.3*                  |
| Coho Salmon        | 7.2 - 15.6*                            | 4.4 - 9.4*                 | 4.4 - 13.3*                  |
| Sockeye Salmon     | 7.2 - 15.6*                            | 10.6 - 12.2*               | 4.4 - 13.3*                  |
| Spring Chinook     | 3.3 - 13.3*                            | 5.6 - 13.9*                | 5.0 - 14.4*                  |
| Summer Chinook     | 13.9 - 20.0*                           | 5.6 - 13.9*                | 5.0 - 14.4*                  |
| Fall Chinook       | 10.6 - 19.4*                           | 5.6 - 13.9*                | 5.0 - 14.4*                  |
| Steelhead Trout    |  | 3.9 - 9.4*                 |                              |
| Cutthroat Trout    |  | 6.1 - 17.2*                |                              |
| <b>Resident</b>    |  |                            |                              |
| Kokanee            |  | 5.0 - 12.8*                |                              |
| Mountain Whitefish |  | 0.0 - 5.6†                 |                              |
| Cutthroat Trout    | -5.0                                   | 4.4 - 12.8†<br>5.5 - 15.5‡ |                              |
| Rainbow Trout      |  | 2.2 - 20.0*<br>4.4 - 12.8† |                              |
| Dolly Varden       |  | 6.1                        |                              |
| Bull Trout         |  | <9.0§<br>4.5†              | 2.0 - 6.0§                   |
| Arctic Grayling    |  | 4.4 - 11.1†                |                              |

\* Bell 1986

† Everest, et al. 1985

‡ Varley and Gresswell 1988

§ Pratt 1992

† Ratliff 1992

Table 5.2.2.2a Water Depths and Velocities Used by Anadromous and Resident Salmonids for Spawning

| Species                                   | Preferred Depth <sup>†</sup><br>(cm) | Preferred Velocity <sup>‡</sup><br>(cm) | Source                     |
|---|--------------------------------------|---|----------------------------|
| Chinook salmon <sup>††</sup>              | 0.15-0.43                            | 0.37-0.69                               | Bovee (1978)               |
| Chinook salmon <sup>††</sup> (Skagit R.)  | 0.52-1.28                            | 0.55-1.13                               | Graybill et al. (1979)     |
| Spring Chinook salmon                     | ≥24                                  | 30-91                                   | Thompson 1972              |
| Summer Chinook salmon                     | ≥30                                  | 32-109                                  | Reiser and White 1981a     |
| Summer Chinook salmon                     | 5-700                                | 10-189                                  | Healey 1991                |
| Fall Chinook salmon                       | 0.24                                 | 0.30-0.91                               | Thompson (1972).           |
| Fall Chinook (Columbia R.) <sup>†</sup>   | 1.22-1.98                            | 0.84-1.14                               | Chambers et al. (1955).    |
| Fall Chinook salmon                       | ≥24                                  | 30-91                                   | Thompson 1972              |
| Spring Chinook salmon <sup>††</sup>       | 0.18-0.38                            | 0.24-0.61                               | Bovee (1978)               |
| Spring Chinook salmon                     | 0.24                                 | 0.30-0.91                               | Thompson (1972).           |
| Spring Chinook (Cowlitz R.)               | 0.30-1.07                            | 0.30-0.44                               | Chambers et al. (1955).    |
| Chum salmon                               | ≥18                                  | 46-101                                  | Smith 1973                 |
| Chum salmon                               | 13-110                               | 0-168 <sup>§</sup>                      | Salo 1991                  |
| Coho salmon                               | ≥18                                  | 30-91                                   | Thompson 1972              |
| Coho salmon                               | 4-33                                 | 30-75                                   | Sandercocock 1991          |
| Coho salmon <sup>††</sup>                 | 0.12-0.35                            | 0.25-0.61                               | Bovee (1978)               |
| Coho salmon                               | 0.18                                 | 0.30-0.91                               | Thompson (1972).           |
| Coho salmon                               | 0.20-0.25                            | 0.25-0.70                               | Li et al. (1979)           |
| Pink salmon                               | ≥15                                  | 21-101                                  | Collings 1974              |
| Pink salmon                               | 10-100                               | 30-104                                  | Heard 1991                 |
| Sockeye salmon                            | ≥15                                  | 21-101 <sup>†</sup>                     | †                          |
| Sockeye salmon                            | 15-300                               | nd                                      | Burgner 1991               |
| Sockeye salmon <sup>††</sup>              | 0.17-0.49                            | 0.34-0.58                               | Bovee (1978)               |
| Sockeye salmon <sup>††</sup>              | 0.15-0.55                            | 0.28-0.79                               | Stober and Graybill (1974) |
| Sockeye salmon                            | 0.30-0.46                            | 0.53-0.55                               | Clay (1961)                |
| Kokanee salmon                            | ≥6                                   | 15-73                                   | Smith 1973                 |
| Kokanee salmon <sup>††</sup>              | 0.06-0.23                            | 0.12-0.41                               | Bovee (1978)               |
| Steelhead trout                           | ≥24                                  | 40-91                                   | Smith 1973                 |
| Steelhead trout <sup>††</sup>             | 0.18                                 | 0.30-0.91                               | Stober and Graybill (1974) |
| Steelhead trout <sup>†</sup>              | 0.12-0.70                            | 0.37-1.09                               | Hunter (1973)              |
| Steelhead trout <sup>††</sup> (Skagit R.) | 0.27-0.88                            | 0.46-0.91                               | Graybill et al. (1979)     |
| Winter Steelhead trout <sup>††</sup>      | 0.24-0.45                            | 0.43-0.87                               | Bovee (1978)               |
| Rainbow trout                             | ≥18                                  | 48-91                                   | Smith 1973                 |
| Rainbow trout                             | 0.15-0.43                            | 0.27-0.79                               | Chambers et al. (1955)     |
| Rainbow trout                             | 0.21-0.30                            | 0.30                                    | Li et al. (1979)           |
| Cutthroat trout                           | ≥6                                   | 11-72                                   | Hunter 1973                |
| Cutthroat trout <sup>†</sup>              | 0.17-0.30                            | 0.15-0.46                               | Chambers et al. (1955)     |
| Mountain whitefish <sup>††</sup>          | ≥0.22                                | 0.30-0.66                               | Bovee (1978)               |
| Mountain whitefish                        | 6.10-12.20                           | ≥0.15                                   | Li et al. (1979)           |

† Range of depths shown in parentheses  
 ‡ Range of velocities shown in parentheses  
 § Zero velocity likely at redds with upwelling  
 † Estimated by Bjornn and Reiser 1991 from criteria for other species  
 †† Utilizes 80% probability range  
 †† Utilizes 50% probability range  
 nd no data

Table 5.2.3.2a Depths and Velocities at Holding Sites of Salmonids in Streams by Age or Size. From Bjornn and Reiser (1991).  
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| Species and source                 | Age <sup>†</sup> or size | Depth (cm) | Velocity (cm/s)    |
|------------------------------------|--------------------------|------------|--------------------|
| Steelhead                          | 31 - 44 mm               | 24         | 40                 |
| Bugert (1985)                      |                          |            |                    |
| Everest and Chapman (1972)         | 0                        | <15        | <15                |
|                                    | 1                        | 60 - 75    | 15 - 30            |
| Hanson (1977)                      | 1                        | 51 mean    | 10 mean            |
|                                    | 2                        | 58 mean    | 15 mean            |
|                                    | 3                        | 60 mean    | 15 mean            |
| Moyle and Baltz (1985)             | 0                        | 35         | 7.3                |
|                                    | Juvenile                 | 63         | 19.4               |
|                                    | Adult                    | 82         | 28.6               |
| Sheppard and Johnson (1985)        | 37 mm                    | <30        | <25                |
| Smith and Li (1983)                | 25 mm                    |            | 4                  |
|                                    | 50 mm                    |            | 8                  |
|                                    | 75 mm                    |            | 18                 |
|                                    | 100 mm                   |            | 24                 |
|                                    | 150 mm                   |            | 24                 |
| Stuehrenberg (1975)                | 0                        | <30        | 14 (range, 3 - 26) |
|                                    | 1                        | >15        | 16 (range, 5 - 37) |
| Thompson (1972)                    | 0                        | 18 - 67    | 6 - 49             |
| Chinook Salmon                     | 0                        | 15 - 30    | <15                |
| Everest and Chapman (1972)         |                          |            |                    |
| Konopacky (1984)                   | 77 - 89 mm               | 55 - 60    | 12 - 30            |
|                                    |                          |            | 18 (dawn)          |
|                                    |                          |            | 12 (midday)        |
|                                    |                          |            | 25 (dusk)          |
| Stuehrenberg (1975)                | 0                        | <61        | 9 (range, 0 - 21)  |
|                                    | 1                        | <61        | 17 (range, 5 - 38) |
| Thompson (1972)                    | 0                        | 30 - 122   | 6 - 24             |
| Steward and Bjornn (1987)          | 78 - 81 mm               | 40 - 58    | 8 - 10             |
| Coho Salmon                        | 40 - 50 mm               | 24         | 39 (flume)         |
| Bugert (1985)                      | 0                        |            | 15                 |
|                                    | 1                        |            | 18                 |
| Nickelson and Reisenbichler (1977) | 0                        | >30        | >30                |
| Pearson et al. (1970)              | 0                        |            | 9 - 21             |
| Sheppard and Johnson (1985)        | 62 mm                    | 30 - 70    | <30                |
| Hanson (1977)                      | 1                        | 51 mean    | 10 mean            |
|                                    | 2                        | 56 mean    | 14 mean            |
|                                    | 3                        | 57 mean    | 20 mean            |
|                                    | 4                        | 54 mean    | 14 mean            |
| Pratt (1984)                       | <100 mm                  | 32         | 10                 |
|                                    | >100 mm                  | 62         | 22                 |
| Thompson (1972)                    | 0, 1                     | 40 - 122   | 6 - 49             |
| Bull trout                         |                          |            |                    |
| Pratt (1984)                       | <100 mm                  | 33         | 9                  |
|                                    | >100 mm                  | 45         | 12                 |

<sup>†</sup> Ages are in years or life stages, without units

Table 5.2.3.3a Lower Lethal, Upper Lethal, and Preferred Temperatures (°C) for Selected Species of Salmon and Trout\*. From Bjornn and Reiser (1991).  
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| Species         | Lethal Temperature (°C)   |                           | Preferred Temperature (°C) | Source                                  | Technique  |
|-----------------|---------------------------|---------------------------|----------------------------|---|------------|
|                 | Lower Lethal <sup>†</sup> | Upper Lethal <sup>‡</sup> |                            |   |            |
| Chinook salmon  | 0.8                       | 26.2                      | 12 - 14                    | Brett 1952                              | ILT        |
| Coho salmon     | 1.7                       | 26.0<br>28.8 <sup>§</sup> | 12 - 14                    | Brett 1952<br>Becker & Genoway 1979     | ILT<br>CTM |
| Sockeye salmon  | 3.1                       | 25.8                      | 12 - 14                    | Brett 1952                              | ILT        |
| Chum salmon     | 0.5                       | 25.4                      | 12 - 14                    | Brett 1952                              | ILT        |
| Steelhead       | 0.0                       | 23.9                      | 10 - 13                    | Bell 1986                               |            |
| Rainbow trout   |                           | 29.4<br>25.0              |                            | Lee & Rinne 1980<br>Charlon et al. 1970 | CTM<br>ILT |
| Cutthroat trout | 0.6                       | 22.8                      |                            | Bell 1986                               |            |

\* Based on techniques to determine Incipient Lethal Temperatures (ILT) and Critical Thermal Maxima (CTM) Data from Bjornn and Reiser 1991 in Meehan, et al. 1991.

<sup>†</sup> Acclimation temperature was 10°C; no mortality occurred in 5,500 min.

<sup>‡</sup> Acclimation temperature was 20°C unless noted otherwise; 50% mortality occurred in 1,000 min.

<sup>§</sup> Acclimation temperature was 15°C

Table 5.2.3.3b Guidance for Relating Dissolved Oxygen Criteria to Use Protection. From Oregon DEQ (1994).

| Current and Alternative Criteria and Use Protection |                        |            |               |         |   |
|---|------------------------|------------|---------------|---------|---|
| Class   | Concentration (mg/l)   |            |               |         | Use/Level of Protection   |
|   | 30-Day Mean            | 7-Day Mean | 7-Day Minimum | Minimum |   |
| Salmonid Spawning                                   |                        | 11         |               | 9<br>6¶ | Salmonid spawning and incubation of embryos. Low risk of impairment to cold-water aquatic community of salmonids, other native fish, and invertebrates.   |
| Cold Water  | 8                      |            | 6.5           | 6       | Principally cold-water communities, salmon, trout, cold-water invertebrates, other native cool-water species, throughout all or most of the year. Juvenile anadromous salmonids may rear throughout the year. Low level risk of impairment for these groups.  |
| Cool Water  | 6.5                    |            | 5             | 4       | Mixed native cool-water species, such as sculpins, and cool-water aquatic life. Provides migratory route for salmon and trout. Salmonids and other cold-water biota may be present during part or all of the year but may not dominate community structure. Slight level of risk to cool-water community. |
| Warm Water  | 5.5                    |            |               | 4       | Native warm-water fish; non-native species, salmonid migration; waterbodies may not naturally support native cool-water communities.  |
| No Risk   | No change from natural |            |               |         | The only criteria that provides no additional risk to the resource is no change from background.  |

30-Day Mean

7-Day Mean

7-Day Minimum

Minimum

¶ Intergravel Dissolved Oxygen

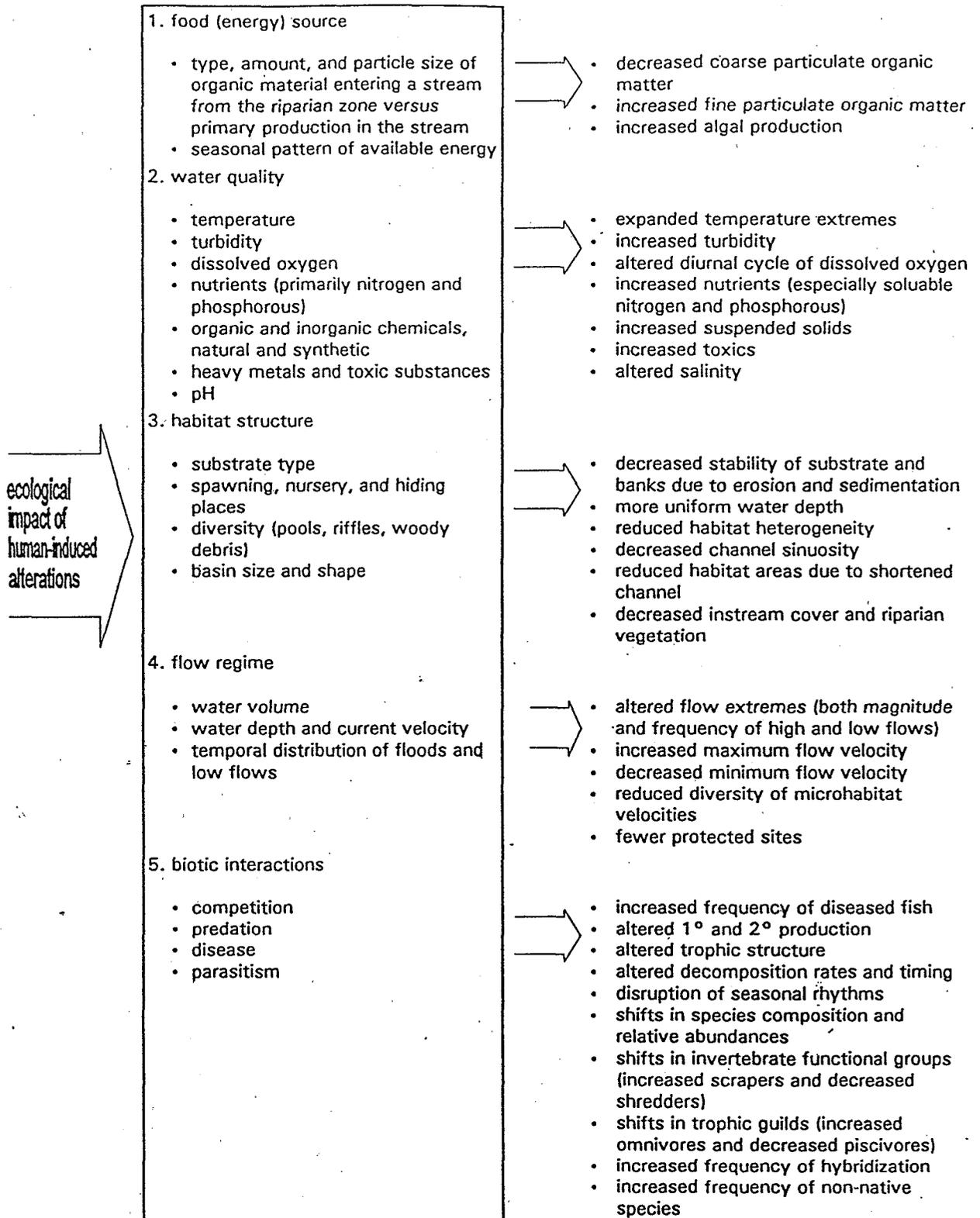


Figure 5.0a

Five major classes of environmental factors that affect aquatic biota (Karr et al. 1983). Arrows indicate the kinds of effects that can be expected from human activities (modified from Karr et al. 1986).

## 6.0 Effects of Human Activities on Watershed Processes and Aquatic and Riparian Habitats

Land-use practices, including forestry, grazing, agriculture, urbanization, and mining, can disrupt aquatic ecosystems by altering watershed processes that ultimately influence the attributes of streams, lakes, and estuaries. In this section, we review specific mechanisms by which human activities directly or indirectly affect aquatic ecosystems. With the exception of direct chemical applications, most effects on watershed processes result from changes in vegetation and soil characteristics, which in turn affect the rate of delivery of water, sediments, nutrients, and other dissolved materials from uplands to stream channels. Within the riparian zone, land-use activities can alter the amount of solar radiation reaching the stream surface, affect the delivery of coarse and fine organic materials to streams, and modify fluvial processes that affect bank stability, sediment transport, seasonal streamflow patterns, and flood dynamics. Disconnecting streams from their floodplains can further impair nutrient dynamics and vegetation characteristics.

Dams and diversions affect salmonids by hindering migrations of fish, by altering the physical (e.g., temperature, flow, sediment routing) and chemical characteristics of streams, and by causing changes in stream biota. Other activities, including wetland removal, harvesting of salmon, introduction of non-native species and hatchery salmonids, eradication of beaver, and activities associated with river, estuarine, and ocean traffic also affect salmonids and their habitat.

### 6.1 Forestry

#### 6.1.1 Background

Forest vegetation covers approximately 46% of the combined land surface of Washington, Oregon, and Idaho, including 34% of nonfederal lands and 58% of federal lands (Jackson and Kimerling 1993).<sup>1</sup> Most commercial harvesting of timber is for softwoods, primarily Douglas-fir, ponderosa pine, hemlock, Sitka spruce, and lodgepole pine. Industry-owned lands, despite constituting only 18% of the softwood growing stock, accounted for 44% of the total softwood harvest in the region in 1986. In contrast, national and state forests constitute 70% of the growing stock, but accounted for only 46% of the softwood harvest (Jackson and Kimerling 1993). Intense production from nonfederal lands is likely to continue or increase as federal timber supplies diminish.

Logging in the Pacific Northwest began in the mid 1800s, and by the 1860s, the timber industry was well established. By 1880, forests along Puget Sound, as well as along many major rivers and streams had been cleared for two or more miles inland, and up to several miles inland along major streams and rivers (Sedell and Luchessa 1981). Throughout the 1900s forest harvest has continued, and the effects of logging have become pervasive across the region. Early forest practices were particularly damaging to stream environments. Splash damming was commonly used to float logs down to the sawmills, a practice that has had long-lasting effects on channel morphology and the abundance of large woody debris. Clear-cuts often included riparian forests, which supported large quantities of wood that were easily transported downstream. Debris jams were routinely removed at the behest of biologists, who believed they hindered migration of anadromous fishes. Today the functional importance of large woody debris to salmonids is well documented (Bisson, Bilby et al. 1987; Hicks, Hall et al. 1991; Naiman, Beechie et al. 1992), and state forest practice rules have been modified to reflect this understanding. But, despite recent improvements in forest practices, a legacy of past practices and cumulative effects will hamper our ability to quickly reverse habitat changes accrued from logging practices.

In the sections that follow, we review the effects of forest practices on watershed processes and salmonid habitats. We use the term "forest practices" to include all activities associated with the access, removal, and reestablishment of forest vegetation, including road construction, timber harvest, site preparation, planting, and intermediate treatments. Understanding the effects of these practices on natural processes will improve our ability to incorporate habitat conservation concepts on forested lands.

#### 6.1.2 Effects on Vegetation

Forest practices directly influence vegetation within a watershed through the removal of trees during harvest, thinning, and road construction, and through manipulations of understory and ground vegetation (e.g., burning, and mechanical and chemical treatments) designed to increase the vigor of desired species and inhibit growth of understory vegetation. In addition, forest vegetation is indirectly affected by changes in site conditions following harvest. Removal

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<sup>1</sup>Percentages listed in Table 7 of Jackson and Kimerling (1993) for nonfederal lands are in error. The correct total acreage for nonfederal lands is 83,066,500 acres (P. Jackson, pers. comm.).

of overstory vegetation can change local microclimate, soil moisture and stability, ground cover, and susceptibility to erosion, all of which may influence the reestablishment of vegetation in the harvested area (Beschta et al. 1994). Soil compaction by ground-based equipment can reduce infiltration of water, thereby hindering the reestablishment of seedlings or growth of established vegetation. In addition, as tree roots die after logging, subsurface spaces (macropores) become compacted or filled with sediment, reducing infiltration of water and reducing aeration in the soils. When porosity is reduced below 20-25%, root growth is retarded. Mixing of mineral and organic soil layers also strongly influences the revegetation process (Beschta et al. 1994).

The magnitude of vegetation change and the succession of vegetation following logging depend on the type and degree of disturbance. For highly disturbed sites, early succession is dominated by colonizing annual and herbaceous species, followed by dominance/codominance by perennial species, and finally dominance by overstory species (Beschta et al. 1994). Where disturbance is less severe, residual species may dominate the early successional stages. Beschta et al. (1994) provide a more thorough review of the effects of forest practices on regeneration of vegetation.

At the landscape level, forest practices have resulted in substantial modification of species and age composition of western forests. Natural forests typically exhibit a mosaic of patches in different states of ecological succession. These mixed-age, multi-species plant communities have been replaced with even-aged forest plantations dominated by a single species. Riparian forests have been especially affected in areas where rapid growth of hardwood species (e.g., alder and maple) and shrubs (e.g., salmonberry) has precluded reestablishment of coniferous species (see e.g., Bisson et al. 1987). In coastal streams, riparian areas outside of wilderness areas are dominated by alder and big leaf maple (FEMAT 1993). Certain conifers, such as western hemlock, regenerate most successfully on partially decomposed nurse logs that are elevated above the forest floor. Because downed trees have been removed from riparian areas, reestablishment of these species is difficult.

### 6.1.3 Effects on Soils

Forest practices can result in significant disturbance to soils, including increased compaction, scarification, and mixing of soil layers. The degree and effects of compaction are influenced by a number of factors, including the total area compacted, the soil type and moisture content, the equipment used, and the number of passes the vehicle makes over the site. Cafferata (1992) in Beschta et al. (1994) reviewed a number of studies and determined that 10-40% of a harvest area may be compacted during tractor logging.

The effects of soil compaction appear to be of long duration. Studies have estimated recovery times from 10-50 years, with estimates as long as 90-110 years in an arid high elevation site (Webb et al. 1986; Cafferata 1992). Duration of compaction depends upon depth of compaction, soil texture, soil temperature and moisture regimes, and biological activity. Recovery time increases with increasing depth of compaction. Soil recovery occurs more rapidly in clay soils that shrink and swell with changing moisture content; high elevation soils that are subjected to freezing and thawing; soils with high organic content that cushions soils from compaction; and soils with high biological activity (e.g., burrowing rodents, earthworms, insects, soil microbes) (Beschta, Boyle et al. 1994).

### 6.1.4 Effects on Hydrology

Timber harvest and the associated road construction and site preparation practices can have significant effects on hydrologic processes that determine streamflow. In most cases, the removal of vegetation increases the amount of water that infiltrates the soil and ultimately reaches the stream by reducing water losses from evapotranspiration. However, in forested systems where fog drip contributes significantly to total precipitation (Harr 1982), harvesting trees may have little effect on the total amount of water reaching and infiltrating the soil. Soil compaction can decrease infiltration and increase the likelihood of surface runoff. Roads can affect the routing of water by intercepting subsurface flow and diverting it down drainage channels, effectively increasing drainage density within a watershed (Sidle, Pearce et al. 1985). King and Tennyson (1984) observed altered hydrology when 4% or more of catchment area was roaded. In snow-dominated systems, logging can influence the spatial distribution of snow on the ground, as well the energy transfer processes that affect melting rate of snowpack (Chamberlin, Harr et al. 1991). The effect of logging on hydrologic processes can change annual water yield, the magnitude and timing of peak flows, and the magnitude of summer low flows. The effects of logging on hydrologic processes are reviewed in three recent syntheses (Hicks et al. 1991; Chamberlin et al. 1991; Beschta et al. 1994), and the material presented below is based primarily on these analyses.

#### 6.1.4.1 Water Yield

In most instances, clearcutting has been found to increase total water yield. In western Oregon, increases in water yield in the first 1-5 years following logging have ranged from a few inches to almost 2.5 inches (Beschta et al. 1994). The largest increases in yield occur in areas of high precipitation and high evapotranspiration. In these areas,

increased yield resulted primarily from reduced evapotranspiration losses. One case study in the Bull Run watershed of Oregon found that water yields decreased after logging (Harr 1982). Apparently, the decrease in evapotranspiration losses was offset by decreases in the amount of fog-drip that normally occurred in the forested watershed.

In forests east of the Cascade Range water yield increases are not as great. In a snow-dominated system in the Blue Mountains of Oregon, no increase in water yield was observed following logging (Fowler, Helvey et al. 1987). The authors suggest that higher wind speeds after clear-cutting increased evaporation from snowpack, offsetting reduced evapotranspiration losses. Elsewhere in eastern Oregon and Washington, forested watershed are likely to experience minor changes in total water yield. Beschta et al. (1994) speculate that in areas with low precipitation (<15–20 inches) increases in water yield are likely to be negligible. Water yields in interior ponderosa pine and douglas fir forests are generally low to begin with because of high evaporation demands. Consequently, reductions in transpiration losses may be compensated by higher evaporation losses. In areas with higher precipitation, increases in water yield may be somewhat greater, but generally still small compared with those of westside systems (Ziemer 1986; Beschta et al. 1994). Small increases in yield from snow-dominated systems in British Columbia and Colorado have also been observed (Hibbert 1967; Cheng 1989).

In addition to being affected by forest vegetation and climatic conditions, increases in water yield also depend on the percentage of the land area that is harvested or roaded (Hewlett and Nutter 1970; Trimble and Weirich 1987). In general, the increase in yield is directly proportional to the size of the area logged. However, for patch cuts, removal of vegetation may result in a smaller increased yield than predicted by area alone because of increased utilization of available moisture by vegetation in surrounding uncut areas (Beschta et al. 1994). For the same reason, selective harvesting or thinning may have minimal effect on water yield (Hibbert 1967). Bosch and Hewlett (1982) reviewed over 90 watershed studies and concluded that increased yield usually occurs only after 20–30 % of a watershed has been harvested.

The effects of logging on total water yield persist until the transpiration demands of recovering vegetation approach those of uncut forests. In forests west of the Cascade Crest, return to natural conditions may take 30–40 years if no further disturbances occur in the watershed (Harr and Cundy 1992; Stednick and Kern 1992). Brush removal by mechanical means, chemical treatments, or burning (to aid reestablishing desired trees) can influence the rate of recovery. However, in general, these activities are practiced only until seedlings attain sufficient height to shade out competing species. Thus effects of these practices on water yield are likely to be short term.

#### 6.1.4.2 *Timing of Runoff*

In rain-dominated systems, the largest increases in water yield due to logging generally occur during periods where both precipitation and transpiration rates of vegetation are relatively high, usually the fall (Chamberlin et al. 1991) and spring (Beschta et al. 1994). With reduced transpiration, soil moisture is rapidly replenished with the onset of rains in the fall and subsurface flow to stream channels commences (Rothacher 1971; Harr et al. 1979). Evapotranspiration losses from mature forests are comparatively small during the winter because of low temperatures and high humidity and, consequently, increased yield in winter is generally smaller (Chamberlin et al. 1991); however, in the spring, the differences between transpiration losses in mature forests and those in clearcuts are again greater, and increases in water yield may be higher than in winter (Beschta et al. 1994).

In snow-dominated systems, increases in water yield generally occur during the early spring snowmelt period. The loss of shading following removal of the forest canopy can accelerate snowmelt, resulting in an earlier peak in the stream hydrograph. In snow-dominated systems, solar radiation is the primary factor influencing rate of snowmelt (Chamberlin et al. 1991). In the snow-rain transition zone of the western Cascades snowmelt is driven primarily by convective transfer of sensible and latent heat to the snowpack (Harr 1986). Opening up the forest canopy can increase wind speed and turbulence, facilitating more rapid melting.

#### 6.1.4.3 *Peak Flows*

A recent review (Beschta et al. 1994) of effects of timber harvest on peak flows in systems in the Pacific northwest indicates a high degree of variability. In rain-dominated systems of the Coast Range, most studies have indicated increases in peak flows following logging, particularly those occurring in fall (Table 6.1.4.3a). In a few cases, increases have been insignificant, and in one case, a decrease in peak flows was observed. Where increases in peak flow occur, they likely result from reduced evapotranspiration losses with removal of the forest canopy and more rapid routing of water to the stream channel because of roading and soil compaction.

In transient-snow systems of western Oregon responses of peak flows are also variable (Table 6.1.4.3b). Several studies have indicated increases of 10–200% in peak flows (Rothacher 1973; Harr et al. 1979) while others have shown no change or decreases (Harr et al. 1979; Harr et al. 1982; Harr and McCorison 1979). Harr (1986) reanalyzed published data and found that studies showing decreases in peak flows were inconclusive. In systems where harvest has increased peak flows in the transient-snow zone, it is believed that vegetation removal increased delivery of

water to the soil from the snowpack during rain-on-snow events (Harr 1986). Coffin and Harr (1992) used lysimeters placed under the snowpack to confirm increased melt rates and delivery of water to the soil during rain-on-snow events. Maximum differences in melt rates between open and forested plots occurred when rain events were accompanied by relatively high temperatures and wind speeds, apparently because of increased transfer of sensible and latent heat to the snow. Coffin and Harr (1992) report that effects of harvesting on peak flows were still evident in 20–25 year-old plantations.

In snow-dominated systems peak flows have generally shown little or no change following logging (Table 6.1.4.3b) although studies are limited in geographic distribution. In most of these studies, the percentage of the watershed cut has been less than 50 %. Despite the lack of conclusive data, it is reasonable to predict increased peak flows following logging. Snow accumulation is generally higher in open patches created by logging (Chamberlin et al. 1991), though it is unclear whether this is merely a redistribution of snow over the watershed or an actual increase in availability. Increased wind speeds in cleared areas may accelerate melting, leading to more rapid runoff and higher peak flows.

#### 6.1.4.4 Low flows

Increases in summer low flows have been observed following logging in a number of systems in the Pacific Northwest. Harr and Krygier (1972) documented average increases in summer flows of 60 % following logging of a Coast Range stream in Oregon. Somewhat larger increases were observed in a western Cascade stream (Rothacher 1970). Keppeler and Ziemer (1990) noted increases in summer flows in a northern California stream, but found that the increases disappeared within five years. Studies in drier, interior climates have been less conclusive. Cheng (1989) reported increases in summer flows that persisted for six years after logging in an interior British Columbia stream. However, Troendle (1983) found no increase in summer low flows following logging in Colorado. Where increases in summer flows occur, they likely result from reductions in evapotranspiration losses.

Few long-term studies of effects of logging on low flows have been performed. A notable exception is Hicks et al. (1991) who found that August streamflows in a central Oregon Cascade stream increased for 8 years following logging, but decreased for 18 of the next 19 years. On average, August streamflows were 25% lower than in pre-logging years. The authors attributed reductions in streamflow to the replacement of coniferous vegetation with more consumptive hardwood species. Thus, the long-term effects of logging on streamflows likely depend on vegetation composition before and after harvest.

#### 6.1.5 Effects on Sediment Transport

Forest practices can substantially increase delivery of sediments to streams through both surface erosion and mass wasting. The effect of forest practices on sediment transport depends on a number of local site conditions including climate, vegetation, topography, and soil type, as well as specific aspects of the activity, including the type and areal extent of disturbance and the proximity of the disturbance to the stream channel. Thus, the relative effects of roading, timber harvest, site preparation, and other forest practices on sediment production vary with location (Beschta et al. 1994).

Furniss et al. (1991) concluded that forest roads contributed more sediment than all other forest activities combined on a per unit area basis. Summarizing results from nine different studies, they reported that mass wasting associated with roads produced 26–346 times the volume of sediment as undisturbed forests. Mass failures were attributed to poor road location, construction, and maintenance, as well as inadequate culverts. Beschta (1978) found that, in three out of eight years, suspended sediment increased significantly from two catchments in the Coast Range, primarily as a result of mass failures from roads. Mass failures associated with roads most commonly occur on cut and fill slopes, but may also initiate where end-haul material is deposited on a hillslope (Dent 1993). In addition, channel constrictions at road crossings may lead to bank sloughing and bank erosion.

Surface erosion from roading also constitutes a significant source of chronic sediment inputs (Beschta et al. 1994). Splash erosion mobilizes sediment on exposed road surfaces, and runoff from compacted surfaces may also facilitate sediment transport. Water diverted into drainage ditches along roads gathers energy and can cause significant erosion at the outlets of cross-drain culverts (Beschta et al. 1994). Montgomery (1993), studying three small catchments, found that roads decreased the drainage area needed to support a channel head and thereby increased the length of the channel. He attributed this phenomenon to lower infiltration or greater runoff on roads. He also demonstrated that channel expansion was a function of catchment area. The combined effects of mass wasting and surface erosion can lead to elevated sediment levels in streams even when only a small percentage of a watershed is roaded. For example, Cederholm et al. (1981) reported increased sediments in salmonid spawning gravels when roads exceeded 3% of the total basin area.

Sediment delivery due to other forest activities, including harvest, yarding and site preparation, may be increased via several mechanisms. Loss of the protective vegetative cover can increase splash erosion and decrease

slope stability (Swanston et al. 1980) (Marcus et al. 1990). Yarding activities cause extensive soil disturbance and compaction that may increase splash erosion and channelized runoff. Ground-based vehicles moving logs from felled trees and skidding the logs along the ground to landing sites compact and scarify the soil. Compaction of the decomposing root systems can reduce the infiltration capacity of these channels, leading to slumps, landslides, and surface erosion (Everest et al. 1987). Loss of the humic layer through mechanical disturbance and fire further increases the potential for surface erosion.

The quantity of sediments delivered to the stream channel depends upon the integrity of the riparian zone, the intensity of disturbance, the areal extent of the disturbance, the proximity of the disturbance to the stream channel, and slope steepness. Buffer strips can trap sediments, significantly reducing delivery rate (Swanston 1991). Site disturbance may be intensified by a hot burn following harvest which creates extensive areas of bare soil (Everest et al. 1987). Piling and burning versus broadcast burning will also intensify the site disturbance and increase sediment delivery rates.

