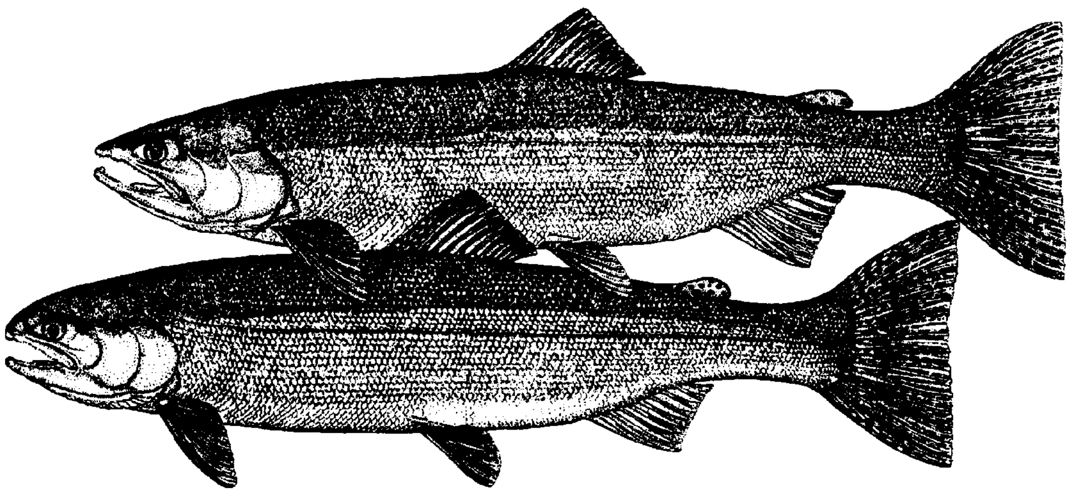
Factors for Decline

A Supplement to the Notice of Determination for
West Coast Steelhead Under the Endangered Species Act



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Introduction to the Endangered Species Act

The Endangered Species Act of 1973 (ESA)[1] provides a framework for identifying species that are in danger of (or threatened with) extinction. The ESA imposes obligations on Federal agencies to prohibit actions that might jeopardize a listed species and directs agencies to use their authorities to promote the conservation of listed species. Further, the ESA imposes restrictions on the activities of all persons that might result in the taking,[2] either directly or indirectly, of listed species.

The ESA[3] divides responsibility for listing species between the Secretary of the Interior and the Secretary of Commerce. Essentially, the Secretary of the Interior is responsible for all terrestrial and freshwater species while the Secretary of Commerce is responsible for all marine species. In some cases, such as for sea turtles, the two departments share jurisdiction. The Secretary of the Interior has delegated his authority under the ESA to the United States Fish and Wildlife Service (FWS). The Secretary of Commerce has delegated his authority to the National Marine Fisheries Service (NMFS).

The NMFS' ESA implementing regulations define a "species" to include any species or subspecies of fish, wildlife, or plant, and any distinct population segment of any vertebrate species that interbreeds when mature.[4] A "threatened" species is defined as any species in danger of becoming endangered in the foreseeable future;[5] an "endangered" species is defined as a species in danger of extinction throughout all or a significant portion of its range.[6]

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1 16 U.S.C. §§ 1531-1544 (1994).

2 "Take" is defined under the ESA as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct." 16 U.S.C. § 1532 (19) (1994).

3 16 U.S.C. § 1532 (15) (1994).

4 50 CFR § 424.02 (k) 1995.

5 50 CFR § 424.02 (m) 1995.

6 50 CFR § 424.02 (e) 1995.

The ESA allows listing of "distinct population segments" of named species. According to NMFS policy, a salmon population or group of populations is considered "distinct" and hence a "species" under the ESA if it represents an ESU of the biological species (Waples 1991). To qualify as an ESU under NMFS policy, a salmon population or group of populations must satisfy the following two criteria: (1) it must be substantially reproductively isolated from other conspecific population units, and (2) it must contribute substantially to ecological/genetic diversity of the biological species as a whole (Waples 1991). The reproductive isolation need not be absolute but must be strong enough to permit evolutionarily important differences to accrue in different population units.

Summary of Events Leading to the Steelhead Status Review

The NMFS' decision to initiate a comprehensive steelhead (*Oncorhynchus mykiss*) status review was prompted by three petitions, culminating in the agency's proposal to list 10 steelhead ESUs as threatened or endangered. On May 6, 1992, NMFS received a petition from the Oregon Natural Resources Council and 10 co-petitioners to list Oregon's Illinois River winter steelhead under the ESA. The NMFS completed a status review, summarized in the May 20, 1993, *Federal Register* (58 FR 29390), and concluded that the Illinois River winter steelhead did not represent a "species" under the ESA. At the same time however, NMFS initiated a status review of coastal steelhead populations to identify the ESU that includes Illinois River winter steelhead. This status review resulted in the identification of a Klamath Mountains Province ESU that includes steelhead from the Illinois River; NMFS proposed listing this ESU on March 16, 1995 (59 FR 14253). The NMFS received a second petition on September 21, 1993, from Washington Trout which requested listing Washington's Deer Creek summer steelhead. As was the case with Illinois River winter steelhead, NMFS determined that Deer Creek summer steelhead did not themselves constitute an ESU (November 21, 1994, 59 FR 59981). The third and most recent steelhead petition was submitted by Oregon Natural Resources Council and 15 co-petitioners on February 16, 1994. In accepting this petition, which requested ESA listing for all steelhead in Washington, Oregon, California and Idaho, NMFS announced that the agency's ongoing steelhead status review would be further expanded to include steelhead populations in

Idaho (May 27, 1994, 59 FR 27527).

On August 9, 1996, NMFS published in the *Federal Register* (61 FR 41541) its initial findings on a comprehensive status review of West Coast steelhead populations in Washington, Oregon, Idaho, and California. The NMFS identified 15 ESUs within this range, and proposed to list 5 ESUs as endangered and 5 ESUs as threatened under the ESA. The endangered steelhead ESUs are located in California (Central California Coast, South/Central California Coast, Southern California, and Central Valley ESUs) and Washington (Upper Columbia River ESU). The threatened steelhead ESUs are dispersed throughout all four states and include the Snake River Basin, Lower Columbia River, Oregon Coast, Klamath Mountains Province, and Northern California ESUs. Additionally, NMFS designated the Middle Columbia River ESU as a candidate species because while there was not sufficient information available to indicate that steelhead in this ESU warrant protection under the ESA, NMFS identified specific risk factors and concerns that need to be evaluated prior to concluding its assessment of the overall health of Middle Columbia River steelhead.

Purpose of Report

In accordance with the ESA[7], NMFS is authorized to list a species as endangered or threatened based upon any one or more of the five following factors: (A) the present or threatened destruction, modification, or curtailment of a species' habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; or (E) other natural or manmade factors affecting the species continued existence. The purpose of this report is to synthesize available scientific information with respect to the factors of decline for west coast steelhead. This information will be used by NMFS in making its listing determinations for west coast steelhead.

To ensure that the best available information was used in this report, NMFS solicited the

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[7] 16 U.S.C. § 1533 (a) (1) 1994.

assistance of state and tribal fisheries agencies in identifying factors of decline for steelhead. This report is in part derived from information provided by these steelhead co-managers. While every attempt was made to capture the most up-to-date information on steelhead factors for decline, NMFS recognizes that some areas may have been overlooked or not dealt with in sufficient detail. The NMFS encourages anyone interested in providing comments on this report to submit materials to NMFS at the addresses below.

In addition to consideration of the factors of decline, the ESA provides that NMFS make listing determinations "after taking into account those efforts, if any, being made by any State or foreign nation, or any political subdivision of a State or foreign nation to protect such species."[8] Toward this end, NMFS has prepared a separate document entitled *Conservation Measures: A supplement to the notice of determination for west coast steelhead* which addresses Federal, state, tribal, and local conservation measures pertinent to steelhead. The Conservation Measures report, in conjunction with this report and NMFS' Status Review of West Coast Steelhead (Busby et al. 1996), serve as the basis for NMFS steelhead listing determinations. For copies of these or other related documents, write to Garth Griffin, NMFS, Protected Species Branch, 525 NE Oregon St. - Suite 500, Portland, Oregon, 97232; or Craig Wingert, NMFS, Protected Species Management Division, 501 W. Ocean Blvd. - Suite 4200, Long Beach, CA 90802.

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[8] 50 C.F.R. § 424.11 (f) 1995.

