
An Ecosystem Approach to Salmonid Conservation

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6 Effects of Human Activities

Land-use practices--forestry, grazing, agriculture, urbanization, and mining--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 chemical contamination, 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 and channel stability, sediment transport, seasonal streamflow patterns, and flood dynamics. Disconnecting streams from their floodplains further alters hydrologic processes, nutrient dynamics, and vegetation characteristics.

Dams and water diversions affect salmonids by blocking or 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 that influence salmonids and their habitats include 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.

6.1 Forestry

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 (Pease 1993)¹. 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 provided only 46% of the softwood harvest (Jensen 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 many rivers and streams, had been cleared for three or more kilometers inland (Sedell and Luchessa 1982). 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 yielded 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 et al. 1987; Hicks et al. 1991a; Naiman et al. 1992) and State forest practice rules have been modified to reflect this knowledge. 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 re-establishment of forest vegetation, including road construction, timber harvest, site preparation, planting, and intermediate treatments. Understanding the effects of these practices on natural processes will foster improved management, providing greater protection to salmonid habitats.

6.1.1 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 designed to increase the vigor of desired species and inhibit growth of understory vegetation (e.g., burning or mechanical and chemical treatments). In addition, forest vegetation is indirectly affected by changes in site conditions following harvest. Removal of overstory vegetation can change local microclimate, soil moisture and stability, ground cover, and susceptibility to erosion, all of which influence the re-establishment of vegetation in the harvested area (Beschta et al. 1995). Soil compaction by ground-based equipment can reduce infiltration of water, thereby hindering the re-establishment of seedlings or the 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. 1995).

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 of perennial species, and finally by dominance of overstory species. Where disturbance is less severe, residual species may dominate the early successional stages. Beschta et al. (1995) 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 assemblages 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 re-establishment of coniferous species (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 and Sitka spruce, frequently regenerate on partially decomposed nurse logs that are elevated above the forest floor (reviewed in Harmon et al. 1986). Removal of downed trees from the riparian zone may affect re-establishment of these species.

6.1.2 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. 1995) 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 them from compaction, and soils with high biological activity (e.g., burrowing rodents, earthworms, insects, soil microbes) (Beschta et al. 1995).

6.1.3 Effects on Hydrology

Timber harvest and its 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 the stream. 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 ditches, effectively increasing drainage density within a watershed (Sidle et al. 1985). King and Tennyson (1984) observed altered hydrology when roads constituted 4% or more of catchment area (i.e., 4% was "roaded"). In snow-dominated systems, logging can influence the spatial distribution

of snow on the ground, as well as the energy transfer processes that affect the melting rate of snowpack (Chamberlin 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 two recent syntheses (Chamberlin et al. 1991; Beschta et al. 1995), and the material presented below is based primarily on these analyses.

Water Yield

In most instances, clear-cutting 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 centimeters to almost 65 cm (25 inches) (Beschta et al. 1995). 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 slightly after logging (Harr 1982). Apparently, the decrease in evapotranspiration losses was offset by a decrease in the amount of fog-drip that normally occurred in the forested watershed.

In forests east of the Cascade Range, increases in water yield after logging 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 et al. 1987). The authors suggested that higher wind speeds after clear-cutting increase evaporation from snowpack, offsetting reductions in transpiration. Elsewhere in eastern Oregon and Washington, forested watersheds are likely to experience minor changes in total water yield. Beschta et al. (1995) speculated that in areas with low precipitation (< 30-51 cm [$< 15-20$ 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 still less than those observed in westside systems (Ziemer 1986; Beschta et al. 1995). Small increases in yield from snow-dominated systems in British Columbia and Colorado have also been reported (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. 1995). 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 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 re-establishing desired trees) can slow 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.

Timing of Runoff

In rain-dominated systems, the largest increases in water yield because of 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. 1995). 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 clear-cuts are again greater, and increases in water yield may be higher than in winter (Beschta et al. 1995).

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 the 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.

Peak Flows

A recent review (Beschta et al. 1995) of effects of timber harvest on peak flows in systems in the Pacific Northwest indicates a high degree of variability among sites. 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). 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 soil compaction and roads.

In transient-snow systems of western Oregon responses of peak flows are similarly variable (Table 6-2). 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) re-analyzed 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 6-2) 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.

Table 6-1. Effects of timber harvesting on peakflows in coastal areas of the Pacific Northwest. Modified from Beschta et al. (1995). Reproduced with permission from the principal author.

Watershed/ Location	Size (acres)	Geology/Soils	Harvest method and percent	Peakflow effect	Reference
OREGON					
Needle Branch (Alsea Watershed)	173	sandstone shallow GL- GCL*	cc† 82%	fall: +50% winter: +19%	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek-Main (Alsea watershed)	748	sandstone shallow GL- GCL	cc 26%	all: ns‡	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek-2 (Alsea watershed)	138	sandstone shallow GL- GCL	cc 90%	fall: +51% winter: +20%	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek-3 (Alsea watershed)	99	sandstone shallow GL- GCL	cc 65%	fall: +50% winter: +30%	Hsieh (1970) Harris (1977) Harr et al. (1975)
BRITISH COLUMBIA					
West Coast-1	57	quartz GSL§	cc 71%	all: -22%	Cheng et al. (1975)

Carnation Creek B	2,470	volcanic coarse soil	cc 41%	all: ns	Hetherington (1987)
Carnation Creek	30	volcanic coarse soil	cc 90%	all: +20%	Hetherington (1987)
Jamieson Creek	739	na [†]	cc 19.2%	winter: +13.5%	Golding (1987)
CALIFORNIA					
South Fork Caspar Creek	1,047	sandstone coarse soil	sc [#] 60%	small: +107% large: ns	Ziemer (1981)

* gravelly loam-gravelly clay loam (GL-GCL)

† clear-cut (cc)

‡ not significant (ns)

§ gravelly sandy loam (GSL)

¶ not available (na)

shelterwood (sc)

Table 6-2. Effects of timber harvesting on peakflows in interior areas of the Pacific Northwest. Modified from Beschta et al. (1995). Reproduced with permission of the principal author.

Watershed/ Location	Size (acres)	Geology/ soils	Harvest method and percent	Peakflow effects	Reference
RAIN-ON-SNOW					
Watershed--1* (OR)	235	basalt/GL [†]	cc [‡] 100%	fall: +200% winter: ns [§]	Rothacher (1973)
Watershed--3 (OR)	249	basalt/GL	cc 25%	mean: +10%	Rothacher (1973)
Watershed--6 (OR)	32	basalt/GL	cc 100%	all: ns	Harr et al. (1982)
Watershed--7 (OR)	52	basalt/GL	sc [¶] 60%	all: ns	Harr et al. (1982)
Watershed--10 (OR)	25	basalt/GL	cc 100%	ROS [#] : -36% rain: ns	Harr and McCorison (1979)

Coyote Creek--1** (OR)	170	basalt/ GL-GCL††	sc 50%	mean: +30% large: +48%	Harr et al. (1979)
Coyote Creek--2 (OR)	168	basalt/ GL-GCL	cc 30%	all: ns	Harr et al. (1979)
Coyote Creek--3 (OR)	121	basalt/ GL-GCL	cc 100%	mean: +44% large: +35%	Harr et al. (1979)

SNOWMELT

High Ridge--1‡ (OR)	73	fractured basalt/SL§§	cc 43%	all: ns	Fowler et al. (1987)
High Ridge--2 (OR)	60	fractured basalt/SL	sc 50%	all: ns	Fowler et al. (1987)
High Ridge--4 (OR)	292	fractured basalt/SL	cc 22%	all: ns	Fowler et al. (1987)
Camp Creek (BC)¶¶	8,373	granite/ coarse soils	cc 30%	annual: -9 to +35%	Cheng (1989)

* Watersheds in H. J. Andrews Experimental Forest, western Cascades region

† gravelly loam (GL)

‡ clear-cut (cc)

§ not significant (ns)

¶ shelterwood (sc)

rain-on-snow (ROS)

** Coyote Creek, South Umpqua Experimental Forest, Western Cascades Region

†† gravelly loam--gravelly clay loam (GL-GCL)

‡‡ High Ridge Evaluation Area, Blue Mountains Region

§§ silt loam (SL)

¶¶ Camp Creek, southern British Columbia, Canada

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 streamflows that persisted for six years after logging of a basin in the interior of British Columbia. However, Troendle (1983) found no increase in summer low flows after 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. (1991b) 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 prelogging years. The authors attributed reductions in streamflow to the replacement of coniferous vegetation with more water-consumptive hardwood species. Thus, the long-term effects of logging on streamflows likely depend on vegetation composition before and after harvest.

6.1.4 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 on 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. 1995).

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 begin 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. 1995). Splash erosion mobilizes sediment on exposed road surfaces, and runoff from compacted surfaces may also facilitate sediment transport. Water diverted into ditches along roads gathers energy and can cause significant erosion at the outlets of cross-drain culverts (Beschta et al. 1995). 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 from 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 which may increase splash erosion and channelized runoff. Ground-based vehicles moving logs from felled trees and skidding logs to landing sites compact and scarify the soil. Compaction of the decomposing root systems reduces 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. 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. Riparian buffer strips and buffer strips below roads can trap sediments, significantly reducing the delivery rate (Swanston 1991).

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 km²) catchment with a relief ratio of 0.10. Presumably, these sediments (2-151 mg·L⁻¹ 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 in-channel structures appear important in determining whether sediments are deposited locally or transported downstream.

Table 6-3. Summary of summer temperature changes associated with management activities on forested 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)	Clear-cut and natural openings	Temperature per 100 m of channel	+0.1 -1.1°C per 100 m; Avg = 0.7°C per 100 m	Meehan (1970)
British Columbia (Vancouver Island)	Logged (Tributary H)	Average Jun-Aug diurnal temperature range	+0.5 -1.8°C over pretreatment levels	Holtby and Newcombe (1982)

	Logged and burned (Tributary J)	Average Jun-Aug diurnal temperature range	+0.7 -3.2°C over pretreatment levels	Holtby and Newcombe (1982)
Oregon (Cascades)	Clear-cut	Average Jun-Aug maximum	+4.4 -6.7°C	Levno and Rothacher (1967)
	Clear-cut and burning	Average Jun-Aug maximum	+6.7 -7.8°C	Levno and Rothacher (1967)
Oregon (Coast Range)	Clear-cut	Average Jul-Sep maximum	+2.8 -7.8°C	Brown and Krygier (1967)
	Clear-cut and burning	Average Jul-Aug maximum	+9 -10°C	Brown and Krygier (1970)
Oregon (Cascades)	Mixed clear-cut and forested reaches	Temperature per 100 m of channel	+0 -0.7°C per 100 m	Brown et al. (1971)
	Tractor stripped area	Temperature per 100 m of channel	+15.8°C per 100 m	Brown et al. (1971)

6.1.5 Effects on Thermal Energy Transfer and Stream Temperature

Logging most directly affects energy transfer by reducing shade provided by riparian vegetation, which increases the amount of direct solar radiation reaching the stream surface. The increase in energy reaching the stream depends 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 (Table 6-3).