The fate of sediments once in the channel also depends on the nature of sediments (coarse versus fine) and local site characteristics. Although surface erosion is less dramatic and less evident than mass wasting, it may be more detrimental to stream biota because the delivery of particles occurs over a longer time and those particles are smaller and more likely to become embedded in coarser substrates. Bilby (1985) found that sediments from road crossings were flushed from a fourth-order stream reach of 2% gradient that drained a small (5.5 km<sup>2</sup>) catchment with a relief ratio of 0.10. Presumably, these sediments (2–151 mg/l above control levels) were deposited downstream. In a separate study, Duncan et al. (1987) reported that first- or second-order channels with high amounts of wood debris retained 55% of road-crossing sediments at flows up to 7% of bankfull. Thus stream gradient and retentive inchannel structures appear important in determining whether sediments are deposited locally or transported downstream.

### 6.1.6. Effects on Energy Transfer and Stream Temperature

Logging most directly affects energy transfer by reducing shading (by removing riparian vegetation), which increases the amount of direct solar radiation reaching the ground and stream surfaces. The increase in energy reaching the ground and stream will depend on the amount of shading lost. Measurements from an old-growth Douglas fir forest in western Oregon indicated shading averaged 84% (Summers 1983). Brazier and Brown (1973) reported that angular canopy densities generally fall between 80 and 90% in old-growth stands in western Washington (cited in Beschta et al. 1987). In eastern Oregon, natural canopy density is somewhat less. Slightly lower shading (75%) has been reported for a stream in northern California. Erman et al. (1977) and Anderson et al. (1993) estimated shading in old-growth forests of the Upper Grande Ronde basin in eastern Oregon to be around 72%. Thus, the magnitude of increase in stream temperatures following canopy removal is likely to differ across the region.

Removal of riparian canopy also affects other energy transfer processes including convection, evaporation, and advection. Convective and evaporative heat exchange are both affected by wind speed (see Section 4.6), which generally increases as riparian vegetation is removed. Consequently, convective exchange as well as evaporative losses tend to increase slightly following logging (Brown 1969). The removal of vegetation from upslope areas generally allows greater heating of the soil surface during the summer months. Rain falling in the early part of the rainy season may pick up additional heat as it passes through the soil and infiltrates subsurface aquifers, resulting in increases in ground water temperature after logging.

Removal of riparian canopy has been shown to have two major effects on temperatures of smaller streams in the Pacific Northwest: increased maximum temperatures (particularly in summer) and increased diel fluctuations (Beschta et al. 1987; Beschta et al. 1994). For coniferous forests of the Coast Range and western Cascades, increases in average summer maximum temperatures due to clearcutting have ranged from about 3 to 8°C (Table 6.1.6a); (Beschta et al. 1987). Increases up to 10°C have been observed when clearcutting has been followed by slash burning (Brown and Krygier 1970). Increases in annual maximum daily temperatures can be substantially greater. Hall and Lantz (1969) reported increases in maximum daily stream temperatures of up to 16°C in years immediately following logging of a small watershed in the Coast Range of Oregon. Holtby (1987) reported that average monthly water temperatures increased from 0.7 to 3.2°C following logging of the Carnation Creek (British Columbia) watershed, with the largest increases occurring in May–September and the smallest increases in December and January. These changes persisted for at least seven years after logging. Average diel temperature fluctuations increased by as much as 3.7°C in two Carnation Creek tributaries that had diel fluctuations of less than 1°C prior to logging (Holtby and Newcombe 1992). Hall and Lantz (1969) reported that midsummer diel fluctuations of 15°C were common in Needle Branch, Oregon, after logging. Documentation of temperature changes resulting from logging east of the Cascade range is sparse. Because the degree of shading provided by more open forest types (e.g. ponderosa pine) is lower than for coastal and western Cascade streams, the increase in temperatures resulting from canopy removal might be expected to be slightly less. Nevertheless, because many streams east of the Cascades approach the maximum thermal tolerance level for salmonids during the summer, smaller increases in temperature might be equally or more detrimental to salmonids.

Although summer stream temperatures have been the focus of most research on the effects of logging on stream temperatures, changes in winter stream temperatures may also occur. Theoretically, the loss of riparian

vegetation allows for greater radiative cooling at night during the winter months, potentially decreasing winter temperatures. However, Holtby (1988) reported increases in February–April mean temperatures of 1–2°C. Increases in groundwater temperatures following canopy removal may have been responsible for the increase in winter temperatures. Hall and Lantz (1969) also noted similar increases in temperatures during the winter in a coastal Oregon stream after the entire basin was clearcut.

The magnitude of temperature change following removal of riparian vegetation depends on the size of stream and channel morphology. Because stream discharge and depth increase downstream, the ability of solar radiation to effect stream temperatures also diminishes with increasing stream size (Beschta et al. 1994). Moreover, the amount of shading provided by riparian vegetation decreases as streams become larger and wider. Consequently, the removal of riparian vegetation effects temperature most in small-to-medium-sized streams, and least in large river systems. Sullivan and Adams (1990) suggest that riparian vegetation has a negligible effect on stream temperatures for streams that are 5th order or larger.

Although the effects of logging on stream temperatures within the logged area are well documented, the cumulative effects of temperature increases both downstream and over time are less well understood. Temperature data from Needle Branch in Oregon's Coast Range indicate that thermal regimes returned to near normal approximately seven years after logging and slash burning (Hall et al. 1987). In this case, alder replaced conifers as the dominant riparian vegetation and provided significant shade to this small stream. However, temperature increases in Carnation Creek showed no sign of diminishing eight years after logging and the author estimated that elevated stream temperatures were likely to persist for an additional decade or more (Holtby 1988). Similarly, in the higher elevation fir zone of the Cascade Range, the degree of shading may not reach pre-logging levels for 40 years or more (Summers 1983). Thus the duration of temperature effects depends on the rate of recovery of riparian vegetation and the level of shading provided.

The cumulative effects of stream temperature changes downstream of logged areas are not well documented. As streams leave harvested areas and reenter forested reaches, temperatures tend to decline as solar radiation is reduced. Similarly, small tributaries generally have a minor effect on the temperatures of larger streams which they enter (Caldwell et al. 1991). Sullivan and Adams (1990) have argued that in streams in western Washington, temperatures approach mean air temperatures at a "threshold distance" downstream from the watershed divide (see section 4.6). The cumulative effects of warming of upstream tributaries may have little effect beyond this distance, though no analysis has been conducted to validate this hypothesis outside of western Washington (Beschta et al. 1994). In a study in the western Cascades of Oregon, Beschta and Taylor (1988) found that stream temperatures increased coincidentally with increased logging and road building in the basin. As logging activity decreased in subsequent years, temperatures also declined, strongly suggesting that the cumulative effects of logging and road building were responsible for the previous temperature increases.

### 6.1.7 Effects on Nutrients

Forest practices can lead to changes in nutrient distribution and dynamics in upland areas, which in turn affect availability in streams. Nutrients are directly lost to the ecosystem through the removal of trees. Harvest intensity (i.e. proportion of forest canopy removed), type of harvest (logs or whole tree) and cutting frequency all affect the rate of nutrient removal from the system (Beschta et al. 1994). Despite the loss of nutrients stored in removed biomass, nutrients are generally more available to stream organisms in the years immediately following harvest (Figure 6.1.7a). This results in part from the addition of slash to the forest floor (Frazer et al. 1990), accelerated decomposition of organic litter resulting from increased sunlight reaching the forest floor (Beschta et al. 1994), increased water availability for leaching of materials, and increased overland runoff and erosion that contributes unbound (nitrate and ammonium) and bound (orthophosphate) nutrients to the stream (Gregory et al. 1987). Where logging reduces riparian vegetation, nutrient supply to the stream (e.g., leaf litter and large woody debris) may be reduced. As soils stabilize and revegetation occurs, the nutrient flux declines, though nutrients from herbaceous plants in the riparian zone add high quality materials that easily decompose. Over time herbs, shrubs, deciduous trees, and conifers provide allochthonous inputs for nutrient uptake (Figure 6.1.7b).

Burning of slash, or the entire harvested area, can temporarily elevate the concentrations of nutrients entering the stream. Grier et al. (1989) suggests that fire effectively accelerates decomposition processes. If a fire is hot, however, much of the nitrogen is volatilized and lost to the system (Gessel and Cole 1973). Other nutrients including potassium, phosphorus, calcium and magnesium increased by 2-8 times, while nitrogen decreased by two thirds following burning (Austin and Baisinger 1955). Herbicide treatments, like burning, can lead to short-term increases in nutrients as deciduous vegetation dies and decomposes.

The significance of forest harvest on nutrient losses depends on the mechanism causing the loss. The most significant losses result from tree removal. Leaching is not considered a major component of losses overall, accounting for less than 1 % of losses due to harvest. Losses due to volatilization resulting from fire can be much more significant (Beschta et al. 1994).

### 6.1.8 Effects of Fertilizer and Pesticide Use

Herbicides, insecticides, and fertilizers are the most commonly used forest chemicals. In forest plantations, herbicides control the invading (herbaceous) vegetation and enhance the suitability of the area for reestablishment of desired tree species. Fertilizers are used to replace nutrients lost following logging and accelerate growth of conifers. Insecticides are used to protect desired species from insect pests. By extending the early phases of catchment recovery, shortening the intermediate (deciduous) phases, and accelerating development of conifers, herbicides and fertilizers can have both positive and negative effects on streams. Slower revegetation of deforested soils may more than double sedimentation rates. If more sunlight reaches the stream it will warm and algal production increases, increasing the food base for invertebrates and fish. Algae growth is also stimulated by fertilizers. Delayed production of deciduous trees and accelerated growth of conifers reduces the delivery of leaves and intermediate sized wood to streams over the short term, but increases the potential for large wood over longer periods. Depending on whether temperature, spawning sites, cover, or food is limiting, these changes may initially hinder or aid salmonid production. If conifers are protected from insect pests, fewer die and fall into streams, thereby reducing the availability of large woody debris and stream and riparian complexity. A similar result occurs if fire retardants protect trees. The general effects of forest chemicals, then, is a function of the degree to which they simplify the stream/riparian ecosystem in both the short and long terms.

Insecticides have shorter term effects on stream ecosystems than herbicides, but occasionally they may be more dramatic. The diminution, death, or catastrophic drift of aquatic invertebrates reduces the salmonid food supply and may take several months to recover. Changes in invertebrate assemblage composition may require several years before returning to pre-spray conditions. If drifting organisms or terrestrial insects that fall into streams contain sufficient pesticide, it may be concentrated by the fishes feeding on them. Because salmonids in forest streams are may be food limited and territorial, reductions in aquatic insect biomass and altered assemblage composition may result in reduced growth and numbers of salmonids.

Direct application of chemicals, particularly insecticides, to stream channels can have direct toxic or indirect effects on aquatic organisms. However, if no-spray riparian zones are identified and spared from spraying, few such incidents would occur (Norris et al. 1991). Given the current small extent of riparian zones in the Pacific Northwest and the density of streams west of the Cascades, there is significant opportunity for direct toxic effects, but few have been documented. Indirect effects of herbicides, insecticides, and fire retardants are far more likely, and some may even be beneficial to salmonids. In addition, less than 1 % of total pesticides used in the United States are used in forestry (Norris et al. 1991), thus contamination from forest practices is likely minor compared to other agricultural practices.

### 6.1.9 Effect on Physical Habitat Structure

Timber harvest has resulted in the removal and decreased recruitment of large woody debris (LWD) in streams throughout the Pacific Northwest (Bisson et al. 1987; Maser et al. 1988; Hicks et al. 1991), which in turn has substantially modified habitat characteristics of streams. Removal of vegetation from the riparian zone has altered sources, mechanisms for delivery, and distribution patterns of wood in stream channels (Bisson et al. 1987). Woody debris provided by deciduous vegetation tends to be smaller, more mobile, and shorter-lived than that derived from conifers (see Section 3.9.5) and, consequently, does not function as well in retaining sediment. Hicks et al. (1991) documented both short and long-term effects of forest practices on stream habitat (Table 6.1.9a). The reduced supply of large woody debris eventually leads to loss of instream cover and pool habitat available for fish (Bisson et al. 1987). In addition, loss of large woody debris results in decreased retention of sediments, including gravels used by salmonids for spawning, as well as organic materials. The lack of debris also simplifies channel hydraulics and leaves banks unprotected, resulting in increased bank erosion. The formation of undercut banks that salmonids frequently use for cover is also diminished with the loss of riparian vegetation.

Road construction also results in changes to instream habitats. Culverts create physical obstructions that fish must negotiate when migrating to and from the ocean and between summer or winter rearing habitats. Poorly designed and installed culverts act as barriers to both anadromous and nonanadromous salmonids. At culvert sites, excessive flow velocities, insufficient water, excessive culvert heights, and the absence of a pools all can impede migration (Evans and Johnson 1980). Culverts also fail frequently when inappropriately designed and installed, resulting in mass wasting of much of the road crossing. Road construction along streams, particularly where revetments are required, can constrain streamflow, thereby facilitating scouring of the channel bed. Dose (1994) found that channels widened as road density in catchments increased.

The net effect of forest practices has been a reduction of complexity of micro- and macrohabitats in streams. The FEMAT (1993) report documents substantial decreases in the number of large, deep pools in river systems west of the Cascade Range, which were attributed to loss of pool-forming structures (e.g., boulders, large wood), filling of pools with sediment, and loss of sinuosity of stream channels. These changes are in part due to logging. Research indicates that the effects of logging on habitats are ongoing, cumulative, and may persist for decades to a century or more. However, the causal linkages between land use, habitat development, and fish impacts are not always clear.

### 6.1.10 Effects on Stream Biota

A substantial volume of literature exists documenting effects of logging on aquatic biota (see reviews in Gregory et al. 1987; Hicks et al. 1991; and Beschta et al. 1994). The response of aquatic communities to logging depends on a variety of factors, thus studies have sometimes predicted seemingly contradictory results. Any of the following factors may influence the specific response of a given system: 1) species and stocks of fishes are diverse and adapted to local conditions, such that response may vary in different portions of each species' range; 2) physical and vegetative conditions, as well as logging methods, vary among regions such that impacts differ in magnitude, persistence, and ecological significance; 3) biotic interactions and long-range fish movements can mediate the effects of habitat alteration, such that most important biotic changes are indirectly and incompletely related to physical effects; 4) impacts of numerous independent factors can accumulate over time or space, or interact in either a compensatory or synergistic way, such that ecological responses are complex and difficult to predict; 5) dynamic, sometimes catastrophic natural events (e.g., large floods, changing oceanic conditions) create variable "baseline" conditions so that additional variability added by habitat alterations is difficult to quantify (Frissell 1991).

Within this context, some common patterns in the response of aquatic organisms to forest practices have been identified, based on both empirical evidence and theoretical expectations (e.g., the river continuum concept). In smaller streams, the removal of riparian vegetation increases light intensity, which stimulates the growth of benthic algae (Gregory 1980; Gregory et al. 1981; Shortreed and Stockner 1983). In contrast, energy inputs from allochthonous sources decrease after harvest of riparian vegetation (Gregory et al. 1987; Bilby and Bisson 1992). Macroinvertebrate communities respond to these changes in food sources. Herbivorous invertebrates, particularly those that scrape algae from the substrate, become more abundant, while those species that feed on detritus (i.e. shredders and collector-gatherers, Merritt and Cummins 1984), decline in numbers (Beschta et al. 1994). Invertebrate and vertebrate predators, including juvenile salmonids, have been shown to increase in response to increased secondary production in streams in Alaska and the Cascades (Murphy et al. 1981; Hawkins et al. 1982). As riparian vegetation recovers, the amount of solar radiation reaching the channel diminishes, algal production decreases, and shredders and collector-gatherers begin to replace scrapers.

This general pattern of change in aquatic communities in response to changing energy sources caused by logging of riparian vegetation can be confounded by other simultaneous changes in habitat conditions. Siltation from mass failures and surface erosion can affect invertebrate production as gravel interspaces are filled by silt and algae are buried or abraded (Beschta et al. 1994). In these instances, invertebrate communities are typically characterized by high numbers of a few tolerant, colonizing species (Newbold et al. 1980; Murphy et al. 1981; Hawkins et al. 1982; Lamberti et al. 1991). Loss of substrate complexity also tends to decrease the diversity of aquatic invertebrates. Similarly, application of insecticides and herbicides may have substantial and long-lasting effects on invertebrate community structure, with stoneflies, mayflies, and caddisflies all being particularly sensitive (reviewed in Beschta et al. 1994). Gregory et al. (1987) suggest that an overall pattern of increased production of a few taxa accompanied by a reduction in overall biodiversity may be common to all invertebrate trophic levels in streams that have been simplified through forest practices.

The composition and diversity of fish populations and communities are affected in numerous ways by logging. Bisson et al. (1992) found that salmonid communities changed in response to the conversion of pools to riffles after logging. The pool habitat was favored juvenile coho while the riffles were preferred by older, larger trout. In a long-term study of the effects of harvesting on a watershed (Alsea Watershed Study, Oregon), number of outmigrating fry from a clearcut section were reduced by more than 50%, while pre-logging and post-logging numbers from a patch cut watershed and unharvested watershed were not significantly different (Hall et al. 1987). Declines in the clear-cut watershed were attributed to a reduction in gravel quality from increased fines that led to a decrease in survival from egg deposition to emergence. In the same study, Moring and Lantz (1975) found reductions in late-summer densities of cutthroat trout in the clear-cut watershed following logging, but no reductions in the other two watersheds. Hartman and Scrivener (1990) reported that numbers of steelhead smolts declined, but cutthroat numbers remained constant following logging of 41% of the Carnation Creek watershed in British Columbia. Holtby et al. (1989) reported that increases in water temperatures in Carnation Creek following logging resulted in earlier outmigration of coho and chum salmon fry, and earlier emigration by coho salmon smolts to the ocean. Similarly, the Alsea Watershed study documented an increase in the number of early (November) coho migrants, although whether the change was due to temperature increases or loss of rearing habitats is uncertain. Nevertheless, these results suggest that small increases in temperature (1–2°C) can result in significant shifts in the timing of important life history events.

Other general effects of logging on fishes identified by Hicks et al. (1991) include reduced growth efficiency, increased susceptibility to disease, increased vulnerability to predation, lower winter survival, blockages to migration from poorly designed culverts, and changes in the structure of fish communities (e.g., increased carrying capacity for age-0 fish but reduced carrying capacity for age-1 and older fish). For Carnation Creek, Holtby et al. (1989) found that outmigrating coho smolts were evenly divided between 1- and 2-year old fish in years preceding logging, but dominated

by 1-year old fish after logging. Such changes potentially increase the vulnerability of specific year classes to environmental fluctuations in both the freshwater and marine environments.

Finally, roads constructed for timber harvest may indirectly affect salmonids by increasing access to previously remote locations. Angling pressure generally decreases with increasing distance from access roads, consequently habitat degradation may be accompanied by increased angling mortality.

## 6.2 Grazing

### 6.2.1 Background

Livestock grazing represents the second dominant land use in the Pacific Northwest, following timber harvesting. In Oregon, Washington, and Idaho combined, over 56.5 million acres of grassland and desert shrubland, approximately 8 million acres of non-federal forest land, and an undetermined amount of federal forest land are grazed by cattle and sheep (Kimerling and Jackson 1985). This acreage represents approximately 41% of the total land base. Rangelands are fairly evenly divided between federal and non-federal lands; federal rangelands total approximately 30.5 million acres (excluding federal forest lands that are grazed) and non-federal rangelands total 34 million acres. Estimates from 1987 indicate that 4.76 million cattle and 0.87 million sheep were produced for sale in Idaho, Oregon, and Washington (Jackson and Kimerling 1993). The majority of rangelands in Washington and Oregon lie east of the Cascade range (Palmisano et al. 1993a), but livestock are also concentrated in the Willamette Valley and Puget Lowlands west of the Cascades, as well as in coastal valleys of Washington and Oregon. No estimates of rangeland area in northern California were readily available. However, from 1966–1980 California produced an average of approximately 4.75 million cattle and 1.0 million sheep annually (Hornbeck et al. 1983), comparable to the livestock production of Oregon, Washington, and Idaho combined. The largest concentrations of livestock in California within the current range of the Pacific salmon occur in the Sacramento-San Joaquin Valley, the Sierra Nevada and Coast Range foothill regions, and in coastal valleys of northern California.

Livestock grazing in the West was already heavy by the mid-to-late 1800s. In 1898, the National Academy of Sciences prepared a report for the Interior Department alleging significant destruction by unregulated grazing in national Forest Reserves (Irwin et al. 1994). By the late 1920s, concern about deterioration of rangelands on national forests was growing (Platts 1991; Heady and Child 1994a). In the 1930s the Forest Service documented widespread degradation of rangeland conditions, concluding that overgrazing had destroyed more than half of all rangelands and that 75% of remaining rangelands were degraded (Heady and Child 1994a). Concern for rangeland conditions prompted Congress to enact the Taylor Grazing Act of 1934, which established 80 million acres of land in grazing districts to be administered by the U.S. Grazing Service, later to become the U. S. Bureau of Land Management (BLM) (Platts 1991; Wilkinson 1992). The percentage of total rangeland in "poor" condition decreased from 36% in 1936 to 18% in 1984, suggesting some improvement in overall range condition (Heady and Child 1994a). However, recent reports have indicated that most riparian areas remain in fair-to-poor condition (USGAO 1991; Chaney et al. 1990). Thus, while upland conditions appear to be improving, riparian areas continue to be degraded. In 1991, BLM began a program to restore riparian areas so that 75% or more are properly functioning by 1997 (Barrett et al. 1993).

Despite the generally poor condition of most riparian areas, the potential for restoration is arguably greater for those damaged by grazing than for those affected by other activities (Behnke 1977; Platts 1991). Recovery of grasses, as well as willows and other woody species can occur within a few years when grazing pressure is reduced or eliminated (Elmore and Beschta 1987; Platts 1991; Elmore 1992). Restoration of fully functioning riparian areas that support a variety of plant species, including older forests of cottonwood and other large tree species, will take considerable time. Nevertheless, many important riparian functions including shading, bank stabilization, sediment and nutrient filtering, and allochthonous inputs may be rapidly restored to the benefit of salmonids, provided the stress of grazing is alleviated and prior damage has not been unduly severe.

### 6.2.2 Effects on Vegetation

Heavy grazing around the turn of the century had significant and widespread effects on upland and riparian vegetation, many of which persist today. Rangelands have experienced decreases in the percentage of ground covered by vegetation and associated organic litter (Heady and Child 1994a). Species composition of plants in upland areas have shifted from perennial grasses towards non-native annual grasses and weedy species (Heady and Child 1994a). East of the Cascade Range, sites that once supported Idaho fescue plant associations are now devoid of native bunchgrasses, which have been replaced with tarweed, gumweed, and other noxious plants (McIntosh et al. 1994). In riparian areas east of the Cascades, willow, aspen, sedge, rush and grass communities have been reduced or eliminated and replaced with annual grasses or sagebrush. Diaries of early trappers in eastern Oregon noted that grasses were as high as seven feet (Wilkinson 1992) and that streams were well lined with willows, aspen, and other woody vegetation (Ogden in Elmore 1992). In eastside meadows, alteration of the vegetation has been so pervasive that little is known about the native vegetation that once inhabited riparian meadow communities. Currently, these meadows are dominated

by Kentucky bluegrass, big sagebrush, and annual brome grasslands (McIntosh et al. 1994). Fleischner (1994) recently reviewed the literature and found numerous examples of changes in species composition, diversity and richness associated with livestock grazing or removal of livestock in western states (Table 6.2.2a).

Much early alteration of rangelands was by settlers who engaged in widespread clearing of grasslands and riparian forests to grow crops, build houses, obtain fuelwood, and increase availability of land for domestic animals (Heady and Child 1994a). Conversion of lands for the purpose of livestock production continues today. Woody shrubs and trees are sometimes removed by chaining and cabling—uprooting of vegetation with anchor chains or cables stretched between tractors—for the purpose of increasing grass production (Heady and Child 1994a). Removal of woody shrubs through chemical application or by mechanical means is also a common range management practice. In addition, suppression of fire on rangelands is responsible for changes in upland vegetation, including encroachment by juniper in many areas of eastern Oregon and Washington (Miller et al. 1989).

Cattle and sheep affect vegetation primarily through browsing and trampling. Grazing animals are selective in what they eat; thus, preferred vegetation types are generally removed first, followed by less palatable species. Heavy, continual grazing causes plants to be partially or wholly defoliated, which can reduce biomass, plant vigor, and seed production (Kauffman 1988; Heady and Child 1994a). Selection of specific plant species may allow other taxa to dominate (reviewed in Kauffman and Krueger 1984; Fleischner 1994). Vegetation may also be lost or damaged through trampling, which tears or bruises leaves and stems, and may break stems of woody plants. Regeneration of some woody vegetation, such as willow, cottonwood, and aspen, is inhibited by browsing on seedlings (Fleischner 1994). Vegetation may also be directly lost when buried by cattle dung. McDiarmid and Watkin (1972) reported that 75% of grasses and legumes under dung piles were killed.

Livestock grazing also influences vegetation through modification of soil characteristics. Hooves can compact soils that are damp or porous and can inhibit the germination of seeds and reduce root growth (Heady and Child 1994a). Changes in infiltration capacity associated with trampling may lead to more rapid surface runoff, lowering soil moisture content and hence the ability of plants to germinate or persist (Heady and Child 1994a). In some instances, trampling may break up impervious surface soils, allowing for greater infiltration of water and helping to cover seeds (Savory 1988 in Heady and Child 1994c). Soils in arid and semi-arid lands have a unique microbiotic surface layer or crust of symbiotic mosses, algae, and lichens that covers soils between and among plants. This "cryptogamic crust" is believed to provide favorable conditions for the germination of vascular plants (St. Clair et al. 1984 in Fleischner 1994) as well as playing important roles in hydrology and nutrient cycling (see sections 6.2.4 and 6.2.7 below). Trampling by livestock breaks up these fragile crusts, and reformation may take decades; Anderson et al. (1982) found recovery of cryptogamic crusts took up to 18 years in ungrazed exclosures in Utah. Finally, livestock may indirectly affect plant species composition by aiding the dispersion and establishment of non-native species, by spreading seeds carried on the fur or in dung (Fleischner 1994).

The effects of livestock grazing on vegetation are especially intense in the riparian zone because of the tendency for livestock to congregate in these areas. Gillen et al. (1984) found that 24–47% of cattle in two pastures in north-central Oregon were observed in riparian meadows constituting only 3–5% of the total land area. Roath and Krueger (1982) reported that riparian meadows that constituted only 1–2% of the total land area accounted for 81% of the total herbaceous biomass removed by livestock. Similar preferences for riparian areas have been observed elsewhere in the west (reviewed in Kauffman and Krueger 1984; Fleischner 1994). Cattle and sheep typically select riparian areas because they offer water, shade, cooler temperatures, and an abundance of high quality food that typically remains green longer than in upland areas (Kauffman and Krueger 1984; Fleischner 1994; Heady and Child 1994a). In mountainous terrain, the preference of cattle and sheep for the riparian zone also appears related to hillslope gradient (Gillen et al. 1984). Heady and Child (1994a) suggest that cattle avoid slopes greater than 10–20%. The intensity of use by livestock in riparian zones exacerbates all of the problems noted above, as well as generates additional concerns. Alteration of flow regime, changes in the routing of water, and incision of stream channels can lead to reduced soil moisture in the floodplain. Many types of riparian vegetation are either obligate or facultative wetland species that are adapted to the anaerobic conditions of permanently or seasonally saturated soils. Stream downcutting and the concomitant lowering of the water table can lead to encroachment of water-intolerant species including sagebrush and bunchgrasses into areas formerly dominated by willows, sedges, rushes, and grasses (Elmore 1992). In addition, flood events may be an important mechanism for seed dispersal throughout the floodplain for woody plants, a function that is diminished as channels are incised.

### 6.2.3 Effects on Soils

Rangeland soils may be compacted by livestock. Soil compaction depends on soil characteristics, including texture, structure, porosity, and moisture content (Platts 1991; Heady and Child 1994a). As a general rule, soils that are high in organic matter, porous, and composed of a wide range of particle sizes are more easily compacted than other soils. Similarly, moist soils tend to be more susceptible to compaction than dry soils, although extremely wet soils may give way and then recover following trampling by livestock (Clayton and Kennedy 1985). The result of soil compaction

is an increase in bulk density (specific gravity) in the top 5–15 cm of soil as pore space is reduced. Because of the loss of pore space, infiltration is reduced and surface runoff is increased, thereby increasing the potential for erosion (see section 6.2.10). The available studies indicate that, in general, compaction increases with grazing intensity, but that site-specific soil and vegetative conditions are important in determining the response of soils to this grazing activity (reviewed in Kauffman and Krueger 1984; Heady and Child 1994a).

Trampling may also displace or break up surface soils. In instances where surface soils have become impervious to water, trampling may increase their ability to absorb water. On the other hand, loosening soils makes them more susceptible to erosion. Heavily pulverized soil (dust) may become hydrophobic, reducing infiltration and increasing surface runoff. In arid and semi-arid climates, the cryptogamic crust has been shown to increase soil stability and water infiltration (Loope and Gifford 1972; Kleiner and Harper 1977; Rychert et al. 1978). Disruption of the cryptogamic crust may thus have long-lasting effects on erosional processes.

Livestock also alter surface soils indirectly by removing ground cover and mulch, which in turn affects the response of soils to rainfall. Kinetic energy from falling raindrops erodes soil particles (splash erosion), which may then settle in the soil interstices resulting in a relatively impervious surface. Livestock grazing can increase the percentage of exposed soil and break down organic litter, reducing its effectiveness in dissipating the energy of falling rain.

#### 6.2.4 Effects on Hydrology

Grazing modifies two fundamental hydrologic processes, evapotranspiration and infiltration, that ultimately affect the total water yield from a watershed and the timing of runoff to streams. Loss of upland and riparian vegetation results in reduced interception and transpiration losses, thereby increasing the percentage of water available for surface runoff (Heady and Child 1994b). Shifts in species composition from perennials to annuals may also reduce seasonal transpiration losses. Reductions in plant biomass and organic litter can increase the percentage of bare ground and can enhance splash erosion, which facilitates clogging of soil pores and decreases infiltration. Similarly, soil compaction (see section 6.2.3) reduces infiltration. Rauzi and Hanson (1966) report higher infiltration rates on lightly grazed plots, compared to moderately and heavily grazed plots in South Dakota. Similar experiments in northeastern Colorado showed reductions in infiltration in heavily grazed plots, but no differences between moderately and lightly grazed plots (Rauzi and Smith 1973). Johnson (1992) reviewed studies related to grazing and hydrologic processes and concluded that heavy grazing nearly always decreases infiltration, reduces biomass, and increases bare soil.

Decreased evapotranspiration and infiltration increases and hastens surface runoff, resulting in a more rapid hydrologic response of streams to rainfall. Some authors have suggested that the frequency of damaging floods has increased in response to grazing; however, there remains uncertainty about the role of grazing in mediating extreme flow events (reviewed in Fleischner 1994).

Reduced stability of streambanks associated with loss of riparian vegetation can lead to channel incision or "downcutting" during periods of high runoff. In naturally functioning systems, riparian vegetation stabilizes streambanks, slows the flow of water during high flow events, and allows waters to spread out over the floodplain and recharge subsurface aquifers (Elmore 1992). Moreover, riparian vegetation may facilitate sediment deposition and bank building (see below) that further increases the capacity of the floodplain to store water, which is then slowly released as baseflow during the drier seasons (Elmore and Beschta 1987). Downcutting effectively separates the stream channel from the floodplain, allowing flood waters to be quickly routed out of the system and leading to lowering of the water table (Platts 1991; Elmore 1992; Armour et al. 1994). Consequently, summer streamflows may decrease even though total water yield increases in response to vegetation removal (Elmore and Beschta 1987). Li et al. (1994) found that streamflow in a heavily grazed eastern Oregon stream became intermittent during the summer, while a nearby reference stream in a similar-sized watershed had permanent flows. They suggested that the difference in flow regimes was a consequence of diminished interaction between the stream and floodplain and resultant lowering of the water table.

#### 6.2.5 Effects on Sediment Transport

Livestock presence in the riparian zone increases sediment transport rates by increasing both surface erosion and mass wasting (Platts 1991); (Marcus et al. 1990); (Heady and Child 1994a). Splash erosion on devegetated, exposed soils may significantly increase sediment loads (Blackburn 1984 in Fleischner 1994). Rills and gullies may form in areas denuded by livestock trails, or grazing, resulting in increased channelized erosion (Kauffman et al. 83a). As gullies expand and deepen, streams downcut, the water table drops, and sediments are transported to depositional areas downstream (Elmore 1992; Fleischner 1994; Henjum et al. 1994). Stream downcutting leads to further desertification of the riparian area and promotes soil denudation and the establishment of xeric flora. This in turn increases the potential for soil erosion. Some evidence suggests that significant channel downcutting in the Southwest occurred prior to the introduction of livestock (Karlstrom and Karlstrom 1987 in Fleischner 1994), however, studies in eastern Oregon and northern California implicate livestock as a major cause of downcutting (Dietrich et al. 1993a, b; Peacock 1994).

Mass wasting of sediment occurs along stream banks where livestock trample overhanging cut banks (Platts and Raleigh 1984 in Marcus et al. 1990; Behnke and Zarn 1976; Fleischner 1994). Grazing also removes vegetation that stabilized streambanks (Platts 1991), causing undercutting and sloughing, increasing sediment loads, filling stream channels, changing pool-riffle ratios, and increasing channel width (Platts 1981, in Fleischner 1994).

### 6.2.6 Effects on Thermal Energy Transfer and Stream Temperature

Riparian vegetation shades streams and thereby regulates stream temperatures. On rangelands east of the Cascades, black cottonwood, mountain alder, and quaking aspen are the dominant deciduous tree species in natural communities, whereas west of the Cascades, black cottonwood, red alder, and big leaf maple are dominant (Kauffman 1988). Shrubby vegetation, such as willows, may also be important sources of shade along smaller streams and in mountainous areas (Henjum et al. 1994; Li 1994), and even tall grasses can provide some measure of shade along narrow first- and second-order streams (Platts 1991).

The removal of riparian vegetation along rangeland streams can result in increased solar radiation and thus increased summer temperatures. Li et al. (1994) noted that solar radiation reaching the channel of an unshaded stream in eastern Oregon was six times greater than that reaching an adjacent, well shaded stream and that temperatures were 4.5°C warmer in the unshaded tributary. Below the confluence of these two streams, reaches that were unshaded were significantly warmer than shaded reaches both upstream and downstream. A separate comparison of water temperatures at two sites of similar elevation and draining comparable land areas found temperature differences of 11°C between shaded and unshaded streams (Li 1994). Warming of streams from loss of riparian vegetation is likely widespread east of the Cascades and may be particularly acute because of low summer flows and a high percentage of cloud-free days.

The effects of riparian canopy in winter on stream temperatures are less well understood and various studies have shown increases, decreases, and no change in water temperature following removal of riparian canopy (reviewed in Beschta et al. 1987). Riparian cover can inhibit energy losses due to evaporation, convection, and long-wave radiation during the winter, and several authors have suggested that removal of vegetation can increase radiative heat loss and facilitate the formation of anchor ice (Beschta et al. 1991; Platts 1991; Armour et al. 1994). This is most likely to occur in regions where skies are clear on winter nights and where snow-cover is inadequate to blanket and insulate streams (Beschta et al. 1987), primarily in mountainous regions and east of the Cascades and Sierra Nevada.