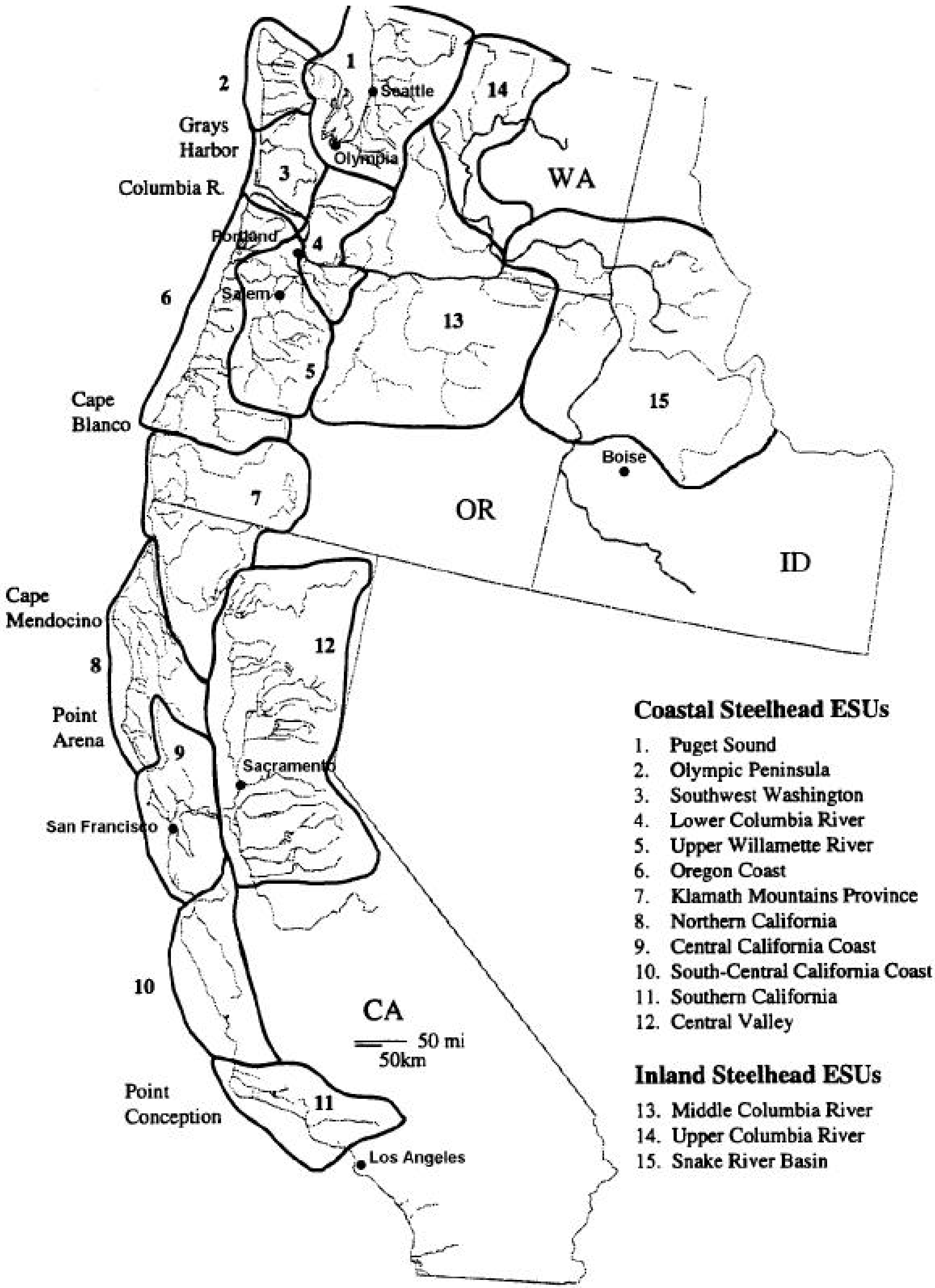

Figure 1. Geographic Distribution of Steelhead ESUs

Factors Contributing to the Decline of Steelhead

I. *The Present or Threatened Destruction, Modification, or Curtailment of Steelhead Habitat or Range.*

A. Hydropower Development

Hydroelectric development has impacted fish stocks in a variety of ways. Construction of dams has blocked access to miles of previously productive habitat. Modification of natural flow regimes by dams has resulted in increased water temperatures, changes in fish community structure, and increased travel time by migrating adult and juvenile salmonids. Physical features of dams such as turbines, have resulted in increased mortality of adults and juvenile salmonids as well. Attempts to mitigate adverse impacts of these structures have to date met with limited success.

Hydroelectric development has substantially reduced the abundance of salmon in the Columbia River Basin and the Pacific Northwest. The Northwest Power Planning Council (NWPPC) has estimated 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 (Northwest Power Planning Council 1987). Approximately half of the 8 million fish loss resulted from curtailment of the fishes range caused by Chief Joseph and Hells Canyon dams in the upper Columbia and Snake rivers, respectively. The remaining 4 million fish loss was attributed to ongoing annual passage losses at and between the mainstem projects below Chief Joseph and Hells Canyon dams. Although the specific number of steelhead lost is unknown, they are included in the overall numbers presented by the NWPPC.

In California, as in the Pacific Northwest, dams which have been constructed on many rivers and streams have adversely impacted anadromous salmonid populations, in particular, steelhead. Most hydroelectric development projects in California have not been required to construct fish bypass facilities; further, projects that have been required to provide fish passage have met with limited success. Dams, such as Copco Dam on the Klamath River, Scott Dam on the Eel River, Shasta Dam on the Sacramento River, Friant Dam on the San Joaquin River, Folsom Dam on the American River, Oroville Dam on the Feather River, Warm Springs Dam and Coyote Dam on Russian River system, Los Padres Dam on the Carmel River, Bradbury Dam on the Santa Ynez River, Robles, Casitas and Matilija dams in the Ventura River system, the Vern Freeman Diversion Facility on the Santa Clara River, Rindge Dam on Malibu Creek and numerous other developments throughout California's Central Valley and coastal rivers, have eliminated or severely hindered access to historical spawning and rearing habitats and have altered the natural flow regimes within the basins.

Juvenile and adult steelhead tend to experience different types of direct and indirect physical impacts as a result of dam configuration and operation. Below we discuss how these two life stages are impacted.

1. Juvenile Steelhead Passage

Juvenile steelhead are subject to increased mortality from passage through dam structures and environmental conditions created by dams such as decreased flow, increased water temperatures, and changes in fish community structure. Sublethal impacts (e.g., stress, injury, descaling, and delay) also occur and can affect survival (Hawkes et al. 1991; Johnsen et al. 1990).

At dams, injury and mortality can occur through all routes of passage (i.e., turbines, ice and trash sluiceways, juvenile bypass systems, adult fish ladders); however, studies have documented that mortalities through turbines are generally higher relative to other routes of passage. Two studies using subyearling fall chinook salmon *Oncorhynchus tshawytscha* showed mean turbine mortalities of 11 to 15 percent (Holmes 1952; Schoeneman et al. 1961).

Screens that deflect a percentage of juvenile steelhead out of turbine intakes and through juvenile bypass facilities have been installed at five of the eight mainstem Columbia River dams through which juvenile steelhead must pass. Juvenile bypass mortalities, excluding outfall mortality, are believed to be in the range of 1 to 3 percent (Brege et al. 1987; Ledgerwood et al. 1990; Monk and Williams 1991).

At dams without screened bypass systems, limited spill is provided on an interim basis to decrease the number of juvenile fish passing through turbines. Spill mortalities have been estimated in the range of 0 to 3 percent at each Columbia River hydroelectric project, though estimated mortality was higher at some projects prior to the implementation of measures to control dissolved gas supersaturation (Columbia Basin Fish and Wildlife Authority 1988).

In reservoirs, the loss of juvenile migrants is related to a number of factors. Low flows during outmigration tend to make juveniles more suspectable to predation due to decreased turbidity and increased time in the river. High water temperatures during the juvenile steelhead migration, which tend to be associated with low flows, also impair steelhead avoidance of predators.

Mortality of juvenile steelhead at dams and in reservoirs may be substantially reduced by collecting juvenile fish at upper river dams, transporting them, and releasing them below the lowest dam. For example, studies using primarily Columbia River summer and fall chinook salmon have estimated that subyearling fall chinook salmon survival for fish transported from McNary Dam to below Bonneville Dam was more than 1.8 times higher than for fish migrating inriver (Matthews et al. 1988).

2. Adult Steelhead Passage.

Cumulative passage losses for adult steelhead passing through dams may be high. From analyses of adult salmon counts in the Columbia and Snake Rivers, it has been estimated that upstream passage results in 3 to 5 percent mortality to adult salmonids (Northwest Power Planning Council 1986). However, this fails to account for an additional unexplained 5 percent mortality which may result from delayed mortality or possibly poaching (Kaczynski and Palmisano 1993). Based on these figures, mortality may range from 5 to 10 percent loss per hydroelectric project on the Columbia and Snake rivers systems (Kaczynski and Palmisano 1993). There were no estimates available for steelhead mortalities associated with California projects.

Delay at dams is an important factor in the survival of adult steelhead. Factors influencing delay include the effectiveness of fish passage facilities, powerhouse and spillway operations, and flow and water quality. Average delay at lower Columbia River mainstem dams may be one to three days when good passage conditions exist (Ross 1983). Delay is important because adult steelhead do not generally feed during their upstream migration. Delays during migration deplete limited energy reserves, which can reduce survival and spawning success.

Delay can be greater when adult passage facilities are not operated in conformance with established criteria (i.e., when reduced hydraulic head and weir depths reduce attraction flows at fishway entrances). Inadequate water velocity inside fishways also increases delay. Adult fishways, on mainstem Columbia River dams, have operated below flow/velocity criteria in one or more areas of the fishways frequently a substantial amount of time (Fish Passage Center of the Columbia Basin Fish and Wildlife Authority 1988, 1989, 1990).

Another important factor to consider regarding steelhead passage is the potential of "fall back" of migrating adults through turbine structures. Research has indicated that mortality rates for steelhead that fall back through turbines may range from 22 to 41 percent (Wagner and Ingram 1973). This may be an important source of mortality for migrating steelhead, however, it has been difficult to quantify the degree to which this has affected steelhead coastwide.

In summary, hydroelectric development and dam operations have to a large degree modified and curtailed steelhead habitat and range. Dam structures through which migrating steelhead are able to pass often contribute to increased mortality through physical injury and delay. Measures implemented to date have failed to effectively mitigate these impacts.

B. Water Withdrawal, Conveyance, Storage, and Flood Control

Depletion and storage of natural flows have drastically altered natural hydrological cycles in the Snake and Columbia River Basins, as well as many rivers and streams in Washington, Oregon, Idaho and California. Alteration of streamflows has resulted in juvenile salmonid mortality for a variety of reasons: migration delay resulting from insufficient flows or habitat blockages; loss of sufficient habitat due to dewatering and blockage; stranding of fish resulting from rapid flow fluctuations; entrainment of juveniles into poorly screened or unscreened diversions; and increased juvenile mortality resulting from increased water temperatures (Bergren and Filardo 1991; California Advisory Committee on Salmon and Steelhead Trout 1988; California Department of Fish and Game 1991; Columbia Basin Fish and Wildlife Authority 1991a; Chapman et al. 1994; Cramer et al. 1995; Palmisano et al. 1993; Reynolds et al. 1993). In addition to these factors, reduced flows negatively affect fish habitats due to increased deposition of fine sediments in spawning gravels, decreased recruitment of new spawning gravels, and encroachment of riparian and non-endemic vegetation into spawning and rearing areas resulting in reduced available habitat.