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 3.7), 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. 1995). For coniferous forests of the Coast Range and western Cascades, increases in average summer maximum temperatures because of clear-cutting have ranged from about 3 to 8°C (Table 6-3); (Beschta et al. 1987). Increases up to 10°C have been observed when clear-cutting 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 (1988) 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 1982). 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 clear-cut.

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. 1995). 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- and 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 prelogging 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 re-enter 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. 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. 1995). In a study in the western Cascades of Oregon, Beschta and Taylor (1988) found that stream temperatures increased 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.

6.1.6 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. 1995). 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). 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 ground (Beschta et al. 1995), 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 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). 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). In another study, 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 from harvest. Losses because of volatilization resulting from fire can be much more significant (Beschta et al. 1995).

[Figure 6-1. Temporal patterns of physical factors \(A\) and riparian vegetation \(B\) after timber harvest \(time is expressed as years on a logarithmic scale.\) From Gregory et al. \(1987\). Reproduced with permission from the publisher.](#)

6.1.7 Effects of Forest Chemicals

Fertilizers, herbicides, and insecticides are commonly used in forest environments to prepare sites for planting, to release and stimulate growth of conifers, and to control diseases and pests. In addition, fire retardants are used to halt the spread of wildfire on forest lands. All of these chemicals can affect salmonids through several direct and indirect pathways. Fertilizers, pesticides, and fire retardants that reach surface waters can be toxic to salmonids or may alter primary and secondary production, influencing the amount and type of food available to salmonids. Fertilizers and pesticides indirectly affect salmonids by influencing the rate of recovery of upland and riparian vegetation following logging, which in turn affects hydrologic processes, delivery of sediment and organic debris, heat transfer, nutrient cycling, and soil biota.

Contamination of surface waters by forest chemicals and the resultant risk of toxic effects on aquatic organisms depend on several factors, including the form and application rate of the chemical (and associated carriers), the application method (i.e., aerial versus ground spraying), soil type, weather conditions during and after application, and the retention of no-spray buffers in the riparian zone. The specific formulation determines the solubility of the chemical in water and its tendency to adsorb onto organic or inorganic matter within the soil, both of which determine the substance's mobility within the soil column. The persistence of forest chemicals in the environment varies. Some chemicals are highly volatile or are rapidly broken down through microbial activity or photodecomposition, whereas others persist for months or years. The volatility and adsorption of chemicals to soil particles are affected by temperature and soil moisture. If chemical application is followed by precipitation events, the likelihood that chemicals will reach surface waters increases. Contamination of surface waters also occurs when chemicals that have been applied to ephemeral channels are later mobilized during rainstorms. Aquatic organisms may be exposed to forest chemicals through direct contact with contaminated water, sediments, or food. No-spray buffers around streams (including ephemeral streams) and riparian areas substantially reduce the risk of contamination (Norris et al. 1991); however, toxic levels of chemicals may reach streams from storm runoff and wind drift even when best management practices are employed (e.g., Rashin and Graber 1993). Indirect effects of chemicals on watershed processes are largely unavoidable because change in vegetation is usually the desired outcome of such applications.

Although both direct and indirect effects of forest chemicals on salmonids may be significant, it is important to note that less than 1% of total pesticides applied in the United States are used in forestry (Norris et al. 1991); thus, contamination from forest practices is likely minor compared to contamination from agricultural practices.

Fertilizers

Fertilizers are used in forest settings to replace nutrients lost during and after timber harvest and to accelerate growth of conifers. Application of fertilizers to a catchment typically results in increased concentrations of nutrients, particularly nitrogen, in streams. A recent review of effects of forest fertilization on water quality and aquatic biota indicates that urea application typically leads to elevated levels of urea-N, ammonia-N, and nitrate-N in surface waters (Bisson et al. 1992a). Urea-N usually dissipates within a few days, whereas ammonia-N may be elevated for months and nitrate-N for a year or more. The concentration of nitrogen within the stream depends on a number of factors, including the percentage of the watershed fertilized, the application rate, the drainage density (stream km relative to total watershed area), the width of unfertilized buffers along streams, and whether or not precipitation occurs following application. Although drinking water and aquatic standards are typically not exceeded with most applications, the elevation of nitrogen has the potential to promote growth of periphyton, which in turn may influence production of invertebrates and fishes. However, Bisson et al. (1992a) concluded that enhanced fish production because of forest fertilization has not been demonstrated in the Pacific Northwest. An indirect benefit of fertilizer applications is more rapid growth of vegetation within the catchment, which in turn accelerates the recovery of natural hydrologic regimes and sediment delivery rates.

Herbicides

In forest plantations, a wide variety of herbicides are used to control the invading hardwoods, herbaceous plants, and grasses to enhance the suitability of the area for re-establishment of desired tree species. Ten herbicides commonly used in forestry are 2,4-D, picloram, hexazinone, atrazine, imazapyr, triclopyr, forsamine, glyphosate, dalapon, and dinoseb. The behavior and toxicity of these substances is reviewed in detail in Norris et al. (1991) and Beschta et al. (1995), from whom much of the information below was excerpted.

The risk of toxicological effects of herbicides on salmonids is greatest when herbicides are directly applied to surface waters or reach surface waters by wind drift. Whether herbicides applied to upland forests will reach surface waters depends on their volatility, mobility in the soil, and persistence in the environment. Of the herbicides commonly used in forest applications, hexazinone, atrazine, imazapyr, and triclopyr are generally the most persistent, with soil half-lives of 2-6 months or more, depending on soil type. The half-lives of most other forest herbicides are generally from 2-5 weeks. Although there is substantial literature on the toxicity of various herbicides to salmonids, most of the available information

comes from laboratory studies rather than the field. These laboratory studies focus on acute lethal doses (Reid 1993). Sublethal effects of herbicides on salmonids include reduced growth, decreased reproductive success, altered behavior, and reduced resistance to stress (reviewed in Beschta et al. 1995). Sublethal exposures of picloram were found to increase mortality by 70% in yearling coho exposed to seawater (Lorz et al. 1979). Information on effects of herbicides in aquatic invertebrates is also scarce. Hartman and Scrivener (1990) reported a 42% reduction in the density of aquatic macroinvertebrates for 1.5 years following application of Roundup. These reductions were attributed to herbicide-induced irritation and drift of invertebrates coupled with high flows and decreases in substrate stability.

Herbicides used to release conifers from competing vegetation can accelerate the long-term recovery of upland and riparian areas. Over the short term, the elimination of deciduous vegetation can affect streams in several ways, both positive and negative. Herbicide applications in upland areas slow the recovery of vegetation, prolonging disruption to hydrologic and sediment delivery processes. Within the riparian zone, removal of deciduous vegetation increases solar radiation reaching streams, which stimulates algal production, potentially increasing the food base for invertebrates and fish. 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 recruitment of large coniferous wood over longer periods. Depending on whether temperature, spawning sites, cover, or food is limiting, these changes may initially hinder or aid salmonid production.

Insecticides

Insecticides are used both to prevent insect infestations and to control insect outbreaks once they have occurred. In general, insecticides are more toxic to fish and other aquatic biota than herbicides; however, they usually are applied at lower rates (Beschta et al. 1995). The greatest effect of insecticide on fish probably arises from effects on terrestrial and aquatic insects that form the salmonids' food base. Forest insecticides cause direct mortality to these insects or may stimulate catastrophic drift of aquatic invertebrates out of the affected stream reach. In addition, benthic algal communities in streams are frequently controlled by grazing invertebrates; consequently, the loss of invertebrates may release primary production, causing fundamental shifts in the trophic structure of streams. Norris et al. (1991) concluded that insecticides generally have shorter term effects on stream ecosystems than herbicides but that the effects may be more dramatic. Populations of invertebrates may take months to recover following insecticide applications, and full recovery of the invertebrate assemblage may take several years (Norris et al. 1991). Because salmonids in some forest streams may be food-limited, reductions in aquatic insect biomass and altered assemblage composition may result in reduced growth and numbers of salmonids. For example, Kingsbury (1983 in Norris et al. 1991) reported a decline in the growth rate of Atlantic salmon parr immediately following treatment with an insecticide; however, by the end of summer, fish in treated and untreated reaches were of similar size. Direct toxic effects may occur if salmonids consume drifting, pesticide-laden, aquatic organisms or terrestrial insects that fall into streams. Other indirect effects of insecticides on salmonid habitats are not well documented; however, protection of trees from insect pests may reduce the number of trees that die and fall into streams,

thereby reducing recruitment of large woody debris.

Fire Retardants

The use of chemical fire retardants plays an important role in the suppression of wildfires in the west. Historically, a variety of chemicals have been used to suppress fires; however, ammonium-based retardants account for nearly all chemical retardants used today (Norris and Webb 1989). Although documentation of adverse effects of fire retardants on salmonids is scarce, quantities of retardant dropped during fires may be significant, and cases of fish mortality caused by retardants have been reported. For example, approximately 5.3 million liters of retardant were used to fight the Yellowstone fire of 1988, and at least two small fish kills (approximately 100 fish each) were reported (Schullery 1989). Fire retardant killed approximately 700 adult salmon, as well as a large number of juveniles, in an Alaskan stream (Hakala et al. 1971 in Norris and Webb 1989). Potential indirect effects of fire retardants on salmonids include mortality of invertebrates and eutrophication of downstream reaches (from phosphates). The extent of effects of retardants on aquatic ecosystems is influenced by application procedures (quantity applied, line of flight of aircraft relative to the stream), site characteristics (stream width-depth ratio, degree of canopy cover), and streamflow.

6.1.8 Effects on Physical Habitat Structure

Timber management activities have resulted in substantial modification of the physical characteristics of stream habitats throughout forested regions of the Pacific Northwest. Many of these changes have resulted from decreased recruitment of large woody debris (LWD) from the riparian zone and intentional removal of LWD from stream channels (Bisson et al. 1987; Maser et al. 1988; Hicks et al. 1991a). Removal of vegetation from the riparian zone has altered sources, mechanisms for delivery, distribution patterns, and stability of wood in stream channels (Bisson et al. 1987). Hardwoods have replaced conifers in many riparian areas following logging; woody debris produced by deciduous vegetation tends to be smaller, more mobile, and shorter-lived than that derived from conifers and, consequently, does not function as well in retaining sediment. The reduced supply of large woody debris decreases channel stability and eventually leads to loss of instream cover and pool habitat available for fish (Bisson et al. 1987). During the winter, salmonids have been reported to abandon reaches that are devoid of large wood (Tschapinski and Hartman 1983). 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, diminishing the heterogeneity that allows fishes to segregate among microhabitats. Loss of riparian vegetation also leaves banks unprotected, increasing bank erosion and reducing the formation of undercut banks that salmonids frequently use for cover. Hicks et al. (1991a) provide a good summary of short and long-term physical effects of forest practices on stream habitats (Table 6-4).