Alteration of stream temperature processes may also result from changes in channel morphology. Streams in areas that are improperly grazed tend to be wider and shallower than in ungrazed systems (reviewed in Platts 1991), exposing a larger surface area to incoming solar radiation (Bottom et al. 1985; Platts 1991). Wide, shallow streams heat more rapidly than narrow, deep streams (Brown 1980). Similarly, wide, shallow streams may cool more rapidly, increasing the likelihood of anchor ice formation. Reducing stream depth may expose the stream bottom to direct solar radiation, which may allow greater heating of the substrate and subsequent conductive transfer to the water.

### 6.2.7 Effects on Nutrients and Other Solutes

Livestock activities can directly affect nutrient dynamics through several mechanisms. The removal of riparian vegetation by grazing can reduce the supply of nutrients provided by organic leaf litter. Livestock may redistribute materials across the landscape. Because riparian areas are favored by cattle and sheep, nutrients that have been ingested elsewhere on the range tend to be deposited in riparian zones, or near other attractors, such as salt blocks (Heady and Child 1994a). The deposition of nutrients in riparian areas increases the likelihood that elements such as nitrogen and phosphorous will enter the stream. Nutrients derived from livestock wastes may be more bioavailable than those bound in organic litter. Elimination of the cryptogamic crust by livestock may also alter nutrient cycling in arid and semi-arid systems. These microbiotic crusts perform the majority of nitrogen fixation in desert soils (Rychert et al. 1978). Loss of these crusts can lead to reduced availability of nitrogen for plant growth, potentially affecting plant biomass in uplands (Fleischner 1994).

Riparian areas play a major role in regulating the transportation and transformation of nutrients and other chemicals (see section 3.9). Altered soil moisture conditions resulting from downcutting stream channels and separation from riparian and floodplain areas changes the quantity and form of nutrients and their availability to aquatic communities. In the anaerobic environments of saturated soils, microbial activity transforms nitrate nitrogen ( $\text{NO}_3$ ) into gaseous nitrous oxide ( $\text{N}_2\text{O}$ ) and elemental nitrogen ( $\text{N}_2$ ) that are liberated to the atmosphere (Figure 6.2.7a) (Green and Kauffman 1989). Under drier soil conditions (oxidizing environments), denitrification does not occur and nitrate-nitrogen concentrations in the soil increase. Because nitrate is negatively charged, it is readily transported by subsurface flow to the stream channel (Green and Kauffman 1989). Thus by altering the hydrologic conditions in the riparian zone, grazing can increase the amount of nitrate nitrogen released to streams. Excessive nitrate concentrations can facilitate algal growth, increase turbidity, and in some cases cause oxygen depletion due to increased biological oxygen demand.

Other elements including manganese, iron, sulfur, and carbon are also dependent on the redox potential of soils. In their reduced form, manganese, iron, and sulfur can be toxic to plants at high concentrations (Green and

Kauffman 1989). Obligate and facultative wetland plant species have special adaptations for coping with these reduced elements that allow them to survive where more xeric plants cannot. Changes in hydrologic conditions resulting from downcutting can therefore modify the form of elements available to plants, modify competitive interactions among plants, and significantly change riparian plant communities.

### 6.2.8 Effects of Vegetation Management

Herbicides, fire, and fertilizers are used in rangeland management to alter vegetation in favor of desired species. If suitable no-spray buffers are protected and limited runoff occurs, the effects on aquatic communities are generally negligible (Heady and Child 1994a). However, the lack of suitable buffers in Pacific Northwest rangelands suggest that reductions in riparian woody plants from herbicides and fire, and enrichment of streams from fertilizer runoff, are likely widespread.

Manipulation of vegetation with chemical treatments may have positive or negative effects on streams. Fertilizer may decrease sedimentation if it increases vegetative cover and it could be useful for stimulating recovery of riparian woody plants. If more sunlight reaches the stream as a result of fire or herbicide use, the water will warm and algae production will increase, increasing the food base for invertebrates and fish. Algae growth is also stimulated by fertilizers reaching the stream. Reduced production of riparian deciduous trees and brush increases sedimentation and reduces undercut and overhanging bank cover and the delivery of leaves and intermediate sized wood to streams. Depending on whether temperature, spawning sites, cover, or food is limiting, these changes may initially hinder or aid salmonid production. Given the lower elevations and higher summer temperatures of many rangelands, the elevated stream temperatures and increased algae production are likely to lead to insufficient nighttime dissolved oxygen levels and afternoon gas supersaturation. Salmonid numbers and production will decrease if the increased invertebrate production is offset by undesirable alterations in the benthos assemblage to less nutritious taxa, reduced cover, increased sedimentation, and lower water quality.

### 6.2.9 Effects on Physical Habitat Structure

Livestock-induced changes in physical structure within streams results from the combined effects of modified hydrologic and sediment transport processes in uplands, and the removal of vegetation within the riparian zone. Platts (1991) and Elmore (1992) review effects of grazing on channel morphology and are the sources of most information presented below. Loss of riparian vegetation from livestock grazing generally leads to stream channels that are wider and shallower than those in ungrazed or properly grazed streams (Hubert et al. 1985; Platts and Nelson 1985a,b in Marcus et al. 1990). Loss of riparian root structure promotes greater instability of stream banks, which reduces the formation of undercut banks that provide important cover for salmonids (Henjum et al. 1994). Furthermore, the increased deposition of fine sediments from bank sloughing may clog substrate interstices, thereby reducing the quality of gravels. Reductions in instream wood reduce the retention of spawning gravels and decrease the frequency of pool habitats. In addition, the lack of structural complexity allows greater scouring of streambeds during flow events, which can reduce gravels available for spawning and facilitate channel downcutting. Figure 6.2.9a illustrates the characteristics of vegetation in functional and dysfunctional riparian zones on rangelands, and the channel modifications that typically result.

### 6.2.10 Effects on Stream Biota

As with forest practices, removal of riparian vegetation by livestock can fundamentally alter the primary source of energy in streams. Reductions in riparian canopy increase solar radiation and temperature, which in turn stimulates the production of periphyton (Lyford and Gregory 1975); Tait et al. (1994) in a study of seven stream reaches in eastern Oregon, reported that thick growths of filamentous algae encrusted with epiphytic diatoms were found in reaches with high incident solar radiation, whereas low amounts of epilithic diatoms and blue-green algae dominated in shaded reaches. Periphyton biomass was found to be significantly correlated with incident solar radiation.

While densities of macroinvertebrates in forested streams typically increase in response to increased periphyton production, the effect of stimulated algal growth in rangeland streams is less clear. Tait et al. (1994) found that biomass, but not density, of macroinvertebrates was greater in reaches with greater periphyton biomass. The higher biomass was a consequence of large numbers of *Dicosmoecus* larvae, a large-cased caddisfly that can exploit filamentous algae. Consequently, any potential benefits of increased invertebrate biomass to organisms at higher trophic levels, including salmonids, may be minimal, as these larvae are well protected from fish predation by their cases. Tait et al. (1994) suggest that in fact these organisms may act as a trophic shunt that prevents energy from being transferred to higher levels.

Evidence of negative effects of livestock grazing on salmonid populations is largely circumstantial, but is convincing nonetheless. Platts (1994a) found that in 20 of 21 studies identified, stream and riparian habitats were degraded by livestock grazing, and habitat improved when grazing was prohibited in the riparian zone. Fifteen of the 21

studies associated decreasing fish populations with grazing. Although they caution that some of these studies may be biased, usually because of a lack of pregrazing data, the negative effects of grazing on salmonids seem well supported. Storch (1979) reported that in a reach of Camp Creek, Oregon, passing through grazed areas, game fish made up 77% of the population in an enclosure, but only 24% of the population outside the enclosure. Platts (1981) found fish density to be 10.9 times higher in ungrazed or lightly grazed meadows of Horton Creek, Idaho, compared to an adjacent heavily grazed reach. Within an enclosure along the Deschutes River, Oregon, the fish population shifted from predominately dace (*Rhinichthys* sp.) to rainbow trout over a ten year period without grazing (Clair and Storch 1983). Platts (1991) cites other examples of improved habitat conditions resulting in increased salmonid populations.

## 6.3 Agriculture

### 6.3.1 Background

Approximately 12 % of the total land area in Washington, Oregon, and Idaho is dry cropland, with an additional 4% devoted to irrigated agriculture (Jackson and Kimerling 1993). Wheat, barley, and hay account for approximately 44% of the total harvested cropland, with fruits, nuts, berries, hops, peppermint, dry peas, and grass seed all contributing significantly to the total acreage (Jackson and Kimerling 1993). Like the other forms of food and fiber production, farming results in massive alterations of the landscape and the aquatic and riparian ecosystems contained therein. In general, the effects of agriculture on the land surface are more severe than logging or grazing, because vegetation removal is permanent and disturbances to soil often occur several times per year. In addition, much agriculture takes place on the historic flood plains or river systems, where it has a direct impact on stream channels and riparian functions. Furthermore, irrigated agriculture frequently requires diversion of surface waters, which decreases water availability and quality for salmonids and other aquatic species (see Section 6.6). Qualitative summaries of the historic effects of agriculture on aquatic ecosystems, have been reported by Cross and Collins (1975), Gammon (1977), Menzel et al. (1984), and Smith (1971).

### 6.3.2 Effects on Vegetation

In the Pacific Northwest, natural grasslands, woodlands, and wetlands have been eliminated to produce domestic crops. For example, in the Willamette Valley of Oregon, the original fire-maintained prairie and floodplain forest were replaced with cropland (Johannessen et al. 1971). Replacement of natural forest and shrubland vegetation with annual crops frequently results in large areas of tilled soil that become increasingly compacted by machinery and are only covered with vegetation for a portion of the year. Commonly, little or no riparian vegetation is retained along streams as farmers attempt to maximize acreage in production. While there is potential to restore agricultural lands to more natural communities, conversions to croplands have by and large been permanent alterations of the landscape.

### 6.3.3 Effects on Soils

Agriculture involves repeated tillage, fertilization, and harvesting of the cropped acreage. The repeated mechanical mixing, aeration, and introduction of fertilizers or pesticides significantly alters physical soil characteristics and soil microorganisms. Further, tillage renders a relatively uniform characteristic to soils in the cropped areas. Although tillage aerates the upper soil, compaction of fine textured soils typically occurs just below the depth of tillage, altering the infiltration of water to deep aquifers. Other activities requiring farm machinery to traverse the cropped lands, and roads along crop margins, causes further compaction, reducing infiltration and increasing surface runoff. Where wetlands are drained for conversion to agriculture, organic materials typically decompose, significantly altering the character of the soil. In extreme cases, the loss of organic materials results in "deflation", the dramatic lowering of the soil surface. Soil erosion rates are generally greater from croplands than from other land uses, but vary with soil type and slope.

### 6.3.4 Effects on Hydrology

The changes in soils and vegetation result in lower infiltration rates, which yield greater and more rapid runoff. For example, forested land may absorb fifty times more water than agricultural areas (Auten 1933). Loss of vegetation increases runoff, peak flows, and flooding during wet seasons (Hornbeck et al. 1970). Reduced infiltration and the rapid routing of water from croplands may also lower the water table, resulting in lower base flows, higher water temperatures, and fewer permanent streams. Typically, springs, seeps, and headwater streams dry up and disappear, especially when wetlands are ditched and drained.

Water that is removed from streams and spread on the land for irrigated agriculture reduces streamflows, lowers water tables, and leaves less water for fish. Often the water is returned considerable distances from where it was withdrawn, and the return flows typically raise salinity and temperature in receiving streams. Extreme examples of this occur in many rivers east of the Cascades and in the Central Valley of California. The flows of these rivers are naturally low in late summer, but the additional losses from irrigation accentuate low flows. The reduced low flows greatly

degrade water quality because the water warms more than normal and causes increased evaporation which concentrates dissolved chemicals and increases the respiration rates of aquatic life.

Streams are typically channelized in agriculture areas, primarily to reduce flooding duration and alter geometry of cropped lands to improve efficiency of farm machinery. Because peak flows pass through a channelized river system more quickly, downstream flood hazards are increased (Henegar and Harmon 1971). When channelization is accompanied by widespread devegetation, the severity of flooding is increased, such as occurred in the Mississippi Valley in 1993. On the other hand, low flows are reduced in channelized streams because of deeper channels and reduced groundwater storage (Wyrick 1968), resulting in dry streams during droughts (Gorman and Karr 1978; Griswold et al. 1978). Reduced flows during summer can limit habitat availability, increase crowding and competition.

### 6.3.5 Effects on Sediment Transport

The loss of vegetative cover increases soil erosion because raindrops are free to dislodge soil particles (splash erosion); splash erosion may also reduce soil permeability, reducing infiltration and increasing overland flow. Agricultural practices typically smooth and loosen the land surface, enhancing the opportunity for surface erosion. When crop lands are left fallow between cropping seasons, excessive erosion can greatly increase sediment to streams. Mass failures are probably rare on most agricultural lands because slopes are generally gentle; however, sloughing of channel banks may occur along riparian zones in response to vegetation removal.

### 6.3.6 Effects on Thermal Energy Transfer

Removal of forests and shrubs for agriculture reduces shading and increases wind speeds, which can greatly increase water temperatures in streams passing through agricultural lands. In addition, bare soils may retain greater heat energy than vegetated soils, thus increasing conductive transfer of heat to water that infiltrates the soil or flows overland into streams. These effects are discussed in greater detail on Section 6.1.6.

### 6.3.7 Effects on Nutrient and Solute Transport

Agricultural practices may substantially modify the water quality of streams. Omernik (1977), in a nationwide analysis of 928 catchments, demonstrated that streams draining agricultural areas had mean concentrations of total phosphorus and total nitrogen 900% greater than those in streams draining forested lands. Smart et al. (1985) found that water quality of Ozark streams was more strongly related to land use than to geology or soil. Exponential increases in chlorine, nitrogen, sodium, phosphorus, and chlorophyll-*a* occurred with increases in percent pasture in streams draining both forested and pastured catchments, and fundamental alterations in chemical habitat resulted as the dominant land use changed from forest to pasture to urban. Stimulation of algal growth by nutrient enrichment from agricultural runoff may affect other aspects of water quality. As algal blooms die off, oxygen consumption by microbial organisms is increased and can substantially lower total dissolved oxygen concentrations in surface waters (Waldichuk 1993). Nutrient enrichment from agricultural runoff has been found to significantly affect water quality in two rivers in interior British Columbia. Die-off of nutrient-induced algal blooms resulted in significant oxygen depletion (concentrations as low as 1.1 mg/l) in the Serpentine and Nicomekl rivers during the summer, which in turn caused substantial mortality of coho salmon.

### 6.3.8 Effects of Fertilizer and Pesticide Use

The two most commonly used agricultural chemicals, herbicides and nitrogen, are frequently found in groundwater in agricultural areas. McBride et al. (1988) report that atrazine is the herbicide most often detected in corn belt groundwater. In Oregon, groundwater nitrogen concentrations at or above health advisory levels were found in Clatsop, Marion, Deschutes, Morrow, Umatilla, Union, and Malheur counties, and elevated levels were reported for Multnomah, Linn, and Lane counties (Vomocil and Hart 1993). Because of the lack of a statistically representative sample of groundwater in the region's agricultural areas, the degree and extent of contamination is unknown.

Unlike native vegetation, agricultural crops require substantial inputs of water, fertilizer, and biocides to thrive. Currently used pesticides, although not as persistent as previously-used chlorinated hydrocarbons, are still toxic to aquatic life. Where biocides are applied at recommended concentrations and rates, and where there is a sufficient riparian buffer, the toxic effects to aquatic life may be minimal. However, agricultural lands are also characterized by poorly-maintained dirt roads and ditches that, along with drains, route sediments, nutrients, and biocides directly into surface waters. Thus, roads, ditches, and drains have replaced headwater streams; but rather than filter and process pollutants, these constructed systems deliver them directly to surface waters (Larimore and Smith 1963).

### 6.3.9 Effects on Physical Habitat Structure

Agricultural practices typically include stream channelization, large woody debris removal, construction of revetments (bank armoring), and removal of natural riparian vegetation. All four reduce physical habitat complexity,

decrease channel stability, and alter the food base of the stream (Karr and Schlosser 1978). Natural channels in easily eroded soils tend to be braided and meander, creating considerable channel complexity as well as accumulations of fallen trees. Large wood helps create large, deep, relatively permanent pools (Hickman 1975), and meander cutoffs; the absence of snags simplifies the channel. Channelization lowers the base level of tributaries, stimulating their erosion (Nunnally and Keller 1979). The channelized reach becomes wider and shallower, unless it is revetted, in which case bed scour occurs that leads to channel downcutting or armoring. Channel downcutting leads to a further cycle of tributary erosion. Richards and Host (1994) reported significant correlations between increased agriculture at the catchment scale and increased stream downcutting. Incised channels in an agricultural region were found to have less wood debris and more deep pools than non-incised channels (Shields et al. 1994).

### 6.3.10 Effects on Stream Biota

Agricultural practices also cause biological changes in aquatic ecosystems. In two states typified by extensive agricultural development and with extensive statewide ecological stream surveys, instream biological criteria were not met in 85% of the sites (Ohio EPA 1990; Maxted et al. 1994). Nonpoint sources of nutrients and physical habitat degradation were identified as causes of much of the biological degradation. Maxted also demonstrated that amount of shading had marked effects on stream temperatures and dissolved oxygen concentrations (e.g., Figure 6.3.10a). In some agricultural stream reaches with riparian vegetation absent and no shading the extremes exhibited in both temperature and DO would preclude the survival of all but the most tolerant organisms. Higher temperatures increase respiration rates of fish, increasing oxygen demand at the same time that oxygen is depleted by stimulated plant respiration at night. During daylight hours, high plant respiration (elevated by greater nutrient concentrations, higher temperatures, and lower flows) may produce gas supersaturation and cause fish tissue damage. Smith (1971) reports that 34% of native Illinois fish species were extirpated or decimated, chiefly by siltation, and lowering of water tables associated with drainage of lakes and wetlands. Although point sources were described by Karr et al. (1985) as having intensive impacts, nonpoint sources associated with agriculture are considered most responsible for declines or extirpations of 44% and 67% of the fish species from the Maumee and Illinois drainages, respectively. declines. Sixty-three percent of California's native fishes are extinct or declining (Moyle and Williams 1990), with those in agricultural areas particularly hard hit. Nationwide, Judy et al. (1984) reported that agriculture adversely affected 43% of all waters and was a major concern in 17%.

Modification of physical habitat structure has been linked with changes in aquatic biota in streams draining agricultural lands. Snags are critical for trapping terrestrial litter that is the primary food source for benthos in small streams (Cummins 1974), and as a substrate for algae and filter feeders in larger rivers Benke et al. (1985) describe the importance of snags to benthos and fish in rivers with shifting (sand) substrates. Such systems, typical of agricultural lands, support the majority of game fish and their prey. Marzolf (1978) estimates 90% of macroinvertebrate biomass was attached to snags. Hickman (1975) found that snags were associated with 25% higher standing crops for all fish and 51% higher standing crop of catchable fish. Fish biomass was 4.8–9.4 times greater in a stream side with instream cover than in the side that had been cleared of all cover (Angermeier and Karr 1984). Gorman and Karr (1978) reported a correlation of 0.81 between fish species diversity and habitat diversity (substrate, depth, velocity). Shields et al. (1994) found that incised channels in agricultural regions supported smaller fishes, and fewer fish species.

On a larger scale, habitat and reach diversity must be great enough to provide refugia for fishes during temperature extremes, droughts, and floods (Matthews and Heins 1987). If refugia occur, fishes in agricultural streams can rapidly recolonize disturbed habitats and reaches. However, loss of refugia, alterations in water tables, simplifications of channels, and elimination of natural woody riparian vegetation symptomatic of agricultural regions creates increased instability and results in stream degradation (Karr et al. 1983).

## 6.4 Urbanization

### 6.4.1 Background

Urban areas occupy only 2.1% of the Pacific Northwest regional land base (Pease 1993), but the impacts of urbanization on aquatic ecosystems are severe and long-lasting. Future projections suggest that urban areas will occupy an increasing fraction of the landscape. From 1982 to 1987, lands devoted to urban and transportation uses increased by 5.2% (123,813 acres) in the Pacific Northwest. In the Puget Sound area, the population is predicted to increase by 20% between 1987 and the year 2000, requiring a 62% increase in land area developed for intense urbanization (Puget Sound Water Quality Authority 1986). As urban areas continue to expand, natural watershed processes will be substantially altered.

Urbanization has obvious effects on soils and natural vegetation that, in turn, affect hydrologic and erosional processes, as well as physical characteristics of aquatic habitats. Urban developments, including roads, buildings, sidewalks, and other impervious surfaces, greatly reduce water infiltration, which alters the routing and storage of water

in the basin. Many of the resulting changes are intended and make the land more amenable to specific human uses (e.g., transportation, human habitation), but other important resource values (e.g., water supplies, fisheries, and wildlife) may be damaged by unintended effects on aquatic ecosystems, including increased peak flows, channel erosion, landslides, pollution, and channelization.

#### 6.4.2 Effects on Vegetation

Urbanization causes severe and permanent alteration of natural vegetation. The total vegetated area in the basin is typically diminished, and replacement vegetation (e.g., lawns, ornamental plants) often requires large quantities of water and fertilizers for growth. In addition, riparian corridors are frequently constricted, disabling or altering riparian processes. The loss of riparian vegetation reduces inputs of large woody debris and smaller organic detritus including leaves. Stream channels and banks are deprived of stability provided by large woody debris and the roots of riparian vegetation.

#### 6.4.3 Effects on Soils

The effects of urbanization on soils can be divided into two phases. During urban construction, significant soil displacement, alteration, and movement occurs associated with grading, filling, and hauling activities. Once land conversion is complete, much of the surface soil is covered with concrete or asphalt. In most residential areas, soils may be exposed, but they are generally altered and fertilized to support domesticated vegetation. Because of this dramatic alteration, the ecological functions that occur in the soil are likely greatly diminished, and these changes are permanent.

#### 6.4.4 Effects on Hydrology

Urbanization can significantly influence hydrologic processes, including increased magnitude and frequency of peak discharges, and reduced summer base flows (Klein 1979; Booth 1991). These changes occur primarily due to increases in the impervious surface and the replacement of complex, natural drainage channels with a network of storm pipes and drainage ditches (Lucchetti and Fuerstenberg 1992). In urban areas, infiltration is reduced as 1) soils are stripped of vegetation, compacted, and or paved; 2) internal draining depressions are graded; 3) subsurface flow is intercepted by drains and discharged to streams; and 4) buildings are erected (Booth 1991). Instead of infiltrating into the soil, storm water is quickly delivered to the channel, resulting in a more episodic flow regime with higher peak flows and reduced base flows. In non-urban areas west of the Cascades, rainfall intensities are lower than the rate of infiltration, and subsurface flows predominate (Dunne et al. 1975). Only a small portion of the watershed contributes overland runoff, the remaining water infiltrates and becomes part of the subsurface regime. In arid and semi-arid eastside systems, overland runoff is more common due to intense rainfall, sparse vegetation, and shallow, less permeable soils. Runoff generally travels quickly from the hillslopes to the channel, and virtually all parts of the watershed contribute to storm runoff. Due to high natural infiltration, evaporation, and transpiration, westside areas are most affected by human activities. Although eastside runoff is primarily overland flow, urbanization increases the efficiency of water delivery to the channel. Culverts and drainpipes are straighter and provide a more direct and more efficient flow to the stream channel.

Increases in storm runoff caused by decreased infiltration also may result in more frequent flood events (Klein 1979). Using a model that incorporated historic storm data for Hylebos Creek, Washington, Booth (1991) found that over a 40 year simulation period, storm flows from an urban area were significantly greater than those from a forested basin. For the fully forested basin, eight floods exceeding the magnitude of a five-year event were simulated for the 40 year period. In contrast, in the urbanized basin, simulated floods equaled or exceeded the discharge of a five-year flood event in 39 years of the 40 years (Figure 6.4.4a).

Water withdrawals for water supply, industry and food processing can alter the flow regimes and quantity and quality of stream water. Muckleston (1993) reports that in populous areas such as Puget Sound, Washington and the Willamette basin, Oregon, public water supply withdrawals total 42% and 84% of the overall withdrawals from surface waters. In the lower Columbia sub-basin, public water supply and industrial usage make up over 80% of total withdrawals. East of the Cascade crest, food processing is generally the most significant industrial use of water though refining primary metals is important locally in the Clark Fork, Kootenai, Spokane and Mid-Columbia sub-basins. The need for water supplies and dependable power has led to numerous impoundments on the major northwest river systems. These reservoirs have altered the natural flow regimes. The flow of the Willamette river, which historically reflected the annual precipitation pattern, now has increased mean summer low flows due to water storage and usage needs of local urban populations. The effect has been an increase in summer water quality.

#### 6.4.5 Effects on Sediment Transport

Loss of vegetation and alteration of soil structure during construction of buildings and roads may increase sediment loading to streams by several orders of magnitude (Klein 1979); however, the effect is likely to be of short duration. Once building and landscaping is complete, surface erosion is reduced, possibly to levels lower than prior to construction because much of the land surface is under concrete or asphalt. Specific effects are likely to vary with degree of urbanization, and whether drainage systems ditches are composed of erodible materials or concrete. Street sweeping and runoff from city streets transports some sediment to storm sewers and ultimately to streams, but the impact of that sediment is negligible. However, contaminants associated with such sediments can have significant impacts on water quality (see Section 6.4.8).

#### 6.4.6 Effects on Thermal Energy Transfer and Stream Temperatures

through the system and avert flood damage. Rip-rap, concrete, and other forms of channel revetment are commonly employed to counteract the increased erosive force associated with higher discharge volumes. In addition, with increased magnitude and frequency of floods in urban streams and rivers, greater within-stream bedload transport occurs and channels become less stable (Bryan 1972; Scott et al. 1986). The rates of disturbance from flood events may accelerate to a point that the stream cannot recover between disturbance events. Lucchetti and Fuerstenberg (1992) noted that urbanized streams take on a clean "washed-out" look as channel complexity is lost. Such stream beds are uniform, with few pools or developed riffles, and with substrates dominated by coarser fractions rather than sand and silts. The lack of large woody debris inputs exacerbates channel simplification (Lucchetti and Fuerstenberg 1992), causing increased bed scour and fill and changing channel hydraulics at a given maximum flow. These highly modified channels generally provide poor habitat for fish.

In unconstrained urban streams, stream channels may become substantially wider and shallower than streams in rural areas due to higher stream energy and increased erosion of streambanks (reviewed in Klein 1979). In other areas, streambed morphology is further modified by channel incision, which leaves exposed, near-vertical channel banks (Lucchetti and Fuerstenberg 1992). In areas near the ocean, this can effectively isolate the estuaries from the surrounding riparian zone and essentially create a non-interacting conduit between upriver areas and the sea. Important interactions between the stream and surrounding floodplain are lost.

#### 6.4.10 Effects on Stream Biota

Biological community structure, abundance and composition is greatly altered by urban impacts on channel characteristics and water quality. Research indicates that stream quality impairment is correlated to the percentage of watershed imperviousness. Impaired water quality becomes noticeable at 8–12% imperviousness and becomes severe above 30% imperviousness (Klein 1979; Pedersen and Perkins 1986; Limburg and Schmidt 1990). In a study of northern Virginia streams, Jones and Clark (1987) found that the taxonomic composition of macroinvertebrates was shifted markedly by urbanization, though development had minor effect on the total insect densities. Relative abundance of Diptera (primarily chironomids) increased at the most developed sites, and more sensitive orders, including Ephemeroptera (mayflies), Coleoptera (beetles), Megaloptera (dobsonflies), and Plecoptera (stoneflies), decreased. The response of Tricotera (caddisflies) was variable. Pedersen and Perkins (1986) showed that a rural stream had twice the functional diversity of an urban stream. Those organisms that persisted were adapted to extreme bed instability.

Fish are also adversely affected by urbanization. Limburg and Schmidt (1990) demonstrated a measurable decrease in spawning success of anadromous species (primarily alewives) for Hudson River tributaries from streams with 15% or more of the watershed area in urban land use. In Kelsey Creek, Washington, urban development resulted in a restructuring of the fish community in response to habitat degradation (Bryan 1972; Scott et al. 1986). Coho salmon appeared to be more sensitive than resident cutthroat trout to habitat alteration, increased nutrient loading, and degradation of the intragravel environment in the stream. In a study of Puget Sound streams, Lucchetti and Fuerstenberg (1992) found that fish assemblages in small urbanized streams have been dramatically altered or lost. They conclude that coho are of particular concern in urbanized areas due to their habitat needs (smaller streams, relatively low velocity niches, and especially large pools). Their study found that as impervious surfaces increased fish species diversity and coho abundance declined and resident cutthroat trout dominated.

Recent studies in the Pacific Northwest suggest that pollution from urban areas may be having insidious effects on anadromous salmonids (Arkoosh et al. 1991; Arkoosh et al. 1994). Arkoosh et al. (1994) found that juvenile chinook salmon that migrate through an urban estuary contaminated with polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) bioaccumulated these pollutants and exhibited a suppressed immune response compared to fish from a non-contaminated rural estuary. In subsequent studies, Arkoosh et al. (1994) exposed juvenile salmon collected from the same two estuaries, as well as their respective releasing hatcheries, to the pathogen *Vibrio anguillarum*. Salmon from the urban estuary exhibited higher mortality rate after 7 days than unexposed fish from the releasing hatchery. In contrast, no difference in mortality rates from this pathogen were observed between the salmon from the uncontaminated estuary and its releasing hatchery. Casillas et al. (1993) found that juvenile chinook exposed to PAHs and PCBs in an urban estuary, showed suppressed immune competence and suppressed growth for up to 90 days after exposure, while juvenile chinook from a non-urban estuary did not develop these symptoms. They suggested that suppressed immune function, reduced survival, and impaired growth, result from increased chemical contaminant exposure of juvenile chinook as they move through urban estuaries on their way to the ocean. The role of contaminants in the overall decline of salmonids is not known; however, these studies indicate that contaminant exposure is perhaps an overlooked cause of mortality for populations that migrate through urbanized streams, particularly since exposure occurs during the physiologically stressful period of smoltification.

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#### 6.4.6 Effects on Thermal Energy Transfer and Stream Temperatures

Changes in riparian vegetation along urban streams can alter the degree of shading provided to the stream, which in turn influences seasonal and diurnal temperature ranges (see Section 3.7). As with other land uses, effects are likely to be greatest for smaller streams that previously had closed canopies. Published examples of changes in temperature regimes caused by urbanization are scarce; however, likely effects are increased maximum temperatures (Klein 1979), greater diel fluctuations, and reduced winter temperatures. Pluhowski (1970 in (Klein 1979)) found that winter stream temperatures in urban areas were 1.5–3°C lower than in non-urbanized streams on Long Island, New York. Although other land use activities alter stream temperatures, in urban areas the loss of riparian function is permanent.

Stream temperatures may also be indirectly affected by changes in hydrology, channel morphology, and the urban microclimate. Klein (1979) suggests that reductions in groundwater inflow may alter natural thermal regimes, resulting in lower winter minimum temperatures and higher summer maximum temperatures. Widening and shallowing of channels caused by greater peak discharges can also influence the rate of energy transfer to and from streams. Air temperatures in urban areas also tend to be warmer than those in surrounding rural areas, which may affect convective and evaporative energy exchange.

#### 6.4.7 Effects on Nutrients and Solutes

The primary changes in nutrient cycling are the type and quantity of materials delivered to the stream channel. Large woody debris and leafy detritus are replaced in importance by nutrient loading from sewage and other sources. Novitzki (1973) reported that effluent from a sewage treatment plant in small town in Wisconsin significantly degraded brook trout habitat downstream of the release point. High nutrient levels from the effluent generally stimulated primary and secondary production; however, under conditions of high temperature and low flow during the summer, heavy oxygen demand from the aquatic vegetation and effluent created critically low dissolved oxygen levels that resulted in fish kills. Omerik (1977) determined that total nitrogen exports from urban areas were second only to agricultural influenced watersheds.

#### 6.4.8 Effects of Chemical Use

Runoff from the urban areas contains many different types of pollutants depending on the source and nature of activities in the area. Wanielista (1978) identifies numerous sources of urban nonpoint source pollution including heavy metals, nutrients (phosphates and nitrates), pesticides, bacteria, organics (oil, grease) and dust/dirt. Heavy metal concentrations found in street runoff were 10-100 times greater than treated wastewater effluent. Grease and oil contributions ranged from 32.8 lb/curb mile/day for industrial areas to 4.9 lb/curb mile/day for commercial areas (Pitt and Amy 1973). Residential areas fell in between (18.6 lb/curb mile/day). Klein (1976) reported that 9% of persons that changed their own engine oil in their cars, disposed of used oil by pouring it into storm drains or gutters. In suburban areas, fertilizers, herbicides, pesticides and animal waste are added to the effluent. For example, Bryan (1972) found that pesticide loadings in runoff from urban areas was three-to-four times greater than for rural areas. In industrial areas, runoff may include heavy metals, PCB's, high pH concrete dust, and other toxic chemicals (Washington State Departments of Ecology 1992). Water quality degrades as a consequence of these pollutants entering our water supplies. Biological oxygen demand is increased with the addition of organic materials, and lethal or sublethal effects may occur with influxes of heavy metals and pesticides (See section 5.1.2).

#### 6.4.9 Effects on Physical Habitat Structure

Urbanization frequently results in gross modification of stream and river channels through road construction, the filling of wetlands, encroachment on riparian areas and floodplains, relocation of channels, and construction and maintenance of ditches, dikes, and levees, and and levees. Urban-related development can influence instream channel structure in a variety of ways. High densities of roads require road crossings, culverts, and other structures that constrain channels and may impede fish migration. Channels are frequently straightened in an attempt to route water quickly

through the system and avert flood damage. Rip-rap, concrete, and other forms of channel revetment are commonly employed to counteract the increased erosive force associated with higher discharge volumes. In addition, with increased magnitude and frequency of floods in urban streams and rivers, greater within-stream bedload transport occurs and channels become less stable (Bryan 1972; Scott et al. 1986). The rates of disturbance from flood events may accelerate to a point that the stream cannot recover between disturbance events. Lucchetti and Fuerstenberg (1992) noted that urbanized streams take on a clean "washed-out" look as channel complexity is lost. Such stream beds are uniform, with few pools or developed riffles, and with substrates dominated by coarser fractions rather than sand and silts. The lack of large woody debris inputs exacerbates channel simplification (Lucchetti and Fuerstenberg 1992), causing increased bed scour and fill and changing channel hydraulics at a given maximum flow. These highly modified channels generally provide poor habitat for fish.

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## 6.5 Mining

### 6.5.1 Sand and Gravel Mining

#### 6.5.1.1 Background

Gravel and sand removal from streams and adjacent floodplains is common in many areas of the Pacific Northwest, particularly near and in low-gradient reaches of rivers west of the Cascade range. In Oregon, permits are required for removal of gravel or sand in excess of 50 cubic yards (OWRRI 1994). The Oregon Division of State Lands has issued over 4,000 permits for gravel removal since 1967 (OWRRI 1994), and between 1987 and 1989, a total 1767 dredge, fill, and aggregate extraction permits were processed, 718 of which were new permits (Kaczynski and Palmisano 1993). Because there are no permit requirements for gravel extraction of less than 50 cubic yards, little information exists regarding the extent of small-scale gravel mining in Oregon. In Washington, large amounts of gravel are associated with glacial deposits and, thus, instream mining has decreased in recent years as extraction has shifted towards glacial and floodplain deposits (Dave Norman, Washington Dept. of Natural Resources, pers. comm.). Nevertheless, mining activity occurs near or in most major rivers west of the Cascade Range (Palmisano, Ellis et al. 1993a, b). Sandecki (1989) reported that production of sand and gravel in California during 1986 exceeded 128 million short tons. The greatest demand for gravel and sand is associated with industrial development, and because of the expense of transporting gravel, mining is most prevalent around urban areas, along highways, or near other major construction sites. Most gravel permit sites in Washington are located near or in urban areas and along the Interstate 5 corridor Figure 6.5.1.1a. In Oregon, gravel production has generally risen between 1940 and 1990; however, gravel mining activity peaked during the 1960s and early 1970s with construction of the John Day, Green Peter, and Foster dams (OWRRI 1994). The majority of gravel mining in Oregon occurs in the Willamette Valley.