Within the Columbia and Snake rivers systems, the largest consumptive use of water is agricultural irrigation. In addition to direct diversion of natural flows, agricultural water use promotes and is supported by water storage in Federal and private reservoirs. For example, total discharge of the Snake River on an annual basis is approximately 36 million acre-feet (MA-ft) (44.4 cubic kilometers (km^3)). Of this, approximately 16 MA-ft (19.73 km^3) is diverted annually, and 6 MA-ft (7.4 km^3) is consumed by agriculture (Hydrosphere 1991). Additionally, total active storage (the amount of water that can be removed from a reservoir) in the Snake River Basin above Hells Canyon Dam (including Brownlee Reservoir) is approximately 11.3 MA-ft (13.94 km^3). The amount of active storage available for use varies from year to year depending on run-off and rainfall. This storage alters timing of peak flows in the Snake River, that would under natural conditions, have occurred during the spring/summer run-off, when juvenile salmonids are migrating.

In California, water withdrawal, conveyance and diversion has resulted in the loss of a significant amount of steelhead habitat. Diversion and transfer of water has resulted in depleted river flows necessary for migration, spawning, rearing, flushing of sediment from the spawning gravels, gravel recruitment and transport of large woody debris (Botkin et al. 1995; California Advisory Committee on Salmon and Steelhead Trout 1988; Reynolds et al. 1993).

There are roughly 1,400 Federal, state, and private dams in California at least 25 feet (7.5 meters) high or holding back a minimum of 50 acre-feet (A-ft) (0.06 cubic meters (m^3)) of water (California Department of Water Resources 1982a). One hundred and one of these dams contain 90 percent of the total capacity of all California dams. Including dams on the Colorado River, these dams collectively can impound 42.2 MA-ft (51.8 km^3) of water. Titus et al. (1994) extensively reviewed current freshwater habitat conditions for steelhead populations along the California coast south of San Francisco Bay. They reported blockages in 12 of 46 tributaries within the southern portion of the central California ESU, blockages in 28 of 66 tributaries within the south-central California coast ESU, and that of 32 tributaries within the southern California ESU, 21 have blockages due to dams and 29 have impaired mainstem passage. Recent drought conditions which persisted from the mid-1980's through the early 1990's have shown that there is little water to spare for instream uses in many areas of California.

The Sacramento River Basin covers an area in excess of 22,000 square miles (57,000 square km) in central and northeastern California and is the largest river system in California. Salmon and steelhead spawning and rearing habitat of the Central Valley of California has been reduced from approximately 6,000 miles (9,677 km) that existed prior to the construction of dams to less than 300 miles (484 km) today, a 95 percent reduction in available habitat (California Advisory Committee on Salmon and Steelhead Trout 1988; Reynolds et al. 1993). However, because steelhead utilize habitats located in the upper tributaries of watersheds, and virtually all of the major dams were built in the lower reaches of the rivers, there is probably a greater reduction in available habitat for steelhead populations.

The operations of the Central Valley Project (CVP) and the State Water Project - Harvey O. Banks Pumping Plant (SWP) in the Sacramento-San Joaquin delta region have had a tremendous negative effect on steelhead. The CVP and SWP cause reverse flows in the delta region which delay migration of juvenile and adult steelhead, entrain fish into the pumping facilities, and increase predation at water facilities (California Department of Fish and Game 1991; Reynolds et al. 1993). The SWP and the California Aqueduct more than doubled the capacity to export water south to southern California. Prior to the installation and operation of the SWP Delta

Pumps, Delta water exports were limited to the quantities the Federal pumps could deliver. The addition of the SWP delta pumps, the magnitude of reverse flows across the delta increased, delta outflow decreased, and the concomitant entrainment of salmonids increased (Reynolds et al. 1993). Many of the salmonid populations in the Central Valley showed a dramatic decrease in abundance when the SWP came online.

Water development and flood control projects have dramatically altered the Sacramento and San Joaquin rivers' natural flow regimes and sediment transport characteristics. The rivers of the Central Valley are now managed, unnatural waterways and are operated to make daily deliveries to irrigation districts and to prevent flooding (California Advisory Committee on Salmon and Steelhead Trout 1988). These projects have also had a major influence on the lower reaches of the river and its associated riparian habitat (see loss of riparian vegetation, section 1.c). Federal- and state-funded structures in the Sacramento-San Joaquin Valley include Shasta and Keswick dams, and the Red Bluff Diversion Dam all in the upper Sacramento River, Whiskeytown Reservoir which stores and diverts water from the Trinity River into the Sacramento River, Folsom Dam on the American River, Oroville Dam on the Feather River, Don Pedro Dam on the Tuolumne River, New Melones on the Stanislaus River, Friant Dam on the San Joaquin River, Exchequer Dam on the Merced River, the SWP, and the Tracey Pumping Plant (CVP). These structures cumulatively have had an enormous negative impact on all anadromous salmonid populations (Moyle and Herbold 1989). The rivers in the Sacramento and San Joaquin systems are regulated to the point that high flows below the dams typically occur in late spring and summer during the irrigation season, and low flows occur in the fall, winter, and early spring during the storage season (Reynolds et al. 1993). This flow regime is completely inverse to the conditions in which salmonids evolved in the Central Valley.

Flood control operations at dams can also conflict with efforts to provide migration flows for juvenile steelhead (Hydrosphere 1991). Flood control constraints at Brownlee Reservoir on the Snake River, Idaho, require that 500,000 AF (0.62 km^3) of storage be available by the end of February, and, if necessary, all 980,000 AF (1.20 km^3) of active storage can be evacuated (Bennett et al. 1979). Thus, water that could be used to aid anadromous fish migration is drafted prior to the migration period. Flood control constraints at Dworshak Reservoir on the Snake River have a similar effect on availability of water for juvenile salmonid migration. Adding to the problem is the inclination toward increased production of hydropower at both projects during the winter (Hydrosphere 1991).

The Sacramento River Flood Control Project extends south from Chico Landing and includes a series of levees, weirs, and overflow channels. The Sacramento River Bank Protection Project was designed to protect the flood control system between Chico Landing and Collinsville; today over 150 miles (242 km) of the Sacramento River's banks have been riprapped (California State Land Commission 1993). The Chico Landing to Red Bluff Comprehensive Bank Stabilization Project was designed to control lateral migration of the river in this reach and is about 54 percent complete (Reynolds et al. 1993).

Flood control operations at the Folsom Reservoir on the American River have also impacted steelhead populations. On several occasions releases from the reservoir have been abruptly reduced resulting in excessive mortality of naturally-produced juvenile salmonids. In the spring of 1995, flows were reduced from 18,000 to 8,000 cubic feet per second (cfs) in a four-hour period, resulting in the loss of several hundred thousand salmon, steelhead, and other juvenile fishes (California Department of Fish and Game 1995).

From this, it can be concluded that water diversion, conveyance, withdrawal, and storage for agriculture, industrial, and municipal uses have drastically altered the natural flow regimes throughout the range of steelhead. This has resulted in decreased juvenile and adult steelhead survival during migration, and in many cases, had resulted in the dewatering and loss of important spawning and rearing habitats.

C. Land use activities

High water quality and quantity are essential for survival, growth, reproduction, and migration of individuals composing aquatic and riparian communities. Important water quality elements for aquatic organisms include water temperatures within the migratory range, rearing and emergence needs of fish and other aquatic organisms (Quinn and Tallman 1987; Sweeney and Vannote 1978). Desired conditions for salmonids include an annual abundance of cool (generally less than 68° Fahrenheit (F) (20° Celsius (C)), well oxygenated water, low suspended sediments (Barnhart 1986; Sullivan et al. 1987) and other pollutants that could limit primary production and benthic invertebrate abundance and diversity (Cordone and Kelley 1961; Lloyd et al. 1987).

Numerous studies have been conducted regarding the impacts of land use activities on salmonid habitat in the states of Washington, Oregon, Idaho, and California. Land use activities associated with logging, road construction, urban development, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality. Associated impacts of these activities include the following: alteration of streambank and channel morphology; alteration of ambient stream water temperatures; degradation of water quality; elimination of spawning and rearing habitat, fragmentation of available habitats; elimination of downstream recruitment of spawning gravels and large woody debris; removal of riparian vegetation resulting in increased stream bank erosion; and degradation of water quality (Botkin et al. 1995; Bottom et al. 1985; Brown et al. 1994; Bryant 1994; California Advisory Committee on Salmon and Steelhead Trout 1988; California Department of Fish and Game 1965, 1991, 1994; California State Lands Commission 1993; McEwan and Jackson 1996; Nehlsen et al. 1991; Titus et al. in prep.; Wilderness Society 1993). The loss of channel complexity, pool habitat, suitable gravel substrate, and large woody debris, and other development activities have caused increased sediment input into spawning and rearing areas (Bottom et al. 1985, Forest Ecosystem Management Assessment Team 1993, Higgins et al. 1992, U.S. Forest Service and U.S. Bureau of Land Management 1994a).

1. Logging and Agricultural Activities

Research indicates that activities associated with logging result in habitat simplification of stream channels through sedimentation, channelization, and loss of riparian vegetation, large woody debris, and habitat complexity (Anderson 1971; Bottom et al. 1985; U.S. Forest Service and U.S. Bureau of Land Management 1994b; Wilderness Society 1993). Further, historical practices, such as splash dams, and wide spread removal of beaver dams, log jams and snags from river channels, have adversely modified fish habitat (Bottom et al. 1985). More recently, logging has reduced the amount of instream, large woody debris, resulting in significant impacts to salmonid habitat.