Excessive sedimentation resulting from logging and associated roads has also played a substantial role in

altering salmonids habitats. Several studies have recorded increased levels of fine sediment in spawning gravels following logging (Cederholm and Reid 1987; Hall et al. 1987; Hartman et al. 1987); reductions in production of salmonids have been attributed in part to sedimentation of redds in each of these instances. Fine sediments generated by logging and roading activities also can fill substrate interstices, decreasing the availability of cover for juvenile salmonids and altering primary production and invertebrate abundance (Cederholm and Reid 1987; Hicks et al. 1991a). Inputs of coarse sediments can fill pool habitats, resulting in channel shallowing and widening, and decreasing channel stability.

Culverts on logging roads 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 resident salmonids. At culverts, excessive flow velocities, insufficient water, excessive culvert heights, and the absence of pools all can impede migration (Evans and Johnson 1980). Culverts also fail frequently when inappropriately designed and installed, resulting in mass wasting of road crossings. Road construction along streams, particularly where revetments are required, can constrain streamflow, thereby facilitating scouring of the channel bed. Dose and Roper (1994) found that channels widened as road density in catchments increased.

The cumulative effect of forest practices has been a reduction in the complexity of stream macro- and microhabitats. FEMAT (1993) 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. Similar declines in pool frequency in river basins of eastern Oregon and Washington are reported in McIntosh et al. (1994a, 1994b). Reductions in pool habitat arise from the combined effects of loss of large woody debris, increased sediment inputs (which fill pools), and hydrologic changes that accompany land use. Although logging-related activities are not the only cause of these changes, they certainly play an important role. Effects of logging on physical habitat structure 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.

6.1.9 Effects on Stream Biota

A substantial volume of literature documents the effects of logging on aquatic biota (see reviews in Gregory et al. 1987; Hicks et al. 1991a; and Beschta et al. 1995). The response of aquatic communities to logging depends on a variety of factors, thus studies have sometimes produced 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; thus, the 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, making ecological responses complex and difficult to predict;

5) dynamic, sometimes catastrophic natural events (e.g., large floods, changing oceanic conditions) create variable "baseline" conditions making it difficult to quantify additional variability caused by habitat alterations (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; Murphy et al. 1981; Shortreed and Stockner 1983; Murphy et al. 1986). 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, are expected to become more abundant (but see Hawkins et al. 1982), while those species that feed on detritus (i.e., shredders, filterers, and collector-gatherers, sensu Merritt and Cummins 1984) typically decline in numbers (Hawkins et al. 1982; Beschta et al. 1995). The abundance of invertebrate predators has been shown to increase in response to increased secondary production in streams in the Oregon 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. It is important to note that these responses are likely typical only of streams where primary production is light-limited (i.e., small streams in dense forests); the response of invertebrates and juvenile salmonids to canopy removal in more open, eastside systems may differ.

Table 6-4. Influences of timber harvest on physical characteristics of stream environments, potential changes in habitat quality, and resultant consequences for salmonid growth and survival. From Hicks et al. (1991a). 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
		Increased fine sediment in spawning gravels and food production areas	Reduced spawning success; reduced food supply
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
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

The 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. Silt deposited from mass failures and surface erosion can affect invertebrate production as gravel interstices are filled by silt, and algae are buried or abraded (Beschta et al. 1995). In these instances, invertebrate assemblages 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, including large woody debris, 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. 1995). Gregory et al. (1987) suggest that an overall pattern of increased production of a few taxa accompanied by a reduction in biodiversity may be common to all invertebrate trophic levels in streams that have been simplified through forest practices.

All of the logging-induced changes in physical and biological characteristics discussed in preceding sections interact to influence the composition and diversity of fish populations and communities; however, few studies in the Pacific Northwest have been designed to address specific relationships between changes in habitat attributes and structure of fish assemblages (Bisson et al. 1992). In addition, changes in one habitat attribute that benefit salmonid productivity (e.g., increased light and primary production) may be compensated by other adverse effects (e.g., loss of rearing habitat, sedimentation, excessive temperatures, change in timing of life-history events), which may not be manifest until subsequent life stages. Similarly, most studies of effects of logging in the Pacific Northwest have been conducted in relatively wet, forested ecosystems of the Coast Range and Western Cascades. In eastside systems, canopy removal, loss of physical structure, and increases in sediment loading may have greater ecological impacts on salmonids because different factors may limit production (e.g., summer temperatures, pool habitats).

Studies in the Cascades and Coast Ranges of Oregon and northern California indicated higher densities and biomass of salmonids in recently clear-cut reaches compared to shaded old-growth and second-growth reaches (Murphy et al. 1981; Hawkins et al. 1983); however, no analysis of age structure was provided. In western Washington, Bisson and Sedell (1984) found that total salmonid biomass was greater in streams that had been logged and cleaned (i.e., large woody debris removed), but that populations were dominated by underyearling trout, with proportionately fewer age 1 and older trout. These differences were attributed in part to a higher frequency of riffles (favored by underyearlings) and a lower frequency of pools (favored by older trout) in clear-cut reaches. Subsequent studies suggest that juvenile coho salmon and older trout, which prefer pool habitats, are adversely affected by conversion of pools to riffles (Bisson et al. 1992b). In a long-term study of the effects of logging in an Oregon watershed (Alsea Watershed Study), numbers of fry migrating from a clear-cut section decreased by more than 50%, while prelogging and postlogging numbers from a patch-cut watershed and an 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. In southeast Alaska, summer abundance of coho salmon increased following canopy removal; however, in winter parr densities were lower in clear-cut reaches than old growth reaches, apparently because of a lack of large woody debris and undercut banks (Heifetz et al. 1986).

Other general effects of logging on fishes include decreased growth efficiency, reduced survival to emergence, increased susceptibility to disease, increased vulnerability to predation, lower winter survival, blockages to migration from poorly designed culverts, changes in the age-structure of fish populations, reduced development time of embryos, and altered timing of life-history events (Hicks et al. 1991a). Scrivener and Brownlee (1989) reported reductions in survival to emergence of coho and chum salmon of approximately 45% as a result of increased fine sediment in spawning gravels following logging of the Carnation Creek watershed. In the same watershed, Holtby et al. (1989) reported that increases in water temperatures following logging resulted in earlier outmigration of coho and chum salmon fry and earlier emigration by coho salmon smolts to the ocean. The change in migration timing of coho salmon smolts is believed to have reduced ocean survival. Furthermore, 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 increase the vulnerability of specific year classes to environmental fluctuations in both the freshwater and marine environments. The Alsea Watershed study documented an increase in the number of early (November-January) coho migrants (Hicks et al. 1991a); whether the change was due to temperature-induced acceleration of growth or loss of rearing habitats remains uncertain. Nevertheless, these results suggest that small increases in temperature (1-2C) can result in significant shifts in the timing of important life history events.

Finally, roads constructed for timber harvest may indirectly affect salmonids by increasing public access

to previously remote locations. Angling pressure generally decreases with increasing distance from access roads; consequently, increased mortality from angling may accompany habitat degradation.

6.2 Grazing

Livestock grazing represents the second most dominant land use in the Pacific Northwest, following timber production. In Oregon, Washington, and Idaho combined, over 22.9 million hectares (56.5 million acres) of grassland and desert shrubland, approximately 3.2 million hectares (8 million acres) of nonfederal forest land, and an undetermined amount of Federal forest land are grazed by cattle and sheep (Pease 1985). This acreage represents approximately 41% of the total land base. Rangelands are fairly evenly divided between Federal and nonfederal lands; Federal rangelands total approximately 12.3 million hectares (30.5 million acres) (excluding Federal forest lands that are grazed), and nonfederal rangelands total 13.8 million hectares (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 (Pease 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 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), a number 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 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 1994). 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 1994). 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 1994). However, recent reports have indicated that most riparian areas remain in fair-to-poor condition (Chaney et al. 1990; GAO 1991). Thus, while upland conditions appear to be improving, riparian areas continue to be degraded. In 1991, BLM began a program to improve riparian management, with a goal of restoring 75% or more of riparian areas to properly functioning condition by 1997 (Barrett et al. 1993).

Despite the generally poor condition of most riparian areas, the potential for restoring those areas

damaged by grazing is arguably greater 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--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 too severe.

6.2.1 Effects on Vegetation

Heavy grazing around the turn of the century had significant and widespread effects, many of which persist today, on upland and riparian vegetation. Rangelands have experienced decreases in the percentage of ground covered by vegetation and associated organic litter (Heady and Child 1994). Species composition of plants in upland areas have shifted from perennial grasses toward non-native annual grasses and weedy species (Heady and Child 1994). East of the Cascade Range, upland sites that once supported plant associations of Idaho fescue are now devoid of native bunchgrasses, which have been replaced with tarweed, gumweed, and other noxious plants (Johnson et al. 1994). In riparian areas, 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 (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 (Johnson 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 6-5).

Table 6-5. Deleterious effects of livestock grazing on plant communities in western North America. From Fleischner (1994). Reprinted with permission of Blackwell Scientific Publications, Inc.

Habitat	Location	Effect	Authority
Sonoran desert scrub	Arizona	Perennial grasses and <i>Krameria</i> (palatable shrub) showed dramatic density decreases with grazing	Blydenstein et al. (1957)

Mojave desert scrub	California	60% reduction in above-ground biomass of annuals, 16%-29% decrease in cover of perennial shrubs with grazing	Webb and Stielstra (1979)
Sagebrush desert	Idaho	Grazed site had one-third species richness of ungrazed site	Reynolds and 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 and Evans (1945)
Riparian	Oregon	Species richness increased from 17 to 45 species 9 years after removal of livestock	Winegar (1977)
Riparian	Arizona	Herbaceous cover of grazed plot less than half that of ungrazed plot	Szaro and Pase (1983)
Riparian	Colorado	Shrub canopy coverage increased 5.5 times, willow canopy coverage 8 times after removal of livestock	Schulz and Leininger (1990)

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 1994). 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 1994). Removal of woody shrubs through chemical application or by mechanical means is also a common practice in range management. 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. 1989a).

Cattle and sheep affect vegetation primarily through browsing and trampling. Grazing animals are selective in what they eat; consequently, 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 1994). Selection of specific plant species may allow other taxa to dominate (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. In a dairy pasture, MacDiarmid and Watkin (1971) found that 75% of grasses and legumes under manure piles were killed.