Two recent reviews focused on effects of gravel removal on hydrology and channel morphology (Sandecki 1989; Collins and Dunne 1990), and a third focused on effects on salmonids in Oregon (OWRRI 1994). Much of the material contained in this section comes from these three sources.

#### 6.5.1.2 Effects on Geomorphology and Sediment Transport

Removal of sand and gravel from within a stream channel may fundamentally alter the way in which water and sediment are carried through a system (Sandecki 1989), resulting in altered channel morphology, increased instability, accelerated erosion, and changes in the composition and structure of the substrate (Sandecki 1989; OWRRI 1994). The extent of effects depends on many site-specific characteristics, including the geomorphic setting (e.g., stream gradient and nature of bed material), the quantity of material extracted relative to the sediment supply, and the hydrologic and hydraulic conditions within the stream reach.

The effects of gravel mining on the stream environment involve complex interplay between the direct effects of channel modification and altered substrate composition, and the resulting alteration of the erosional and depositional processes, which in turn feed back to cause further changes in channel configuration. Excavation of materials from the stream bed results in immediate changes to channel morphology. Newly created mining pits within streams are highly unstable and tend to migrate up or downstream in response to scouring and deposition of sediments (Lee et al. 1993). Thus, the physical effects of mining pits propagate away from the immediate excavation site (Sandecki 1989; OWRRI 1994). In undisturbed stream channels, coarser materials have a tendency, through hydraulic sorting, to "armor" the stream bed, increasing its resistance to scour (Lagasse et al. 1980). Finer materials work their way into deeper layers. Gravel mining disrupts the armor layer, leaving smaller materials at the bed surface that are more easily mobilized by streamflow; thus bedload movement occurs at lower stream velocities following gravel mining (Sandecki 1989).

Removal of bed material and increased bedload transport can combine to cause downcutting of the stream channel in both upstream and downstream directions (Sandecki 1989; OWRRI 1994). Downstream progression may result from reduced bed material discharge or decreased size of bed material, while upstream progression occurs when gravel extraction increases the river gradient (OWRRI 1994). In some cases, downcutting may occur until sand, gravel, and cobble are completely removed and underlying bedrock is exposed. Downcutting may cause streambanks to collapse, introducing additional sediments into the stream (Collins and Dunne 1990).

Collins and Dunne (1990) recently reviewed case histories on the effects of gravel extraction on downcutting and found several examples where stream channels lowered 4 to 6 meters in response to gravel mining (Table 6.5.1.2a). In several cases, downcutting occurred over several decades; however, in one instance, a drop in bed elevation of 4.5 meters occurred during two flood events that spilled into a large mining pit in Tujunga Wash, California, demonstrating that downcutting can occur rapidly under extreme circumstances. Kondolf and Swanson (1993) reported that gravel extraction below a dam in a Sacramento River tributary resulted in downcutting of more than 5 meters and caused a shift from a highly braided channel to a single channel. Downcutting was severe in part because the dam prevented recruitment of gravels from upstream areas; however, reduced peak flows may have compensated for reduced sediment recruitment by reducing scouring. This example highlights the fact that effects of gravel mining depend on the cumulative effects of other activities in the watershed.

While the effects of off-channel mining are likely to be less direct, they may nevertheless be significant. Frequently, berms, dikes, or revetments are constructed to prevent flood flows from spilling into the excavation area and to reduce bank erosion. These structures prevent lateral migration of the stream channel, which may be important in recruiting gravels from streambanks. During high flows, water is constrained to a narrow channel, which increases the velocity and, hence, the erosive potential of the discharge. Artificially restricted channels, like excavations, may thus lead to degradation of the stream bed. Bar scalping also may affect erosion processes. When bar height is reduced, instream bars may be more prone to erosion when water levels rise (Collins and Dunne 1990).

In summary, the effects of gravel extraction on stream channels may include local adjustments, increased meandering or widening of the stream channel, changes in thalweg configuration, altered pool-riffle sequences, shifts from braided to single-thread channels, and downcutting of the channel bed (Sandecki 1989). Gravel mining may also change the frequency and extent of bedload movements, and increase the amount of suspended fine sediments and turbidity in the water column. Turbidity caused by excavation generally decreases shortly after mining activity ceases; however, turbidity caused by changes in erosion potential may persist until the streambed restabilizes (reviewed in OWRRI 1994). Fine sediments may settle in gravel pits or travel downstream to settle in other slow-water areas. As a result, downstream substrates may be covered with sand, mud, and silt.

#### *6.5.1.3 Effects on Hydrology*

Gravel mining likely has little effect on the total amount of water moving through a stream system; however, it may significantly affect the routing and timing of streamflow. Both downcutting and channel simplification increase the hydraulic efficiency of the stream—water is routed more quickly through the system, especially during periods of high flow (Sandecki 1989). This increased efficiency may reduce the probability of overbank flooding (Collins and Dunne 1990). The elimination of overbank flows prevents the recharge and subsequent release of water from the floodplain, which in turn results in flashier streamflows. In addition, channel downcutting may drain shallow groundwater, and lower the water table (Sandecki 1989; Collins and Dunne 1990; OWRRI 1994). Loss of shallow groundwater storage can reduce summer low flows. It may also lead to loss of riparian vegetation (Sandecki 1989).

#### *6.5.1.4 Effects on Thermal Energy Transfer*

The most likely changes in heat transfer processes resulting from gravel mining are increased heat exchange due to loss of riparian vegetation and alteration of the surface-to-volume ratio of the stream (OWRRI 1994). As discussed in Section 3.8, heat exchange is greater in wide, shallow streams than in narrow deep channels, so temperatures may increase or decrease depending on the specific change in channel morphology that follows gravel extraction. Stream temperatures may also increase due to inputs of heated water from off-channel ponds created by excavation (OWRRI 1994).

#### *6.5.1.5 Effects on Nutrients and Solutes*

We found no published information regarding the effects of instream gravel mining on nutrient cycling or availability. However, if the water table in the floodplain is lowered, floodplain soils may shift from reducing environments to oxidizing environments. Because the form of nitrogen and other solutes depends on the redox potential of the subsurface environment (Section 6.2.7) the availability of nitrate nitrogen and other solutes may increase in response to the oxidizing environment.

#### *6.5.1.6 Effects on Habitat Development*

Most concern regarding the effects of gravel and sand mining on salmonids has focused on spawning habitats. Extraction of gravels may also directly eliminate the amount of gravels available for spawning if the extraction rate exceeds the deposition rate of new gravels in the system. The areal extent of suitable spawning gravels may be reduced in areas where degradation reduces gravel depth or exposes bedrock. In addition, decreases in the stability of streambeds can potentially increase embryo and alevin mortality due to scouring of gravel beds. Deposition of fine sediments downstream of mining activities may reduce the quality of these areas as spawning habitats (Kondolf 1994). Increased fine sediment concentrations in gravel redds impede the flow of oxygen-rich water into the redd slow the removal of metabolic wastes, and hinder the ability of fry to reach the surface (Koski 1966; Everest et al. 1987).

OWRRI (1994) found few studies that address other effects of gravel mining on salmonids; however, qualitative inferences can be drawn from studies of the effects of logging, grazing, and other activities where physical changes are comparable. Salmonids require clean, well-oxygenated waters for successful incubation of embryos and alevins. Mechanical disturbance of spawning beds by mining equipment can potentially lead to high mortality rates of embryos and alevins. The OWRRI (1994) report cites one study where angler wading caused high mortality (43–96%) of alevins with only one to two passes per day. It is likely that gravel mining equipment would be substantially more damaging to incubating embryos and alevins than anglers.

Widening and shallowing of stream channels in response to gravel mining may affect the suitability of stream reaches as rearing habitat for juveniles, particularly during summer low-flow periods, when deeper waters may be critical for survival. Similarly, a reduction in pool frequency may adversely affect migrating adults that require holding pools during their upstream migrations.

#### 6.5.1.7 *Effects on Stream Biota*

Gravel mining can change the abundance and composition of species at lower trophic levels. Increased turbidity reduces light penetration, thereby affecting the production of benthic algae (OWRRI 1994). Aquatic invertebrates, which are an important prey for stream-dwelling salmonids, can be disrupted by disturbance of the substrate during mining (AFS Water Quality Section 1988) or by changing substrate composition or covering of substrate with fine sediments (Hicks et al. 1991). Potential effects on invertebrates include changes in species composition, reduced biomass, and slowed biotic colonization (OWRRI 1994).

Turbidity reduces the reactive distance of fish during foraging (Barrett et al. 1992), clogs or damages buccal or gill membranes, and inhibits normal activities (Hicks et al. 1991; Barrett, Grossman et al. 1992). Sigler and Bjornn (1984) reported that turbidity reduced growth rates of steelhead trout. Other direct effects of turbidity on fish are discussed in greater detail in Section 5.1.2.2. Potential effects on fish assemblages include reduced salmonid production, reductions in total biomass, decreased species diversity, and shifts away from species preferring clear waters towards species that are tolerant of high turbidities (OWRRI 1994). Those species that are most susceptible to increased fine sediments are those that rely heavily on benthic organisms for food or clean gravels for spawning, such as salmon and trout (OWRRI 1994).

### 6.5.2 Mineral Mining

#### 6.5.2.1 *Background*

In the Pacific Northwest and California, mining has had substantial influence on environmental conditions and patterns of human settlement. Mining provided the initial driving force for the ecological transformation of the interior Northwest. The discovery of gold in California and the western interior region in the 1860s catalyzed the large influx of people intent on extracting minerals from streams and mountainous slopes. This provided a wedge into the interior-montane ecosystems from the coastal regions for the cultural transformation of the Pacific northwest. Mining as practiced in the 1800s was especially disruptive to stream ecosystems. Hydraulic mining sluiced hillslopes down into streams, causing siltation of waterways, and degradation of riparian habitats. Extensive cutting of inland forests was undertaken to provide trusses for mine tunnels and wooden viaducts, sluices, and flumes. By 1870, cattle and sheep that had been brought in to feed miners grazed throughout the intermontane Northwest (Robbins and Wolf 1994). Hydraulic mining of the main river valleys of California's Salmon River from 1870 to 1950 is estimated to have produced about 15.8 million cubic yards of sediments (PFMC Habitat Committee 1994). The effects here and elsewhere are still being felt today as sediments and pollutants derived from mine tailings continue to enter streams. The Pacific States Marine Fisheries Commission (1994) reports that mining is responsible for polluting 12,000 miles of rivers and streams in the western United States. Recovery rates of degraded streams vary, ranging from 20 years for areas with no acid drainage to generations for coal mines (with acid drainage), and radioactive phosphate and uranium mines (AFS 1988). Prior to the Surface Mining Control and Reclamation Act of 1977, little thought was given by many to post-mining effects and reclamation efforts (Nelson et al. 1991). However, some states (e.g., Oregon) have enacted more stringent laws regulating certain types of mineral extraction (Jackson and Kimerling 1993).

Minerals are extracted by several methods that can be combined into two broad categories. Surface mining includes dredging, hydraulic mining, strip mining, and pit mining. Underground mining utilizes tunnels and or shafts to extract minerals by physical or chemical means. Surface mining probably has greater potential to affect aquatic ecosystems, although pollution associated with all forms of mining activities may be damaging to aquatic life. Specific effects on aquatic systems depend on the extraction and processing methods employed and the degree of disturbance.

#### 6.5.2.2 *Effects on Geomorphology and Sediment Transport*

Like sand and gravel mining, mineral mining can have a significant effect on channel morphology, depending upon the extraction method. General effects of mining, including increased sedimentation, accelerated erosion, change in substrate, and increased streambed and streambank instability have been discussed in Section 6.5.1.2. Mineral mining can have some additional effects on channel formation and stability. During dredging operations, gravels are removed from rivers but are not hauled away from the channel; gold is extracted and waste gravels are piled along the banks, covering the riparian vegetation. These piles may eventually revegetate but remain unstable and leave banks with a high potential for erosion (Nelson et al. 1991). Dredging for gold in the early 1900s left extensive mine tailings, which continue to constrict stream channels and serve as sediment sources (McIntosh et al. 1994). In the Grande Ronde river of eastern Oregon, extensive alteration of the river is evident in the upper headwater reaches. McIntosh et al.

(1994) report that many upriver sections currently flow underground through the rubble left behind from mining earlier in the century. Hydraulic mining, which involves washing of unconsolidated ore-bearing alluvial gravels out of river banks or from hillslope areas down into the river, is uncommon today, yet effects are still being propagated throughout many river systems from long-abandoned operations (AFS 1988). Several forms of mining (strip, open-pit, quarry) remove the vegetation and topsoils from the site creating the potential for erosion and increased sedimentation. If topsoils are not retained to cover mine spoils, revegetation may be inhibited for extended time periods, especially if mine spoils are acidic (Butterfield and Tueller 1980; Fisher and Deutsch 1983).

#### 6.5.2.3 *Effects on Hydrology*

Mineral mining may alter the timing and routing of surface and subsurface flows. Surface mining may increase streamflow and storm runoff (Sullivan 1967; Collier et al. 1970), as a result of compaction of mine spoils, reduction of vegetated cover, and the loss of organic topsoils, all of which reduce infiltration (Nelson et al. 1991). Merz and Finn (1951) in Nelson et al. (1991) reported infiltration rates of 452.1 cm/h on undisturbed soils versus 43.2 cm/h on adjacent graded spoils banks. Lower infiltration rates increase overland runoff and streamflow, particularly during storm events. Increasing flows may cause channel adjustments, including increased width and depth. Pit and strip mining may also affect groundwater by physical disruption of aquifers (Nelson et al. 1991). Large amounts of water are needed for processing mining products, and in arid regions east of the Cascades, withdrawals for mining may significantly affect the limited water supplies. Lindskov and Kimball (1984) estimated that extraction of 400,00 barrels of oil annually from oil shales in Utah, Colorado and Wyoming would require 86 million m<sup>3</sup> of water per year, which would be pumped from groundwater aquifers (Nelson et al. 1991).

#### 6.5.2.4 *Effects on Thermal Energy Transfer and Stream Temperature*

Dredging and other mining practices may cause loss of riparian vegetation and changes in heat exchange, leading to higher summer temperatures and lower winter stream temperatures (McIntosh et al. 1994). Bank instability can also lead to altered width-to-depth ratios, which further influences temperature (see Section 3.7).

#### 6.5.2.5 *Effects on Nutrients and Pollutants*

No published information was found regarding the effects of mineral mining on nutrient cycling or availability. However, surface mining and dredging likely affect inputs of nutrients where vegetation is removed or buried, and may increase nutrient spiraling length within streams where structure is simplified and nutrient retention is diminished.

Perhaps the most important effect of mining on aquatic ecosystems is contamination of surface waters from mine spoils. Acidification of surface waters by mining operations is generally considered to be the most serious consequence of mining. Water is acidified due to oxidation of iron-containing waste products, which are then carried with runoff into local drainages (Nelson et al. 1991). In the western USA, much of the mineral recovery occurs from granitic deposits containing pyrite (Nelson et al. 1991). When exposed to atmospheric oxygen, pyrite is readily oxidized in water to produce sulfuric acid, which lowers the pH of mine spoils. Other metallic sulfides including chalcopyrite (CuFeS<sub>2</sub>), sphalerite (ZnS), galena (PbS), and greenockite (CdS), undergo similar acid-generating processes (Nelson et al. 1991). Reductions in pH significantly increase the mobility of certain heavy metals (e.g., arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, nickel, zinc) by altering their chemical form, particularly if drainage is through waste piles. High acidity also facilitates formation of ferric hydroxide (FeOH<sub>3</sub>), a noxious precipitate often called "yellow boy" (Nelson et al. 1991). The process of acidification is ongoing, and increased soil acidity converts metals into forms that are more bioavailable.

Nelson et al. (1991) reviewed the literature and found several examples of pollution associated with mine wastes. Levels of copper and zinc were 4–10 times above background levels 560 km from the major source of contamination on the Clarks Fork River, Montana (Johns and Moore 1985). Duamie et al. (1985) reported that copper and zinc loadings from an abandoned mine in Montana were 13.6 kg/d and 1.6–145.5 kg/d, respectively. Acid mine drainage and copper loadings of 41–147 kg/d have been documented for Panther Creek, Idaho (Reiser 1986).

Heap leach mining is a form of open-pit mining used to extract gold from low-grade ore deposits. Extracted ore is crushed and placed into piles called pads where a dilute solution of sodium-cyanide (NaCN) is sprayed over the ore. As the cyanide solution percolates through the pad, gold is bonded to solutes and is collected in catch basins. With further processing, termed flotation, the gold is recovered. Cyanide is a well known toxicant, and any that leaches into local streams is potentially lethal to all aquatic organisms.

#### 6.5.2.6 *Effects on Physical Habitat Structure*

The effects of surface mineral mining on physical structure of salmonid habitats are similar to the effects of gravel and sand mining. Elevated levels of erosion increase sedimentation which in turn affects the structure and composition of instream gravel beds. Spaulding and Ogden (1968) estimated that hydraulic mining for gold in the Boise

river basin, Idaho produced 116,500 tons of silt in 18 months. They also reported that dredging in the Salmon River produced enough silt to cover 20.9 km of stream bottom with 0.16 cm of silt every 10 days and reduced salmon spawning by 25%. Other effects of increased sedimentation include shallowing and widening of channels and reduction in pool frequency.

Dredging and placer mining practices have significantly altered the stability of habitats for fish and other organisms. An unnatural forced meander pattern was created along some sections of the Crooked River in Idaho, while another section was straightened. All along these disturbed sections, meadows and riparian vegetation were lost as a result of gold dredging (Nelson et al. 1991).

#### 6.5.2.7 Effects on Stream Biota

Aquatic communities are affected by mining activities primarily through the alteration of physical processes (e.g., increased sediment inputs, greater channel instability, and simplification of channel structure) and chemical characteristics (e.g., acidification, heavy metals). Toxic effects of metals and acid can affect growth, reproduction, behavior, and migration of salmonids, resulting in the loss of sensitive species, changes in productivity, and alterations in population structure (AFS 1988). Increased turbidity reduces light penetration and decreases production of benthic algae (Nelson et al. 1991). Acidification of surface waters precipitates ferrous hydroxide, further decreasing benthic algal production and degrading macroinvertebrate habitat.

Stream acidification affects organisms that are sensitive to low pH, including salmonids. Salmonids experience reduced egg viability, fry survival, growth rate, development of pigmentation, ossification and heart rate in brook trout exposed to low pH (Trojnar 1977; Nelson 1982). Johnson and Webster (1977) have reported that spawning brook trout avoid areas of low pH, and speculate that recruitment is likely affected. Reduced numbers and diversity of benthic invertebrate taxa were found below an abandoned gold and silver mine on Coal Creek, Colorado (Reiser et al. 1982). Ephemeropterans (mayflies), plecopterans (stoneflies), and trichopterans (caddisflies) were found most sensitive to lowered pH in a study by Roback and Richardson (1969).

Lowered pH also enhances the availability and toxicity of heavy metals or metaloids. Arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, and zinc, are all toxic to fish, affecting growth, metabolism, respiration, reproduction, and numerous other biological functions (reviewed in Nelson et al. 1991). These substances may act singly, in combination, synergistically, or antagonistically (to reduce toxicity). Since many of these metals tend to bioaccumulate, increased toxicity is seen in higher-level trophic organisms for a given "background level" in surface waters. Numerous studies have developed LD<sub>50</sub> levels for these toxicants using numerous invertebrate and fish (Table 6.5.2.7a) test organisms. Other studies have also shown that continuous exposure to sublethal levels may produce effects that are just as important for determining ultimate species survival in the affected habitat (EPA 1986).

Effects of chronic pollution from mine wastes have been documented for several streams in the west. Mining wastes containing arsenic, cadmium, copper and zinc have been contaminating the Clark Fork River in Montana for more than 125 years. These metals have resulted in elevated metal concentrations in stream biota (Woodward et al. 1993) and are believed to be affecting benthic invertebrate communities and trout productivity in the river (Pascoe et al. 1993). Laboratory experiments in which rainbow trout fry were exposed to metal concentrations in water and food comparable to those in the Clark Fork indicate that uptake through the diet was the more important source of exposure (Woodward et al. 1993). Exposed fish experienced reduced growth and survival compared to control fish. Other examples of exposure of salmonids and other aquatic organisms to pollution from mine wastes are reviewed in (Nelson et al. 1991).

### 6.6 Effects of Hydroelectric Dams

Hydroelectric dams have contributed substantially to the decline of salmonids in the Pacific Northwest, particularly anadromous stocks in the Columbia, Snake, and Sacramento River systems. The Northwest Power Planning Council (NPPC) estimates that current annual salmon and steelhead production in the Columbia River Basin is more than 10 million fish below historical levels, with 8 million of this annual loss attributable to hydropower development and operation (NPPC 1987). They conclude that approximately half of these losses occur during fish passage through the mainstem projects below Chief Joseph Dam (upper Columbia River) and Hells Canyon Dam (Snake River), and the remaining 4 million in losses are due to the restriction of the fishes' range caused by dams. Access to approximately 55% of the total basin area and 33% of the linear stream miles has been blocked by dams (Thompson 1976; PFMC 1979).

Dams influence salmonids and their habitats in a variety of ways. They impede migration of juvenile and adult fish, delaying migration (Raymond 1979) and thereby increasing the duration of exposure to predators. Juvenile or adult fish that pass through turbines may be killed outright (Bevan et al. 1994a) or may be injured or disoriented, becoming

easy prey for aquatic and terrestrial predators. Attempts to bypass dams through barging or trucking may stress fish and increase disease transmission among individuals, which ultimately may reduce survival (Matthews et al. 1988).

Hydroelectric operations alter natural flow regimes, including daily and seasonal flow patterns. Unnaturally large daily fluctuations in flow occur downstream of dams during peaking operations. Seasonal flow fluctuations tend to be dampened, with water stored during periods of high flow in the winter or spring and released in summer when natural flows are lower (Marcus et al. 1990). These changes can affect migratory behavior of juvenile salmonids. Water-level fluctuations associated with hydropower peaking operations may reduce habitat availability, inhibit the establishment of aquatic macrophytes that provide cover for fish, and in some cases result in stranding of fish or desiccation of spawning redds (Palmisano et al. 1993a). The impoundment of water behind dams creates slackwater environments that are less favorable to salmonids, and more favorable to certain predators of salmonids, including northern squawfish (Faler et al. 1988). The lower and mid-Columbia River has been changed from a free-flowing river to a series of ponded reaches, with little fast water.

Hydroelectric dams also modify sediment transport, natural temperature regimes, and the concentration of dissolved gases. Water storage at dams may prevent flushing flows that are needed to scour fine sediments from spawning substrate and move wood and other materials downstream. Behind dams, suspended sediments settle to the bottoms of reservoirs, covering coarser substrate and depriving downstream reaches of needed sediment inputs (Marcus et al. 1990). The reduction in sediments can result in changes in channel geometry (Sedell and Everest 1991). Reservoirs also modify temperature regimes in streams and rivers. Below larger reservoirs that thermally stratify and that have hypolimnetic discharges, seasonal temperature fluctuations generally decrease; temperatures are cooler in the summer as cold hypolimnetic waters are discharged, but warmer in the fall as energy stored in the epilimnion during the summer is released. Finally, dams have resulted in changes in concentrations of dissolved oxygen and nitrogen concentrations (Bevan et al. 1994a). Behind dams, slow-moving water has lower dissolved oxygen levels than in faster, turbulent waters. Water that spills over dams entrains air, and supersaturation of dissolved gases results. Supersaturation of nitrogen gas can cause gas bubble disease in salmonids, resulting in mortality, or weakening fish such that they become more vulnerable to predation (Parametrix 1975; Blahm et al. 1975).

An exhaustive review of effects of dams on salmonids is beyond the scope of this document. A more thorough discussion of effects of dams on endangered salmonids in the Columbia Basin can be found in the recovery plan for Snake River salmon (Bevan et al. 1994a).

## 6.7 Effects of Irrigation Impoundments and Withdrawals

Damming and diversion of streams and rivers for agricultural purposes began in earnest in the mid-1800s as settlers moved into the region (Palmisano et al. 1993). In the Pacific Northwest, withdrawals for agriculture (crop irrigation and stock watering) currently account for the vast majority (80-100%) of offstream water uses in all major sub-basins east of the Cascades (Muckleston 1993) and in the upper Klamath Basin. In addition, agriculture accounts for 62% of offstream water use in the coastal basins of Oregon, and 28% of the use in the Willamette Valley.

Water for irrigation is withdrawn in several ways. For major irrigation withdrawals, water is either stored in impoundments or diverted directly from the river channel at pumping facilities. Individual irrigators commonly construct smaller "push-up" dams from soil and rock within the stream channel, to divert water into irrigation ditches or to create small storage ponds from which water is pumped. In addition, pumps may be submerged directly into rivers and streams to withdraw water.

Many of the effects of irrigation withdrawals on aquatic systems are similar to those associated with hydroelectric power production, including impediments to migration, changes in sediment transport and storage, altered flow and temperature regimes, and in some cases water level fluctuations. In addition, aquatic organisms may be affected by pollutants from agricultural runoff and reduced assimilative capacity of streams and rivers from which substantial volumes of water are withdrawn. Alterations in physical and chemical attributes in turn affect many biological components of aquatic systems including vegetation within streams and along reservoir margins, as well as the composition, abundance, and distribution of macroinvertebrates and fishes.

### 6.7.1 Fish Passage

For many early irrigation dams, no fish passage facilities were constructed, resulting in the loss of several significant salmon runs. For example, irrigation dams in the Yakima Basin blocked sockeye runs estimated at 200,000 adult fish (Palmisano et al. 1993). At some older irrigation impoundments (e.g., the Savage Rapids Dam on the Rogue River in Oregon), adult passage is hindered by poorly designed fish ladders. Smaller instream diversions may also impede the migrations of adult fish or cause juveniles to be diverted into irrigation ditches. Salmonid juveniles and smolts are also lost through entrainment at unscreened diversions or impingement on poorly designed screens.

### 6.7.2 Flow Modifications and Water-Level Fluctuations

The volume of water diverted for agriculture is substantial. Muckleston (1993) reports that withdrawals in the Snake River Basin total approximately 45,000 acre feet per day (equivalent to approximately 22,500 cfs); because this value is an annual average, daily diversions during the peak irrigation season are likely much higher. Diversion from individual rivers may also be great. For example, the Wapato Canal on the Yakima River has a capacity of 57 m<sup>3</sup>/s (2,000 cfs), with operation usually extending from March to mid-October (Neitzel et al. 1990).

Irrigation withdrawals affect both the total volume of water available to fish and the seasonal distribution of flow. Dams for irrigation typically store water during periods of high runoff in the winter or spring, and release water during the summer when flows are naturally low. Consequently, these impoundments tend to moderate streamflows, reducing winter and spring peak flows. Most direct diversions from rivers occur from spring to fall, during the peak growing season of agricultural crops. Because irrigation of crops coincides with periods of maximum solar radiation, evapotranspiration losses are greater than would occur under normal rainfall-runoff regimes, resulting in reduced summer flows in streams and rivers.

Changes in the quantity and timing of streamflow alters the velocity of streams which, in turn, affects all types of aquatic biota. Water velocity is a major factor controlling the distribution of periphyton and benthic invertebrates in streams (Hynes 1970; Kennedy 1967; Gore 1978; Horner 1978). At low velocities, diatom-dominated periphyton communities may be replaced by filamentous green algae (McIntire 1966). In western Washington streams, periphyton growth rates increased as velocity increased up to 0.1 m/s (Gore 1978); however, as velocities increase above that level, erosion of periphyton exceeds growth. Reduced velocity may eliminate invertebrate species that require high velocities (Trotzky and Gregory 1974). The abundance and composition of fish species may also be determined by the water velocity in their environment (Powell 1958; Priimachenko 1961; Fraser 1972). Changes in velocity influence incubation and development of eggs and larval fish by affecting oxygen concentrations within the gravel (Silver et al. 1963). Reduced water velocities in the Columbia River, which are in part a result of agricultural diversions, may delay downstream migration of salmon smolts. If temperatures become excessively warm, smolts may discontinue migration and revert to a pre-smolt physiology (Ebel 1977). Survival of these holdovers (fish delaying seaward migration for a year or more) is only about 20% (Adams et al. 1975), and very few may survive to return as adults (CRFC 1979).

Where irrigation water is withdrawn from smaller streams, seasonal or daily flow fluctuations may affect fish, macroinvertebrates in littoral areas, aquatic macrophytes, and periphyton (reviewed in Ploskey 1983). Lowered water levels may concentrate fish, which potentially increases predation and competition for food and space (Aggus 1979). Fluctuating water levels may delay spawning migrations, impact breeding condition, reduce salmon spawning area (Aronin and Mikheev 1963; Beiningen 1976); dewater and expose developing embryos, strand fry (CRFC 1979), and delay downstream migration of smolts. Water level fluctuations in reservoirs also reduce the density of bottom-dwelling organisms (Fillion 1967; Stober et al. 1976; Kaster and Jacobi 1978) through stranding, desiccation, or exposure to freezing temperatures (Powell 1958; Kroger 1973); (Brusven and Prather 1974). In the littoral zone, frequent changes in water level can eliminate aquatic macrophytes that provide habitat for fish (Munro and Larkin 1950; Aas 1960). Loss of periphyton (attached algae) in the stream margins because of desiccation has been observed below hydroelectric dams (Neel 1966; Radford and Hartland-Rowe 1971; Kroger 1973), and may occur along the margins of streams below pumping facilities. Reductions in periphyton production may affect other levels in the food web, particularly in large, unshaded rivers, where periphyton can be an important energy source.

### 6.7.3 Changes in Sediment Transport

Irrigation withdrawals and impoundments can affect the quantity of sediments delivered to streams and transported down river. In general, siltation and turbidity in streams both increase as a result of increased irrigation withdrawals because of high sediment loads in return waters. Unlined return canals contribute heavier silt loads than lined canals or subsurface drains (Sylvester and Seabloom 1962). Turbidity in the Wenatchee River doubled over a 45-year period, because of increased agriculture and other human activities (Sylvester and Ruggles 1957). Once in the stream channel, the fate of sediments depends on hydrologic conditions. In systems where total water yield or peak discharge are reduced, sediments may accumulate in downstream reaches, affecting the quality of salmonid habitats. In the Trinity River in California, extreme streamflow depletion (85-90% of average surface runoff) has allowed sediments to accumulate downstream, covering gravels and filling in pools that chinook salmon use for spawning and rearing, respectively (Nelson et al. 1987). The lack of flushing flows during the winter has exacerbated this problem. In other systems, concentrations of suspended sediments below irrigation impoundments may be lower because lower water velocities allow sediments to settle (Sylvester and Ruggles 1957). The deposition of coarse, gravel sediments may be essential for developing high quality spawning gravels downstream of impoundments. Downstream reaches may become sediment starved, and substrate are dominated by cobble and other large fractions unsuitable for spawning.

Iwamoto et al. (1978) report that algae, phytoplankton, zooplankton, benthos, aquatic insects, and fish are all adversely affected by suspended and shifting sediments. In addition, sediments deposited into reservoirs, coupled with

reduced streamflows, may improve habitat for intermediate hosts of several fish parasites. The impacts of suspended sediments, turbidity, and siltation are discussed in greater detail in Section 5.1.2.2.

#### 6.7.4 Changes in Stream Temperature

Irrigation impoundments and withdrawals may increase water temperatures by increasing the surface area of rivers (i.e., reservoirs), reducing discharge volume, and returning heated irrigation waters to streams. In systems with irrigation impoundments, the seasonal thermal regime may also be altered. Reservoirs allow heating of surface waters that, depending on whether releases are from the epilimnion or hypolimnion, can result in increased or decreased temperatures. Below Lost Creek Dam on the Rogue River in Oregon—a multipurpose dam from which irrigation waters are withdrawn—temperatures decreased during summer due to hypolimnetic discharges, but increased during the autumn and winter as water that had been heated during the summer was released (Satterthwaite et al. 1992). The increases in fall and winter temperatures accelerated embryonic development of chinook salmon, resulting in earlier emergence. Typically, return flows of surface water from irrigation projects are substantially warmer after passage through the canals and laterals common to irrigated agriculture (Sylvester and Seabloom 1962). The degree to which water temperatures are affected by withdrawal of irrigation water ultimately depends on the proportion of water removed from and returned to the system and on the seasonal hydrologic regime. Water withdrawals in years of low flow are likely to have greater thermal effects on the fishes and other aquatic biota compared with similar withdrawals during years of high flow.

#### 6.7.5 Changes in Dissolved Oxygen

Dissolved oxygen (DO) concentrations may decrease in both summer and winter in systems with irrigation withdrawals or impoundments. During summer, high solar radiation and warm air and ground temperatures combine to raise the water temperature of irrigation return flow, which diminishes the ability of water to hold DO. Increased water temperatures of irrigation return flows have been shown to reduce DO levels in the Yakima River (Sylvester and Seabloom 1962). Low summer flows can allow greater diel temperature fluctuations, which may exacerbate reductions in DO (McNeil 1968). In addition, higher concentrations of nutrients associated with irrigation returns may reduce DO by increasing biochemical oxygen demand. The extent and period of reduced DO concentrations depends on the quantity of water withdrawn and the quality of the return flow. In winter, low DO levels may occur in irrigation impoundments that have been drawn down. Fish kills can occur through anoxia if lowered water level facilitates freezing, which in turn inhibits light penetration and photosynthesis (Ploskey 1983; Guenther and Hubert 1993).

#### 6.7.6 Impacts of Impoundment and Water Withdrawal on Disease

Impoundment and water withdrawal for off-stream use may facilitate disease epizootics in salmonids by altering temperature regimes, lowering water levels, reducing flow velocities, creating habitat for intermediate hosts of parasites, and concentrating organisms, thereby facilitating the transmission of certain pathogens. Pathogen virulence and salmonid immune systems are greatly affected by water temperature (reviewed in Section 4.3.3); thus increasing temperatures by impoundment, flow reduction, or return of heated irrigation waters will affect disease susceptibility and prevalence in fish populations. Becker and Fujihara (1978) emphasize that extended periods of warm temperature and low flow increase the epizootiology of *F. columnaris* in Columbia River fish populations, and they warn that increasing withdrawal of Columbia River water for offstream use increases the potential for disease. Bell (1986) suggests that fish populations inhabiting lakes and reservoirs tend to experience more disease epizootics than fish species found in free-flowing rivers. Diseases in impoundments generally occur as a result of widespread parasite infections (Bell 1986). Decreasing water depth may provide additional habitat for intermediate hosts of parasites. Snail populations, as well as parasitic trematodes that use snails as intermediate hosts (e.g., *Diplostomum* and *Posthodiplostomum*), are more abundant in shallow waters (Hoffman and Bauer 1971). Consequently, reductions in flow may increase the likelihood of parasite epidemics. Finally, return flows from irrigated fields may transport parasitic nematodes and viruses from infested fields into streams (USBR 1976), thereby increasing the potential for epidemics.