Agricultural practices have also contributed to the degradation of salmonid habitat on the West Coast through irrigation diversions, overgrazing in riparian areas, and compaction of soils in upland areas from livestock (Botkin et al. 1995; Palmisano et al. 1993). Grazing has been identified by Bottom et al. (1985) as having the foremost impact on riparian vegetation in Oregon streams. Livestock grazing patterns in and around riparian areas can alter the vigor, composition, and diversity of natural vegetation. This in turn can affect the site's ability to control erosion, provide stability to stream banks, and provide shade, cover, and nutrients to the stream. Mechanical compaction can reduce the productivity of the soils and cause bank slough and erosion. Mechanical bank damage often leads to channel widening, lateral stream migration and excess sedimentation.

Both logging and agriculture activities result in many similar impacts on salmonid habitat. Major impacts common to both activities include loss of large woody debris, sedimentation, loss of riparian (streamside) vegetation, and loss of habitat complexity, all of which affect water quality and the biotic communities. Summarized below are the effects of these activities on steelhead and their habitat.

a. Loss of Large Woody Debris

Large quantities of downed trees are a functionally important component of many streams and estuaries (Naiman et al. 1992; Sedell and Luchessa 1982; Sedell and Maser 1994; Swanson et al. 1976). Large woody debris influences channel morphology by affecting longitudinal profile, pool formation, channel pattern and position, and channel geometry (Bisson et al. 1987). Downstream transport rates of sediment and organic matter are controlled in part by storage of this material behind large wood (Betscha 1979). Large wood affects the formation and distribution of habitat units, provides cover and complexity, and acts as a substrate for biological activity (Bisson et al. 1987; Sedell and Maser 1994; Swanson et al. 1976). Wood enters streams inhabited by salmonids either directly from adjacent riparian zones or from riparian zones in adjacent non-fish bearing tributaries.

Prior to the 1970's, there was so much debris resulting from poor logging practices that many streams were completely clogged and were thought to have been total barriers to fish migration. As a result, in the 1960's and early 1970's it was common practice among fishery management agencies to remove woody debris thought to be a barrier to fish migration (Bisson and Sedell 1984). However, it is now recognized that too much large woody debris was removed from the streams resulting in a loss of salmonid habitat (Bottom et al. 1985, California Department of Fish and Game 1994). Botkin et al. (1995) reported that the routine, large scale removal of woody debris prior to 1980 had major, long-term negative effects on rearing habitats for salmonids in southern Oregon and northern California. Areas that were subjected to this removal of large woody debris are still limited in the recovery of salmonid stocks; this limitation could be expected to persist for 50 to 100 years following removal of debris. Botkin et al. (1995) also stated that a primary goal in salmonid ecosystem recovery would be correcting the insufficient amount or loss of large woody debris. Botkin et al. (1995) further stated that the most important element of woody debris are large, decay-resistant conifers.

Past and present harvesting practices have eliminated large trees, large logs, and other woody debris from streamside areas which could have otherwise been recruited to the channel. Kreissman (1991) reported that California had lost 89 percent of the state's riparian woodland. There has been an 83 to 90 percent loss of old-growth forests in Douglas Fir regions of Oregon and Washington (Harris 1984; Norse 1990; Spies and Franklin 1988). Kellogg (1992) reported that 96 and 75 percent of the original coastal temperate rainforests in Oregon and Washington, respectively, have been logged. This is particularly a problem for redwood, which takes many decades to decay and could have provided long term benefits to fish habitat and watershed stability. Wilburn (1985) reported that California had lost greater than 85 percent of its coastal redwood *Sequoia sempervirens* forests by the early 1980's. Repeated entries into the riparian zones for sanitation salvage and harvesting under exemptions and emergency notices continue to further limit recruitment of large woody debris (Bisson et al. 1987; Bryant 1980; California Department of Fish and Game 1994). Consequently, there is now very little recruitment of large or other woody debris in most coastal streams needed to replace old logs that have been washed out of the system, buried during flood events, or removed decades ago to provide fish passage. Bottom et al. (1985) and Seddell et al. (1988) reported that large logs are no longer available to replace old logs that are still buried in some stream reaches due to logging in stream side areas.

b. Sedimentation

The U.S. Environmental Protection Agency reporting the results of its assessments found many streams throughout Washington, Idaho, Oregon, and California to be either moderately or severely impacted by increases in water temperature and sedimentation (Edwards et al. 1992). Sedimentation resulting from logging, mining, urban and agricultural activities is a primary cause of habitat degradation in the range of steelhead. Quantitatively, sediment has been identified as the greatest single pollutant in the nation's waters (Barnhart 1986, Poon and Garcia 1982, Ritchie 1972, U.S. Environmental Protection Agency 1988). Suspended solids can have damaging physical and biological effects. Cordone and Kelley (1961) and Herbert and Merkens (1961) reported that suspended sediment occasionally reaches concentrations high enough to directly injure steelhead. Sigler et al. (1984) reported that chronic turbidity in streams during emergence and rearing of steelhead affects the numbers and quality of fish production. In general, effects of sedimentation on salmonids are well documented and include: clogging and abrasion of gills and other respiratory surfaces; adhering to the chorion of eggs; providing conditions conducive to entry and persistence of disease-related organisms; inducing behavioral modifications; entombing different life stages; altering water chemistry by the absorption of chemicals; affecting useable habitat by scouring and filling of pools and riffles and changing bedload composition; reducing photosynthetic growth and primary production; and affecting intergravel permeability and dissolved oxygen levels (Koski and Walter 1978) (Appendix A).

Increased turbidity decreases photosynthesis of aquatic plants and can clog the respiratory surfaces and feeding mechanisms of aquatic animals. Turbidity results when fine silt, part of the overall sediment transport, remains suspended for long periods of time. Turbidity causes light to be scattered and absorbed, reducing light penetration and thus diminishing or even eliminating aquatic plant growth. Loss of aquatic plants leads to the loss of associated snails and aquatic invertebrates and serve as a food source for young fish. Barnhart (1986) reported that deposited sediment directly reduces the carrying capacity of the stream by reducing available rearing habitat; indirectly, sediments reduce the production of invertebrate food resources for rearing juvenile steelhead. The authors concluded that rearing habitats of juvenile salmonids in streams, as well as spawning gravels, require protection from excessive quantities of fine sediments. Chronic turbidity in streams was shown by Sigler et al. (1984) to affect the number and quality of fish produced. Further, high concentrations of fine sediment were demonstrated by Phillips et al. (1975) to result in premature emergence of coho salmon. Turbidity generally reduces feeding by fish even if there is an abundance of prey (Noggle 1978). Some salmonid species have complex reproductive and social behaviors that depend on visual signals which may be obscured in turbid waters (Berg and Northcote 1985).

The steelhead's environment can be impaired by particles deposited as bedload sediment. Bjornn et al. (1977) found significant reductions in the numbers of juvenile steelhead in stream channels where boulders were embedded in sediment. Embedded sediment also decreases the ability of juvenile steelhead to migrate into the substrate during high winter flows to avoid being flushed out of the system. Crouse et al. (1981) reported significant decreases in fish production in streams where cobbles were embedded 80-100 percent and where sediment (2.0 mm or less) composed 26-31 percent (by volume) of the total substrate composition.

Evidence is emerging that stability of spawning gravels may be a critical limiting factor for salmonids. Nawa et al. (1991) found that scour and fill of aggraded stream beds caused by minor storms (two year events) in southwest Oregon was sufficient to cause mortality of salmonid eggs and alevins. Payne and Associates (1989) reported that gravels are extremely unstable in the lower Klamath River tributaries; therefore, mortality of eggs similar to that noted by Nawa et al. (1990) is likely occurring there. California Department of Water Resources (1982b) reported that decreasing stability of spawning gravels due to aggradation was the major cause of declines of salmon runs in the South Fork Trinity River. Smith (1992) reported the loss of several salmon redds in Waddell Creek, California, due to late winter storm's scouring flows.

Sedimentation has also been shown to increase stream temperatures. Hagans et al. (1986) reported water temperatures to be adversely impacted by increased sedimentation of gravels and pools. This impact is caused by (1) the loss of a reflective bottom with darker sediment (as opposed to clean gravels) which stores and transfers heat into the water column from direct solar radiation; and (2) reduced water flow through interstitial gravel spaces increases exposure of the water column to direct solar radiation and thus more heat.

Logging conducted in west coast watersheds prior to the early 1970's induced the damage described above to many coastal streams. Accelerated rates of erosion and sedimentation are a consequence of most forest land management activities. Road networks in many upland areas of the Pacific Northwest are the most important source of management-accelerated sediment delivery to anadromous fish habitats (Forest Ecosystem Management Assessment Team 1993). The sediment contribution to streams from roads is often much greater than that from all other land management activities combined (Gibbons and Salo 1973). Federal lands (Forest Service and Bureau of Land Management) within the range of the northern spotted owl *Stix occidentalis caurina* contain approximately 110,000 miles (177,000 km) of road, with an estimated 250,000 stream crossings (e.g., culverts, wet crossings) in the road network (Forest Ecosystem Management Assessment Team 1993). Road densities on private lands are considered to be higher. Fisk et al. (1966) provided testimony to California's State Interim Committee on Stream and Beach Erosion in 1956 and indicated that over 1,000 miles (1,600 km) of streams in California had been damaged or destroyed by 1956. In 1962, Calhoun and Seeley (1963) found that 33 streams totalling about 55 miles (89 km) were damaged that year. Fisk et al. (1966) reported that surveys on the Garcia River and Redwood Creek, California, showed that the Garcia River was severely to moderately damaged by ongoing logging and road building along 52 (84 km) of 104 miles (168 km) of available salmonid habitat while 68.5 (110 km) of 84 miles (135 km) of habitat in Redwood Creek were similarly damaged. Holman and Evan (1964) estimated all of the 70 miles (113 km) of potential habitat in the Noyo River during the late

1950s had been damaged by past logging activities prior to the mid 1940s. In addition, the U.S. Bureau of Reclamation (1973) surveyed Redwood Creek in Humblodt County, Ten Mile, and Big Rivers in Mendocino County, and Gualala River in Sonoma County, California, and found that all had been negatively affected by logging activities, road building, livestock grazing, or urbanization.