Livestock grazing also influences vegetation through modification of soil characteristics. Hooves compact soils that are damp or porous, which inhibits the germination of seeds and reduces root growth (Heady and Child 1994). Changes in infiltration capacity associated with trampling may lead to more rapid surface runoff, lowering moisture content of soil and hence the ability of plants to germinate or persist (Heady and Child 1994); however, 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 1994). 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" plays important roles in hydrology and nutrient cycling (see Sections 6.2.3 and 6.2.6) and is believed to provide favorable conditions for the germination of vascular plants (Fleischner 1994). 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 indirectly affect plant species composition by aiding the dispersion and establishment of non-native species; seeds may be carried on the fur or in the dung of livestock (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 1994). 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 (1994) 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 and 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 such as 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.

6.2.2 Effects on Soils

Rangeland soils are frequently compacted by livestock. The degree of soil compaction depends on soil characteristics, including texture, structure, porosity, and moisture content (Platts 1991; Heady and Child 1994). 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 6.2.4). The available studies indicate that compaction generally 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 1994).

Trampling by livestock may also displace or break up surface soils. In instances where surface soils have become impervious to water, light trampling may increase the soil's 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.

6.2.3 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 1994). 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 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 vegetative 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 facilitates sediment deposition and bank building, increasing 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, well vegetated 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 with resultant lowering of the water table.

6.2.4 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 1994). Devegetation and exposure of soil by grazing facilitates detachment of soil particles during rainstorms, thereby increasing

overland sediment transport. Rills and gullies often form in areas denuded by livestock trails or grazing, resulting in increased channelized erosion (Kauffman et al. 1983). 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. 1993; Peacock 1994).

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

6.2.5 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), 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 summer 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 in watersheds of comparable size found temperature differences of 11°C between shaded and unshaded streams (Li et al. 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 from 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, 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.

6.2.6 Effects on Nutrients and Other Solutes

Livestock activities can directly affect nutrient dynamics through several mechanisms. The removal of riparian vegetation by grazing reduces the supply of nutrients provided by organic leaf litter. Livestock also 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 1994). 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 reduce the 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 3.9.6). As stream channels incise and streams are separated from their floodplains, soil moisture is reduced, which in turn alters the quantity and form of nutrients and their availability to aquatic communities. In the anaerobic environments of saturated soils, microbial activity transforms nitrate nitrogen (NO_3) into gaseous nitrous oxide (N_2O) and elemental nitrogen (N_2) that are liberated to the atmosphere (Figure 6-2) (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 facilitate algal growth, increase turbidity, and in some cases cause oxygen depletion because of increased biochemical oxygen demand.

The form of other elements including manganese, iron, sulfur, and carbon also depends 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. Thus, changes in hydrologic condition caused by downcutting can modify the form of elements available to plants, thereby altering competitive interactions among plants and changing riparian plant communities.

6.2.7 Effects of Vegetation Management

Fertilizers, herbicides, mechanical treatments, and prescribed fire are commonly used in rangeland management to alter vegetation in favor of desired species. In principle, the potential effects of these activities on salmonids and their habitats are no different than similar activities in forested environments; however, because the physical and biological processes that regulate the delivery of water, sediments, and chemicals to streams differ on forests and rangelands, so may the response of aquatic ecosystems.

Fertilizers are used on rangelands to increase forage production, improve nutritive quality of forage, and enhance seedling establishment, although the high costs and varied results have led to a decline in fertilizing rangeland in the past 20 years (Heady and Child 1994). Fertilizers that reach streams through direct application or runoff can adversely affect water quality. Nutrient enrichment (especially nitrogen) promotes algal growth, which in turn can lead to oxygen depletion as algae die and decompose. Conversely, fertilizer applied to rangelands may reduce sedimentation, hydrologic, and temperature effects by stimulating recovery of vegetation, including woody riparian shrubs. Herbicides are typically used to target unpalatable or noxious weeds that compete with desired forage species. Many of the herbicides commonly used in forestry (e.g., 2,4-D, picloram, glyphosate, tricopyr) are used in range management as well, although other highly selective herbicides may be used to control particular weeds common to rangelands, including unpalatable woody shrubs. Direct toxic effects on aquatic biota may occur where herbicides are applied directly to stream channels; however, risks of contamination can be minimized if adequate no-spray buffers are maintained (Heady and Child 1994). Herbicide applications to upland areas may decrease total groundcover, increasing the potential for surface erosion. In the riparian zone, use of herbicides may reduce production of deciduous trees and shrubs, opening streams to greater direct solar radiation, which in turn leads to elevated stream temperatures and increased algal production. These conditions can lead to insufficient nighttime dissolved oxygen concentrations and afternoon gas supersaturation. The loss of riparian vegetation also decreases the amount of organic litter and large wood delivered to streams. Furthermore, without the root structure of woody vegetation, banks are prone to collapse, increasing sedimentation and reducing the amount of cover for fish.

[Figure 6-2. Nitrogen cycling pathways in undisturbed \(left\) and disturbed \(right\) riparian zones of northeastern Oregon, as indicated by redox potential \(Eh\). From Green and Kauffman \(1989\). Reproduced with permission from the principal author.](#)

The influence of mechanical treatment and prescribed fire on aquatic ecosystems in rangelands depends on the type and intensity of disturbance. The use of tractors with dozer blades, brush rakes, cables, or rolling cutters for vegetation removal all can lead to compaction of rangeland soils (Heady and Child 1994) and thereby increase surface runoff and erosion. Disking of soils may break up relatively impervious soils, allowing greater infiltration of water; however, unless the area is rapidly revegetated, raindrop splash on exposed soils is likely to facilitate surface erosion and increase sediment delivery to streams. Disking and dozer use also rearrange soil layers, mixing topsoil with woody debris, which may affect re-establishment of vegetation. Positive effects of mechanical vegetation removal are also possible. Removal of vegetation with high evapotranspiration rates (e.g., juniper woodlands that have encroached because of grazing and lack of wildfire) may potentially increase the amount of water available during the summer, although documentation of this effect is poor. Prescribed fire is most likely to affect aquatic ecosystems through increased surface runoff and erosion resulting from the removal of vegetation and formation of hydrophobic soils.

In summary, manipulations of vegetation on rangelands can influence salmonid habitats through both direct and indirect pathways. These changes may harm or benefit salmonids depending on whether temperature, spawning sites, cover, or food limit the production of salmonids. Salmonid abundance will decrease if the increased invertebrate production is offset by undesirable alterations in the benthos assemblage to less nutritious species, reduced cover, increased sedimentation, and lower water quality.

6.2.8 Effects on Physical Habitat Structure

Livestock-induced changes in physical structure within streams result 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, 1985b 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 both invertebrate production and the quality of spawning gravels. Over the long-term, reductions in instream wood diminish 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 high-flow events, which can reduce gravels available for spawning and facilitate channel downcutting. Figure 6-3 illustrates the characteristics of vegetation in functional and dysfunctional riparian zones on rangelands, and the channel modifications that typically result.

6.2.9 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). In a study of seven stream reaches in eastern Oregon, Tait et al. (1994) 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.

[Figure 6-3. General characteristics and functions of a\) disturbed and b\) undisturbed riparian areas on rangelands. From Elmore \(1992\). Reproduced with permission from the publisher.](#)

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, because these larvae are well protected from fish predation by their cases. Tait et al. (1994) suggest that these organisms may act as a trophic shunt that prevents energy from being transferred to higher trophic levels.

Evidence of negative effects of livestock grazing on salmonid populations is largely circumstantial, but is convincing nonetheless. Platts (1991) 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 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 (Claire and Storch 1983). Platts (1991) cites other examples of improved habitat conditions resulting in increased salmonid populations.

6.3 Agriculture

Approximately 12% of the total land area in Washington, Oregon, and Idaho is dry cropland, with an additional 4% devoted to irrigated agriculture. 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 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 historical floodplains of 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 6.8). Qualitative summaries of the historical effects of agriculture on aquatic ecosystems have been reported by Smith (1971), Cross and Collins (1975), Gammon (1977), and Menzel et al. (1984).

6.3.1 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 prairies and floodplain forests were replaced with croplands (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 in most instances have been permanent alterations of the landscape.

6.3.2 Effects on Soils

Agriculture involves repeated tillage, fertilization, pesticide application, 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 (see Section 6.3.4), but vary with soil type and slope.

6.3.3 Effects on Hydrology

Changes in soils and vegetation on agricultural lands typically result in lower infiltration rates, which yield greater and more rapid runoff. For example, Auten (1933) suggested that forested land may absorb fifty times more water than agricultural areas. Loss of vegetation and soil compaction increase 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 summer 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. Reductions in summer base 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 flood duration and to 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, channelization of streams leads to decreases in summer base flows because of reduced groundwater storage (Wyrick 1968), which can limit habitat availability for fish and increase crowding and competition. In more extreme cases, streams may dry completely during droughts (Gorman and Karr 1978; Griswold et al. 1978).

6.3.4 Effects on Sediment Transport

Because of the intensity of land use, agricultural lands contribute substantial quantities of sediment to streams. The Soil Conservation Service (SCS 1984) estimated that 92% of the total sediment yield in the Snake and Walla Walla River basins of southeastern Washington resulted from sheet and rill erosion from croplands--lands that accounted for only 43% of the total land area. The loss of vegetative cover increases soil erosion because raindrops are free to detach soil particles (splash erosion). Fine sediments mobilized by splash erosion fill soil interstices, which reduces infiltration, increases overland flow, and facilitates sheet and rill erosion. 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 delivery to streams (SCS 1984). Mass failures are probably rare on most agricultural lands because slopes are generally gentle; however, sloughing of channel banks may occur in riparian zones in response to vegetation removal.

6.3.5 Effects on Thermal Energy Transfer and Stream Temperature

Removal of riparian 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. In areas of irrigated agriculture, temperature increases during the summer are exacerbated by heated return flows (Dauble 1994). These effects are discussed in greater detail on Section 6.1.5.

6.3.6 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 $1.1 \text{ mg}\cdot\text{L}^{-1}$) in the Serpentine and Nicomekl rivers during the summer, which in turn caused substantial mortality of coho salmon.

6.3.7 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).

6.3.8 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. Each of these activities reduces physical habitat complexity, decreases channel stability, and alters 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 woody debris and more deep pools than nonincised channels (Shields et al. 1994).

6.3.9 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. 1994a). Nonpoint sources of nutrients and physical habitat degradation were identified as causes of much of the biological degradation. In another study, Maxted et al. (1994b) also demonstrated that the amount of shading had marked effects on stream temperatures and dissolved oxygen concentrations (e.g., Figure 6-4). In some agricultural stream reaches without riparian vegetation, 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) reported 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 were considered most responsible for declines or extirpations of 44% and 67% of the fish species from the Maumee and Illinois drainages, respectively.