### 6.8 River, Estuary, and Ocean Traffic (Commercial and Recreational)

Within a few decades of settlement, estuaries and low gradient rivers of western California, Oregon, and Washington were channelized; eventually even the Columbia was tamed. Major rivers and estuaries of the Pacific Northwest were altered fundamentally, first for riverboat navigation and then for log rafting, barges, and ports (Maser and Sedell 1994). Navigation channels and pools continue to be maintained by dredging, removing snags, installing revetments, and operating locks and dams. Consequently, salmonids evolved in rivers and estuaries much different from what we now see in most of the Pacific Northwest.

What once was an incredible complex of channels, islands, bays, and wetlands connected with the sea are now highly simplified conduits. These complex mazes of shifting channels and bars laden with enormous snags and jams

impeded navigation, but they were a haven for resident and migrating salmonids. Braided channels under gallery forests and flowing through alluvial plains with high water tables had abundant inflows of cool ground water during the summer. Water was stored in extensive floodplains (instead of behind dams) during the wet season and entered the channel via subsurface flows during the dry season. Such a network of essentially small, partially shaded rivers offered much more productive rearing habitat than the present navigation channels. What were nutrient-rich sloughs in the summer became rearing ponds in the winter if protected from high velocity flows. These changes are reflected in the 80% reduction in the number of upper mainstem Willamette River channels (Sedell and Froggatt 1984)

High flows that once signaled migrations, offered passage over falls, and transported smolts rapidly to the estuaries have been moderated to facilitate the year-round boat traffic. For example, a 1938 Oregon statute directs that a year-round minimum of 6,000 cfs be maintained in the Willamette River at Salem (Muckleston 1993). As a consequence, numerous flood control reservoirs are operated to ensure this flow. The harbors, docks, and marinas offer salmonids some limited cover, but only to species that can tolerate high loads of wood preservatives, petroleum, and organic wastes, as well as human predators. Noise pollution from boat traffic in estuaries and the open ocean may disrupt the navigation and communication of sharks and toothed whales and thereby indirectly affect salmon by effectively eliminating the chief natural predators of seals and sea lions that feed on salmon.

These channel changes have markedly altered the abundance and distribution of salmonids by making the physical habitat less suitable to resident and migrating fish. Losses of these salmon produced losses of particular life-history strategies from the population. Moreover, because these low gradient habitats were also among the most productive freshwater areas for salmon, their degradation has resulted in the loss of more fish than higher gradient reaches of similar size.

## 6.9 Wetland Loss/Removal

In 1989, Congress directed the Secretary of the Interior to assess the estimated total acreage of wetlands in each State in the 1780s and in the 1980s. The study (Dahl 1990) estimated that approximately 221 million acres of wetland functioned in the conterminous United States in the 1780s and that 53% of those acres had been lost by the 1980s. Wetlands lost during this period included 4,546,000 acres in California, 491,300 acres in Idaho, 868,100 acres in Oregon, and 412,000 acres in Washington. This lost acreage meant wetland area decreased from 4.9% to 0.4% of the land area in California, 1.6% to 0.7% of the land area in Idaho, 3.6% to 2.2% of the land area in Oregon, and 3.1% to 2.1% of the land area in Washington. These losses changed the function of ecosystems at the landscape scale because wetlands affect the transport and character of water in watersheds, lakes, and streams.

Wetlands provide a moderated climate compared to the adjacent uplands (cooler in summer and warmer in winter) because of the ground water (at relatively constant temperature) supplied to the site and the microclimate that develops within the vegetation occupying the wetland. Activities that modify the ground water supplied to the site, or modify the plant community, can impair the wetland's ability to moderate climate. Wetlands typically occur as a transition between upland and aquatic ecosystems, for example, at the edge of streams or between the stream and the adjacent valley walls. Wetlands require the surplus water that distinguishes them from uplands (Federal Register 1980). Because wetlands may be only slightly wetter than adjacent upland, they are often targeted for drainage—either by ditching or tiling. These activities change the timing and duration of wetness of the site and modify or impair the wetland's functions. Diking may cause wetlands to be drier where the dike prevents floodwater from entering the wetland. Diking also may eliminate some functions performed by the wetland, for instance, floodwater storage; however, most wetlands do not exclusively depend on floodwater for their existence. Consequently, diking may not totally eliminate other normal wetland functions (as described below). Building, paving, or other permanent changes to the wetland's surface usually eliminate the majority of its functions, although some functions (e.g., floodwater storage) may continue at the site. Wetlands perform several functions related to hydrology, water quality, and habitat; these functions ultimately support salmonids.

### 6.9.1 Wetlands and Hydrology

Wetlands store water during runoff events, thereby reducing flood volumes and flood stages downstream. Further, floodwaters slow as they move into wetlands, reducing damage associated with scour and erosion caused by high velocity flows and allowing sediments, particulate organic matter, and other materials to be deposited in the wetland. Water quality improves with deposition of sediments, and some dissolved materials are either trapped within sediment deposits or utilized by vegetation and organisms in the wetlands. Movement of water through the wetland may also redistribute organic and inorganic particulates as well as import or export plant propagules or organisms. Infiltration of the flood waters into wetland soils supports other wetland functions, such as nutrient cycling, the retention and processing of elements and compounds, and the support of microbial communities adapted to survival in anaerobic conditions. And finally, because of unique hydrologic characteristics and soils, wetlands support unique floral and

faunal communities. Wetlands are an integral component in the hydrologic cycle locally and of the habitat provided by the total watershed.

Novitzki (1979) reported that wetlands had a pronounced influence on flood peaks and seasonal distribution of streamflow: in Wisconsin, flood flows were 80% lower in watersheds with 40% lake-and-wetland area than in watersheds with no lake-or-wetland area. (Wetlands occupied 14.8% of the land area of Wisconsin in the 1980s (Dahl: 1990), so wetlands may have a greater influence on streamflow in Wisconsin than in the Pacific Northwest.)

Wetlands also modify the rate of ground-water discharge to streams (Novitzki et al. 1993). Wetlands, particularly those occurring adjacent to streams, usually exist because of ground-water discharge. Wetland soils typically are less permeable than upland soils, especially where prolonged wetness fosters the accumulation of organic material (Novitzki 1989). Because the wetland soils are less permeable, the rate of ground-water discharge from upgradient sources, through the wetland, and to the stream is slowed. The net effect is to reduce the rate of ground-water discharge to the stream but to increase the length of time that discharge occurs. Ground water typically discharges at a relatively constant, cool temperature, and it has a major influence on the temperature regime in streams. Changing the amount and timing of ground-water discharge may change the temperature regime of the stream significantly, affecting the suitability of the stream as salmon habitat. Ground-water upwelling into streams through gravels is a determinant in spawning redd selection for some species. The constant upwelling of fresh, oxygenated water may be necessary to egg and fry survival. Loss of wetlands will likely change the rate of ground-water discharge at critical times and may reduce spawning success in streams.

### 6.9.2 Wetlands and Water Quality

Wetlands retain particulate materials transported into them by overland flow or river flooding. Wetlands typically are flat areas adjacent to streams, and as floodwaters enter, flow velocities decrease and sediment loads are deposited. This phenomenon manifests as berms, often wooded, that build up next to the river channel in wide river valleys. Wetlands tend to stabilize stream banks because of the robust plant community that grows there. Wetlands tend to be wet through a larger part of the growing season, fostering plant growth that in turn provides sufficient root mass to stabilize soils. Where banks are stabilized by the lush wetland vegetation, stream channels tend to be somewhat deeper, and undercutting provides shelter to salmonids and other aquatic biota. Logging, grazing, farming, or other activities that change the wetland plant community can significantly reduce the wetland's ability to stabilize stream banks. Moreover, the velocity of water moving through wetlands is further reduced by dense vegetation, especially shrubs and trees, which in turn increases sediment deposition in the wetland. Thus, wetlands tend to reduce the amount of sediment transported to streams. Loss or removal of wetland areas may result in increased sediment loads (especially clays and silts) in receiving streams.

Wetlands also retain and process dissolved materials contained in overland flow or floodwaters. For example, they process nutrients (e.g., nutrient cycling), and other dissolved materials as well as retaining materials attached to suspended solids (e.g., phosphorus). Loss or removal of wetland areas may result in increased nutrient and contaminant loading to receiving streams.

### 6.9.3 Wetlands and Salmonid Habitat

Wetlands may contribute significantly to certain characteristics required by salmonids in their aquatic ecosystems, such as variable, but moderate streamflows; cool, well oxygenated, unpolluted water; relatively sediment-free streambed gravel; an adequate food supply; and instream structural diversity provided by woody debris (Carey et al. 1994). Because wetlands affect flood flows and springtime flows, they also influence the streamflow characteristics of the streams and aquatic habitat that support salmonids. Loss of wetlands likely increases the amount of individual flood peaks but reduces the duration of high flow events. Streams in the Pacific Northwest may require the infrequent (i.e., the 100-year) flood to reset; however, they may also require stability between extreme events to recover and re-establish equilibrium. Wetland loss may reduce the time between significant (e.g., 5- to 50-year frequency) floods and impair the stream's ability to recover. The timing and amount of springtime streamflows triggers the movement of spawning salmon into nursery streams. Changing the timing of peak flows may change the timing of migration and spawning. Changing the timing of spawning may result in minor, but significant, changes in the size and condition of salmon smolts returning to the ocean. These changes may have pronounced impacts on survival of young salmon in the ocean phase of their life cycle.

Wetlands support unique floral and faunal communities. The unique biota supported in wetlands contribute to the food web supporting the salmonids and associated biota in the streams, both adjacent to and downstream of the wetlands. Riparian vegetation, including that in wetlands, regulates the exchange of nutrients and material from upland forests to streams and wetlands (Carey et al. 1994). Wetlands and ponds have been found to provide critical habitats for both juvenile salmonids (Peterson 1982; Cederholm and Scarlett 1982), and a variety of wildlife species (Zarnowitz and Raedeke 1984). Species that frequent riparian areas include amphibians, reptiles, birds, mammals, and mollusks

(FEMAT 1993). Activities (e.g., draining, diking) that prevent the normal wetland functions or impair the connectivity of the wetland to the aquatic ecosystem may prevent the transport of materials into and out of the wetland and may prevent the wetland's contribution to an important element of the aquatic ecosystem. Interrupting or otherwise changing the connections between the wetland and the stream can impede the exchange of nutrients, organic detritus, insects, or other materials supporting the food web of the aquatic ecosystem.

Wetlands often provide refugia within the landscape. Especially in urban areas, agricultural areas, or other disturbed environments, wetlands are least suitable for conversion to other use: they often are left intact until all other lands have been converted. During this time, they may be the only natural areas left in the modified landscape to provide needed refuge to birds, mammals, and other biota. Because they typically occur at points of ground-water discharge, and reduce the rate but prolong the duration of ground-water discharge, wetlands also provide survival areas to aquatic species sensitive to high or low temperatures during hot summer periods or cold winter periods. In addition, a wetland may offer the only wet habitat available during periods of prolonged drought or during fires to protect those biota able to seek refuge within it.

Wetlands function as an integral component of the local watershed. They tend to be highly productive areas, often serving as a source of organic detritus to adjacent water bodies. Wetlands also provide nursery areas for salmon and habitat for organisms that provide food to salmon and associated biota. The wetland contributes to the ecological balance within the watershed/ecosystem within which it occurs. Destroying, draining, or otherwise impairing the wetland's function alters the hydrologic, sediment, chemical, and biological balance in the watershed.

## 6.10 Salmon Harvest

There are multiple alternatives for the management of salmon harvest. The NMFS Snake River Salmon Recovery Team. Bevan et al (1994a, b, c) recommended terminal area fishing and selective fishing as the best harvest schemes where mixed-stock fisheries include weak, depressed, or endangered stocks. Open-ocean harvest management is complicated by differential distribution patterns of different stocks, resulting in widely varying impacts on various stocks by fishing fleets. Mixed stock fisheries management becomes a sociopolitical question, requiring management agreements and treaties between various state, federal, tribal, and international entities, with complicated discussions concerning management regimen, harvestable surplus, and allocation. The management of open-ocean harvest also requires a greater understanding of the distributions of different stocks in order to determine their vulnerability to the fishing fleets. The presence of stocks of differing degrees of robustness can be dealt with by managing for indicator stocks or for weak stocks. The use of indicator stocks presents the risk that the chosen indicators are not truly representative of all stocks in the fishery, resulting in serious harm to more fragile stocks when healthier indicator stocks are performing well, or, conversely, losing potential harvest from healthy stocks by managing strictly for the weakest stocks (Kope 1992; Restrepo et al. 1992). Weak stock management has the latter risk (unnecessarily-high levels of yield foregone), as well as the difficulty that smaller populations tend to have less data available concerning their production and recruitment mechanisms and fishery impacts (Peterman 1978; Peterman and Steer 1981; Lestelle and Gilbertson 1993). In practice, a mixture of the two techniques is attempted, managing for indicator stocks or complexes of stocks, but also including attention to weaker stocks vulnerable to the fishery.

The definition of mixed-stock fisheries can extend to wild-types versus hatchery fish. Concerns have been raised about the release of large numbers of hatchery fish, usually to mitigate for habitat deterioration and enhance numbers available for harvest. Such large numbers of fish can have several effects, possibly including competition for rearing areas and density-dependent competition in the ocean. Studies have suggested that carrying capacity can be exceeded during the outmigration (Steward and Bjornn 1990). Indications of density-dependent age and size composition have been found for various Pacific salmonids (Ricker 1981; Ishida et al. 1993). Peterman (1978) found that only a few salmonid stocks exhibit density-dependent marine survival patterns, and that these effects were predominantly within or between cohorts—little or no marine density-dependence was found between different stocks, whether derived from nearby or distant spawning areas. Whether hatchery fish can be considered the same stock as wild-types from the same area is being debated, and should be decided separately for each wild vs. hatchery situation. It is probable that density-dependent marine survival only becomes a factor in years of drastically reduced ocean productivity. During recent El Niño events, not only have warm ocean surface temperatures along the Pacific Northwest coast reduced available salmonid food supply, but the numbers of warmwater fish which prey on young salmon, but are usually not found in this area, have increased (Pearcy 1992).

Terminal area and selective fishing patterns provide the best method of weak stock management currently available (Reisenbichler 1987; Kope 1992). Fisheries can be opened in terminal areas where run sizes are adequate to provide harvestable surplus without endangering more fragile stocks returning to other areas; in addition, returns to terminal areas provide more precise and immediate data on run size for that stock than available from less-terminal areas. Selective fishing principles include stock-directed timing and location of fishing, and gear types that allow size selection or live release. Fishing seasons can be shaped so as to avoid areas or times of year when sensitive species or stocks are present, thereby reducing or preventing incidental take of those stocks.

The magnitude of impact of a fishery on a stock can vary greatly due to size or age selection. In the ocean, the older fish tend to be more vulnerable to fisheries because of their longer exposure to the fishery (Moussalli and Hilborn 1986), particularly for those stocks frequenting coastal waters rather than only passing through coastal waters on their way to spawning areas. The average size of fish increases with age, and average fecundity increases with size, so that preferential take of larger fish will result in fewer eggs laid and ultimately a lower juvenile run than for a harvest pattern taking the same number of adults but no size selection (except possibly for populations which exceed the carrying capacity of their habitat) (Ricker 1972; Jaenicke and Celewycz 1994). Larger females also tend to dig deeper redds than smaller females, and select nest locations with larger gravel, with both behaviors combining to provide a greater egg-to-smolt survival (Hankin and Healey 1986; Hankin et al. 1993).

Gear types allowing live release provide the ability to operate fishing seasons in the presence of sensitive stocks with relatively little mortality to incidentally-caught fish (Mongillo 1984; Bendock and Alexandersdottir 1993). This can be applied especially well to species for which no fishing is allowed (e.g., chinook salmon caught by coho or sturgeon fisheries), and for the directed take of hatchery fish. The latter use requires that all harvestable hatchery fish be externally recognizable (by fin clips, visual implant tags, branding, etc.) and that all harvesters be instructed in the recognition of these marks, but also allows that some hatchery fish be left externally unmarked (with only a coded-wire tag, for example) to maintain hatchery brood stock levels.

There are indications that salmon carcasses can contribute significant amounts of nitrogen and phosphorus compounds to headwater streams (Cederholm and Peterson 1985). This nutrient source would be decreased by removal of returning fish via harvest below terminal areas. However, a greater potential source of loss of these nutrients is lack of large organic debris to hold the carcasses in place. The management of the riparian area to provide fallen trees and other sufficient instream obstructions for gravel enhancement, particulate organic matter retention, and pool frequency and shading, would also help retain salmon carcasses in the headwater area (Bilby and Bisson 1987; Bilby and Ward 1991; Botkin et al. 1994).

Northern squawfish, smallmouth bass, channel catfish, and walleye are important predators of juvenile salmon (Poe et al. 1991; Tabor et al. 1993). Common mergansers, gulls, terns and other birds also consume juvenile salmonids (Wood 1987a, b; Bevan et al. 1994c); (Bevan et al. 1994b). Harbor seals and California sea lions feed on both juvenile and adult salmon (Park 1993; Bevan et al. 1994b). Evidence of attempted predation on salmonids by pinnipeds is also seen in the prevalence of tooth and claw abrasions observed on adult salmonids at upstream locations (Harmon et al. 1991)—20% of the fish at the Lower Granite Dam—had injuries from pinnipeds with one-third of fish with abrasions having open wounds. Predator control efforts in the Columbia and Snake rivers to date have focused on removing northern squawfish, evaluating the behavior and distribution of predators in dam tailrace areas, and limiting avian predation by stringing monofilament lines across tailrace areas where juvenile salmonids that may be disoriented from dam passage are vulnerable (Beamesderfer et al. 1990).

### 6.11 Fish Introductions and Hatchery Management

Throughout history, humans have introduced fish into streams, rivers, and lakes in order to increase commercial and recreational fishing opportunities. Introduction of non-native fishes into waters of the Pacific Northwest began prior to the turn of the century and continues today. Four primary sources of introductions include: fishery management manipulations (stocking native or non-native fish); intentional introductions of gamefish by anglers; intentional or unintentional baitfish liberation by anglers; and bilge pumping of ballast water, particularly in estuaries and large rivers. Although there are few well documented studies identifying conditions both before and after species introductions, effects on native fishes may include elimination, reduced growth and survival, and changes in community structure. For example, brown trout replaced brook trout in a Minnesota stream over 15 years (Waters 1983), and cutthroat trout were replaced by more aggressive rainbow trout and brown trout in the Great Basin of western North America (Moyle and Vondracek 1985). Redside shiner was found to compete with young rainbow trout in Paul Lake, British Columbia, leading to decreased growth and survival of the young trout (Johannes and Larkin 1961). Ratliff and Howell (1992) reported that for 65 bull trout populations in Oregon considered at risk of extinction or already extinct, brook trout were the most important stressor in 26% of those populations, and a contributing factor in 22%.

Six mechanisms have been identified that allow introduced fish to dominate or displace native fish including competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization (Moyle et al. 1986). Moyle et al. (1986) suggest that introduced species may thrive best where extensive environmental modification has already occurred. In the Columbia river—a system where temperature and stream velocities have been substantially altered—predator species introduced for recreational fishing, including walleye (*Stizostedion vitreum*), channel catfish (*Ictalurus punctatus*), and smallmouth bass (*Micropterus dolomieu*), are feeding on outmigrating smolts (Palmisano et al. 1993a, b). Whirling disease has spread throughout the Columbia and Colorado river basins and has decimated several important trout fisheries in the west (*Oregonian*, March 12,

1995). This disease was brought over from Europe via a shipment of frozen fish, was identified in a trout hatchery in Pennsylvania in 1956, and was subsequently introduced into our native waters (Marnell 1986).

Artificial propagation of native species has been used for decades as a means of replacing lost natural production resulting from various development activities and to provide increased returns for harvest. Although artificial propagation may increase salmon and trout abundance, hatchery introductions may result in a number of unintended and undesirable consequences for wild salmon and trout populations (Johnson et al. 1991). Interactions between hatchery and natural fishes may result in greater competition for food and habitat (Nickelson et al. 1986), or mates. Transmission of disease between hatchery and wild populations is also possible (Marnell 1986; Steward and Bjornn 1990). In 1987, the incidence of BKD infection in hatchery spring chinook from two Snake River hatcheries was 92 to 99% (Bevan et al. 1994a). Because many fish may carry BKD for extended periods without exhibiting symptoms, cross-transmission may be substantial. The likelihood of transmission may be particularly high when fish are aggregated for transport in raceways, trucks, and barges. Hatchery supplementation can also increase predation rates on wild stocks, as well as increasing harvest pressure on wild populations in mixed stock and terminal fisheries (Palmisano et al. 1993a); (Lichatowich and McIntyre 1987).

Genetic changes in wild populations is another possible consequence of hatchery introductions (Hindar et al. 1991; Waples 1991a). Introduction of hatchery stocks can eliminate unique genomes in local stocks. Straying and subsequent crossbreeding may result in loss of genetic variability between populations and depressed fitness where introgression occurs. Low rates of natural straying may be beneficial in maintaining genetic variability in natural populations, but these rates may become elevated through artificial propagation (Bams 1976; Withler 1982), with potentially serious consequences for locally adapted populations.

Finally, hatchery operations may cause habitat changes that adversely affect wild populations. Discharge of nutrient effluent waters from hatcheries may negatively affect water quality. In addition, the construction of hatchery weirs or diversion structures can impede the migration of wild stocks.

## 6.12 Recreation

Although the primary influence of recreation on salmonids is fishing, there are also indirect effects related to boating, log removal, parks, and campgrounds. Stream and lake banks, riparian vegetation, and spawning redds are disturbed wherever humans become concentrated (Johnson and Carothers 1982); however these effects are generally localized. Human concentrations at campgrounds or vacation areas may also lead to impaired water quality by elevating coliform bacteria and nutrients in streams (Aukerman and Springer 1976; Potter et al. 1984). Recreational boaters, kayakers, and rafters have less obvious, but more far-reaching effects, by removing snags from rivers and lakes. This is done for reasons of aesthetics and safety, but popular whitewater rivers and many recreational lakes are nearly devoid of snags. Removal of this wood potentially affects salmonids by reducing habitat complexity in rivers and in estuaries into which they enter. The reduced number of logs lowers estuarine and marine habitat quality for fishes just as it does habitat in rivers (Maser and Sedell 1994).

## 6.13 Beaver Trapping

Other than humans, the mammal that most shaped North American waterways was probably the beaver. In pre-Columbian times, their numbers were estimated to be 4–26/km<sup>2</sup> across the U.S. (Naiman et al. 1986), and they provided the initial economic base for European exploration and settlement west of the Appalachians. However, because of widespread trapping in the 1800s and early 1900s, their numbers have dwindled to a fraction of their historical abundance (0.4–0.8/km<sup>2</sup> today (Naiman et al. 1986). Beavers have both negative and positive effects on water bodies and riparian ecosystems. Their feeding results in the loss of woody riparian vegetation and increased retention of fine sediments, but increases the input of large woody debris to streams. Beaver ponds increase the surface-to-volume ratio of the impounded area, thereby increasing summer temperatures. Marcus et al. (1990) suggest that in the east, temperature increases may be detrimental to trout populations, but that in the Rocky Mountains, increased temperature where waters are colder, may benefit salmonids. Beaver ponds also supplement summer low flows (Marcus et al. 1990), and provide critical over-wintering habitat for salmonids. Bank dens and channels increase erosion potential, but also offer juvenile salmonids protection from high winter flows. Beaver ponds frequently fill with sediments and become wetlands, but they retard erosion upstream and reduce sedimentation downstream. A high frequency of ponds may reduce the amount of spawning gravel through siltation (Marcus et al. 1990). When channels are once again established, these reaches are very productive of fish. Beaver ponds in the Rocky Mountain West were found to support larger and more numerous trout, and greater concentrations of aquatic invertebrates than undammed sections of the stream (Naiman et al. 1984). Beaver ponds may also provide a sink for nutrients from tributary streams, enhancing pond productivity, and increasing retention time (Maret et al. 1987; Naiman et al. 1986). While it is difficult to generalize about the overall effect of beaver on salmonids, Naiman et al. (1986) suggest beaver act as a keystone

species "to affect ecosystem structure and dynamics far beyond their immediate requirements for food and space." It is clear that their removal has fundamentally altered aquatic ecosystem function.

Table 6.1.4.3a. Effects of Timber on Peakflows in Coastal Areas of the Pacific Northwest. From Beschta et al. (1994).

| Watershed/<br>Location       | Size<br>(acres) | Geology/Soils                            | Harvest<br>method and<br>percent | Peakflow<br>effect            | Reference                                     |
|------------------------------|-----------------|--|----------------------------------|-------------------------------|---|
| NB <sup>1</sup><br>OR        | 173             | sandstone<br>shallow GL-GCL <sup>2</sup> | cc <sup>3</sup> 82%              | fall: +50%<br>winter:<br>+19% | Hsieh 1970<br>Harris 1977<br>Harr et al. 1975 |
| DC <sup>1</sup> -Main<br>OR  | 748             | sandstone<br>shallow GL-GCL              | cc 26%                           | all: ns <sup>4</sup>          | Hsieh 1970<br>Harris 1977<br>Harr et al. 1975 |
| DC-2<br>OR                   | 138             | sandstone<br>shallow GL-GCL              | cc 90%                           | fall: +51%<br>winter:<br>+20% | Hsieh 1970<br>Harris 1977<br>Harr et al. 1975 |
| DC-3<br>OR                   | 99              | sandstone<br>shallow GL-GCL              | cc 65%                           | fall: +50%<br>winter: 30%     | Hsieh 1970<br>Harris 1977<br>Harr et al. 1975 |
| WCBC <sup>**</sup> -1<br>BC  | 57              | quartz GSL <sup>11</sup>                 | cc 71%                           | all: -22%                     | Cheng et al. 1975                             |
| Carn. <sup>**</sup> -B<br>BC | 2,470           | volcanic coarse soil                     | cc 41%                           | all: ns                       | Hetherington 1987                             |
| Carn.<br>BC                  | 30              | volcanic coarse soil                     | cc 90%                           | all: +20%                     | Hetherington 1987                             |
| Jamieson<br>BC               | 739             | na <sup>**</sup>                         | cc 19.2%                         | winter:<br>13.5%              | Golding 1987                                  |
| SFCC <sup>11</sup><br>CA     | 1,047           | sandstone coarse<br>soil                 | sc 60%                           | small:<br>+107%<br>large: ns  | Ziemer 1981                                   |

<sup>1</sup> NB Needle Branch, Alesia Watershed

<sup>2</sup> GL-GCL gravelly loam-gravelly clay loam

<sup>3</sup> cc clearcut

<sup>1</sup> DC Deer Creek, Alesia Watershed

<sup>4</sup> not significant

<sup>\*\*</sup> WCBC West Coast British Columbia

<sup>11</sup> GSL gravelly sandy loam

<sup>\*\*</sup> Carn. Carnation Creek

<sup>\*\*</sup> na not applicable

<sup>11</sup> SFCC South Fork Caspar Creek

<sup>\*\*</sup> sc shelterwood

Table 6.1.4.3b Effects of Timber Harvesting on Peakflows in Interior Areas of the Pacific Northwest. From Beschta et al. (1994).

| Watershed/<br>Location      | Size<br>(acres) | Geology/<br>soils                    | Harvest<br>method and<br>percent | Peakflow<br>effects                    | Reference                  |
|-----------------------------|-----------------|--------------------------------------|----------------------------------|--|----------------------------|
| Rain-on-snow                |                 |                                      |                                  |  |                            |
| WS-1 <sup>1</sup><br>OR     | 235             | basalt/GL <sup>2</sup>               | cc <sup>3</sup> 100%             | fall: +200%<br>winter: ns <sup>4</sup> | Rothacher 1973             |
| WS-3                        | 249             | basalt/GL                            | cc 25%                           | mean: +10%                             | Rothacher 1973             |
| WS-6                        | 32              | basalt/GL                            | cc 100%                          | all: ns                                | Harr et al. 1982           |
| WS-7                        | 52              | basalt/GL                            | sc 60%                           | all: ns                                | Harr et al. 1982           |
| WS-10                       | 25              | basalt/GL                            | cc 100%                          | ROS <sup>5</sup> : -36%<br>rain: ns    | Harr and<br>McCorison 1979 |
| CC-1 <sup>**</sup><br>OR    | 170             | basalt/<br>GL-GCL <sup>**</sup>      | sc <sup>**</sup> 50%             | mean: +30%<br>large: +48%              | Harr et al. 1979           |
| CC-2<br>OR                  | 168             | basalt/<br>GL-GCL                    | cc 30%                           | all: ns                                | Harr et al. 1979           |
| CC-3<br>OR                  | 121             | basalt/<br>GL-GCL                    | cc 100%                          | mean: +44%<br>large: +35%              | Harr et al. 1979           |
| Snowmelt                    |                 |                                      |                                  |  |                            |
| HREA <sup>55</sup> -1<br>OR | 73              | fractured<br>basalt/SL <sup>11</sup> | cc 43%                           | all: ns                                | Fowler et al.<br>1987      |
| HREA-2<br>OR                | 60              | fractured<br>basalt/SL               | sc 50%                           | all: ns                                | Fowler et al.<br>1987      |
| HREA-4<br>OR                | 292             | fractured<br>basalt/SL               | cc 22%                           | all: ns                                | Fowler et al.<br>1987      |
| Camp Cr.<br>BC              | 8,373           | granite/<br>coarse soils             | cc 30%                           | annual:<br>-9 to +35%                  | Cheng 1989                 |

<sup>1</sup> WS Watersheds, H.J. Andrews, Western Cascades Region

<sup>2</sup> GL gravelly loam

<sup>3</sup> cc clearcut

<sup>4</sup> ns not significant

<sup>5</sup> ROS rain-on-snow

<sup>\*\*</sup> CC-1 Coyote Creek, South Umpqua Experimental Forest, Western Cascades Region

<sup>\*\*</sup> GL-GCL gravelly loam-gravelly clay loam

<sup>\*\*</sup> sc shelterwood

<sup>55</sup> HREA-1 High Ridge Evaluation Area, Blue Mountains Region

<sup>11</sup> SL silt loam

Table 6.1.6a Summary of Summer Temperature Changes Associated with Forest Management Activities on Forest Watersheds in the Pacific Northwest. From Beschta et al. (1987). Reproduced with permission from the publisher.

| Location                            | Treatment                           | Stream temperature variables                  | Temperature change(°C)                        | Reference                |
|-------------------------------------|-------------------------------------|---|---|--------------------------|
| Alaska (Southeast)                  | Clearcut and natural openings       | ΔTemperature per 100 m of channel             | 0.1–1.1°C/100m<br>Avg = 0.7°C/100m            | Meehan 1970              |
| British Columbia (Vancouver Island) | Logged (Tributary H)                | Average June-August diurnal temperature range | 0.5–1.8°C increase over pre-treatment levels  | Holtby and Newcombe 1982 |
|                                     | Logged and burned (Tributary J)     | Average June-August diurnal temperature range | 0.7°–3.2°C increase over pre-treatment levels | Holtby and Newcombe 1982 |
| Oregon (Cascades)                   | Clearcut                            | Average June-August maximum                   | 4.4–6.7°C                                     | Levno and Rothacher 1967 |
|                                     | Clearcut and burning                | Average June-August maximum                   | 6.7–7.8°C                                     | Levno and Rothacher 1967 |
| Oregon (Coast Range)                | Clearcut                            | Average July-Sept. maximum                    | 2.8–7.8 °C                                    | Brown and Krygier 1967   |
|                                     | Clearcut and burning                | Average July-August maximum                   | 9–10° C                                       | Brown and Krygier 1970   |
| Oregon (Cascades)                   | Mixed clearcut and forested reaches | Δ Temperature per 100m of channel             | 0–0.7°C/100m                                  | Brown et al. 1971        |
|                                     | Tractor stripped area               | Δ Temperature per 100m of channel             | 15.8°C/100m                                   | Brown et al. 1971        |

Table 6.1.9a Influences of Timber Harvest on Physical Characteristics of Stream Environments, Potential Changes in Habitat Quality, and Resultant Consequences for Salmonid Growth and Survival. Reproduced with permission from the publisher.

| Forest Practice                              | Potential change in physical stream environment                                    | Potential change in quality of salmonid habitat   | Potential consequences for salmonid growth and survival  |
|--|--|---|--|
| Timber harvest from streamside areas         | Increased incident solar radiation   | Increased stream temperature; higher light levels; increased autotrophic production   | Reduced growth efficiency; increased susceptibility to disease; increased food production; changes in growth rate and age at smolting                            |
|  | Decreased supply of large woody debris   | Reduced cover; loss of pool habitat; reduced protection from peak flows; reduced storage of gravel and organic matter; loss of hydraulic complexity | Increased vulnerability to predation; lower winter survival; reduced carrying capacity; less spawning gravel; reduced food production; loss of species diversity |
|  | Addition of logging slash (needles, bark, branches)                                | Short-term increase in dissolved oxygen demand; increased amount of fine particulate organic matter; increased cover                                | Reduced spawning success; short-term increase in food production; increased survival of juveniles  |
|  | Erosion of streambanks   | Loss of cover along edge of channel; increased stream width, reduced depth  | Increased vulnerability to predation; increased carrying capacity for age-0 fish, but reduced carrying capacity for age-1 and older fish                         |
| Timber harvest from hillslopes; forest roads | Altered streamflow regime  | Short-term increase in streamflows during summer  | Short-term increase in survival  |
|  |  | Increased severity of some peak flow events   | Embryo mortality caused by bed-load movement   |
|  | Accelerated surface erosion and mass wasting                                       | Increased fine sediment in stream gravels   | Reduced spawning success; reduced food abundance; loss of winter hiding space  |
|  |  | Increased supply of coarse sediment   | Increased or decreased rearing capacity  |
|  |  | Increased frequency of debris torrents; loss of instream cover in the torrent track; improved cover in some debris jams                             | Blockage to migrations; reduced survival in the torrent track; improved winter habitat in some torrent deposits  |
| Increased nutrient runoff                    | Elevated nutrient levels in streams  | Increased food production   |  |
| Increased number of road crossings           | Physical obstructions in stream channel; input of fine sediment from road surfaces | Restriction of upstream movement; reduced feeding efficiency  |  |

Table 6.1.9a Influences of Timber Harvest on Physical Characteristics of Stream Environments, Potential Changes in Habitat Quality, and Resultant Consequences for Salmonid Growth and Survival. Reproduced with permission from the publisher.

| Forest Practice   | Potential change in physical stream environment | Potential change in quality of salmonid habitat   | Potential consequences for salmonid growth and survival |
|---|---|---|---|
| Scarification and slash burning (preparation of soil for reforestation) | Increased nutrient runoff                       | Short-term elevation of nutrient levels in streams  | Temporary increase in food production                   |
|   | Inputs of fine inorganic and organic matter     | Increased fine sediment in spawning gravels and food production areas; short-term increase in dissolved oxygen demand | Reduced spawning success                                |

Table 6.2.2a Deleterious Effects of Livestock Grazing on Plant Communities in Western North America. From Fleischner (1994). Reprinted by permission of Blackwell Scientific Publications, Inc.