Wetlands, estuaries and lagoons provide critical nursery habitat for all juvenile salmonids migrating to the ocean and are essential to all anadromous salmonids. These critical habitats play an important role as a feeding area for juvenile salmonids and also in their acclimatization to higher salinities (Cooper and Johnson 1992). Loss of these habitats may limit food resources for juvenile salmonids. Therefore, juveniles may tend to migrate to open water at a smaller size and thus be more susceptible to predation (Thom 1991). The ocean survival for juvenile salmonids is greatly increased if rearing fish are able to attain larger size for an extended period in the estuary (Simenstad et al. 1982).

The Oregon and California coasts have a naturally low shoreline/coastline ratio (Bottom et al. 1986). As a consequence, there are few well-developed estuaries and other nearshore rearing areas. Almost all of the west coast estuaries and wetlands that have been scrutinized have been subject to significant degradation. These habitats are subject to degradation from a variety of causes including the following: diking, filling, erosion, artificial breaching, chemical pollution, sewage and livestock runoff, and water withdrawals (surface and ground water extraction). Busby (1991), Hofstra (1983), Puckett (1977), and Smith (1987, 1990) reported that many estuaries remain filled with sediment and debris washed in from upstream areas and are no longer capable of supporting the numbers of salmonid juveniles they once did. Higgins (1991) reported that species diversity dramatically declined in the Eel River estuary and that the estuary decreased considerably in size between 1950 and 1977. The lack of habitat in estuaries due to sedimentation may be forcing juvenile salmonids into the marine environment at a less than optimal size thus reducing their ocean survival (Nicholas and Hankin 1988b).

Variability in ocean conditions and paucity of high quality near-shore habitats makes freshwater environments more crucial for the survival and persistence of steelhead trout in the southern one-half of the steelhead range. Compared to northern areas with more stable ocean conditions and better developed near-shore habitats, steelhead in the southern one-half of their range are more dependent on freshwater environments to achieve larger sizes, which increases probability of marine survival (Pearcy 1992; Shapovalov and Taft 1954).

As streams and pools fill in with sediment, flood flow capacity is reduced and meandering increases, resulting in wider and shallower streams with less structure and undercut banks. Such changes cause decreased stream stability and increased bank erosion which exacerbates existing sedimentation problems. All of these sources contribute to the sedimentation of spawning gravels and filling of pools, wetlands and estuaries used by all anadromous salmonids. Although steelhead are resilient and are capable of tolerating suspended sediments for short periods of time, prolonged exposure to high sediment levels can result in simplification of critical habitats. This can lead to starvation, predation, or reproductive failure of the species.

c. Loss of Riparian Vegetation

Many watersheds in the range of west coast steelhead have been subjected to land use activities which have resulted in the reduction of riparian vegetation. The type and structure of streamside vegetation reflects both climate and the disturbance regime of the area. The reduction in tree shade canopy along with the initial and continued loss of trees adjacent to riparian zones can produce significant increases in water temperatures (Bottom et al. 1985; California Department of Fish and Game 1994; Forest Ecosystem Management Assessment Team 1993). Riparian vegetation protects stream banks from erosion through soil binding by root masses and the presence of ground litter and dense overstory canopy, which impedes the rate of surface runoff (Forest Ecosystem Management Assessment Team 1993). Riparian vegetation promotes deposition of silt as new soil during periods of flood, without which key riparian species such as alders, willows, and cottonwoods could not reproduce. Also, riparian vegetation provides important substrates for aquatic invertebrates, cover for predator avoidance, and resting habitat for many fish species. The dead organic matter or detritus from the riparian vegetation is an important source of nutrients to streams, estuaries, and the marine environments. Riparian vegetation that is carried from upland areas and deposited in estuaries is a major source of food and habitat for obigatory, wood-boring marine invertebrates which break down and pass usable carbon into the water's current where it enters the detrital-based marine food web (Sedell and Maser 1994). As much as 99 percent of the annual energy input, the food base for all aquatic communities, comes from riparian vegetation (Reynolds et al. 1993). Removal of streamside vegetation simplifies channel banks and destroys shelter for rearing steelhead, simplifies channel shape so that are fewer pools and riffles, and eventually leads to a widening of channels that are more prone to warming by sunlight (Botkin et al. 1995; California Advisory Committee on Salmon and Steelhead Trout 1988; California Department of Fish and Game 1994).

Reduction in tall tree shade canopy brought about by the initial and continued logging of stands adjacent to and in riparian zones, has resulted in significant increases in water temperatures (Bisson et al. 1987; California Department of Fish and Game 1994; Forest Ecosystem Management Assessment Team 1993). The shading effect of riparian vegetation along fish-bearing and smaller tributary streams provides significant temperature-moderating effects to the adjacent watercourses. Such moderation in temperatures can determine the suitability of rivers and streams for anadromous salmonids. The lack of, or removal of, shading along streams can increase water temperature by 11.7 to 18° F (7.8 to 11.3° C) (Reynolds et al. 1993). Shading can also significantly diminish daily temperature variations in streams. Many Pacific North American watersheds supporting anadromous salmonids have been logged more than once resulting in cumulative removal of most of the original dense conifer overstory covering streams.

High stream temperatures become a chronic problem for fish populations due to a lack of adequate shading. Temperature increases can shift ecological relationships allowing fish species, such as sunfish, suckers, dace, squawfish, and shiners, to become numerically dominant over salmonids (Higgins et al. 1992; Reeves, 1985). Water temperatures in many streams throughout California and the Pacific Northwest are now approaching upper lethal temperatures for anadromous salmonids. A review of U.S. Geological Survey gauging station data for selected major river basins throughout the west coast (Busby et al. 1996) indicates that summer water temperatures have increased in nearly all basins. Kubicek (1977) and the U.S. Fish and Wildlife

Service (1991) reported that main river channels have become increasingly unsuitable for all salmonids during the summer months due to high stream temperatures. Removal of riparian vegetation can also make streams in interior areas more subject to freezing and anchor-ice formation during winter months (Higgins et al. 1992).

Cumulative effects of past and present human activities have degraded aquatic and associated riparian systems substantially. Activities such as mining, timber and fuelwood harvesting, channelization, dam and levee construction, bank protection, and stream flow regulation have altered the riparian system and contributed to vegetation loss (Reynolds et al. 1993). As a result, few high-quality aquatic ecosystems remain in the United States. In 1982, the U.S. National Park Service completed the "Nationwide Rivers Inventory" and found that of 3.25 million stream miles (5.24 million km) examined in the lower 48 states, less than 2 percent were considered of "high natural quality" (Behnke 1990). Edwards et al. (1992) reported that approximately 55 percent of the 27,000 stream miles (43,000 km) in Oregon are either severely or moderately impacted by nonpoint-source pollution. Historically, the Sacramento River was bordered by up to 500,000 acres of riparian forest, with bands of vegetation spreading four to five miles (6 to 8 km) wide. In the last century and one-half, agricultural conversion and urbanization have been the primary factors eliminating riparian habitat (California State Lands Commission 1993; Forest Ecosystem Management Assessment Team 1993; Reiner and Griggs 1989). Conversion of riparian woodlands by agriculture and urbanization in California's Central Valley has reduced the present habitat to less than 2 percent of the original acreage (Nature Conservancy 1990). Martin (1986) reported a 99.9 percent loss of Central Valley riparian oak forest.

The phenomenon of diminishing aquatic ecosystem quality is not limited to riverine environments. Dahl (1990) and Tiner (1991) reported that between the 1780's and 1980's, the lower 48 states lost approximately 53 percent of all wetlands. Wetlands in Washington and Oregon have diminished in area by over 33 percent (Dahl 1990) and many of the remaining wetlands are degraded. The riparian wetland habitat within California's Central Valley has been reduced by about 95 percent and 91 percent statewide (Barbour et al. 1991; Dahl 1990; Jensen et al. 1990; Reynolds et al. 1993). However, most of these studies only examined wetland loss and did not assess the health of those remaining, thus, the actual area of high quality wetlands may likely be much lower than the total reported acres.

d. Loss of Habitat Complexity and Connectivity

In Pacific Northwest and California streams, habitat simplification has lead to a decrease in the diversity of anadromous salmonid species complex (Bisson and Sedell 1984; Hicks 1990; Li et al. 1987; Reeves et al. 1993). Habitat simplification may result from various land-use activities, including timber harvest, grazing, urbanization (California State Lands Commission 1993; Forest Ecosystem Management Assessment Team 1993; Frissel 1992) and agriculture (Forest Ecosystem Management Assessment Team 1993). Timber harvest and range management activities can result in a decrease in the number and quality of pool habitats (Sullivan et al. 1987). Reduction of wood in the stream channel, either from past or present activities, generally reduces pool quantity and quality, alters stream shading which can affect water temperature regimes and nutrient input, and can eliminate critical stream habitat needed for both vertebrate and invertebrate populations. Removal of vegetation also can destablize marginally stable slopes by increasing the subsurface water load, lowering root strength, and altering water flow patterns in the slope (Forest Ecosystem Management Assessment Team 1993). Skid trails, logging roads, and road crossings can be direct sources of sediment to the stream and can act as direct conduits for water yield and sediment from other sources. Constricting channels with culverts, bridge approaches, and streamside roads can reduce stream meandering, partially constrict or channelize flows, reduce pool maintenance, and can preclude passage of anadromous salmonids (Forest Ecosystem Management Assessment Team 1993).