Sixty-three percent of California's native fishes are extinct or declining (Moyle and Williams 1990), with species in agricultural areas being particularly affected. 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 crops for 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

[Figure 6-4. Diel fluctuations in temperature \(top\) and dissolved oxygen \(bottom\) in shaded and unshaded reaches of Mudstone Branch/Wharton Branch. From Maxted et al. \(1994b\). Reproduced with permission from the principal author.](#)

(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

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% (45,346 hectares [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 (PSWQA 1988). 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.

6.4.1 Effects on Vegetation

Urbanization causes severe and essentially 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.

6.4.2 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 buildings, concrete, and 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 may be permanent.

6.4.3 Effects on Hydrology

Urbanization significantly influences hydrologic processes, increasing the magnitude and frequency of peak discharges and reducing summer base flows (Klein 1979; Booth 1991). These changes occur primarily because of 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 1993). 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 nonurban 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 because of higher rainfall intensities, 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. 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 historical 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, seven floods exceeding the magnitude of a 5-year event were simulated for the 40-year period. In contrast, in the urbanized basin, simulated floods equaled or exceeded the discharge of a 5-year flood event in 39 of the 40 years (Figure 6-5).

Water withdrawals for water supply, industry and food processing can alter the flow regimes and quantity and quality of stream water. Muckleston (1993) reported that public water supplies accounted for 42% and 84%, respectively, of the total withdrawals from surface waters in the Willamette Basin, Oregon, and Puget Sound, Washington; these areas have the highest population densities found in these two States. 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, dependable power, and flood control has led to numerous impoundments on the major Northwest river systems. These reservoirs have altered the natural flow regimes and fish habitats. For example, flows in the Willamette River, which historically reflected annual precipitation patterns, have been substantially altered to accommodate urban water needs. On average, summer low flows are higher than in predevelopment periods because water is now stored during the wet season and released during the summer.

[Figure 6-5. Hydrologic Simulation Program Fortran simulation of the Hylebos Creek basin in southwest King County, Washington, under fully forested land cover \(top\) and fully urbanized condition \(bottom\) \(approximately 40% effective impervious area\). Bars show the number of years separating discharge events of 5-year recurrence or greater. For forested condition, the average separation is 5 years \(40 years of simulation, 8 events\), but the actual spacing varies from 1 year \(i.e., successive years\) to 14 years. For urbanized condition, discharges exceeding 5-year forested event occur every year except one. Figure reprinted with permission from Booth \(1991\). ©1991 by the Institute for Environmental Studies, University of Washington.](#)

6.4.4 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 impervious surfaces. Specific effects are likely to vary with degree of urbanization, and whether drainage 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 6.4.7).

6.4.5 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 1.5-3C lower than in nonurbanized streams on Long Island, New York. Although other land-use activities alter stream temperatures, in urban areas the loss of riparian function is long lasting. Stream temperatures may also be indirectly affected by changes in hydrology, channel morphology, and the urban microclimate. Klein (1979) suggested 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.

6.4.6 Effects on Nutrients and Other 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 a 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. Omernik (1977) determined that total nitrogen exports from urban areas were second only to agriculturally influenced watersheds.

6.4.7 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) identified numerous types 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. Contributions of grease and oil ranged from 32.8 lb·curb mile⁻¹·day⁻¹ for industrial areas to 4.9 lb·curb mile⁻¹·day⁻¹ for commercial areas (Pitt and Amy 1973). Residential areas fell between (18.6 lb·curb mile⁻¹·day⁻¹). Klein (1979) reported that 9% of persons who 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 were three-to-four times greater than for rural areas. In industrial areas, runoff may include heavy metals, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), high Ph concrete dust, and other toxic chemicals (Birch et al. 1992). Water quality degrades as a consequence of these pollutants entering streams, lakes, and estuaries. Biological oxygen demand is increased with the addition of organic materials, and lethal or sublethal effects may occur with influxes of heavy metals, pesticides, PCBs, and PAHs (see Sections 5.1.2 and 6.4.9).

6.4.8 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. 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 (1993) 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 have substrates dominated by coarser fractions rather than gravel, sand, and silt. The lack of large woody debris inputs exacerbates channel simplification (Lucchetti and Fuerstenberg

1993), 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 because of higher stream energy and increased erosion of streambanks (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 noninteracting conduit between upriver areas and the sea. Important interactions between the stream and surrounding floodplain are lost.

6.4.9 Effects on Stream Biota

The structure of the biological community and the abundance of aquatic organisms are 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 Tricoptera (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 assemblage 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 (1993) found that fish assemblages in small urbanized streams have been dramatically altered or lost. They concluded that coho are of particular concern in urbanized areas because of their specific 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) found that juvenile chinook salmon that migrate

through an urban estuary contaminated with PCBs and PAHs bioaccumulated these pollutants and exhibited a suppressed immune response compared to fish from an uncontaminated 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 nonurban 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 because exposure occurs during the physiologically stressful period of smoltification.

6.5 Sand and Gravel Mining

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 38.3 m³ (50 yr³) (OWRRI 1995). Since 1967, the Oregon Division of State Lands has issued over 4,000 permits for gravel removal (OWRRI 1995), and between 1987 and 1989, a total of 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 38.3 m³, 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 Department of Natural Resources, personal communication). Nevertheless, mining activity occurs near or in most major rivers west of the Cascade Range (Palmisano et al. 1993a, 1993b). 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 6-6). 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 1995). 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 1995). Much of the material contained in this section comes from these three sources.

[Figure 6-6. Sand and gravel operations of Washington, 1979. From Palmisano et al. \(1993a\). Reproduced with permission from the author.](#)

6.5.1 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, resulting in altered channel morphology, decreased stability, accelerated erosion, and changes in the composition and structure of the substrate (Sandecki 1989; OWRRI 1995). 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 a complex interplay between direct effects of channel modification and altered substrate composition, and the resulting alteration of 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 1995). 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 1995). 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 1995). 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 m in response to gravel mining (Table 6-6). In several cases, downcutting occurred over several decades; however, in one instance, a drop in bed elevation of 4.5 m occurred during two flood events that spilled into a large mining pit in Tujung 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 m 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 constricted 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 level rises again (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 OWRI 1995). 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.

6.5.2 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; OWRI 1995). Loss of shallow groundwater storage reduces summer base flows and may also lead to loss of riparian vegetation (Sandecki 1989).

Table 6-6. Case histories relating the effects of gravel extraction on channel morphology and hydrology of streams in Washington, Oregon, and California.

Location	Activity	Effects
WASHINGTON		
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.
CALIFORNIA		
Cache Creek†	In-channel extraction during dry season	Channel degradation up to 9 m (avg 5 m) over 21-year 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.
Tujunga 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.

OREGON

Willamette River¶ # Sand and gravel extraction

Channel degradation of approximately 0.3 m per year over 20- to 30-year period. Degradation because of 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).

OWRRRI (1995).

6.5.3 Effects on Thermal Energy Transfer and Stream Temperature

The most likely changes in heat transfer processes resulting from gravel mining are increased heat exchange from the loss of riparian vegetation and alteration of the surface-to-volume ratio of the stream (OWRRRI 1995). As discussed in Section 3.7, 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 because of inputs of heated water from off-channel ponds created by excavation (OWRRRI 1995).

6.5.4 Effects on Nutrients and Other 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 6.2.6) the availability of nitrate

nitrogen and other solutes may increase in response to the oxidizing environment. Nutrient inputs are also affected where riparian vegetation is modified or eliminated.

6.5.5 Effects on Physical Habitat Structure

Most concern regarding the effects of gravel and sand mining on salmonids has focused on spawning habitats. Extraction of gravels may 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 where degradation reduces gravel depth or exposes bedrock. In addition, decreases in the stability of streambeds can potentially increase embryo and alevin mortality because of scouring of gravel beds. Deposition of fine sediments downstream of mining activities may reduce the quality of these areas as spawning habitats (Kondolf 1994).

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 are important for survival. Similarly, a reduction in pool frequency may adversely affect migrating adults that require holding pools during their upstream migrations.

6.5.6 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 1995). Aquatic invertebrates, which are an important prey for stream-dwelling salmonids, can be disrupted by disturbance of the substrate during mining (AFS 1988) or by changing substrate composition or covering of substrate with fine sediments (Hicks et al. 1991a). Potential effects on invertebrates include changes in species composition, reduced biomass, and slowed biotic colonization (OWRRI 1995).

OWRRI (1995) 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 (1995) 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.

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. 1991a; Barrett et al. 1992). Sigler et al. (1984) reported that turbidities ranging from 25-50 NTU (nephelometric turbidity units) reduced growth and increased the tendency of young coho salmon and steelhead trout to emigrate from

laboratory streams. Other direct effects of turbidity on fish are discussed in greater detail in Section 5.1.2. 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 1995). 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 1995).

6.6 Mineral Mining

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 portions 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 12.1 million m³ (15.8 million yd³) of sediments (PFMC Habitat Committee 1994). The effects there and elsewhere are still being felt today as sediments and pollutants derived from mine tailings continue to enter streams. The PSMFC (1994) reports that mining is responsible for polluting 19,350 km (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). Before 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 (Field 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 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.

6.6.1 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. 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 chronic sediment sources (McIntosh et al. 1994b). Records from the 1940s indicate that substantial portions of the upper Grande Ronde river flowed beneath the extensive rubble left behind by early mining operations (McIntosh et al. 1994b). 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).

6.6.2 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 $452.1 \text{ cm}\cdot\text{h}^{-1}$ on undisturbed soils versus $43.2 \text{ cm}\cdot\text{h}^{-1}$ on adjacent graded spoils banks. Lower infiltration rates mean overland runoff and streamflow increase, 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 400,000 barrels of oil annually from oil shales in Utah, Colorado and Wyoming would require 86 million m^3 of water per year, which would be pumped from groundwater aquifers (Nelson et al. 1991). It was the tremendous demand for water by mining operations that stimulated the water law of prior appropriation in the West (Wilkinson 1992).

6.6.3 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. Bank instability can also lead to altered width-to-depth ratios, which further influences temperature (see

Section 3.7).

6.6.4 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 by 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_2), sphalerite (ZnS), galena (PbS), and greenockite (CdS), undergo similar acid-generating processes (Nelson et al. 1991). Reductions in pH increase the mobility of many heavy metals (e.g., aluminum, 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_3), 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 Clark Fork River, Montana (Johns and Moore 1985). Duamie et al. (1985) reported that loadings from an abandoned mine in Montana were $13.6 \text{ kg}\cdot\text{d}^{-1}$ for copper and $1.6\text{-}145.5 \text{ kg}\cdot\text{d}^{-1}$ for zinc. Acid mine drainage and copper loadings of $41\text{-}147 \text{ kg}\cdot\text{d}^{-1}$ 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 or is released from storage lagoons is potentially lethal to all aquatic organisms.