| Habitat               | Location   | Effect  | Authority               |
|-----------------------|------------|---|-------------------------|
| Sonoran desertscrub   | Arizona    | Perennial grasses and <i>Krameria</i> (palatable shrub) showed dramatic density decreases with grazing                    | Blydenstein et al. 1957 |
| Mojave desertscrub    | California | 60% reduction in above ground biomass of annuals, 16-29% decrease in cover of perennial shrubs with grazing               | Webb & Stielstra 1979   |
| Sagebrush desert      | Idaho      | Grazed site had ¼ species richness of ungrazed site   | Reynolds & Trost 1980   |
| Desert grassland      | New Mexico | Grass density increased by 110% after 30 years of protection from grazing   | Gardner 1950            |
| Semidesert grassland  | Arizona    | Species richness increased as did canopy cover for midgrass, shortgrass, shrub and forb groups after removal of livestock | Brady et al. 1989       |
| Semidesert grassland  | Arizona    | Woody plants significantly more abundant after removal of livestock   | Bock et al. 1984        |
| Ponderosa pine forest | Washington | Decreased species richness on grazed sites  | Rummell 1951            |
| Mountain canyon       | Utah       | Absence or near absence of 10 grass species on grazed sites   | Cottam & Evans 1945     |
| Riparian              | Oregon     | Species richness increased from 17 to 45 species nine years after removal of livestock                                    | Winegar 1977            |
| Riparian              | Arizona    | Herbaceous cover of grazed plot less than half that of ungrazed plot  | Szaro & Pase 1983       |
| Riparian              | Colorado   | Shrub canopy coverage increased 5.5 times, willow canopy coverage 8 times after removal of livestock                      | Schulz & Leininger 1990 |

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Table 6.5.1.2a Case Histories Relating the Effects of Gravel Extraction on Channel Morphology and Hydrology of Streams in Washington, Oregon and California.

| Location                                  | Activity   | Effects   |
|---|--|---|
| <b>WASHINGTON</b>                         |  |   |
| Humptulips, Wynoochee, and Satsop Rivers* | Gravel bar scalping  | Minimum rates of gravel extraction exceeded replenishment rate. Channel degradation (lowering) occurred at some sites.  |
| White River†                              | Gravel extraction (partly for flood prevention), diking, and straightening | Aggradation in lower reaches, degradation in upper reaches.   |
| Skykomish River‡                          | In-channel gravel mining   | Diminished size of gravel bars that were mined, as well as upstream and downstream sites. Reduced rate of bank erosion.   |
| <b>CALIFORNIA</b>                         |  |   |
| Cache Creek†                              | In-channel extraction during dry season                                    | Channel degradation up to 9 meters (avg 5 m) over 21 yr period. Increased flood capacity has eliminated overbank flooding and is preventing soils from being deposited on flood plain. Drop in ground water table has shifted system from a "drain" system to recharge system. Loss of aquifer storage potential. |
| Russian River†                            | Gravel extraction  | Channel degradation up to 6 m (avg 4 m). Exposure of bedrock substrate.   |
| Dry Creek†                                | Gravel extraction  | Channel degradation up to 4 m. Riparian vegetation has died, probably in response to lowering water table.  |
| Tujung Wash‡                              | Off-channel gravel mining  | Gravel pit was inundated by 1969 floods. Headward scour up to 4.5 m extended 790 - 914 m upchannel.   |
| Redwood Creek‡                            | Channelization, levee construction, gravel mining to low water level       | Alternating lowering of bed by mining and raising of bed from redeposition. Shift in thalweg. Gravel bars removed annually by mining contributing to channel destabilization. Headward degradation of channel.  |
| Stony Creek§                              | In-channel gravel mining   | Channel shifted from braided configuration to single, incised, meandering channel. Degradation up to 5 m. Obliteration of natural low-flow channels. Effects modified by changes in flow regime due to construction of dam upstream.  |
| <b>OREGON</b>                             |  |   |
| Willamette River¶                         | Sand and gravel extraction   | Channel degradation of approximately 0.3 meters per year over 20 - 30 yr period. Degradation due to combined effects of sand and gravel extraction, bank stabilization, dams, watershed changes and natural geological events.  |

\* Collins and Dunne 1989  
 † Collins and Dunne 1990  
 ‡ Scott 1973  
 § Kondolf and Swanson 1993  
 ¶ Klingeman 1993  
 ¶ OWRI 1994

Table 6.5.2.7a Reported Toxicities of Metals in Soft Water (<45 mg/L as CaCO<sub>3</sub>). From Nelson et al. (1991).  
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| Substance     | Species              | Reported toxicity |                | Source                         |
|---------------|----------------------|-------------------|----------------|--------------------------------|
|               |                      | Method*           | Concentration  |                                |
| Aluminum (Al) | Brook trout          | LC50              | 3.6–4.0 mg/L   | Decker and Menendez (1974)     |
| Arsenic (As)  | Rainbow trout        | LC50              | 10.8 mg/L      | Hale (1977)                    |
| Cadmium (Cd)  | Rainbow trout        | LC50              | 6.6 µg/L       | Hale (1977)                    |
|               | Brook trout          | MATC              | 1.7–3.4 µg/L   | Benoit et al. (1976)           |
| Chromium (Cr) | Rainbow trout        | LC50              | 24.1 mg/L      | Hale (1977)                    |
|               | Rainbow trout        | LC50              | 69.0 mg/L      | Benoit (1976)                  |
|               | Rainbow trout        | MATC              | 0.2–0.35 mg/L  | Benoit (1976)                  |
|               | Brook trout          | LC50              | 59.0 mg/L      | Benoit (1976)                  |
|               | Brook trout          | MATC              | 0.2–0.35 mg/L  | Benoit (1976)                  |
| Copper (Cu)   | Coho salmon          | LC50              | 46.0 µg/L      | Chapman and Stevens (1978)     |
|               | Rainbow trout        | LC50              | 253.0 µg/L     | Hale (1977)                    |
|               | Rainbow trout        | LC50              | 125.0 µg/L     | Wilson (1972)                  |
|               | Rainbow trout        | LC50              | 57.0 µg/L      | Chapman and Stevens (1978)     |
|               | Rainbow trout        | ILL               | 37.0 µg/L      | Sprague and Ramsay (1965)      |
|               | Atlantic salmon      | ILL               | 32.0 µg/L      | Sprague and Ramsay (1965)      |
|               | Atlantic salmon      | ILL               | 0.52 mg/L      | Sprague (1964)                 |
|               | Brook trout          | MATC              | 9.5–17.4 µg/L  | McKim and Benoit (1971)        |
| Copper-zinc   | Atlantic salmon      | TU                | 1.0 mg/L       | Sprague and Ramsay (1965)      |
| Iron (Fe)     | Brook trout          | LC50              | 1.75 mg/L      | Decker and Menendez (1974)     |
| Lead (Pb)     | Rainbow trout        | LC50              | 8.0 mg/L       | Hale (1977)                    |
|               | Rainbow trout (eggs) | MATC              | 4.1–7.6 µg/L   | Davies et al. (1976)           |
| Mercury (Hg)  | Rainbow trout        | LC50              | 33.0 mg/L      | Hale (1977)                    |
| Nickel (Ni)   | Rainbow trout        | LC50              | 35.5 mg/L      | Hale (1977)                    |
| Uranium (U)   | Brook trout          | LC50              | 2.8 mg/L       | Parkhurst et al. (1984)        |
| Zinc (Zn)     | Coho salmon          | LC50              | 905.0 µg/L     | Chapman and Stevens (1978)     |
|               | Rainbow trout        | LC50              | 1,755 µg/L     | Chapman and Stevens (1978)     |
|               | Rainbow trout        | LC50              | 0.18–0.39 mg/L | Finlayson and Ashuckian (1979) |
|               | Rainbow trout        | ILL               | 560 µg/L       | Sprague and Ramsay (1965)      |
|               | Atlantic salmon      | ILL               | 0.092 mg/L     | Sprague (1964)                 |
|               | Atlantic salmon      | ILL               | 0.15–1.0 mg/L  | Zitko and Carson (1977)        |
|               | Atlantic salmon      | ILL               | 420 µg/L       | Sprague and Ramsay (1965)      |
|               | Brook trout          | MATC              | 534–1,360 µg/L | Holcombe et al. (1979)         |

\*LC50 = lethal concentration for 50% of test organisms; MATC = maximum acceptable toxic concentration; ILL = incipient lethal level; TU = toxic units.

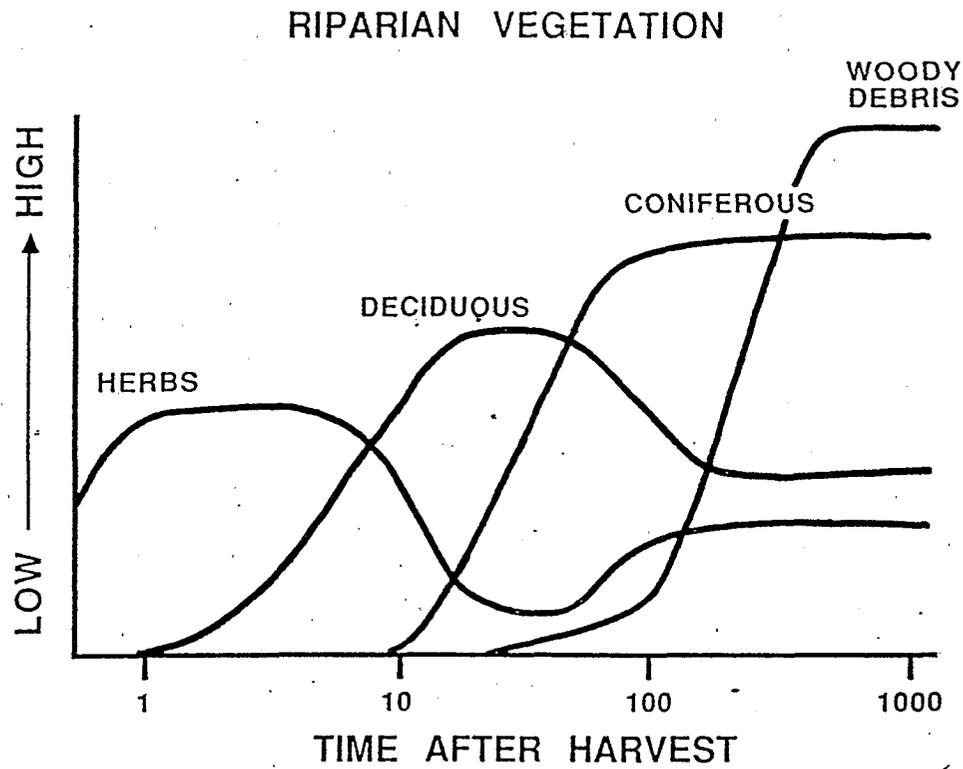


Figure 6.1.7b Temporal patterns of relative dominance of riparian vegetation after timber harvest (time is expressed as years on a logarithmic scale). From Gregory et al. (1987). Reproduced with permission from the publisher.

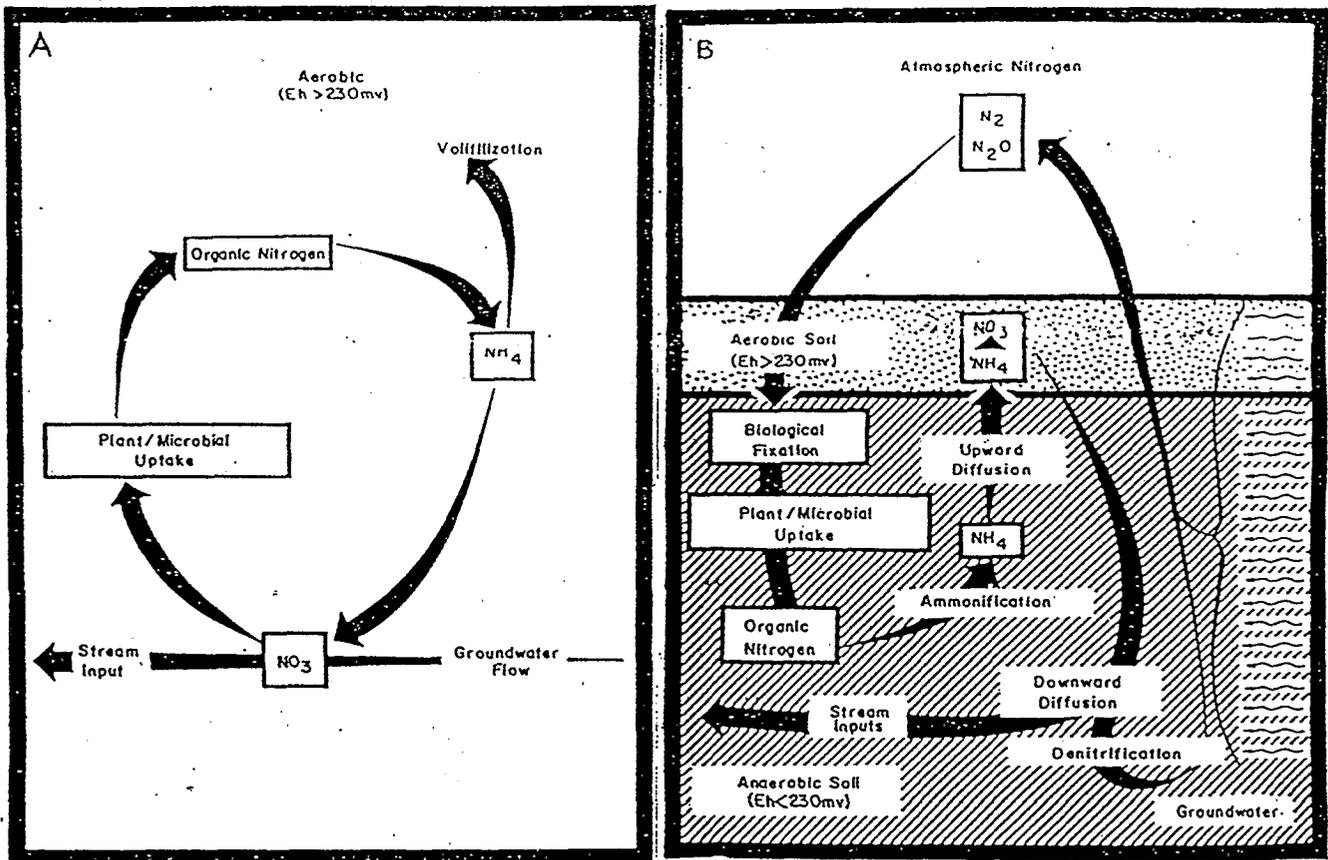


Figure 6.2.7a Nitrogen cycling pathways in a) disturbed and b) undisturbed riparian zones of northeastern Oregon, as indicated by redox potential (Eh). From Green and Kauffman (1989). Reproduced with permission from the author.

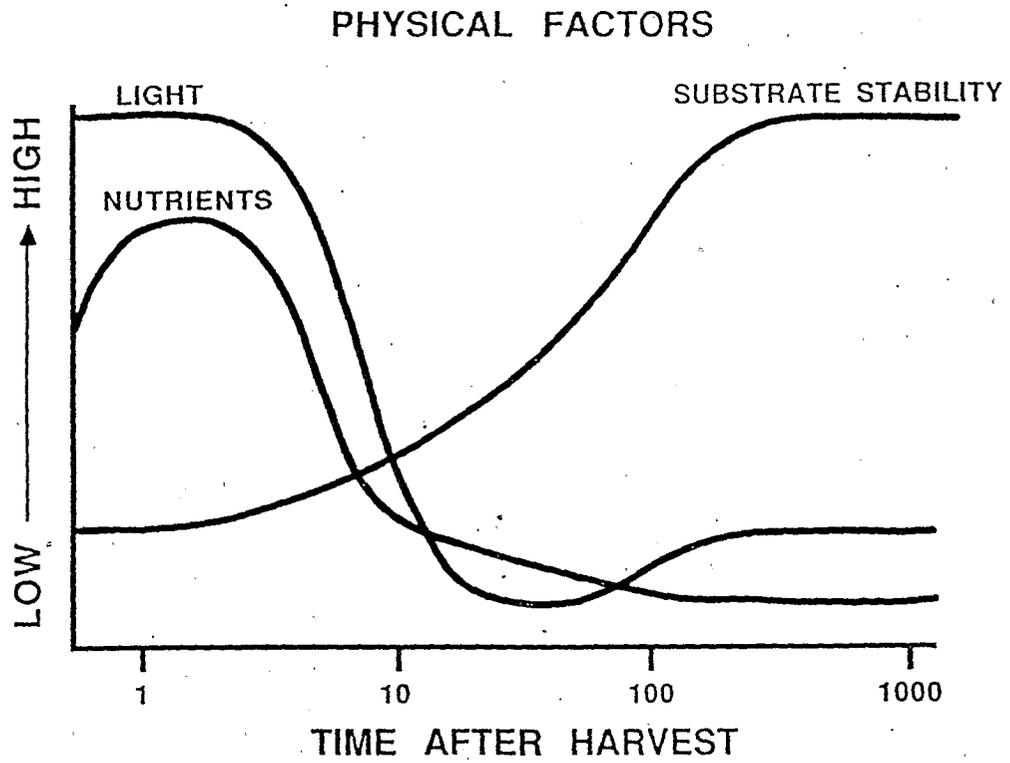


Figure 6.1.7a

Temporal patterns of solar radiation, nutrients, and the degree of substrate stability after timber harvest (time is expressed as years on a logarithmic scale.) From Gregory et al. (1987). Reproduced with permission from the publisher.

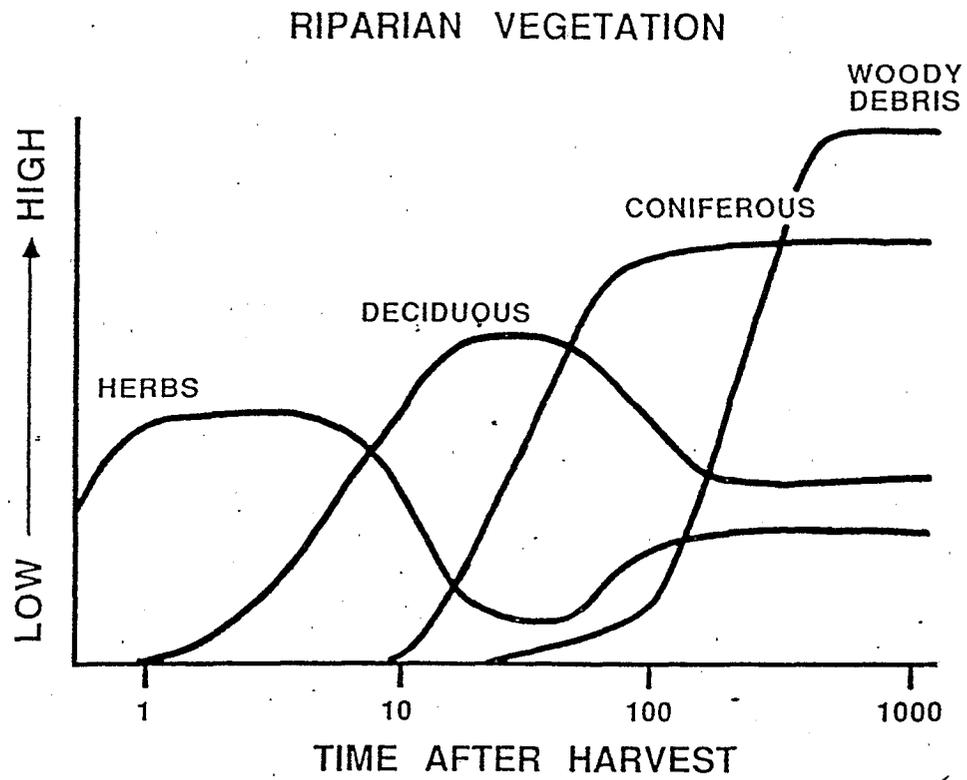


Figure 6.1.7b Temporal patterns of relative dominance of riparian vegetation after timber harvest (time is expressed as years on a logarithmic scale). From Gregory et al. (1987). Reproduced with permission from the publisher.

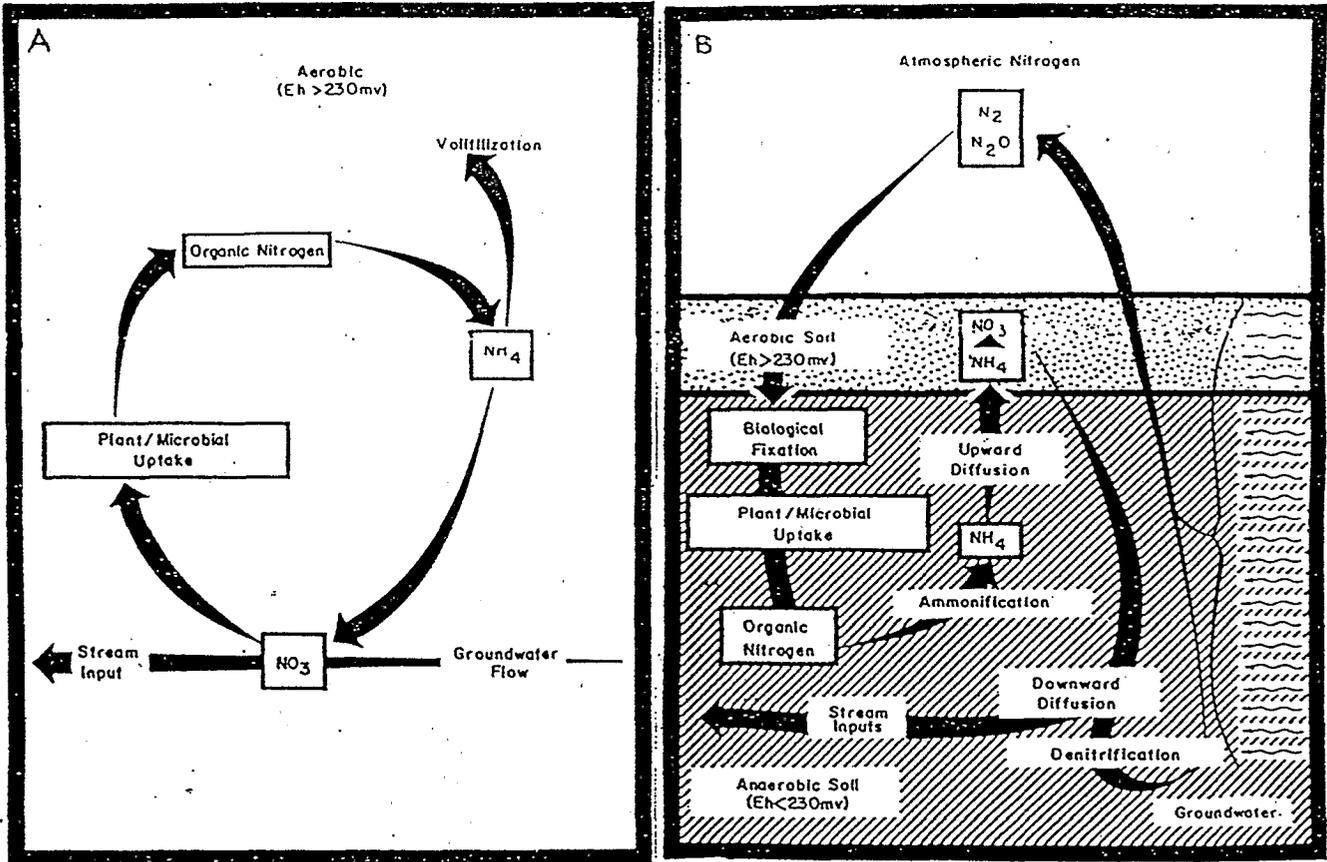


Figure 6.2.7a

Nitrogen cycling pathways in a) disturbed and b) undisturbed riparian zones of northeastern Oregon, as indicated by redox potential (Eh). From Green and Kauffman (1989). Reproduced with permission from the author.

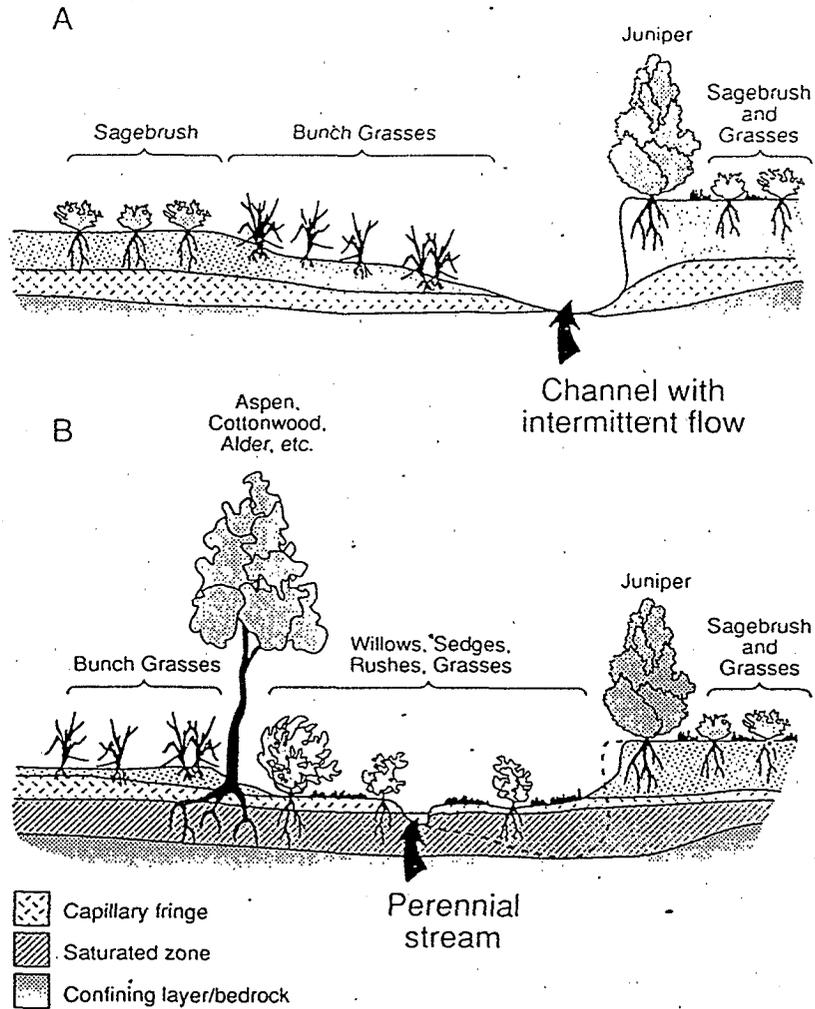


Figure 6.2.9a General characteristics and functions of a) disturbed, and b) undisturbed riparian areas on rangelands. From Elmore 1992. Reproduced with permission from the publisher.

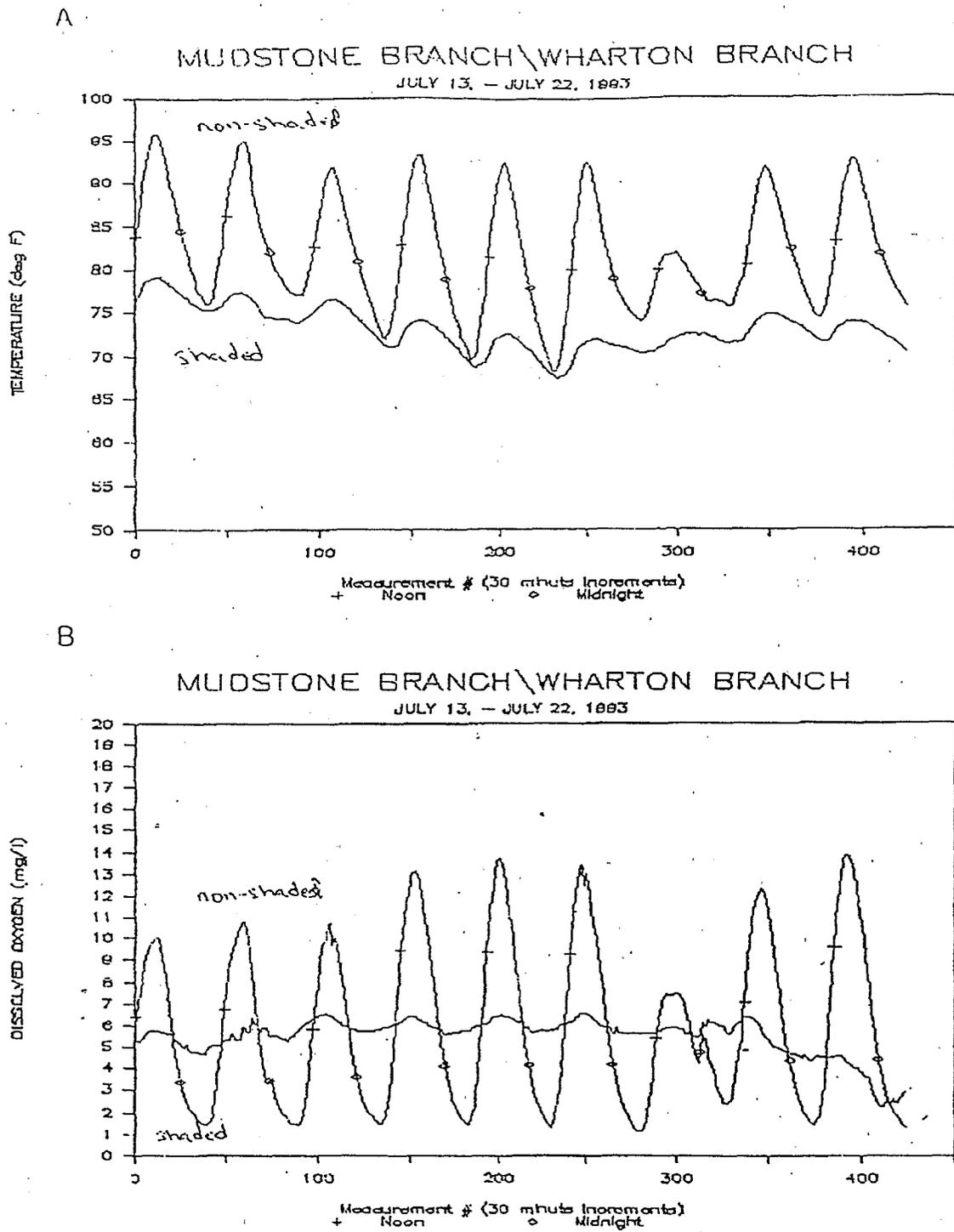


Figure 6.3.10a Diel Fluctuations in a) temperature and b) oxygen, in shaded and unshaded reaches of Mudstone Branch/Wharton Branch. Maxted (N.y.). Reproduced with permission from the author.

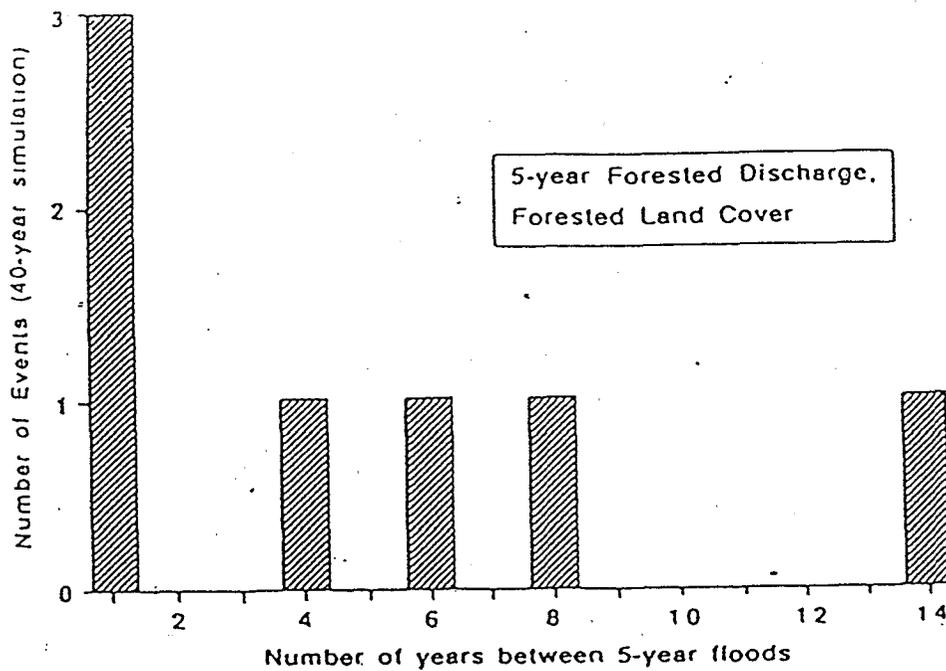


Figure 6.4.4a Hydrologic Simulation Program Fortran simulation of the Hylebos Creek basin in southwest King County, Washington, under fully forested land cover. Bars show the number of years separating discharge events of 5-year recurrence or greater. The average separation is 5 years (40 years of simulation, 8 events), but the actual spacing varies from one year (i.e., successive years) to 14 years.

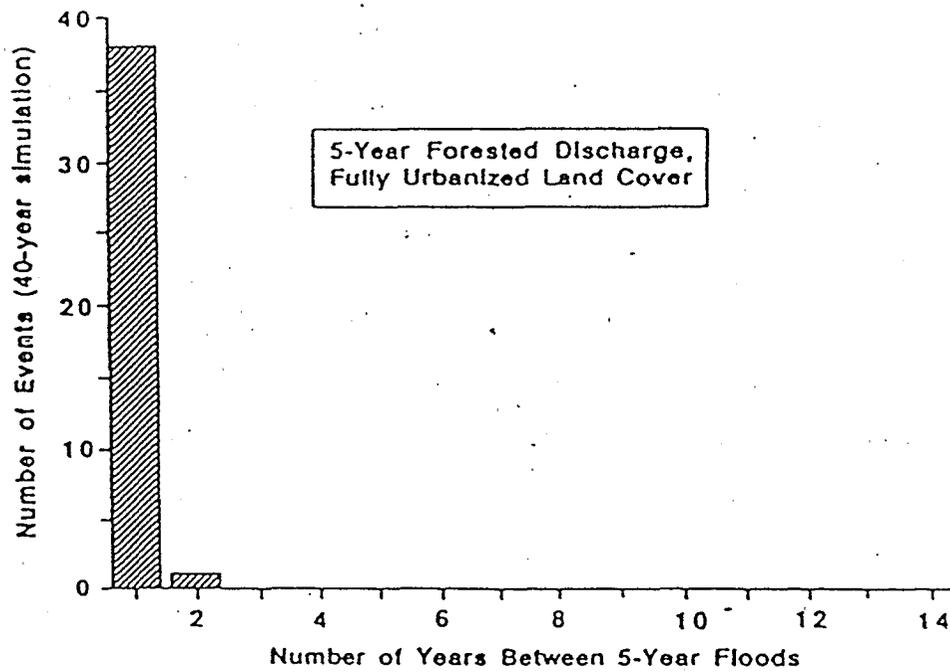


Figure 6.4.4b

Forty years of Hydrologic Simulation Program Fortran simulation of the Hylebos Creek basin under fully urbanized conditions (about 40 percent effective impervious area). Discharges at or greater than the 5-year forested event occur in every year except one (compare with Fig. 6.4.4a). Reproduced with permission from author.