Diverse habitats support diverse species assemblages and communities. This diversity contributes to sustained production and provides stability for the entire ecosystem. Further, habitat diversity can also mediate biotic interactions such as competition (Hartman 1965) and predation (Schlosser 1988). Attributes of habitat diversity include a variety and range of hydraulic parameters (Kaufman 1987), abundance and size of wood (Bisson 1987), and variety of bed substrate (Sullivan et al. 1987).

A primary characteristic of high quality aquatic ecosystems is an abundance of large pool habitats. In many tributaries within the range of steelhead, the number of large, deep pools have decreased. In National Forests within the range of the northern spotted owl in western and eastern Washington, there has been a 58 percent reduction in the number of large, deep pools (Forest Ecosystem Management Assessment Team 1993). A similar trend has been observed in streams on private lands in coastal Oregon, where large, deep pools have decreased by as much as 80 percent (Forest Ecosystem Management Assessment Team 1993). Primary reasons for the loss of pools are: filling by sediments (Megahan 1982), loss of pool-forming structures such as boulders and large wood (Sullivan et al. 1987), and loss of sinuosity by channelization (Benner 1992; Furniss et al. 1991).

An important consideration for maintaining aquatic and riparian ecosystem functions is the degree of spatial and temporal connectivity within and between watersheds (Naiman et al. 1992). Lateral, vertical, and drainage network linkages are critical to aquatic system function. Important connections within basins include linkages among headwater tributaries and downstream channels as paths for water, sediment and disturbances (Forest Ecosystem Management Assessment Team 1993). Further, linkages among floodplains, surface water, and ground water systems as exchange for water, sediment and nutrients are also important (Forest Ecosystem Management Assessment Team 1993). Unobstructed physical and chemical paths to areas critical for fulfilling life-history requirements of aquatic and riparian dependent species must also be maintained to ensure ecosystem stability. Logging, agricultural, and urbanization activities have reduced the connectivity of aquatic and riparian habitats, thereby decreasing and limiting the relative diversity of numerous salmonid-producing watersheds.

2. Mining

The impacts of historical mining operations are evident in many streams of the Pacific Northwest and California. Past mining activities routinely resulted in the removal of spawning gravels from streams, channelization of streams from dredging activities, and leaching of toxic effluents into streams (Bottom et al. 1985; California State Lands Commission 1993). Many of the effects of past mining operations still impact steelhead habitat today. Current mining practices include suction dredging, placer mining, lode mining and gravel mining. Present day mining practices are typically less intrusive than historic operations (hydraulic mining); however, adverse impacts to salmonid habitat still occur as a result of present-day mining activities.

Sand and gravel are used for a large variety of construction activities including base material and asphalt, road bedding, drain rock for leach fields, and aggregate mix for buildings and highways. Since the end of World War II, rapid human population growth and the consequent construction boom has maintained high demand for aggregate material. In 1986, the production of sand and gravel in California alone, primarily derived from river channels and their flood plains, was estimated at 128.5 million tons (116 million kilograms (kg)) with an estimated value of nearly $500 million (Sandecki 1989); nearly double the estimated production of 65 million tons (59 million kg) in 1955 (California State Lands Commission 1993).

Most aggregate is derived principally from pits in active flood plains, pits in inactive river terrace deposits, or directly from the active channel. Other sources include hard rock quarries and mining from deposits within reservoirs. Extraction sites located along or in active flood plains present particular problems for anadromous salmonids. Physical alteration of the stream channel may result in the destruction of existing riparian vegetation and the reduction of available area for seedling establishment (California State Lands Commission 1993). As discussed previously, loss of vegetation impacts riparian and aquatic habitat by causing a loss of the temperature-moderating effects of shade and cover, and habitat diversity. Extensive degradation may induce a decline in the alluvial water table, as the banks are effectively drained to a lowered level, affecting riparian vegetation and water supply (Woodward-Clyde Consultants 1976). Altering the natural channel configuration will reduce salmonid habitat diversity by creating a wide, shallow channel lacking in the pools and cover necessary for all life stages of anadromous salmonids.

There are no accurate records of the number of inactive mines on the West Coast; however, many abandoned mines contribute toxic substances into rivers and streams. In many cases, past hydraulic and explosive mining have exposed rock and metal ores to weathering conditions which has resulted in the formation of acidic compounds (California State Lands Commission 1993). Further, waste products resulting from past and present mining activities, include cyanide (an agent used to extract gold from ore), copper, zinc, cadmium, mercury, asbestos, nickel, chromium, and lead. These products are extremely hazardous to both humans and aquatic life. An example of an inactive mine that is causing severe impacts to water quality is the Iron Mountain Mine in the Sacramento River watershed, California (California Department of Fish and Game 1995). Toxic substances are released directly into the Sacramento River, and water stored in an upstream reservoir (Shasta Reservoir) must be released to dilute toxic spills from the mine (California Department of Fish and Game 1995). These water releases could provide alternative benefits, such as for migration flows, temperature reduction, or high delta outflows if the releases were not required for mitigating mining impacts (California Department of Fish and Game 1995).

3. Urbanization

Urbanization has led to degraded steelhead habitat through stream channelization, floodplain drainage, and riparian damage (Botkin et al. 1995). The distribution of large floods over time reflects the precipitation and runoff region of the watershed, and large floods are natural and necessary for the drainage of the watershed and maintenance of the river channel. When watersheds are urbanized, problems may result simply because structures are placed in the path of natural processes, or because the urbanization itself has induced changes in the hydrologic regime, which in turn impact structures. Point source (PS) and nonpoint source pollution (NPS) occurs at almost every point that urbanization activity influences the watershed. Impervious surfaces (i.e. concrete) reduce water infiltration and increase runoff, thus creating greater flood hazard (Leopold 1968). Flood control and land drainage schemes may increase the flood risk downstream by concentrating runoff. A flashy discharge pattern results in increased bank erosion with subsequent loss of riparian vegetation, undercut banks and stream channel widening. Sediments washed from the urban areas and deposited in river waters include trace metals such as copper, cadmium, zinc, and lead (California State Lands Commission 1993). These, together with pesticides, herbicides, fertilizers, gasoline, and other petroleum products, contaminate drainage waters and destroy aquatic life necessary for steelhead survival. The

California State Water Resources Control Board (1991) reported that NPS pollution is the cause of 50 to 80 percent of impairment to water bodies in California.

In most western states, about 80 to 90 percent of the riparian habitat has been eliminated. While historical uses of riparian areas (e.g., fuelwood cutting, clearing for agricultural uses) have substantially decreased, urbanization still poses a serious threat to remaining riparian areas. Riversides are desirable places to locate homes, businesses, and industry. Further, development within the flood plain results in vegetation removal, stream channelization, habitat instability, and point and nonpoint source pollution.

II. *Over-utilization.*

A. Commercial, Recreational, and Tribal Harvest.

Historically, steelhead were abundant in many western coastal and interior streams of the United States and have supported substantial tribal and sport fisheries, contributing millions of dollars to numerous local economies (Nickelson et al. 1992). Over-fishing in the early days of the European settlement led to the depletion of many stocks of salmon and steelhead even before extensive habitat degradation. However, following the degradation of many west coast aquatic and riparian ecosystems, exploitation rates may have been higher than populations could sustain. Therefore, harvest may have contributed to the further decline of some populations.

Steelhead are harvested by both non-Indian and Indian fishermen in Washington and Oregon. Prior to 1974, steelhead were primarily harvested in recreational fisheries. After the mid-1970's, two federal court rulings governed the allocation between these two groups. United States v. Washington[9] set the harvest allocation criteria for the Puget Sound and the Washington coast north of Willapa Bay. United States v. Oregon[10] set the management criteria for runs returning upstream of Bonneville Dam in the Columbia River system. Only runs returning to Columbia River tributaries downstream of Bonneville Dam, streams entering Willapa Bay, and certain

[9] United States v. Washington, 443 U.S. 658 (1979).

[10] United States v. Oregon, 657 F.2d 1009 (9th Cir. 1981).

Indian reservations are not covered by these decisions or are not subject to sharing between the parties (WDFW 1995).

The Northwest Power Planning Council (1986) estimated that the aboriginal catch ranged from about 4.5 to 5.6 million salmon and steelhead annually in the Columbia River Basin. The harvest of steelhead in the Columbia River peaked in 1892 at over 4.9 million pounds (19 million kg) (WDF and ODFW 1994). Commercial harvest of steelhead has been limited to Native Americans in the Columbia River since 1975, and in the Pacific Southwest since 1924 (Barnhart 1986, WDF and ODFW 1994). Tribal harvest of steelhead at Celilo Falls on the Columbia River ranged from 25,000 to 60,000 fish annually from 1938 to 1974 (WDF and ODFW 1994). During the 1980's, Native American commercial catches in the Columbia River ranged from 10,000 to 70,000 while ceremonial and subsistence fisheries have ranged from less than 1,000 to about 10,000. As recent as 1993, the Native American tribal fishery harvested a total of about 27,500 steelhead during the winter and fall treaty commercial fishing seasons in the Columbia River (WDF and ODFW 1994). This was a decrease from a harvest of about 50,000 in 1992.

In California, steelhead are taken during the Yurok tribe's fall chinook salmon subsistence fishery in the Klamath River. From 1984 through 1992, an estimated 2,350 steelhead were captured, with a range of 472 in 1984 to 68 in 1992, and an estimated mean of 260 steelhead per year (Craig and Fletcher 1994). No data was available from the Hoopa or Klamath tribes net fisheries in the Klamath Basin.