6.6.5 Effects on Physical Habitat Structure

The effects of surface mineral mining on the 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 substrate. Spaulding and Ogden (1968) estimated that hydraulic mining for gold in the Boise River basin, Idaho produced 116,500 tonnes 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, which 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).

6.6.6 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 exposed to low pH have been shown to experience reduced egg viability, fry survival, growth rate, development of pigmentation, ossification, and heart rate (Trojnar 1977; Nelson 1982). Johnson and Webster (1977) reported that spawning brook trout avoid areas of low pH, and speculated 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 metalloids. 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). Because 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 LD₅₀ levels for these toxicants using various invertebrate and fish test organisms (Table 6-7). 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. Hughes (1985) found that Montana streams subjected to periodic mine effluents had fundamentally altered benthos assemblages and reductions or elimination of trout and sculpins. Other examples of exposure of salmonids and other aquatic organisms to pollution from mine wastes are reviewed in (Nelson et al. 1991).

6.7 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 1986). 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). 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 or may be injured or disoriented, becoming easy prey for aquatic and terrestrial predators. Attempts to bypass dams by barging or trucking may stress fish and increase disease transmission among individuals (Bevan et al. 1994a), which ultimately may reduce survival.

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 strand fish or allow desiccation of spawning redds (Palmisano et al. 1993a). The impoundment of water behind dams creates slackwater environments that are less favorable to salmonids. With the exception of the Hanford reach, virtually the entire lower and mid-Columbia River has been changed from a free-flowing river to a series of ponded reaches with little fast water, and significant spawning areas have been lost. The slow-moving water behind impoundments are also favorable to certain predators of salmonids, including northern squawfish (Faler et al. 1988).

Table 6-7. Reported toxicities of metals in soft water (< 45,000 g·L⁻¹ as CaCO₃). From Nelson et al. (1991). Reproduced with permission from the publisher.

Substance	Species	Reported Toxicity		
		Method*	Concentration (g·L ⁻¹)	Source
Aluminum (Al)	Brook trout	LC50	3,600 - 4,000	Decker and Menendez (1974)
Arsenic (As)	Rainbow trout	LC50	10,800	Hale (1977)
Cadmium (Cd)	Rainbow trout	LC50	6.6	Hale (1977)
	Brook trout	MATC	1.7 - 3.4	Benoit et al. (1976)
Chromium (Cr)	Rainbow trout	LC50	24,100	Hale (1977)
	Rainbow trout	LC50	69,000	Benoit (1976)
	Rainbow trout	MATC	200 - 350	Benoit (1976)
	Brook trout	LC50	59,000	Benoit (1976)
	Brook trout	MATC	200 - 350	Benoit (1976)
Copper (Cu)	Coho salmon	LC50	46.0	Chapman and Stevens (1978)
	Rainbow trout	LC50	253.0	Hale (1977)
	Rainbow trout	LC50	125.0	Wilson (1972)
	Rainbow trout	ILL	37.0	Chapman and Stevens (1978)
	Atlantic salmon	ILL	32.0	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	520	Sprague and Ramsay (1965)
	Brook trout	MATC	9.5 - 17.4	Sprague (1964)
	Brook trout	MATC	9.5 - 17.4	McKim and Benoit (1971)
Copper-zinc (Cu-Zn)	Atlantic salmon	TU	1,000	Sprague and Ramsay (1965)

Iron (Fe)	Brook trout	LC50	1,750	Decker and Menendez (1974)
Lead (Pb)	Rainbow trout	LC50	8,000	Hale (1977)
	Rainbow trout (eggs)	MATC	4.1 - 7.6	Davies et al. (1976)
Mercury (Hg)	Rainbow trout	LC50	33,000	Hale (1977)
Nickel (Ni)	Rainbow trout	LC50	35,500	Hale (1977)
Uranium (U)	Brook trout	LC50	2,800	Parkhurst et al. (1984)
Zinc (Zn)	Coho salmon	LC50	905.0	Chapman and Stevens (1978)
	Rainbow trout	LC50	1,775	Chapman and Stevens (1978)
	Rainbow trout	LC50	180 - 390	Finlayson and Ashuckian (1979)
	Rainbow trout	ILL	560	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	92	Zitko and Carson (1977)
	Atlantic salmon	ILL	150 - 1,000	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	420	Holcombe et al. (1979)
	Brook trout	MATC	534 - 1,360	

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

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. The reduction in sediments downstream of dams leads to changes in channel morphology (Marcus et al. 1990). 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 faster, turbulent waters. Water that spills over dams entrains air, and supersaturation

of dissolved gases results. Gas supersaturation can cause gas bubble disease in salmonids, resulting in mortality, or weakening fish such that they become more vulnerable to predation or infection (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).

6.8 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 (Wilkinson 1992; Palmisano et al. 1993a). 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 and in the upper Klamath Basin (Muckleston 1993). 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 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.

6.8.1 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 River basin blocked sockeye runs estimated at 200,000 adult fish (Palmisano et al. 1993a). 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 (BR 1995). 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.

6.8.2 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 $636.8 \text{ m}^3 \cdot \text{s}^{-1}$ (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 has a capacity to withdraw $57 \text{ m}^3 \cdot \text{s}^{-1}$ (2,000 cfs) from the Yakima River, 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; 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 $0.1 \text{ m} \cdot \text{s}^{-1}$ (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 and assemblages is also regulated by the water velocity (Powell 1958; 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 presmolt 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 (Beiningen 1976), dewater redds 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 affects other levels in the food web, particularly in large, unshaded rivers, where periphyton can be an important energy source.

6.8.3 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 spawning gravels and filling in pools that chinook salmon use for rearing (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 slower 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 is frequently dominated by cobble and other large fractions unsuitable for spawning.

Iwamoto et al. (1978) reported 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.

6.8.4 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 because of 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.

6.8.5 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).

6.8.6 Influence of Impoundment and Water Withdrawal on Fish Diseases

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 (see Section 4.3.4); 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 (BR 1976).

6.9 River, Estuary, and Ocean Traffic (Commercial and Recreational)

Within a few decades of settlement, many estuaries and large low gradient rivers of western California, Oregon, and Washington were channelized. Eventually significant portions of major rivers, including the Columbia and Sacramento, were radically transformed. These systems were first altered for riverboat navigation and later to accommodate 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. Networks of essentially small, partially shaded rivers offered much more productive rearing habitat than the present navigation channels. These complex channels were nutrient-rich rearing habitats in the summer and provided refugia from high streamflows during the winter. Changes in channel complexity are exemplified by the 80% reduction in the number of upper mainstem Willamette River channels documented by 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 some limited cover, but salmonids may be exposed to high levels of wood preservatives, petroleum, and organic wastes, as well as intense angling pressure. Noise pollution from boat traffic in estuaries and the open ocean may disrupt the navigation and communication of sharks and toothed whales; this may indirectly affect salmonids because sharks and whales feed on seals and sea lions, which in turn prey on salmon.

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 perhaps resulted in the loss of more fish than alterations of higher gradient reaches of similar size.

6.10 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 89.44 million hectares (221 million acres) of wetland functioned in the conterminous United States in the 1780s and that 53% of that area had been lost by the 1980s. Wetlands lost during this period included 1,839,741 hectares (4,546,000 acres) in California, 198,826 hectares (491,300 acres) in Idaho, 351,315 hectares (868,100 acres) in Oregon, and 166,734 hectares (412,000 acres) in Washington. These losses 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 (EPA 1980b). 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.

6.10.1 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, especially low-flow periods in summer. 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 selection of spawning redd for some species. The constant upwelling of fresh, oxygenated water may be necessary for 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.

6.10.2 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. Some nutrients, as well as toxic substances, are taken up by plants, while others are bound to suspended solids, which subsequently settle to the bottom. Thus, loss or removal of wetland areas may result in increased nutrient and contaminant loading to receiving streams.

6.10.3 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 (Cederholm 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. For some salmonids, the timing and amount of streamflow triggers the movement of adult salmon into spawning streams, as well as the movement of fry and smolts downstream. Changing the timing of flows may thus subtly change the timing of migration and spawning, resulting in disruption of natural biological cycles. 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 (Cederholm 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 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, altering important elements of aquatic ecosystems. 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. In highly modified landscapes, they may be the only natural areas left 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.

6.11 Salmonid Harvest

Although this document focuses on the effects of human activities on salmonid habitats, it is essential to recognize the effects fisheries have had on salmonid populations in the Pacific Northwest. The harvest of salmonids by humans constitutes a significant source of mortality for both anadromous and resident species. For thousands of years preceding settlement of the West by Euro-Americans, Native Americans depended on salmonids as an important source of food, and salmonids continue to be central to the culture and economy of many tribes. Since the mid-1800s, large number of salmonids have been taken in off-shore and in-river commercial and recreational fisheries. In the late 1800s and early 1900s, chinook salmon dominated commercial landings off Washington, Oregon, and California (Deimling and Liss 1994), as well as in-river fisheries in the lower reaches of the Columbia, Sacramento-San Joaquin, and other large river systems. These fish were targeted for their large size and high food quality. Subsequently, salmonid fisheries have become progressively more diverse with other anadromous salmonids, particularly coho and pink salmon and steelhead trout, accounting for growing fractions of the total catch.

The relative importance of different fisheries varies across the region. From central California to Cape Flattery, Washington, ocean commercial troll and recreational fisheries account for the highest catch of anadromous salmonids, although substantial in-river sport and tribal harvest occurs in some river systems. The Pacific Fishery Management Council (PFMC 1995) estimates that between 1971 and 1990, combined catch of coho salmon in the commercial troll and recreational fisheries off Washington (Puget

Sound included), Oregon, and California averaged over 1.9 million fish annually. Catch of chinook in the region during the same period averaged 1.3 million fish, and average catch of pink salmon in odd-numbered years (primarily in Washington) was about 200,000 fish. The Oregon Department of Fish and Wildlife (ODFW 1982) estimates that from 1971-1975 the commercial troll fishery accounted for 67% of the total coho salmon harvest in the Oregon Production Index (OPI) area (Columbia River south to central California), with the ocean recreational fishery and Columbia River gill-net fisheries accounting for 23% and 8%, respectively. In-river recreational and tribal fisheries made up the remaining 2% of the catch. The allocation of chinook salmon among various fisheries in Oregon varies with region and life-history types. For north-migrating stocks, ODFW allocates approximately 50%-55% of total annual harvest to the ocean troll fishery and approximately 45% to the in-river recreational fishery. For south-migrating stocks, ocean troll fisheries account for 60%-67% of the total fish harvested, whereas in-river fishery targets are approximately 15%-37% of total harvest (ODFW 1991).