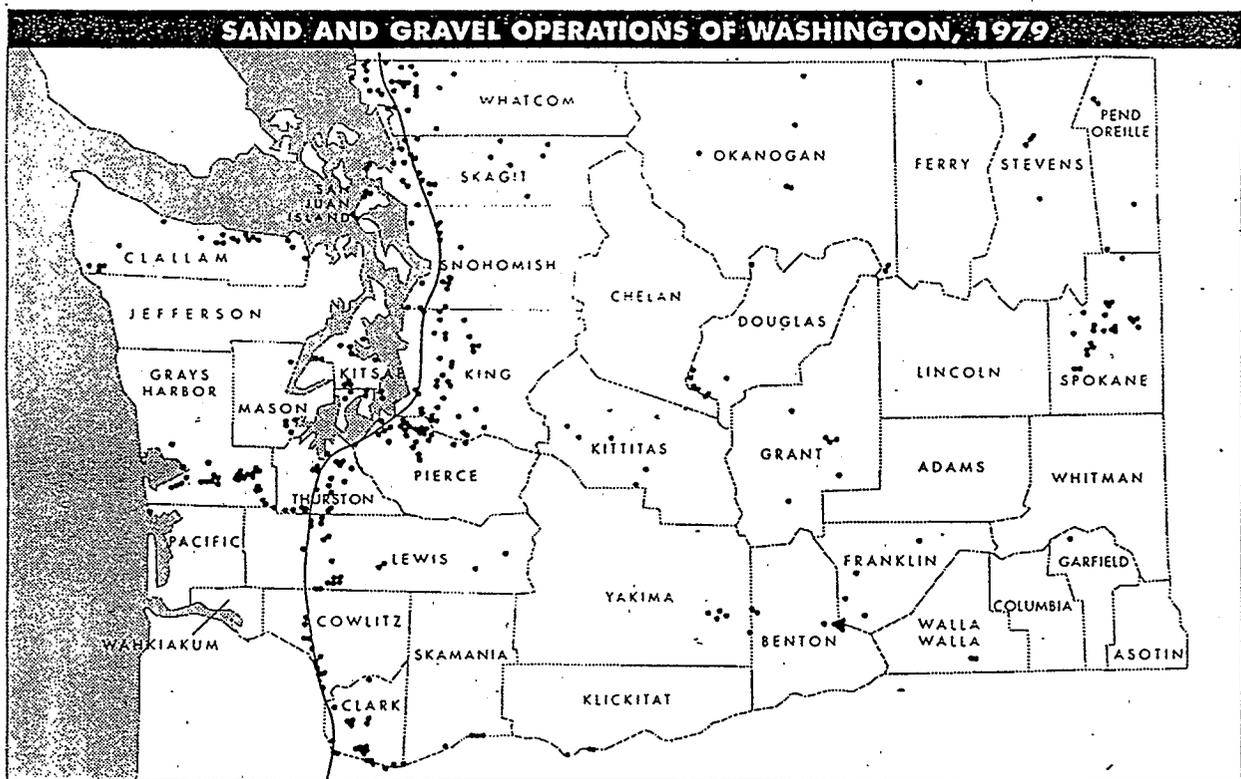


Figure 6.5.1.1a Sand and gravel operations of Washington, 1979. From Palmisano et al. (1993a). Reproduced with permission from the author.

## 7.0 Oceanic and Atmospheric Circulation

Until the mid-1970s, little was known about the effect of oceanic conditions on anadromous salmonids. Most research on salmonid biology focused on the freshwater environment and fishery biologists generally attributed variation in population size to conditions in fresh water. Recent work strongly suggests that marine productivity is dependent on atmospheric and oceanic circulation patterns, and that the abundance of salmonids and other fishes may be greatly affected by short and long-term variation in those patterns (Mysak 1986; Roesler and Chelton 1987; Francis and Sibley 1991; Ware and Thomson 1991). Growing evidence suggests that conditions in the Northeast Pacific Ocean shifted abruptly in the mid-1970s and that salmonid populations along the entire west coast of North America have responded to these large-scale changes (Francis and Sibley 1991; Percy 1992). It is increasingly clear that efforts to restore freshwater habitats of salmonids must be considered in the context of larger-scale fluctuations in numbers brought on by climatic and oceanic conditions.

In this section we briefly review general circulation patterns and the dominant physical processes controlling conditions in the northeast Pacific Ocean. We then discuss hypothesized mechanisms by which salmonid abundance and life histories may be influenced by changing oceanic conditions. Finally, we discuss some implications of long-term variability in marine conditions on strategies for restoring salmonids in the Pacific Northwest.

### 7.1 General Ocean Circulation

Circulation in the northeast Pacific is dominated by the behavior of the Subarctic Boundary Current, or "West Wind Drift," a large west-to-east surface current situated at approximately 46-49° north latitude. This current bifurcates as it approaches North America, with the Alaska Current flowing north, and the California Current flowing south (Figure 7.1a) (Ware and McFarlane 1989). These surface currents interact with prevailing wind patterns and rotation of the earth to produce distinct upwelling and downwelling patterns along the coast. Wind blowing across the ocean surface causes displacement of surface waters at an angle 90° to the right of the direction of the wind. South of Vancouver Island (BC), northwest winds generally blow along shore from May to September, causing surface waters to be transported offshore and resulting in the upwelling of cold water along the continental margin (Ware and McFarlane 1989). North of Vancouver Island, the movement of surface water is generally directed towards the shore by prevailing winds, and downwelling conditions persist for most of the year (Ware and McFarlane 1989).

The behavior of this large-scale oceanic circulation varies from year to year and at longer time scales, depending on atmospheric conditions and particularly the strength of the Aleutian low-pressure system off the coast of Alaska. In years when the Aleutian Low is well-developed, the subarctic boundary is shifted to the south (Mysak 1986), and a greater proportion of water in the West Wind Drift is diverted northward to the Alaska Current (Percy 1992). These conditions are characteristic of El Niño years, when warm waters from the subtropics shift to the north. Conversely, when the Aleutian Low is poorly developed, the subarctic boundary shifts to the north and the California Current receives a higher fraction of the total water. Ware and Thompson (1991) have proposed that long-period oscillations (40-60 years) in wind-induced upwelling significantly influence oceanic conditions along the coast. Thus, short-term variations in the strength of coastal upwelling and the occurrence of El Niño events appear to be layered on oscillations of longer periodicity associated with atmospheric circulation.

### 7.2 Ocean Conditions and Salmonid Production

Variation in oceanic circulation patterns along the North American coast greatly affects characteristics of seawater, including surface water temperatures, salinity, sea level height, and nutrient concentrations, which in turn affect the abundance and distribution of aquatic organisms. High temperatures, reduced upwelling, and inshore depression of the thermocline during the strong El Niño event of 1982-83 together resulted in significant declines in phytoplankton production along the coast of Oregon (reviewed in Percy 1992). Similarly, zooplankton biomass was greatly reduced and species composition shifted to taxa more commonly found in more southern waters. Roesler and Chelton (1987) attribute interannual differences in spring zooplankton biomass off the coast of northcentral California to differences in advective transport of zooplankton from arctic waters by the California Current. Thus changing ocean conditions can significantly affect the amount of food available to juvenile salmonids as they enter the ocean.

In addition to affecting food supply, changing oceanic conditions may also affect the distribution and abundance of predators and competitors. Holtby et al. (1990) speculated that warmer ocean temperatures off the coast of Vancouver Island may lead to northward shift in populations of large piscivorous predators such as Pacific Hake (*Merluccius productus*). Increases in predator abundance, and concomitant decreases in alternative prey species (e.g., Pacific herring) may result in greater mortality to salmonids off the California, Oregon, and Washington coasts during El Niño years. Another hypothesis attributes fluctuations in ocean survival of salmonids to changes in the off-shore transport of juveniles as they enter the ocean. During years of high upwelling, smolts may be transported off-shore where they are less vulnerable to sea birds and other predators that are abundant along the coast line, whereas in years

of poor upwelling salmonids may remain in near-shore areas (reviewed in Pearcy 1992). Migration routes of Fraser River sockeye salmon juveniles differ substantially in El Niño versus La Niña years (Mysak 1986).

Regardless of the specific mechanisms controlling salmonid abundance in the ocean, there is fairly clear evidence that oceanic conditions play a significant role in regulating survival. Numerous studies have linked marine survival of coho salmon in the ocean with the strength of upwelling (Gunsolus 1978; Nickelson 1986; Fisher and Pearcy 1988). Francis and Sibley (1991) demonstrate long-term fluctuations in catch of coho salmon off the coast of Washington, Oregon, and California, which they attribute to changes in the marine environment caused by climatic change. Interestingly, pink salmon catches in Alaska have oscillated out of phase with coho salmon in the Oregon Production Area, indicating that conditions that lead to high production of salmonids in the Coastal Upwelling Domain have adverse effects on salmon in the Coastal Downwelling Domain and vice versa (Francis and Sibley 1991).

In addition to affecting the survival and productivity of salmonids in the Northeast Pacific, variability in marine conditions has likely influenced the evolution of life history characteristics of salmonids (Holtby et al. 1989). Spence (1995) examined migration timing of coho salmon smolts from 50 populations along the coast of North America and found distinct regional differences in migration characteristics. Coho populations in the northern part of the range typically migrate during a relatively short and predictable period during the late spring. In contrast, southern populations generally exhibit a more protracted migration that peaks earlier in the spring, but is more variable from year to year. Spence (1995) suggests that, in part, these differences likely reflect adaptation to differences in the degree of predictability in oceanic conditions in the northern and southern parts of the coho salmon's range. As knowledge of the marine ecology of salmonids increases, additional patterns in life-history characteristics of salmonids will undoubtedly emerge.

### 7.3 Implications for Restoration

Cycles in marine productivity have the potential to mask the effects of degradation in freshwater habitats. Lawson (1993) presented a conceptual model for considering the combined effects of oceanic cycles and habitat degradation in fresh waters (Figure 7.3a). As freshwater habitats are degraded, salmon populations do not decline in linear fashion. Instead, a general downward trend is masked by long-term oscillations in ocean productivity. During periods of unfavorable ocean conditions, the consequences of degradation in freshwater habitats becomes most evident and the risk of local extinction becomes greatest. As Lawson (1993) points out, there may be a tendency for fishery managers and politicians to relax as populations begin to recover—which they eventually will do provided they do not go extinct during a poor ocean phase—even though the quality of freshwater habitats continue to decline. Similarly, ill-conceived restoration strategies may appear to be successful as salmonid numbers increase, even though those increases are merely the fortuitous result of improving oceanic conditions.

Long-term oscillations in ocean productivity also have a significant bearing on harvest and hatchery management. Harvest projections and limits are typically based on maximum sustained yield models that assume a constant environment. Over the term of their prediction, these models assume linear relationships between production and yield. Such models are particularly problematic in a changing environment, or in one that is tending in a direction different from that in which the model was developed. Similarly, the survival and production of hatchery fish may vary significantly with conditions at sea (Pearcy 1992). In the 1950s and 1960s coho salmon hatcheries in Oregon were enthusiastically endorsed by commercial fishers (who tripled in number over a 10 year period) and fishery managers because of early success that was largely the result of favorable oceanic conditions. When environmental conditions shifted in the mid-1970s, survival of hatchery coho decreased, and the overcapitalized fishery took an increasing toll on wild stocks (Pearcy 1992). The results were significant economic hardship for coastal communities and precipitous decline in wild coho populations.

Lawson (1993) concludes that, in the face of natural variation in ocean productivity, salmonid restoration should proceed in three phases: 1) short-term projects, 2) long-term projects, and 3) monitoring. Short-term projects should be directed at immediate and readily identifiable habitat problems where manipulation can temporarily enhance production (e.g., creation of off-channel pools, cleaning of gravels). Long-term projects should be directed at restoring natural ecological processes, and include such things as replanting of riparian zones or reestablishment of wetlands. Monitoring is essential to ensure that both short and long-term projects are effective, but the metrics used to gauge success may be substantially different. Using numerical abundance of salmonids (particularly juveniles) to measure success has obvious pitfalls with a highly fluctuating ocean environment. The focus should be whether or not restoration strategies are effective over decades or even centuries, not years. For long-term monitoring, indicators should measure restoration of ecological functions or processes, rather than solely counts of fish.

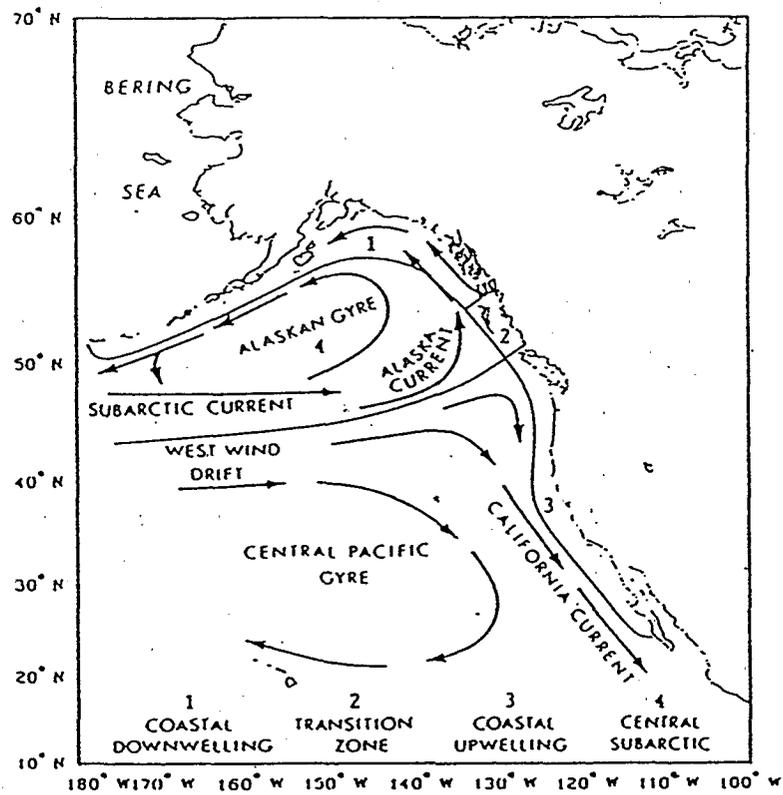


Figure 7.1a. Approximate Areas of Oceanic Domains and Prevailing Current Directions in the Northeast Pacific Ocean. From Ware and McFarlane (1989). Reproduced with permission from the publisher.

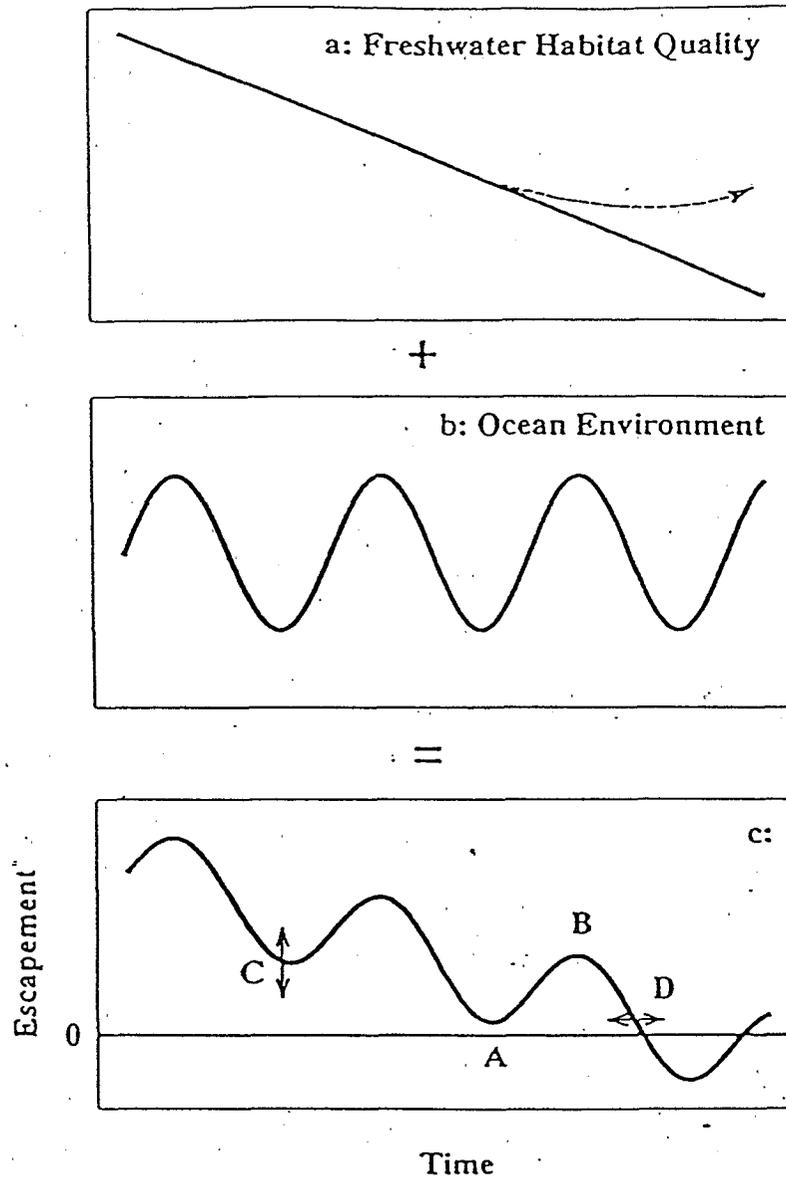


Figure 7.3a. Conceptual Model of Effects of Declining Habitat Quality and Cyclic Changes in Ocean Productivity on the Abundance of Oregon's Coastal Natural Coho Salmon\*. From Lawson (1993). Reproduced with permission from the publisher.

\*Dotted line represents possible effects of habitat restoration projects. a: Trajectory Over time of Habitat Quality. Dotted Line Represents Possible Effects of Habitat Restoration Projects. b: Generalized time series of ocean productivity. c: Sum of top two panels where Letters Represent the Following: A = Current Situation, B = Situation in the Future, C = Change in Escapement from Increasing or Decreasing Harvest, and D = Change in Time of Extinction from Increasing or Decreasing Harvest.

## 8.0 Programs Designed to Restore or Protect Salmonids and Aquatic and Riparian Habitat

In this section we describe existing programs and proposals for restoring or protecting salmonid stocks and habitats. Most such programs have focused primarily on one problem area (e.g., stream bank, stream reach) and one stressor (e.g., grazing, agriculture, forestry, harvest) but have not tried to integrate other stressors into a comprehensive solution for the watershed. Although, we discuss the programs separately, we emphasize the need to integrate them into a true ecosystem management approach at multiple spatial scales from catchments to regions. In addition, wherever possible, the primary focus should be on protecting existing high quality habitats and populations rather than on restoration of already decimated systems (Bradbury et al. 1995). We first discuss programs that focus on fish and water bodies, then move to land use practices that influence water body integrity, and end with the societal values and policies that ultimately determine ecosystem integrity.

### 8.1 Harvest and Hatchery Practices

Two direct stressors leading to the loss of salmon are overharvest and hatcheries. Overharvest reduces both the abundance and individual size of salmon; both conditions now occur in this region. There are also numerous examples of exploiting fisheries to the point of extinction, and few have fully recovered. A mixed stock fishery, such as the ocean troll or lower Columbia fishery, harvests strong and weak stocks at the same rate, meaning that weak stocks decline and eventually may become extinct. For example, average harvest rates of 88% are set for Columbia River coho, which are of largely hatchery origin, versus 69% for wild Oregon coho. The higher harvest rate hinders wild coho escapement, while a lower harvest rate increases potential straying of hatchery fish and subsequent gene pool dilution.

Terminal and bay fisheries, (i.e., fisheries that target adults as they return to their natal streams) providing greater protection for weak stocks by allowing late-maturing fish like chinook to reach maturity, and reducing the incidental mortality of subadults. Near-shore fisheries would also create fewer hazards to fishermen, reduce fishing costs, and increase competition for permits. Such a system could also be easily opened and closed to increase escapement at particular times during the run. If traps, fish wheels, and barbless hooks were used instead of gillnets, incidental mortality could be further reduced by limiting catch to hatchery fish. Also, more accurate monitoring of escapement levels of specific stocks is essential for establishing exploitation levels that ensure the long-term persistence of individual stocks (e.g., Cooney and Jacobs 1994).

There is also growing evidence of the need to greatly decrease or eliminate hatchery releases to protect wild fish from genetic pollution wherever possible. Although this would be disruptive of harvest levels, direct competition and gene pool dilution by hatchery fish is an extremely serious threat to continued persistence of wild salmonids. Targeting of hatchery fish for exploitation is an additional option.

In deciding whether to use artificial propagation to conserve endangered species, a key consideration is the likelihood that such efforts will actually benefit the listed species (Waples 1991b). Although artificial propagation of Pacific salmon has been carried out on a large scale for several decades, almost all these efforts have been directed at fisheries harvest enhancement. Attempts to increase natural production through the use of artificial propagation is a relatively recent enterprise that has, to date, produced mixed results (Miller 1990). The use of artificial propagation to conserve listed species should be viewed as experimental and highly risky given the historical effects of hatcheries on reducing salmonid biological diversity (Bottom 1995).

One of the most disruptive influences of hatcheries has been the introduction of non-native species to Pacific Northwest waters. These include large piscivores such as walleye and bass, together with efficient competitors like brook trout. Management strategies that encourage elimination of non-native fishes include direct removal by piscicides and electrofishing, and indirect removal through use of unrestricted catch limits. Barriers to migration have reduced the spread of these species, but a return to more natural flow fluctuations has been found to restrict their distribution and abundance. Finally, it is important to realize that it is the fish management policies of state agencies and citizen commissions that have resulted in the introductions of non-native stocks and species. As the attitudes of managers and the citizenry change, introductions of non-native fishes will continue to decline.

### 8.2 Waterway Modification

Throughout human history, waterways have been used for water supply, waste discharge, power generation, and transportation, and cities were established along them. With growth in populations and commerce, streams, rivers, and estuaries were increasingly stabilized and simplified by dams, channelization, revetments, snagging, and removal of wetlands and side channels. Seasonal peak and low flows were stabilized by dams, although daily fluctuations were increased to meet power demands. Water quality deteriorated as a result of the above changes plus waste discharges and water withdrawals. Particularly damaging to aquatic life were increased temperature, turbidity, sedimentation, toxics, nutrients, and oxygen-demanding wastes. The progressive commercial, agricultural, and urban development

stimulated additional alterations, often at increasingly greater distances. In places such as western Europe, where these changes have been taking place for centuries, natural channels and high water quality remain in only the most remote areas.

We now recognize the need for healthy waterways. U.S. rivers no longer burn, impede boat traffic with floating wastes, or preclude activities because of nauseating odors. Sensitive forms of aquatic life and recreationalists have begun to return to all but the worst of our water bodies, and the frequency of waterborne disease has been dramatically reduced. Nationally these changes were stimulated and financed by the Federal Water Pollution Control Act and its subsequent amendments; however, several progressive states and dischargers preceded the Act in cleaning up point source discharges. Several lessons in waterway protection and restoration can be learned from the Act, nonfederal responses, and related legislation: (1) A clear set of goals and objectives are needed; (2) Funding must be set aside for applied research and training; (3) Cost-sharing grants are often necessary to construct treatment works; (4) Numerical criteria are required for various pollutants; (5) Both site-specific and basin-wide permitting, monitoring, and reporting systems are useful for controlling pollution and evaluating compliance; (6) Improved land-use effectively controls diffuse pollution; (7) Ambient biological criteria and biological surveys are necessary for evaluating the biological effectiveness of controls; (8) Reducing wastes frequently saves dischargers money; (9) The federal government can provide overall guidance and direction, but states can most effectively conduct the monitoring and enforce the regulations; (10) Ultimately, it is the character and creativity of the discharger that determines the effectiveness of waste reduction.

Recently, researchers have identified the vital role of channel complexity, riparian zones, and floodplains in the productivity and diversity of the aquatic habitat and its organisms (Sedell and Luchessa 1981; Maser et al. 1988; Gregory et al. 1991; NRC 1992). Armed with this new knowledge, numerous programs are underway to increase river-floodplain interactions for the benefit of instream habitat and riparian zones. Approaches range from active natural channel restoration to non-structural passive methods for floodplain protection. The Kissimmee River, Florida restoration project has been called a model for watershed restoration. It calls for physical reconfiguration of the river to reestablish backwaters and contact with the floodplains, and reestablishment of historical inflows from the river's upper watershed. A similar effort has been planned for portions of the Missouri River (Hesse and Sheets 1993). The approach includes an evaluation of pre-dam and pre-channelization conditions, which are considered essential for successful reconnection of historic hydraulic features.

Other researchers have identified strategies to minimize adverse affects of channelization through stream renovation. McConnell et al. (1980) suggest that snagging only one side of a stream halves costs and deleterious impacts to the environment. Nunnally et al. (1978) propose minimizing erosion and reducing hydraulic efficiency by maintaining natural meanders when channelizing streams. The more natural stream channels and riparian vegetation also improve biological and aesthetic benefits. Alternative strategies such as these involve significant environmental tradeoffs, but are less destructive than traditional channelization.

Programs are now beginning to focus on dam removal to restore natural ecological processes. The Maine Legislature requested that Edwards Dam be removed from the Kennebec River to improve migration of Atlantic salmon. Despite fish passage modifications the dam still blocks salmon migration, as well as migratory sturgeon, shad, and smelt, yet provides electricity to fewer than 2000 homes (NRC 1992). The Elwha River Ecosystem and Fisheries Restoration Act (Public Law 102-495) was passed in 1992 to restore the ecosystem and historic runs of five Pacific salmon species (once the most prolific runs on the Olympic peninsula of Washington). The dams also are associated with the disappearance of 22 bird and mammal species from that area of Olympic National Park and annual recreational revenue losses of \$500,000, while the electricity is sold to a paper company. An analysis determined that removal of both dams is the only option that will allow full restoration of the watershed (NRC 1992). In Oregon, two dams on the Rogue River have been identified for potential removal. The Bureau of Reclamation (1994) concluded that removal of Savage Rapids Dam would cost taxpayers nearly \$8,000,000 less than dam retention and produce 24,000 more adult salmon and steelhead. Additional dam removals are called for throughout the Pacific Northwest because of inadequate or no fish passage, excessive sedimentation, structural deterioration, and hazardous or unsafe conditions—especially when federal funds subsidize only local interests.

A variety of programs have been aimed at maintaining the viability of fish stocks by mitigating the impacts of dams. These programs provide upstream passage for mature fish and downstream passage of smolts. The Northwest Power Planning Council's Strategy for Salmon identifies immediate actions to aid juvenile passage (NPPC 1992a, b): screen all turbines, improve bypass systems, lower reservoir pools during smolt migration, barge past dams, place a bounty on predators, and boost flows during out-migration. The Snake River Recovery Team made similar final recommendations to NMFS (Bevan et al. 1994a, b, c). Beaver dams have been recommended for improving stream habitat complexity, slowing stream incision, and increasing dry season flows. Although beaver ponds eliminate spawning areas, warm the water, and reduce dissolved oxygen, they create additional areas for rearing, over-wintering, escaping freshets, and trapping sediments. Their introduction to higher gradient salmonid streams is associated with higher salmon densities and significantly greater overwinter survival (Phillips 1987; Swanston 1991).

The Fish and Wildlife Service has developed the Instream Flow Incremental Methodology (IFIM), to provide a means to systematically evaluate alternative flows for the protection or enhancement of aquatic resources (Armour and Taylor 1991). IFIM's development has allowed quantitative habitat comparisons of different flow regimes to determine minimum allowable instream flows. This is significant for eastside streams where irrigation needs may severely deplete base flows. Though widely used, a number of concerns of IFIM concepts have been expressed. Nestler (1990) summarizes these concerns arguing that depth, velocity, and substrate are inadequate estimates of habitat quality and that habitat quality is only a coarse estimate of population density. Stalnaker (1990) argues that IFIM has been used to establish minimum flows that are frequently violated and that ignore other necessary flows such as floods. Armour and Taylor (1991) point out that the methodology was designed to evaluate alternate flow regimes, not as the definitive answer for flow disputes. The results simply provide a framework for negotiating flows to be maintained.

During the past two decades, increasing effort and resources have been committed to instream artificial structures designed to improve fish habitat. The National Research Council (1992) provides a summary of 22 habitat improvement evaluations deemed successful, based on increased fish density during the period evaluated. No time period for evaluation is given nor are metrics provided that were used as the basis for the evaluation. The results from these studies should be interpreted with caution since they originated in systems east of the Rocky Mountains, systems having different climate, geology, sediment transport, hydrology, and gradient than Northwest streams. In contrast, Frissell and Nawa (1992) suggest that "commonly prescribed structural modifications often are inappropriate and counterproductive." They report damage to structures from streams throughout western Oregon and southwest Washington to be frequent in low grade reaches and widespread in streams with recent watershed disturbance when evaluated for 5-10 year damage rates. Overall median failure rate was found to be 14% and median damage rate (impairment plus failure) was 60%. They specify that streams with high or elevated sediment loads, high peak flows, or highly erodible bank materials, such as many found in the Northwest, are not good candidates for structural modifications. Restoration of fourth order and larger alluvial valley streams, areas identified as having the greatest potential for fish production in the Pacific Northwest, will require natural watershed and riparian processes to be reestablished over the long term. Reeves et al. (1991) take a neutral stance in describing numerous structure and habitat manipulations (gravel cleaning, gabions, weirs, log sills, cedar baffles, fishways, boulders, log structures), and provide an evaluation of their use and applicability for variable life history requirements and differing watershed settings. They caution that much work has been done with very little pre- and post-evaluation of the results, and that successful future projects will depend upon careful evaluation of existing projects. Reeves et al. (1991) conclude that "habitat rehabilitation must never be viewed as a substitute for habitat protection, and ... prevention of initial habitat degradation is more economical of total resources than repairing it, and some damage simply is not reversible".

### 8.3 Forestry Practices

As this nation was settled forests were cut to provide building materials, open up cropland, supply fuelwood, and to be converted into paper. The great coniferous forests of first New England, then the Upper Midwest, and lastly the Pacific Northwest were logged. As the forests disappeared from these regions so have the salmonids. In the process, growing numbers of people inside and outside the timber industry have come to recognize the need for changes in logging. In this section, we briefly review methods for minimizing the effects of forest practices on aquatic ecosystems. Because of the greater importance of forestlands to salmonids, more information is provided in this section than in those for other land uses.

The influence of forest practices on watershed processes and aquatic ecosystems is affected by the harvest schedule, harvest type, harvest equipment, road construction and maintenance, site preparation, and intermediate treatments. To minimize impacts, practices can be selected that are least disruptive to natural watershed processes. Forest practices that will most effectively protect stream ecosystems vary with local biological and physical characteristics. Although riparian activities pose the greatest risk to salmonids, upland practices affect surface erosion, mass wasting, hydrologic processes, and nutrient dynamics, and therefore must be considered.

#### 8.3.1 Upland Forest Management

Rotation schedule influences watershed dynamics by determining the frequency of disturbance to the watershed, affecting the area disturbed, and influencing the materials delivered to a water body. Most commercial forests in western Washington, Oregon, and California have been harvested on a rotation of 45-100 years (Frissell 1991; Hicks et al. 1991). Growth rates of conifers east of the Cascade Crest are lower because of less water, so rotations are generally longer.

Effects of harvest on stream temperatures, hydrology, surface erosion, and the probability of mass failures are generally greatest in the years immediately following logging because the degree of devegetation and soil disturbance is highest during this period. Lengthening the harvest rotation decreases the time that the landscape is disturbed and reduces the probability of catastrophic events. For example, if the risk of landslides is increased for a period of 15 years

after logging, then a stand managed on a 60 year rotation will be vulnerable for 25 years per century, versus 15 years for a stand managed on a 100 year rotation (Frissell 1991). If the average rotation is 60 years, on a regional scale, this means that 25% of the landscape is vulnerable to landslides at any time, versus 15% with a 100 year rotation or 5% with a 300 year rotation.

Recruitment of large wood into streams takes much longer than 15 years. In western Oregon, where deciduous trees have largely replaced conifers in riparian zones following logging, recruitment of significant quantities of woody debris may take 50 years (Grette 1985; Heimann 1988). For coniferous species, complete recovery of large wood recruitment may take 100 years or more (Gregory et al. 1987). Moreover, the larger the wood in a stream, the greater its stability and ability to perform critical functions. Presently, many waterways are starved of large wood and little is available in riparian zones to contribute significantly to the channels for many decades. Again assuming either a 50 or a 100 year rotation, this means that streams would be continuously unstable and scoured. Clearly, rotations must be substantially longer than the current norm to ensure that high-quality large wood remains an important functional element of aquatic ecosystems.

Harvest can generally be divided into "even-aged" and "uneven-aged" methods (Young and Giese 1990). Even-aged methods are those in which the timber stand consists of trees of similar age and size and includes such methods as clearcutting, seed tree methods, and shelterwood cuts. Uneven-aged methods consist of those where trees are selectively harvested and where the resulting stand consists of trees of varied ages.

Clearcutting has been the dominant harvest method in forests of the Pacific Northwest since the turn of the century (FEMAT 1993). Clearcutting is potentially more disruptive of natural watershed processes—including hydrology, sediment transport, energy transfer, nutrient cycling, and stream habitat development—than other methods because virtually all vegetation is removed and soil usually is highly disturbed.

Patch cuts are relatively small clearcuts distributed over the landscape. The effectiveness of patch cutting in mitigating effects on watershed processes depends on the size and location of the harvest units and the total percentage of the watershed harvested. Small patches may reduce hydrologic impacts because vegetation in areas surrounding the cut may take-up some of the additional available water. Patch cuts may facilitate greater snow deposition, however, and result in greater runoff during rain-on-snow events than a single large clearcut. While several patch cuts may be less disruptive to soil and hydrologic processes than a single clearcut of equivalent total area, the resulting fragmentation of the watershed may be more detrimental. Dispersed logging and roading increases the probability of multiple chronic landslides across the landscape, rather than focusing such disturbances in particular catchments. Also, there is a growing consensus in the scientific community that large, intact systems function better as wildlife habitats than highly fragmented systems. Multiple patch cuts may require a higher degree of roading than a single harvested area, further diminishing the ability of habitats to support certain wildlife species. These tradeoffs should be considered when developing timber harvest strategies.

Seed tree and shelterwood cuts differ from clearcuts in that some trees are left on site to provide seed sources for regeneration and, in the case of shelterwood cuts, to provide some shade for seedlings. Following the establishment of seedlings, the remaining large trees are removed, leaving an even-aged stand. Both of these methods can potentially reduce hydrologic effects and surface erosion. Trees that remain on site may exhibit compensatory growth in response to increased water availability, thereby minimizing increases in runoff. Standing vegetation also serves to reduce erosion, although seed trees are likely to have minimal effect on the probability of mass failures.

Selective harvest methods entail the removal of only a portion of the merchantable trees from a cut. In some circumstances, only the largest trees are removed. Alternatively, harvest may involve selective removal of younger trees while leaving large, older trees standing. Proponents of new forestry techniques argue that removal of younger trees hastens the development of the characteristics typical of old-growth and late-successional forest that are desirable to some forms of wildlife, including the northern spotted owl. However, it is not known whether such measures will benefit other late-successional forest species (FEMAT 1993). The benefits of selective harvest vary depending upon the percentage of the basal area removed and the composition of the remaining stand. If properly done, selective harvest can maintain stream shading and input of allochthonous materials, minimize disturbance to soils, reduce soil compaction by ground-based equipment, and minimize effect on hydrologic processes. Disadvantages of selective harvest may include increased frequency of disturbance, suboptimal regeneration of trees, increased density of roads, and increases in harvest costs under some circumstances.

Considerable disturbance occurs during skidding or yarding of logs. Several systems have been devised for yarding that differ in their impact to soils. Ground-based operations using tracked or rubber-tired skidders generally result in the greatest disturbance to soils both in terms of degree of compaction and the percentage of total area affected. Such disturbances increase surface runoff and erosion potential. Because most compaction occurs with the first few passes of heavy equipment, soil disturbance can be reduced by establishing designated skid trails, thereby minimizing the area receiving traffic. Because compaction is highest when soils are moist, impacts may be reduced by skidding either during the dry season or when the ground is frozen.