Steelhead have remained important fisheries for recreational purposes. In the Columbia River recreational fishery, the majority of sport catch occurs in the tributaries located in Oregon and Washington. Most tributaries are limited to hatchery-marked steelhead harvest only. Sport catch of upriver steelhead in Oregon, Washington and Idaho has increased from 21,700 in 1979 to an average of about 90,000 since 1985, while sport catch of winter steelhead in Oregon and Washington has ranged from about 70,000 in 1980 to about 25,000 in 1994 (WDF and ODFW 1994). The combined Oregon, Washington, and Idaho sport catch of steelhead in the Columbia River and tributaries from the 1992-93 run was about 150,000, the highest on record (WDF and

ODFW 1994).

Estimates of steelhead catch in California's rivers by sport fishers are estimates based on limited monitoring. In the early 1960's, there was an estimated harvest of 122,000 adult steelhead per year and an unknown quantity of harvested juvenile steelhead (California Department of Fish and Game 1965). Harvest rate estimates for the Klamath River for the 1977-78 through the 1982-83 seasons ranged from 7.4% to 19.2% and averaged 12.1%. Harvest rates for wild steelhead are similar: wild steelhead in the Trinity River were harvested at rates of 28.0%, 12.5% and 17.3% in 1978-79, 1980-81, and 1982-83 seasons, respectively (DFG, unpublished data), and sport harvest of wild steelhead in the South Fork Trinity River was estimated to be 5.9%, 18.0%, 8.0%, and 20.2% during the 1988-89, 1989-90, 1990-91, and 1991-92 seasons, respectively (Mills and Wilson 1991, Wilson and Mills 1992, Wilson and Collins 1992, Collins and Wilson 1994). The average estimated harvest rate on adult steelhead above Red Bluff Diversion Dam on the Upper Sacramento River for the three year period 1991-92 through 1993-94 was 16.0% (DFG, unpublished data).

In 1991, there were an estimated 99,700 steelhead anglers in California (California Department of Fish and Game 1991). The California Department of Fish and Game (CDFG) has just recently (as of 1993) developed and required a "steelhead catch report card." Approximately 77,500 report cards were purchased by steelhead anglers in 1993, and sales of the 1994 report card were about the same. Preliminary results for 1993 show that an estimated 40,500 steelhead were harvested state-wide in California, with 71 percent of the effort occurring along the northern California coast, primarily in the Smith, Klamath, Trinity, and Mad Rivers (T. Jackson pers. comm.). Sport fishing catch rates are low everywhere in the state indicating declining steelhead population numbers, irrespective of reliable steelhead population estimates (McEwan and Jackson 1996).

Illegal harvest can be a serious problem for salmonids on their spawning beds and on their summer rearing/holding habitats. Roelofs (1983) cited poaching as a serious problem on summer steelhead in northern California streams. Large portions of spring run chinook salmon and summer steelhead runs often congregate in just a few cool pools, increasing their vulnerability to poaching. Rivers considered to have a serious poaching problem include most tributaries of the North Umpqua, Klamath and Trinity, and the Middle Fork Eel Rivers, Redwood Creek, several tributaries of the Sacramento River, and several coastal rivers south of San Francisco Bay.

B. Scientific Utilization.

Fishery agencies and Native American tribes, in cooperation with the U.S. Army Corps of Engineers (COE), the Bonneville Power Administration (BPA), and others, annually conduct a coordinated program to monitor the downstream migration of natural and hatchery produced juvenile salmonids in the Columbia River Basin and in coastal Washington and Oregon. Direct sampling is conducted on a daily basis at the upper Snake River and Clearwater River traps, Lower Granite Dam, Little Goose Dam, and Lower Monumental Dam on the Snake River, and McNary Dam, John Day Dam, The Dalles Dam, and Bonneville Dam on the Columbia River. Data collected during this sampling are used to monitor bypass performance and to manage the water budget, spill, and fish transportation programs. Until recently, sampling of juvenile salmonids at Lower Granite, Little Goose, and McNary Dams has been limited to 8 percent of the total outmigration, providing an additional 2 percent of the outmigration for remaining sampling sites (Columbia Basin Fish and Wildlife Authority 1991b). At the present, outmigrant sampling has been decreased and limited to 3 percent of the total outmigration at all sites (CBIT and SFFWA 1993). While this sampling may result in the delay and handling of juveniles, it does not necessarily result in the mortality of those individuals sampled (USFWS 1995).

In California, most of the scientific collection permits are issued to environmental consultants, federal resource agencies, and universities by the CDFG. The Department controls scientific utilization of steelhead through the issuance of Scientific Collector's Permits. Take of steelhead in excess of sportfishing limits is prohibited unless specifically authorized in the permit. Regulation of take is controlled by conditioning individual permits. The CDFG does require reporting of any steelhead trout taken incidental to other monitoring activities; however, no comprehensive total or estimate of steelhead mortalities related to scientific sampling are kept for any watershed or steelhead stock in the state (D. McEwan pers. comm.). The CDFG does not believe that indirect mortalities associated with scientific utilization have had, or are having, a detrimental effect on any steelhead population in California (D. McEwan pers. comm.).

C. Ocean Harvest

Steelhead are not generally caught by commercial or recreational fishers in the ocean. However, although little documented evidence exists, high seas driftnet fishing has been implicated as a cause for decline of steelhead from coastal streams along the North American Pacific Coast (Light et al. 1988). Observations of returning steelhead to Rowdy Creek Fish Hatchery on the Smith River in 1992 showed healed gillnet scars on 30 of 155 adults (Higgins et al. 1992), and many of the observed returning adults to several Santa Cruz County streams have also shown a high incidence of gillnet scars (D. Streig pers. comm. 1995).

Based on recoveries of marked and tagged North American steelhead, high seas steelhead distribution and driftnet fisheries overlap (Light et al. 1988, Burgner et al. 1992). The recent decline in steelhead abundance may be partially attributed to the harvest of steelhead in high seas driftnet fisheries (Anonymous 1989 cited in Cooper and Johnson 1992). Cooper and Johnson (1992) reported that the authorized Japanese mothership salmon driftnet fishery was largest between 1973-1977 when a total of 21.4 million salmon were harvested per year. The authorized Japanese land-based salmon fishery harvested 30.2 million fish per year, which included a steelhead catch ranging from 2,761 in 1990 to 28,900 in 1983 (Cooper and Johnson 1992). However, Cooper and Johnson (1992) estimated that less than 1 percent of the total salmonid catch in both fisheries were steelhead.

In the past, an authorized high seas squid fishery was operated by Japan, the Republic of Korea, and Taiwan. Benton (1990, as cited in Cooper and Johnson 1992) estimated that by 1988, approximately two million miles (3.2 million km) of squid driftnet were set per year. Pella et al. (1991) estimated the salmonid bycatch and the number of salmonid dropouts during net retrieval in the 1990 authorized squid fishery. They reported that the Japanese bycatch was 210,000 fish caught and 21,000 dropout fish, with the estimated mean steelhead harvest portion of 9,200 fish or roughly 4 percent of the total bycatch. The Korean and Taiwanese steelhead bycatch was estimated at 35 and two steelhead harvested, respectively. The combined authorized high seas driftnet fisheries caught less than 3 percent of the estimated 1.6 million steelhead adults that return to the Pacific coast of North America from 1983 through 1990 (Light 1987, Cooper and Johnson 1992, Burgner et al. 1992). Japan, with largest North Pacific driftnet fleet, ceased such activities in May of 1992. Furthermore, the United Nations continues efforts to halt drift gillnet fishing by South Korea and Taiwan.

Unauthorized fishing on the high seas may result in a substantial level of salmonid mortality (Pella et al. 1991, Cooper and Johnson 1992). Cooper and Johnson (1992) reported that a total of 71 and 165 foreign vessels were observed outside authorized fishing areas in 1990 and 1991, respectively. It was estimated that the unauthorized high seas driftnet fisheries harvest between 2 percent (32,000) and 28 percent (448,000) of the steelhead that are destined to return to the Pacific coast of North America (Cooper and Johnson 1992). However, even if the high seas driftnet fisheries harvested a combined 31 percent (3 percent authorized and 28 percent unauthorized) of the steelhead, the greater than 50 percent decline in North American steelhead runs observed between 1986 through 1991 cannot be solely attributed to this fishery (Cooper and Johnson 1992).

III. *Disease or Predation.*

A. Disease.

Infectious disease is one of many factors which can influence adult and juvenile survival. Steelhead are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environments. Specific diseases such as bacterial kidney disease (BKD), ceratomyxosis, columnaris, Furunculosis, infectious hematopoietic necrosis (IHNV), redmouth and black spot disease, Erythrocytic Inclusion Body Syndrome (EIBS), and whirling disease among others are present and are known to affect steelhead and salmon (Rucker et al. 1953, Wood 1979, Leek 1987, Foott et al. 1994, Gould and Wedemeyer undated). Very little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases for steelhead.

However, studies have shown that native fish tend to be less susceptible to pathogens then hatchery-reared fish (Buchanon et al. 1983, Sanders et al. 1992).
Wild steelhead may contract diseases which are spread through the water column (i.e., waterborne pathogens) (Buchanan et al. 1983). Disease may also be contracted through interbreeding with infected hatchery fish (Fryer and Sanders 1981, Evelyn et al. 1984 and 1986). A fish may be infected yet not be in a clinical disease state with reduced performance. Salmonids typically are infected with several pathogens during their life cycle. However, high infection (number of organisms per host) and stressful conditions (crowding in hatchery raceways, release from a hatchery into a riverine environment, high and low water temperatures, etc.) usually characterize the system before a disease state occurs in the fish.

Recently, USFWS and CDFG monitored the health and physiology of natural and hatchery chinook salmon and steelhead trout in the Klamath and Trinity River basins (Foott et al. 1994). The bacterium, *Renibacterium salmoninarum*, causative agent of BKD, and the trematode parasite, *Nanophyetus salmincola*, were identified as the most significant pathogens affecting both natural and hatchery smolt health in the basin. Natural steelhead smolts in the Trinity River were found to have a incidence of *R. salmoninarum,* much higher than in the Trinity River hatchery stock. In 1991, over twice the percentage of natural steelhead (21 percent) as hatchery fish (10 percent) were found to be infected with *R. salmoninarum* (Foott et al. 1994). Sampling conducted in 1992 showed that 53 percent of the natural steelhead tested positive for the bacterium. This is among the highest positive percentage ever recorded; however, no signs of clinical BKD were observed in either hatchery or natural steelhead.