In the Puget Sound area (Strait of Juan de Fuca, Puget Sound, Hood Canal, Nooksack-Samish, Skagit, and Stillaguamish-Snohomish units) commercial fisheries, including Indian and non-Indian, gill-net, purse seine, and troll fisheries, dominate the catch of salmonids; gill-nets and purse seines accounted for greater than 91% of the commercial catch in 1989 and 1990 (Palmisano et al. 1993a). Total commercial harvest in the Puget sound area compares with the combined troll and recreational catch from coastal waters along Washington to California, with pink salmon and sockeye salmon being numerically dominant. From 1971-1990, combined annual catch of pink salmon (odd-years only) in treaty and nontreaty commercial fisheries averaged approximately 2.9 million fish, while catch of sockeye salmon averaged over 1.8 million. During this same period, catch of coho, chum, and chinook salmon averaged 972,000, 768,000, and 211,000 fish, respectively (PFMC 1993). From 1979-1990, sport harvest in the Puget Sound area averaged approximately 8.6% of the commercial catch (Palmisano et al. 1993a).

Estimating total harvest rates on specific stocks or species of anadromous salmonids is difficult. These calculations require accurate estimates of 1) total ocean and in-river harvest (including harvest of fish originating in Oregon, Washington, and California by fisheries in Alaska and British Columbia), 2) spawning escapement (sometimes direct counts, but often estimated from index streams), 3) indirect hooking and dropout mortality, and 4) rates of natural mortality for species with a variable period of ocean residence. Despite uncertainty associated with each of these estimates, calculations of total harvest rates for several anadromous salmonid populations provide some indication of the magnitude of fishing effects. Between 1960 and 1983, harvest rates of coho salmon in the OPI area ranged from 57% to 87%, with a mean exploitation rate of 71%. Harvest rates were lower from 1984-1993, ranging from 27%-62% with a mean of 43% (T. Nickelson, ODFW, personal communication). Further reductions in harvest rates occurred in 1994 and 1995 because of the closure of the coho fisheries. Ocean harvest rates of chinook salmon originating from the Sacramento-San Joaquin system in California ranged from 50%-79% between 1970 and 1992, with a mean harvest rate of 67% (PFMC 1993); in-river fisheries were not included in these estimates. In its management plan for coastal chinook salmon stocks, the Oregon Department of Fish and Wildlife (ODFW 1991) concludes that most chinook populations can sustain harvest rates below 67% without compromising long-term conservation goals. Actual target harvest rates for various stocks of chinook and coho salmon are adjusted depending on stock strength and specific escapement goals. Recent evidence indicates that biased selection of index streams has resulted in

overestimation of spawning escapement and, hence, underestimation of harvest rates (Cooney and Jacobs 1994). In addition, harvest targets consider only numerical abundance and do not address other long-term effects discussed below.

Adverse effects of harvest on salmonids are particularly difficult to control in mixed-stock fisheries, where multiple species, stocks, and age classes are harvested together. Mixed-stock fisheries occur primarily in the ocean and lower river reaches, before stocks segregate into discrete spawning runs. Mixed-stock fisheries are difficult to manage because escapement goals and harvest rates of different stocks constituting the fishery cannot be controlled. Consequently, strong and weak stocks are harvested at comparable rates, as are fish of wild and hatchery origin. For example, in the Columbia River system, where 90%-95% of the coho salmon are of hatchery origin, harvest rates approaching 90% still allow adequate escapement for hatchery brood-stock purposes, whereas ODFW (1982) estimates that harvest rates should be less than 69% to meet escapement goals for wild coho. Thus, where hatchery and wild coho salmon commingle in the ocean, wild fish are likely to be harvested at an excessive rate. Mixed-stock fisheries are especially detrimental to naturally small populations or populations that have been depressed by human activities. In these populations, escapement may be insufficient to maintain genetic diversity, and the probability of undesirable founder effects increases.

In addition to reducing total escapement of adult salmonids, harvest alters the age- and size-structure of salmonid populations. For example, Ricker (1981) provided evidence that mean sizes of all five Pacific salmon species harvested in British Columbia have decreased over the past 30 to 60 years. Similarly, between 1935 and 1989, the average weight of coho salmon caught in commercial fisheries off the coast of Washington declined by almost 30%; over the same period, mean weight of chinook, pink, and sockeye salmon decreased by 24%, 19%, and 14%, respectively (Palmisano et al. 1993a). Changes in size and age-structure arise for several reasons. For long-lived species that spend several years at sea, such as chinook salmon and steelhead trout, decreases in average size and age occur because immature individuals are harvested by troll fisheries over a number of years. Thus, larger and older individuals are harvested at a higher rate than individuals that mature earlier and at smaller size (Moussalli and Hilborn 1986), particularly for those stocks frequenting coastal waters rather than only passing through coastal waters on their way to spawning areas. Changes in size structure may also result from size-selective fishing gear. Ricker (1981) attributed decreases in average size of coho and pink salmon adults to cumulative genetic effects caused by selective removal of larger individuals in troll and gill-net fisheries. Selective removal of larger fish may also increase the percentage of "jacks" in sockeye salmon populations. In freshwater fisheries, size limits and gear restrictions also alter size and age-structure of salmonid populations. For example, Gresswell and Varley (1988) reported that mean length of cutthroat trout caught in Yellowstone Lake rose from a low of 365 mm in 1966-1967 to a high of 395 mm in 1983-1984 following a change in fishing restrictions from three fish of any size to two fish with a 330 mm maximum. Average age of spawners in Clear Creek, a major spawning tributary of the lake, increased from 4 years to almost 5.5 years during this period, with a concomitant increase in the frequency of fish aged 7-9.

Changes in average size and age of individuals influences success of salmonid populations in several ways. Large size provides salmonids with the energetic reserves needed to undertake extensive

migrations as well as the ability to negotiate large barriers that are impassable to smaller fish; thus, the elimination of large individuals through harvest can effectively diminish the ability of populations to use particular spawning habitats. In addition, because the fecundity of salmonids typically increases with size, the selective removal of larger fish results 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 that exceed the carrying capacity of their habitat) (Ricker 1972; Jaenicke and Celewycz 1994). Larger females also tend to dig deeper redds than smaller females, which reduces the likelihood that eggs will be destroyed by bedload movement during freshets. Larger females also select nest locations with larger gravel, which increases exchange of water and oxygen. Both of these behaviors combine to provide a greater egg-to-smolt survival (Hankin and Healey 1986; Hankin et al. 1993).

Harvest of salmonids also influences the timing of certain life history events, including adult migrations, spawning, and juvenile migrations. Frequently, fisheries are restricted to a relatively narrow window of time, particularly as stocks dwindle in numbers. Selective removal of early or late migrants can potentially result in shifts in the timing of peak migration and spawning within a population. Studies indicate that disproportionate representation of early migrants in hatchery broodstocks can cause a shift in migration timing within only a few generations (Millenbach 1973; Alexandersdottir 1987); harvesting only at the beginning or end of a run may have similar effects. Gharrett and Smoker (1993) reported that early and late-migrating adult pink salmon produce young that migrate to sea at different times. Consequently, removal of predominately early or late migrants can also alter the migration characteristics of the juvenile population.

Finally, the harvest of salmonids by humans can fundamentally alter the structure of stream ecosystems through reduction of nutrient inputs from salmon carcasses as populations decline and average size of fish decreases. Carcasses contribute significant amounts of nitrogen and phosphorus compounds to headwater streams (Cederholm and Peterson 1985; Bilby et al. 1996), the nutrients that most often limit production in oligotrophic systems. The role of carcasses in providing nutrients to stream systems is discussed in greater detail in Section 3.8.2.

6.12 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. These introductions have included both non-native species, primarily from the eastern United States and Europe, and artificially propagated native salmonids.

6.12.1 Introductions of Non-native Species

Introduction of non-native fishes into waters of the Pacific Northwest began before the turn of the century and continues today. Four primary sources of introductions include fishery management

manipulations (stocking of 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 studies documenting conditions both before and after species introductions, effects of introductions on native fishes may include elimination, reduced growth and survival, and changes in community structure. For example, brown trout (*Salmo trutta*) 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 were 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%.

Moyle et al. (1986) identified six mechanisms 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. They 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). Introduced grass carp (*Ctenopharyngodon idella*) and common carp (*Cyprinus carpio*) destroy beds of aquatic macrophytes, which reduces cover for juvenile fishes, destroys complex substrates that support diverse invertebrate assemblages, and increase the turbidity of water to the detriment of fishes that locate prey by sight (Moyle et al. 1986). Whirling disease--which was introduced to North American waters from Europe via shipments of frozen fish containing spores of *Myxobolus cerebralis* (Marnell 1986)--has been implicated in the decline of several important trout fisheries in the intermountain West, particularly in the upper Colorado River basin (Nehring and Walker 1996; Vincent 1996). Although this disease has been found in hatcheries within the Pacific Northwest, there is little evidence that it has affected wild trout populations in the region (Nehring and Walker 1996). Cutthroat and rainbow trout freely hybridize in the wild, with the rainbow trout phenotype becoming dominant (Behnke and Zarn 1976); similarly, the various subspecies of these trouts also interbreed.

6.12.2 Artificial Propagation of Native Salmonids

Artificial propagation of native salmonids has been used for decades as a means of replacing lost natural production resulting from various development activities and to increase returns for harvest. Hatchery programs continue to dominate expenditures of State fishery agencies in the Pacific Northwest. White et al. (1995) report that the State of Washington spent \$31.3 million--35% of their total fishery budget--on salmon culture in the 1991-1992 fiscal year. Similarly, propagation of fish accounted for 42.5% of Oregon's \$90.6 million budget for fisheries for the 1993-1995 biennium, whereas only 3% was devoted to management for natural production.

Although artificial propagation may in some instances increase salmon and trout available for harvest, hatchery introductions can result in a number of unintended and undesirable consequences for wild salmon and trout populations (Lichatowich and McIntyre 1987; White et al. 1995). In freshwater, interactions between hatchery and natural fishes may result in greater competition for food, habitat, or mates (Nickelson et al. 1986). Studies have suggested that carrying capacity can be exceeded during the outmigration of smolts to the ocean (Steward and Bjornn 1990). Once in the ocean, large numbers of hatchery smolts may result in density-dependent decreases in survival and growth, although evidence of density-dependent effects in ocean environments is mixed. Indications of density-dependent age and size composition have been found for various Pacific salmonids (Ricker 1981; Peterman 1987; 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. Holtby et al. (1990) found no evidence for density-dependent marine survival of coho salmon migrating from Carnation Creek, British Columbia. It is possible that density-dependent ocean survival may only be manifest in years of low marine productivity.

Other adverse effects of hatchery introductions include transmission of disease between hatchery and wild populations (Marnell 1986; Steward and Bjornn 1990), alterations of fish behavior, and increased predation on wild fish. 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. Alteration of behavior of wild fish, including stimulation of early migration of juveniles (Hillman and Mullen 1989), has been observed in response to hatchery introductions. Hatchery supplementation can also increase predation rates on wild stocks either directly, through predation of hatchery fish on wild fish, or indirectly by attracting predators.