In cable-yarding systems, logs are attached to a cable and dragged to the landing. In general, a lower percentage of the logged area is disturbed with this method; however, skid marks may channel water and erode, thereby facilitate erosion. High lead systems are similar, with the cable running through an elevated pulley. Skyline systems lift part of the log off the ground as it is dragged to the landing site, further reducing the impact to soils. Full suspension methods lift logs completely off the ground. Balloon and helicopter logging have also been employed to reduce the need for roads and ground-based equipment. In general, the area affected by logging equipment and percentage of bare soil remaining is greatest for tractor and cable logging, intermediate for highlead systems, and least for full suspension, skyline, and balloon logging (Pritchett and Fisher 1987).

Logging roads are primary sources of sediments to streams, both through chronic erosion and as initiation points of mass failures. In addition, the higher the road density, the greater the probability of significant alteration to hydrologic processes. Impacts can be greatly reduced by careful placement, construction, and maintenance of roads. Guidelines for minimizing the effects of road construction are discussed in detail in Furniss et al. (1991) and are specified in the forest practice rules of most western states.

Minimizing total roaded area is the most fundamental means for reducing sediment inputs to channels. Long-range planning of road systems within a watershed helps minimize total roaded area, and reduces construction costs (Furniss et al. 1991). In general, roads should be located away from stream channels, particularly in steep terrain where the likelihood of fill material washing into the stream is high. However, it may be preferable to construct roads in valley bottoms rather than on slopes that have a high probability of failure. In these instances, a buffer strip between the road and stream can reduce disturbances to the channel. Locating roads on ridgelines, as opposed to mid-slope areas, and on dry soils instead of in wet areas, also minimizes erosion risks (Furniss et al. 1991). Knowledge of local soils and geology is essential to preventing unwise placement of roads. Hummocky ground, jack-straw trees, and sag ponds are often good indicators of unstable hillslopes subject to slumping or slides. Creek crossings are frequently the sites of significant erosion. Minimizing the number of crossings is both desirable and required under forest practice rules of some states (e.g., California). When crossings are unavoidable, they should be located where the amount of channel modification and fill material is minimized (Furniss et al. 1991).

The principal considerations in designing and constructing roads to minimize effects on salmonid habitats are ensuring adequate drainage, preventing excessive sedimentation, and providing for fish passage at stream crossings. Specific recommendations of Furniss et al. (1991) aimed at protecting aquatic resources are summarized in Table 8.3.4.2a. Sediment transport is generally tightly coupled with the routing of water on the landscape. Thus, most sound design and construction techniques are devoted to maintaining natural drainage patterns and preventing the concentration of runoff.

Regular and timely maintenance of logging roads helps ensure that drainage and erosion control structures are functioning properly and allows identification of problems that could have adverse consequences. The costs associated with maintenance are generally low compared with reconstruction costs after a significant failure. Grading roads to ensure outsloping surfaces, and clearing of drainage ditches and culverts can ensure that drainage occurs as intended. Where problems are observed, installation of additional ditch-relief culverts or large culverts may alleviate erosion and drainage problems. Seasonal road closures may also be an effective way to reduce sediment delivery to streams.

Once harvesting has been completed at a site and the road is no longer needed, reseeded of the road bed with grasses reduces the amount of exposed soil and thereby diminishes surface erosion. However, this practice has little effect on the potential for deep mass wasting. Recently, there has been growing support for revegetating or decommissioning roads by pulling sidecast material back onto the road bed and reforming the natural slope (Harr and Nichols 1993). Decommissioning of roads involves disturbance to restore natural morphology. Nevertheless, Harr and Nichols (1993) reported that decommissioned roads and landings sustained little or no damage following two significant rain-on-snow events that caused substantial damage to main haul roads in northwest Washington. Similarly, Weaver et al. (1987) reported that obliteration of problem road surfaces and fills, deconstruction of stream crossings, and re-contouring of disturbed slopes were effective techniques for reducing sediment input to streams in Redwood National Park. Other techniques aimed at surface and rill erosion problems were less successful at reducing sediment delivery and tended to be more costly. Both Harr and Nichols (1993) and Weaver et al. (1987) noted that a careful survey of road conditions that allowed them to identify significant problem areas was essential to successful cost-effective application of rehabilitation techniques.

Regeneration of coniferous vegetation in some regions involves reducing shrubs and deciduous trees, eliminating logging debris, and preparing soils for planting. Three techniques have been used for site preparation: burning of slash, mechanical clearing of vegetation, and chemical treatments. As noted earlier, the effects of burns on soil characteristics vary with the intensity of the burn. High-intensity fires can eliminate litter layers and create hydrophobic conditions in surface soils, thereby increasing the amount of exposed soil and the potential for surface runoff. Low-intensity burns, on the other hand, generally cause minimal damage to soils (Pritchett and Fisher 1987), and are therefore less likely to result in changes to hydrologic or erosional processes.

Mechanical clearing potentially has the greatest effect on soil conditions and hydrologic processes, and often causes greater damage to soils than the initial logging. Because heavy equipment must travel over most or all of a site, the potential for soil compaction and exposure of bare soils is high, which can lead to increased surface runoff and erosion (Pritchett and Fisher 1987). In addition, valuable topsoil may be redistributed, with much of it ending up in burn piles.

Chemical treatments may be least physically damaging to a logged site; however, care must be taken to ensure that chemicals do not reach stream systems. Limiting spray operations to calm days, using the minimum effective concentrations, and refraining from spraying in riparian zones can minimize the risk of exposure of aquatic organisms to toxic chemicals. In addition, applications can be timed so as not to overlap with sensitive life-history stages. Norris et al. (1991) conclude that the herbicides used in forestry are relatively immobile in soils and that leaching into subsurface waters is less likely in forested soils than in other environments. From a hydrologic standpoint, application of herbicides is likely to extend the period of increased water yield from a site by reducing evapotranspiration losses. Conversely, at sites where deciduous vegetation will replace coniferous trees if untreated, chemical treatments may prevent reductions in summer streamflow that may occur over longer periods of time.

Intermediate treatments are actions designed to enhance tree growth, and include thinning, pruning, and fertilization. The impacts of thinning and pruning activities are generally related to the type of equipment used and the care demonstrated by the operator. Lighter equipment will generally cause less disturbance to soils and ground cover than heavy equipment. In a well-planned thinning, the use of equipment will be restricted to designated roads and skid trails, thereby minimizing additional soil compaction. Indiscriminate use of equipment, on the other hand, can result in compaction of soils that were left intact during the original harvest operation. Provided that additional soil compaction does not occur, thinning or pruning is unlikely to have a significant effect on water balance in a forest stand because remaining vegetation will take up additional water that becomes available. The effects of fertilization on aquatic systems can be minimized by refraining from fertilizing riparian zones. Fertilizing in ephemeral channels can lead to high concentrations of nitrogen in downstream areas when rainfall begins in the fall and fertilizers are mobilized (Moore 1971). Thus, foresters should avoid applying fertilizers near all permanent streams, ephemeral streams, and drainage channels.

### 8.3.2. Riparian Forest Management

Floodplain and riparian forests in the Pacific Northwest once supported some of the largest and fastest growing trees, and they were among the first that were logged because of the relative ease of transporting logs via waterways. Recent ecological research, however, has indicated the importance of floodplains, floodplain wetlands, and riparian zones for storing and slowing floodwaters, absorbing pollutants from runoff, reducing sediment delivery to streams, maintaining channel complexity, supplying shade and large woody debris, providing shallow water areas for foraging and spawning fish and amphibians, and supporting a highly diverse community of plants and animals (BLM et al. 1994; Cederholm 1994). The practice of leaving riparian buffer strips along streams is now widely applied and is viewed as perhaps the most important aspect of protecting stream habitats from the effects of logging and other land-use activities (Cummins et al. 1994). Three important considerations in establishing buffer zones are: 1) the width of the buffer zone, 2) the level of activity allowed within the riparian zone, and 3) whether riparian buffers are needed for tributary streams that do not contain salmonids. Appropriate buffer widths are the topic of much debate and a number of alternative approaches for determining adequate buffer widths have been proposed (FEMAT 1993; Cederholm 1994; FS and BLM 1994). The appropriate width of buffer zones depends on the specific functions that are being considered. Figure 3.9.1a illustrates generalized curves for the zones of influence of riparian vegetation (FEMAT 1993). Litter inputs and bank stability are generally provided by trees within 0.5 potential tree heights of the channel. Shading and large woody debris are provided by trees farther from the streams. The effect of vegetation on sediment and nutrient inputs may extend even farther from the channel, though these influences are more difficult to define. Complete protection of salmonid habitats requires that all of these functions be maintained.

The influence of riparian vegetation also depends on physical and biological characteristics of the watershed, including topography, soil type, geology, and vegetative cover. For example, the likelihood that large logs will end up in the stream channel is greater on steep slopes compared to gentle slopes. Similarly, topographic shading in canyons may reduce the importance of shading by vegetation. The FEMAT approach to riparian buffers establishes buffer widths, depending on stream and land classifications, that can be adjusted following additional watershed analysis. This approach acknowledges that variable buffer widths determined in accordance with site conditions can maintain critical instream characteristics.

The second significant consideration in the management of riparian zones is the level or intensity of disturbance allowed within them. State forest practice rules allow harvesting of timber within riparian areas but at lower levels than in surrounding uplands, which are usually clear-cut. Usually, specific criteria establish the number of trees, species composition, basal area, canopy cover, or other measures of the vegetation to be left. For example, Washington's Forest Practices Rules and Regulations (Rashin and Graber 1992) establish streamside buffers that

provide some level of shading for the purpose of maintaining suitable water temperatures, providing wildlife and fish habitat, and protecting the physical integrity of the stream—despite logging activities in the watershed. The rules require that 50% to 75% of the pre-harvest shade be maintained in temperature sensitive streams, which may not protect some ecological processes (see Section 4). In addition, because the focus is on stream temperature, other processes (e.g., sediment transport, large wood delivery) may not be protected. Furthermore, a 1990 evaluation indicated that water temperature criteria were met at only 3 of 13 sites evaluated. In general, the objectives of forest practice rules are to prevent violations of water quality standards. The implicit assumption of these rules is that low levels of disturbance are acceptable and have minimal affect on salmonids and their habitats. An alternative view is that the target of riparian management should be no impairment of riparian function and that downstream and cumulative effects must be considered. Cederholm (1994) proposed that riparian zones should be identified and buffer zones should be established around the riparian zone to prevent modification of riparian function.

Finally, impacts of logging can be reduced if buffer zones are left around small headwater streams that themselves do not support salmonids (Cummins, Botkin et al. 1994). In particular, steep headwater drainages are frequently the trigger points of landslides. Minimizing road construction and logging around first order and temporary channels can prevent frequent mass soil movements that propagate downstream, to the detriment of salmonids.

## 8.4 Range Practices

Rangelands worldwide have been decimated by overgrazing. Livestock grazing occurs on 70% of the land in the western U.S., including wildlife refuges and wilderness areas, making it the most widespread land use in the western states (Fleischner 1994). Since the 1930's western U.S. rangelands have benefited from less intensive grazing and less severe droughts; however, 64% of western grazing lands are in only fair or poor condition, producing less than 50% of potential forage (Busby 1979; Heady and Child 1994a). Given such catchment conditions, it is little wonder that rangeland streams have incised or widened and that riparian zones have deteriorated.

### 8.4.1 Upland Range Management

Although the greatest potential for improvement is in rangeland riparian areas, the hydrological changes that accompanied decades of overgrazing and the current state of rangelands call for further alterations in upland range management as well. Three possibilities are suggested by Heady and Child (1994a): lower livestock stocking rates; greater recognition of the values of wild ungulates, and increased evaluation of grazing plans. Large gains could come from lower stocking rates, in fact this is a component of the other two approaches. At high stocking rates livestock do not gain weight quickly or at all, or they lose weight and condition. At the same time, the range deteriorates or fails to recover. These all cost ranchers money. In addition, high stocking rates maximize financial losses when livestock prices fall between the time of calf acquisition and sale. As with fishery catch per effort curves, yield rises constantly with increased livestock density (assuming a constant environment) towards some optimum, then falls sharply slightly beyond that optimum production level. This makes it very difficult to select an optimum stocking rate in a predictable environment, let alone a highly variable one. Heady and Child (1994a) report that, for both environmental and financial reasons, increasing numbers of managers are shifting toward lower stocking levels.

Although common in eastern and southern Africa, where native diseases limit livestock, game ranching and cropping have not been as widely adopted in this country. Livestock grazing benefits wildlife species that prefer habitats altered by livestock, but harms those species preferring natural habitats, or those competing with livestock for food (Heady and Child 1994a). Therefore, depending on the desired species of wildlife, livestock may need to be reduced or removed. If livestock are removed, it should be remembered that native wildlife populations can sustain high harvest levels, for example, some deer populations have been harvested at rates of 33-50% annually for decades without detrimental effects (McCullough 1979; Heady and Child 1994a). Various game management and harvest programs are also possible on these lands, depending on whether the objectives are trophies, meat, or viewing wildlife. Selling prices range from several thousand dollars for a trophy animal to a few dollars for wildlife viewing. However, wildlife viewing is a non-consumptive, repeatable activity. Benefits in addition to range improvement include lower management costs, leaner meat supply, and higher biological integrity.

Four general types of grazing plans have been developed for range improvement: continuous, repeated seasonal, deferred, and rest (Heady and Child 1994a). As the name implies, continuous grazing is grazing throughout the growing season and usually some part of the dormant period, thus the length of time varies with climate. Repeated seasonal grazing refers to annually grazing the same pasture during a specific season, similar to the patterns of migratory wildlife. Deferred grazing means no grazing is conducted until key plants have completed reproduction. Rest once referred to a year without grazing but has since been generalized to any specified period. Where range condition is less than excellent or in arid and semiarid regions, range recovery may require many years of deferment or rest. The success of each is a function of site characteristics, periodic monitoring, and low stocking densities. These will determine which pastures to graze, which season or year to graze them, and for how long.

#### 8.4.2 Riparian Range Management

Western riparian areas are among the most productive ecosystems in North America, yet their present condition is believed to be the worst in American history, largely because of livestock grazing (Fleischner 1994). Acknowledging the need, and including a strategy to manage the entire watershed, Barrett et al. (1993) establish a BLM goal of 75% or more properly functioning riparian wetlands by 1997. They define proper functioning condition as adequate vegetation, landform, or large woody debris to: dissipate stream energy associated with high flows, thereby reducing erosion and improving water quality; filter sediment, capture bedload, and aid floodplain development; improve floodwater retention and groundwater recharge; develop root masses that stabilize streambanks against cutting action; develop diverse ponding and channel characteristics to provide the habitat and the water depth, duration, and temperature necessary for fish production, waterfowl breeding, and other uses; and support greater biodiversity.

Elmore (1992) and Platts (1991) summarize a number of grazing strategies, as well as differences in foraging use by sheep and cows (Table 8.4). Heady and Child (1994a) and Platts (1991) identify seven options to be considered singly or in combination for achieving grazing goals while maintaining or improving fish habitat. They include, rest from grazing, control livestock distribution, control livestock numbers, control forage use, control timing of forage use, determine the kind of livestock best suited for a given area, and artificial rehabilitation of stream riparian ecosystems. One-to-two years of rest out of three provided improved riparian vegetation when forage consumption was 25–60% (Kauffman and Krueger 1984; Heady and Child 1994a), but daily herding of livestock from riparian areas is also successful. Heady and Child note that successful grazing systems usually require fencing, offstream water, or both.

In a review of livestock-riparian interactions, Kauffman and Krueger (1984) report that better livestock management was a less costly strategy than instream structures for restoring channel integrity. They add that the need for structures is negated by rest from grazing and that structures are ineffective when associated with grazing. Channel structures were considered by Heady and Child (1994a) to be treating the symptoms of improper grazing rather than the cause.

Like ecosystems everywhere, each stream reach or riparian zone has a certain gradient, geology, suite of soils, climatic regime, and level of anthropogenic stress (Elmore 1992). Rangeland streams, however, typically suffer from low or highly fluctuating flows and rocky or highly alkaline soils that severely limit riparian vegetation (Crouse and Kindschy 1981). If other preexisting stress levels are high, the capacity to absorb additional grazing stresses or to recover when they are removed may be low.

Grazing stress is a function of how and when a given strategy is used and not simply the total number of animals grazed on a given lot. Selective timing for rotation grazing and strategies to allow growth of riparian vegetation during critical periods are just as effective as minimal grazing effort in some areas (Elmore 1992). To be effective, strategies must integrate the natural potential and the expected grazing stress for a given stream reach (Elmore 1992). Therefore, prescription grazing for a given area based on present natural stresses offers the best strategy for minimizing impacts and allowing some level of grazing. Fleischner, (1994) however, argues that total removal of livestock from riparian zones is necessary for their restoration. In the absence of site specific information, riparian grazing should be deferred wherever possible until vegetation and channels have returned to pre-grazed conditions.

### 8.5 Agricultural Practices

Agriculture and human civilizations began on floodplains because of the availability of water and the fertility of soil there, but they quickly spread to the uplands. Floodplains and riparian areas in agricultural lands remain some of the most disturbed areas in the landscape. Regulations, management practices, and any other activities that completely protect floodplains, riparian areas, and uplands in a natural state ensure that channel and riparian functions are unimpaired. Recognizing that totally protecting large numbers of agricultural watersheds or floodplains is impractical with our current exploitation rates, management strategies are recommended that preserve most critical functions while allowing some resource consumption.

#### 8.5.1 Upland Cropland Management

Current agricultural practices offer ample opportunities for conservation. Of all the water diverted and consumed in the Pacific Northwest, 90% is used to irrigate crops (Pacific Northwest River Basins Commission 1979; Wilkinson 1992). Irrigation is the major stressor in reduced salmon runs in the Yakima, Walla Walla, and Umatilla Rivers (NPPC 1986). Where irrigation withdrawals are substantial, one of the most important current management actions for restoring salmonids is instream water rights. Without sufficient flows, other restoration activities are futile. State and federal fish and wildlife agencies have established most of these rights, but land owners in some states, including Oregon, have also granted or sold water rights to fishermen's groups or the state and generated more net income than they did from marginal crops (Wilkinson 1992). One of the potentially most useful tools for maintaining or restoring instream water is the 1908 U.S. Supreme Court decision on *Winters vs. United States*, which decreed that Indian tribes possessed water rights that were superior to those established by state law. Another legal approach

involves the public trust doctrine of English common law, which holds that the rights to water on larger water bodies cannot be controlled by a single part of the population (Wilkinson 1992).

Water conservation is among the best ways to provide more water for aquatic life, because so much is currently wasted. Effective tools include installation of meters to monitor and tax water use, and graduated pricing of water consumed (Wilkinson 1992). Ditch lining, drip irrigation and plastic strips greatly reduce water consumption as well as the need for weed control. The growing reluctance to build additional dams heightens the need for water conservation because supplies are limited. Screening of irrigation canals and pump intakes also saves salmon, but a recent investigation of intakes along the Columbia River indicates a large percentage are missing screens.

Nationwide, five of the six most popular soil conservation programs funded by the Agricultural Stabilization and Conservation Service (1992) in recent decades involved increased vegetative cover. Establishment of permanent vegetation or hay and pasture were predicted to have 0.1–0.1 times the erosion rates as row crops and small grains in southeast Washington, while various forms of conservation tillage reduced soil erosion by 13–95% depending on precipitation (SCS et al. 1984). Grassed waterways continue to be popular as a means to limit soil erosion and many enlightened farmers leave riparian buffers along surface waters. Dairy farms typically have wet weather controls for limiting run-off from manure heaps and cattle are fenced from stream access. Currie (1994) recommends greater use of existing incentives and disincentives. Among incentives, he includes greater tax abatements through the Washington Open Space Program, and higher federal subsidies for implementing best management plans. At the same time, Currie proposes that farms not implementing best management plans be subjected to reduced Open Space exemptions and higher fines for farm pollution. Watershed analysis, including risk assessment, management plans, and monitoring, is also proposed as a successful tool that could be adapted from forestland management to farmlands (Currie 1994).

Organic farming and integrated pest management are also growing in popularity on small agricultural operations. Demand for biocide-free crops continues to rise along with greater concerns with the dietary and ecological effects of pesticides. In addition, the cost of biocides and problems with neighboring land owners and ground water has restricted their use in some areas. These changes have mostly occurred on small farms that can more effectively implement integrated pest management.

### 8.5.2 Riparian Cropland Management

Just as it is now acknowledged that large woody debris is critical for maintaining channel complexity, it is also apparent that floods are necessary to prevent channels from incising, redistribute coarse sediments, build floodplains, introduce large wood, and propagate natural riparian vegetation. Traditional state and local floodplain zoning and easements allow some measure of control over the type of activities that can occur on floodplains. This approach may be quite successful if those allowed activities cause only minimal disruption of the floodplain ecosystem (e.g., natural parks, fishing access points). Typically, however, floodplain zoning restricts only those activities that incur extensive damage during floods (e.g., structures), but allows other activities that significantly change the characteristics of floodplain ecosystems (e.g., logging, grazing, farming), and hence the functions they perform (Kusler 1979).

An example of a more protective approach is the Banner Drainage and Levee District in Illinois, which is being restored to lakes and wetlands (NRC 1992). Another approach is the Willamette River greenway in western Oregon that includes 255 river miles and includes sloughs and side channels (NRC 1992) however, in many reaches there is no natural floodplain or only a narrow strip of native vegetation. Oregon also has a program to provide tax relief to landowners that maintain natural riparian zones. Water Quality 2000 (1992) stresses such efforts to protect wetlands rather than create them. Recognizing the ecological need for riparian and wetland areas to flood frequently, and the great cost to humans when they do, it is wiser to relocate activities from floodplains than to subsidize their continued settlement through such practices as channel maintenance, dams, levees, federal flood insurance, and "disaster" relief (NRC 1992). The benefits include saving money, allowing natural processes to reestablish habitat, and reducing hazards to human residents.

### 8.6 Mining Practices

In 1872, hard rock mining was encouraged by the federal government so that miners could easily obtain mining claims, produce metals, and settle the West. Now there is considerable reason to ban patenting (obtaining ownership to federal lands by simply paying \$2.50–\$5.00 per acre and investing \$100 per year), set strict reclamation requirements (including liability bonds), charge royalties on minerals removed, and prohibit mining in sensitive areas (Wilkinson 1992). He also recommends leasing, rather than selling, the land to miners and evaluating whether there is a net public benefit of the mining. In other words, hard rock mining would come under the same regulations as mining for energy minerals (e.g., the Surface Mining Control and Reclamation Act). Nelson et al. (1991) add that riparian and stream enhancement should also be part of the reclamation process.

### 8.6.1 Upland Mining Practices

To allow restoration of natural vegetation on mined lands it is critical that topsoil be set aside before mining begins (Meehan 1991). Toxic materials should be buried below the root zone and so that ground water does not pass through them to streams or the water table. When the area is returned to its natural contours (this can be problematic when the volume of spoils exceeds that of the original ore), the soil is replaced and revegetated with the original flora or acceptable substitutes. Revegetation may require seeding or introduction of vegetative propagules, as well as tilling, mulching and fertilization. It is critical that lands be stabilized as soon as possible to limit erosion. Mining-generated solids and seepage or runoff from mines should be kept from streams by proper planning and control structures such as erosion barriers and lined ponds.

### 8.6.2 Riparian Mining Practices

Mining in or near streams requires additional precautions to those for uplands (Nelson et al. 1991). Effluents may be treated with hydrated lime and sulfite then aerated to raise the pH and allow the metals to precipitate. Reverse osmosis and electrochemical precipitation are also effective following acid neutralization. If the channel form and substrate have been altered, a channel and riparian zone should be developed that allows normal ecological processes to occur. Nelson et al. (1991) stress that such channels are not static; they should resemble the preexisting channel in their bed, banks, riparian vegetation, and flows. Reestablishment of riparian woody plants may require transplanting.

## 8.7 Urban Practices

As Smart et al. (1985) indicated, urbanization fundamentally alters water quality in streams. Also, the high amounts of impervious surfaces and increased runoff make hydrographs much flashier. For example a 20% increase in impermeable surfaces can double runoff in a storm event (Washington State Department of Ecology 1992). Channel morphology and riparian vegetation are modified by transforming channels into drains and gallery forests and shrubs into lawns. Both further modify water quality. Protecting and restoring lakes and streams from the effects of urbanization, therefore, involve reducing the amount of urbanization, removing pollutants from the waste stream, and conserving natural channels (Wanielista 1978).

Increasingly land-use planning is used to restrict urban development from most sensitive areas, although past errors in settlements remain. Relatively successful examples of such planning include the California Coastal Commission; the Oregon Land Conservation and Development Commission, and the Tahoe Regional Planning Agency (Wilkinson 1992). The impact of urbanization also can be reduced by discouraging immigration and reproduction, by favoring higher density housing, by greater utilization of bicycles and mass transit, and by placing major transportation networks underground. In other words by designing cities for people instead of automobiles (Doxiadis 1971). Not only do these more rational city designs decrease the amount of impervious surfaces, they also decrease the amount of pollutants collected by and discharged from roads. As with other land uses, rigorous basin planning is incorporated in successful urban planning (Birch et al. 1992).

The major way of removing urban pollutants from the waste stream is by industrial and sewage treatment plants. Most U.S. cities now have secondary sewage treatment and many industries have tertiary or secondary. Secondary treatment, however, only transforms wastes into nutrients that are then discharged into rivers. As water purification and waste treatment costs rise and as rapidly increasing population increase the demand for limited water, more households and municipalities initiate water rationing and recycling. Household rationing has taken the form of xeriscaping in place of lawns, limiting lawn watering, smaller toilet tanks, low-flush toilets, composting toilets, rinse-only showers, and low-discharge shower nozzles (Wilkinson 1992). The major opportunity for domestic recycling is in the reuse of washwater on gardens and lawns and in toilets (Wagner 1971). Municipal rationing involves higher and progressive water and sewer rates, and lawn watering restrictions. Cities also recycle sewage water for irrigation or find it less expensive to purify tertiary-treated (3°) effluent for reuse than to withdraw lower quality water from rivers. Both approaches are in use elsewhere in this country (Wagner 1971) and in Europe. Industrial rationing and recycling have become more common as discharge permits became more restrictive.

Another substantial source of polluted waters is from stormwater runoff from lawns, roofs, parking lots, and streets. These sources can be reduced by decreasing their surface areas as discussed above, and by point source discharge permits on outfalls. More typical practices can be broken into nonstructural and structural approaches (Wanielista 1978). Nonstructural approaches include street cleaning (especially mechanical broom and vacuum sweeping), cleaning of catch basins, dust control, restrictions on dog-walkers and on the use of lawn chemicals, erosion control at construction sites, and the use of natural wetland systems (vegetated floodplains, marshes, ponds riparian zones). Structural management practices for reducing stormwater runoff problems include retention basins, constructed wetlands, land injection, rooftop and parking lot storage, and sediment traps. Illicit connections to storm drains are very common in cities (Washington State Department of Ecology 1992). Improper connections can be located by associating chemicals in the effluent with likely producers, dye studies, and TV inspections. Washington State Department of

Ecology (1992) provides a thorough set of guidelines for controlling erosion and sedimentation from construction sites, including matting and mulching open soil, erosion barriers, sediment traps, interceptors and drains on cut-fill slopes, and removal of sediment from roads. Monitoring and maintenance of control structures at such sites, especially during storms, is important.

## 8.8 Regional Planning and Management Efforts

Most management efforts for protecting and restoring salmonid populations have focused on the fish (harvest restrictions, fish passage, hatchery supplementation) or on aquatic habitats (water quality criteria, physical habitat structure, flow) as is proper. These efforts, however, must fall short without serious consideration of statewide or multistate planning.

FEMAT and PACFISH are examples of coordinated federal land use planning, and the states have begun similar efforts. For example, Oregon's statewide land use planning law is designed to protect forest, agricultural, and coastal lands from urbanization. Goal 5 of that law requires conservation and protection of lands needed for fish and wildlife habitats, water areas, wetlands, watersheds, and groundwater. Although it is statewide in scope, it is implemented and monitored at the county level with little statewide assessment of status or trends by the Department of Land Conservation and Development. Oregon's riparian set aside law, overseen by the Department of Fish and Wildlife, offers landowners tax deductions for protecting such areas. In response to Senate Bill 1125, Oregon Department of Forestry (1994) recently developed rules providing increased riparian protection for all fish bearing forest streams. Levels of protection vary with water body use, type, and size. A higher design and maintenance standard for new stream crossing structures was also promulgated. Aquatic diversity areas (Henjum et al. 1994), similar to FEMAT's key watersheds, have been mapped by the Oregon Chapter of the American Fisheries Society for the entire state. These areas, together with locations of unusually high salmonid production, have been incorporated into a framework for allocating salmon restoration funds (Bradbury et al. 1994). Protection and restoration of such areas throughout the region are necessary to preserve and expand salmonid populations which can support sustainable harvests.

The other states in the region have developed similar planning and management systems. For example, Washington Department of Natural Resources has rules for riparian protection that vary with water body use, type, and size. In addition, Washington promotes watershed analysis as a means of identifying sensitive and high risk areas within watersheds, or to minimize disturbance to aquatic ecosystems resulting from forest practices. California's Coastal Zone Management Act restricts development on sensitive coastal and estuarine areas. All four states in the region have water quality (temperature and dissolved oxygen) standards for the protection of salmonids. Generally they would be protective if monitored and enforced. In addition, the states have local zoning laws restricting building types and densities. The water quality and land use standards, however, differ from state to state and lack a statewide planning and monitoring design, let alone a regional one.

## 8.9 Individual and Societal Practices

Direct alteration of habitat by humans remains the single greatest threat to both terrestrial and aquatic biodiversity (Noss 1992). Most habitat alterations affecting salmonids relate to resource consumption of some sort—the use of water, electricity, wood and wood products, meat and wool, food and non-food crops, and mineral resources. Per capita consumption of resources in the United States is an order of magnitude greater than that in much of the world. Therefore, each of us can minimize our indirect effects on salmonids by markedly reducing consumption of all resources. There are a number of things that we can do individually and as a society to begin these changes and reduce our environmental impact.

### 8.9.1 Short-term Individual and Governmental Actions

The amount of water available for aquatic life in streams and lakes can be increased by reducing the amount diverted for domestic, industrial, and agricultural uses. Water conservation begins at home in how we shower, launder, flush toilets, landscape, irrigate, and use electricity. Those same functions of cleaning, cooling, waste disposal, irrigation, and power consumption offer opportunities for water conservation in industry and agriculture as well. Potential for increased efficiency in these water uses has been demonstrated by various voluntary and mandatory water conservation measures implemented during recent droughts in California, Oregon, and Washington. Key aspects in conserving water and electricity are accurate monitoring of uses, internalizing environmental costs associated with water use (e.g., dam impacts, hatchery operations, wastewater treatment), and progressive pricing so that greater use results in proportionately higher rates.

Wiser use and conservation of metals, particularly aluminum and heavy metals, would also reduce the demand for hydropower and hence the adverse effects of dams on anadromous salmonids. Over 40% of the aluminum used this country is produced in the Pacific Northwest, and fully 20% of the total energy sold by BPA is used by aluminum smelters and other energy-intensive industrial processes. Excessive packaging results in enormous waste of aluminum.

Every three months, Americans discard enough aluminum to rebuild the nation's entire commercial air fleet. Recycling of aluminum requires approximately 5% as much energy as refining the metal from bauxite. Because aluminum is an important component of many car parts, demand for aluminum can also be curtailed through reduced auto use and ownership and greater reliance on mass transit and other forms of transportation.

As with aluminum, wood products have considerable conservation potential. Worldwide, humans used over 30% more wood per person in 1991 than in 1950, mostly as fuel, but in the overdeveloped countries per capita wood consumption has been declining for most of the century (Durning 1994). Other trends are less encouraging. The U.S. produces 26% of the world's industrial wood with Russia a distant second. Average house size in the United States has increased from 100 square meters in 1949 to 185 square meters in 1993. This is 50–100% more space per person than West Europeans and Japanese, respectively. World paper consumption has increased twenty-fold since 1913, mostly in the highly industrialized nations. In 1960, the average family in the United States spent \$500 per year on packaging and the nation as a whole paid \$190 million for junk mail (Packard 1960). Currently, over half a million trees are used each week to print this nation's Sunday newspapers, much of which consist of advertisements that many readers discard and that promote consumption of unneeded products. Demand for wood products can be reduced by creating "paperless" offices, decreasing packaging, recycling paper, and developing alternative sources of fiber. Recent development of chipboard has increased fiber supply options to include wood waste, previously undesirable weed trees, and agricultural wastes such as straw and hemp. Agricultural waste fibers and weed trees also hold promise for paper manufacturing. Current methods to reduce waste and increase recycling and manufacturing efficiency could halve present U.S. wood consumption (Postel 1994).

Because livestock production and commercial fish harvest have substantial effects on salmonids, it is useful to examine ways to reduce consumption of beef and fish. Alternative protein sources, such as grains and legumes, would reduce the demand for salmon harvest and the need for range-fed livestock. Only 10% of the protein ingested by cattle is converted to tissue; consequently, beef is a relatively inefficient source of protein for humans compared with grains and legumes. The growing number of vegetarians (currently estimated as 4%) in the United States and the continuing popularity of wildlife hunting as sources of protein are both desirable trends, as is reduced meat consumption in general. Interestingly, a diet that reduces the risk of death and disease resulting from over-consumption of animal fats is also more beneficial to salmonid habitats.

Development of alternative energy sources could reduce dependence on hydropower and potentially allow for the removal of some hydroelectric dams. Energy conservation is a major source of new energy, but wind farms have considerable potential near the coast and in the Columbia Gorge. In addition, solar power and fuel cell units in individual buildings are likely to become more popular as their unit costs decrease and hydropower rates increase.

Perhaps one of the most effective ways in which our culture could conserve salmonids and their environments is to remove many direct and indirect subsidies that encourage resource use and consumption. Many of these subsidies were initially intended to facilitate the development of the west, long before the environment was a significant societal concern, and they continue at substantial economic and environmental expense. For example, postal customers subsidize both the delivery of junk mail they do not want and its disposal in landfills. Taxpayers indirectly pay for building in high risk areas (flood plains, faults, fire prone lands, ocean shores) through costs of fire suppression and disaster relief. Farmers are aided by taxpayers through drought and crop insurance and federal price supports. In the West, the Bureau of Reclamation may have spent as much as \$70 billion on water projects for agriculture since 1902 (DeBonis 1994). Automobile use is promoted rather than discouraged through subsidies to oil and gas industries that result in lower fuel prices—which encourages consumption—and through federal and state fuel and licensing taxes that foster road improvements, which in turn stimulate more driving, taxes, and roads. Major electric power consumers are subsidized with lower rates for greater consumption, and the Bonneville Power Administration is subsidized by U.S. taxpayers, allowing it to provide extremely inexpensive electric power to its customers. Taxpayer subsidies on public lands are estimated at \$700 million for below-cost timber sales, \$95 million for below-cost grazing fees and wildlife extirpation, and \$2.50/acre land for mining (DeBonis 1994). In summary, all of these subsidies provide disincentives for conserving energy and resources, and in many cases promote excessive use and consumption. Obviously, many of these subsidies provide values that benefit some or all segments of society. But it should be made clear to the public that these programs have associated environmental costs and directly or indirectly influence the ability of aquatic systems in the Pacific Northwest to produce salmonids.

In addition to the above changes, we need to reconsider fundamental policies in four areas of our culture: population, economics, ethics, and education. The first three are the cultural forces that are the root causes of environmental degradation and salmonid extirpation. Education is the method by which we begin to change our minds, and of which this document and others like it are a part. Although these forces are closely interconnected, we list and discuss them separately.