In addition to ecological effects, introduction of hatchery fish may lead to genetic changes in wild populations (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.

The operation of hatchery facilities may adversely affect wild salmonid populations and their habitats in several ways (reviewed in White et al. 1995). Effluent waters from hatcheries may contain high concentrations of nutrients or disinfectant chemicals that negatively affect water quality. Disease organisms can also be introduced to streams via hatchery effluent. The construction of hatchery weirs or diversion structures impedes the migration of wild stocks and diversions of water for hatchery use reduces the amount available for wild stocks. Removal of wild fish for brood stock may threaten the genetic integrity of wild stocks, particularly for small or depleted stocks. And lastly, the removal of fish for brood stock decreases the amount of nutrients available in upstream reaches, since salmon carcasses

are not deposited.

Hatchery supplementation has social repercussions that influence wild salmonids directly, as well as the ability of managers to restore salmonid populations. Hatchery supplementation increases harvest pressure on wild populations in mixed-stock and terminal fisheries ((Palmisano et al. 1993a; Lichatowich and McIntyre 1987), particularly during years when survival of hatchery fish is low due to poor environmental conditions. For example, the overcapitalization of the coho salmon fishery and subsequent overharvest of wild stocks in Oregon in the late-1970s and 1980s was stimulated in part by successful hatchery supplementation during the 1960s and early 1970s (Lichatowich and McIntyre 1987). In addition, once commercial and sport fishers have invested large sums of money in fishing boats and gear, they may become resistant to increased fishing restrictions, making it difficult for managers to enact stricter protections for wild stocks.

Finally, the long history of hatchery programs in the United States has instilled a perception in the public that habitat losses or degradation can be mitigated through artificial propagation (White et al. 1995), or that maintenance of salmon populations depends on hatcheries (Hilborn 1992). The disproportionate spending of State and Federal dollars on hatchery programs compared with protection of natural habitat and wild populations is indicative of the reliance that managers and the public places on artificial propagation. As White et al. (1995) point out, political pressure for stocking has driven management decisions even in cases where scientific evidence has indicated stocking is not needed or detrimental. This pressure has also diverted much-needed funds from other important and more ecologically sound restoration activities.

6.13 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 human use is 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 for habitat in rivers (Maser and Sedell 1994).

6.14 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⁻² across the United States (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⁻² 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 to 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 produce large numbers of fish. Beaver ponds in the Rocky Mountain West were found to support larger and more numerous trout, as well as greater densities 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) suggested beaver act as a keystone species "to affect ecosystem structure and dynamics far beyond their immediate requirements for food and space." Their removal has fundamentally altered aquatic ecosystem function.

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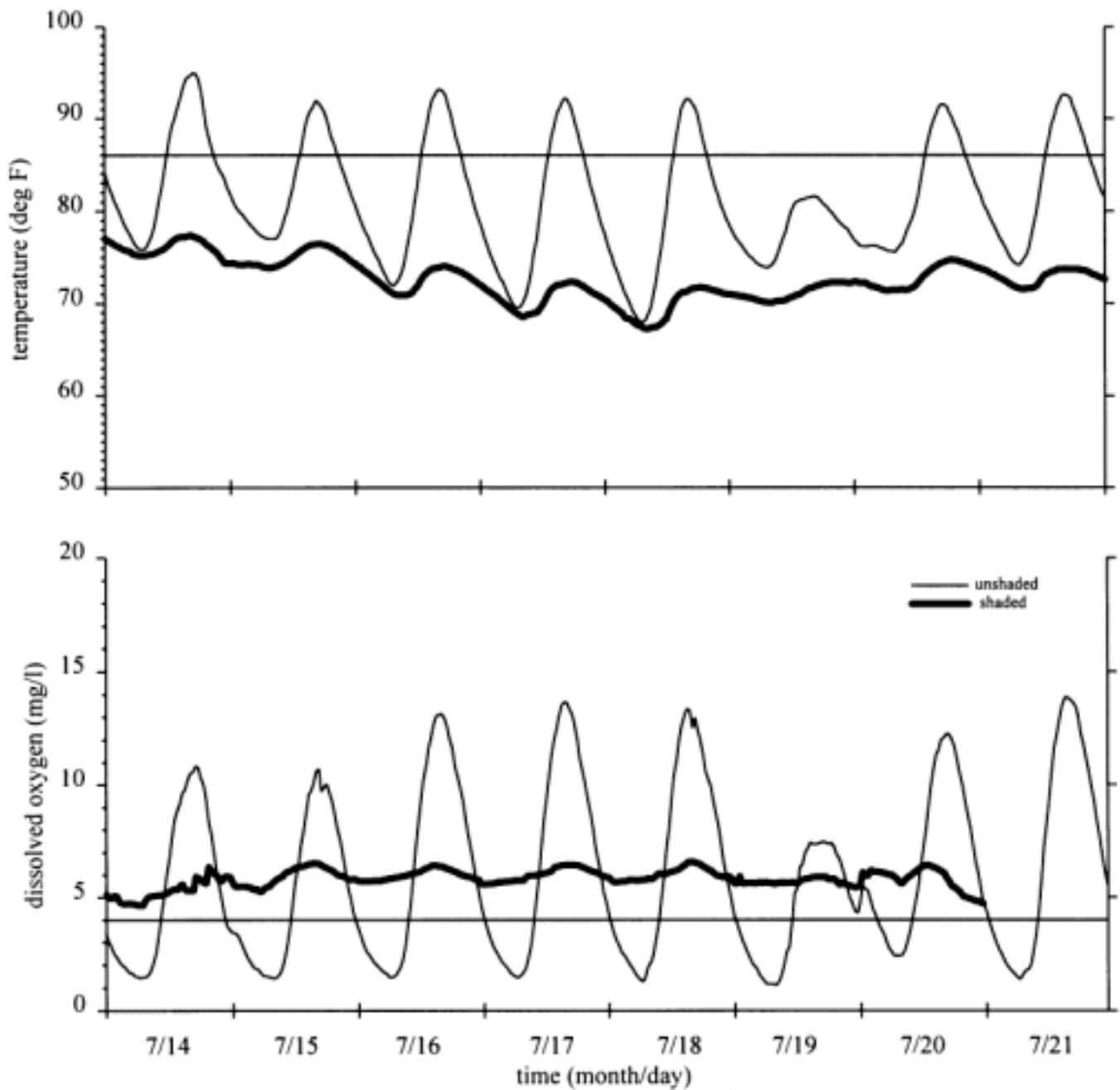


Figure 6-4. Diel fluctuations in temperature (top) and dissolved oxygen (bottom) in shaded and unshaded reaches of Mudstone Branch/Wharton Branch. From Maxted et al. (1994b). Reproduced with permission from the principal author.

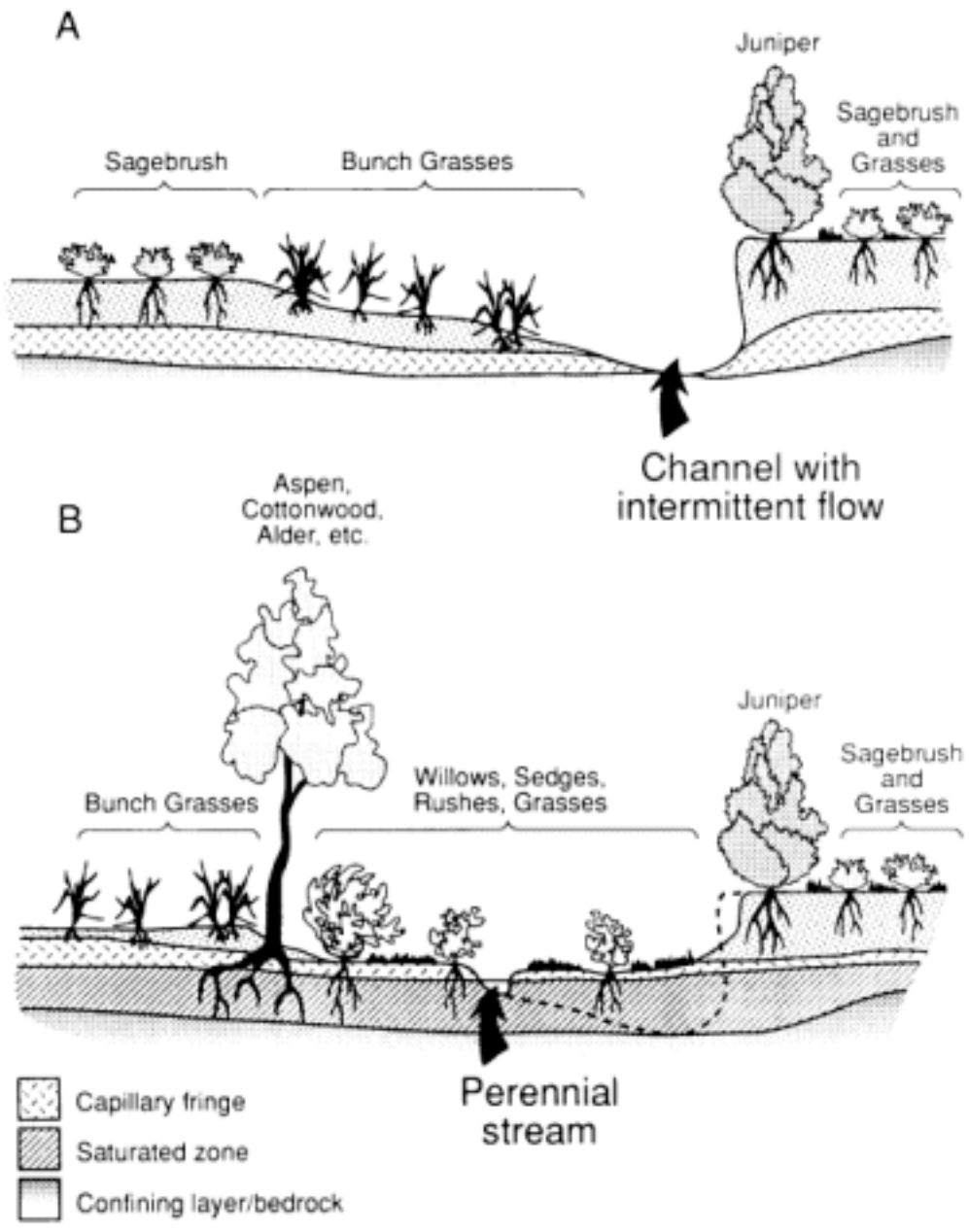


Figure 6-3. General characteristics and functions of a) disturbed and b) undisturbed riparian areas on rangelands. From Elmore (1992). Reproduced with permission from the publisher.