It is possible that steelhead can tolerate *R. salmoninarum* infection better than chinook or coho salmon (Foott et al. 1994). Many of the natural and hatchery steelhead populations throughout California's coast and central valley have tested positive for *R. salmoninarum* (Foott 1992). The impacts of BKD disease are subtle. Juvenile salmonids may survive well in their journey downstream, but may be unable to make appropriate changes in kidney function for a successful transition to sea water (Foott 1992).

Stress during migration may also cause this disease to come out of remission (Schreck 1987). Passage through dams, bypasses, and spillways increase physiological stress and physical injury in migrating juvenile salmonids (Matthews et al. 1986, Maule et al. 1988); in turn, this may increase the susceptibility of migrating salmonids to pathogens (Maule et al. 1988). Adult steelhead are also subject to increased stress from fallback at dams, crowding in ladders, and skin abrasion. Another critical factor in controlling disease epidemics is the presence of adequate water quantity and quality during late summer. As water quantity and quality diminishes, and freshwater habitat becomes more degraded, many previously infected salmonid populations may experience large mortalities since added stress can trigger the onset disease. These factors, in combination with high water temperatures common in various rivers and streams, may increase anadromous salmonid susceptibility and exposure to diseases (Holt et al. 1975, Wood 1979).

Until the late 1970's, *Ceratomyxa shasta* was thought to be confined to waters below the Deschutes River (Wood 1979). Recent investigations on adult summer chinook salmon indicate that upper Snake River waters are also infected (Chapman 1986). Operational problems associated with *C. shasta* began to occur shortly after the opening of Iron Gate Hatchery, located on the Klamath River in California (CH2M Hill 1985). Periodic outbreaks of this parasite continued into the early 1980's (CH2M Hill 1985). *C. shasta* is often found in reservoir environments (Wood 1979); therefore, impounding of the upper Columbia, Snake, and Klamath Rivers may have contributed to the spread of the parasite.

In many cases, disease outbreaks have occurred as a result of introduced, non-native steelhead populations susceptible to disease (KRBFTF 1991). High straying rates of non-native fish exacerbate the situation by spreading pathogens throughout the native community (KRBFTF 1991). In the early 1970's, many Trinity River Hatchery steelhead strayed to the Iron Gate Hatchery. Excess steelhead adults in the Iron Gate Hatchery were then transferred to Shasta, Scott, and other small Klamath River tributaries (KRBFTF 1991). Carlton (1989) found that chinook salmon at Iron Gate Hatchery had a 4 percent susceptibility to *C. shasta* while the Trinity River Hatchery chinook salmon had roughly a 12 percent susceptibility. Hubbell (1979) found similar resistance of Iron Gate Hatchery steelhead and Trinity River Hatchery steelhead (12 percent). Hendrickson et al. (1989) reported that *C. shasta* was endemic to the Klamath River, however, Foott et al. (1994) rarely found either spores or other life stages of *C. shasta* in fish collected in the Klamath River.

B. Freshwater Predation.

Predation on juvenile salmon has increased as a result of water development activities which have created ideal habitats for predators and non-native species. Turbulent conditions near dam bypasses, turbine outfalls, water conveyances, and spillways disorient juvenile steelhead migrants and increase their avoidance response time, thus improving predator success (Sigismondi and Weaver 1988). Increased exposure to predators has also resulted from reduced water flow through reservoirs; a condition which has increased juvenile travel time (Columbia Basin Fish and Wildlife Authority 1991a). For example, Northern squawfish (*Ptychocheilus oregonensis*) and avian predator populations have increased with the formation of ideal predator foraging areas created by dam impoundments. Results from numerous studies indicate that in many reservoirs, northern squawfish are the primary predator of juvenile salmon.

Predators such as walleye (*Stizostedion vitreum*), smallmouth bass (*Micropterus dolomieui*), channel catfish (*Ictalurus punctatus*), and northern squawfish (*Ptychocheilus oregonensis*) have been found to consume significant numbers of juvenile salmon. In the Columbia and Snake Rivers, these predators have been found to consume between 9 and 19 percent of the juvenile salmonids entering reservoirs, with northern squawfish accounting for approximately 78 percent of this loss (Rieman et al. 1991). Squawfish consumption rate tends to be highest during the summer months, which coincide with the juvenile steelhead migration (Poe et al. 1988). Several studies have documented squawfish population increases in the Columbia and Snake River steelhead migration corridor. The estimated squawfish population in the upper half of Lower Monumental reservoir increased from 120,000 in 1975 to 133,000 in 1976 (Sims et al. 1978), and from 68,947 in 1984 to 102,888 in 1986 in the John Day pool (Beamsderfer and Rieman 1988). Lynch (1993) estimated squawfish abundance near The Dalles Dam tailrace and cul-de-sac area to range from 160,000 to 1.7 million in 1991 and from 150,000 to 500,000 in 1992.

The Bonneville Dam forebay squawfish population was estimated in 1980 at between 6,701 and 23,700 individuals (Uremovich et al. 1980) and again in 1989 at between 43,302 and 108,960 (NMFS unpublished).

Sacramento squawfish (*Ptychocheilus grandis*) is a species native to the Sacramento River Basin and has evolved with the anadromous salmonids in this system. However, rearing conditions in the Sacramento River today (e.g., warm water, low-irregular flow, standing water, diversions) compared to its natural state and function 70 years ago, are more conducive to warmwater species such as Sacramento squawfish and striped bass than native salmonids. In the early 1980's, an illegal introduction of Sacramento squawfish occurred in the Eel River Basin via Pillsbury Lake. Today, in little over a decade, Sacramento squawfish have spread to most areas of the Eel River Basin illustrating the fact that this species is better adapted than native salmonids to the artificially warm water conditions. As a result, Sacramento squawfish have been found to constitute a serious problem for native salmonid populations (Higgins et al. 1992, California Department of Fish and Game 1994). If increased water temperature and altered ecosystem trends continue, a shift towards the dominance of warmwater species can logically be expected (Reeves 1985).

In addition to the predators mentioned above, striped bass (*Marone saxatilis*) are often thought to be a significant predator of juvenile salmonids. Around the turn of the Century, striped bass were introduced into the Sacramento River as a forage and recreational fishery. Attempts to plant striped bass in several California coastal tributaries have been unsuccessful. Presently, striped bass abundance is quite low relative to the earlier part of this century; however, striped bass are distributed throughout the California Aquaduct system and associated reservoirs and have been noted in Lake Mendocino and the Russian River system. There are no reliable data available regarding predation rates of striped bass on any steelhead trout population in California (D. McEwan pers. comm.).

In addition to predation by freshwater fish species, avian predators have also been shown to impact juvenile salmonids. Such predation may occur in freshwater areas as well in nearshore marine environments. Ruggerone (1986) estimated that ring-billed gulls (*Larus delawarensis*) consumed 2 percent of the salmon and steelhead trout passing Wanapum Dam during the spring smolt outmigration in 1982. Wood (1987) estimated that the common merganser (*Mergus merganser*), a known freshwater predator of juvenile salmonids, were able to consume 24 to 65 percent of coho salmon production in coastal British Columbia streams. Known avian predators in the nearshore marine environment include herons and diving birds such as cormorants and alcids which include auklets, murres, murrelets, guillemots, and puffins (Allen 1974). Manuwal (1977) estimated that in Washington, about 5 percent of auklet prey biomass was juvenile salmon. Further, Mathews (1983) found that the common murre can consume several smolts per day. With the decrease in riverine and estuarine habitat quality, increased predation by avian predators will occur. Salmonids and avian predators have co-existed for thousands of years, but with the decrease in avoidance habitat (e.g., deep pools and estuaries, large woody debris, and undercut banks), avian predation may play a role in the reduction of some localized steelhead stocks. However, Botkin et al. (1995) stressed that overall predation rates on steelhead should be considered a minor factor for their decline.

C. Marine Predation.

The NMFS has noted that predation by marine mammals in some Northwest salmonid fisheries has increased as marine mammal numbers, especially harbor seals and California sea lions increased on the Pacific Coast (NMFS 1988). Harvey (1988) noted that harbor seal numbers on the Oregon coast had increased at a rate of 6 to 8.8 percent per year between 1975 and 1983. In 1990, 19.2 percent of the adult spring and summer chinook salmon observed in the Snake River at Lower Granite Dam exhibited wounds attributable to marine mammals, primarily harbor seals (Harmon et al. 1989). Prior to 1990, injury of adult salmonids resulting from marine mammal attack was thought to be on the order of a few percent annually (NMFS 1988).

Botkin et al. (1995) reported that marine mammal predation on anadromous salmonid stocks in southern Oregon and northern California was only a minor factor for their decline. In California at the mouth of the Russian River, Hanson (1993) reported that the foraging behavior of California sea lions and harbor seals with respect to anadromous salmonids was minimal.

Hanson (1993) also stated that predation on salmonids appeared to be coincidental with the salmonid migrations rather than dependent upon it. Cooper and Johnson (1992) reported that marine mammal predation does occur on some local steelhead populations, however, believed that it was not an important factor in the decline of coastwide steelhead populations. Although, Roeffe and Mate (1984) found that pinnipeds fed opportunistically on fast swimming salmonids, less than 1 percent of the adult Rogue River (Oregon) summer steelhead were preyed on during their upriver spawning migration. Therefore, salmonids appear to be a minor component of the diet of marine mammals (Scheffer and Sperry 1931, Jameson and Kenyon 1977, Graybill 1981, Brown and Mate 1983, Roffe and Mate 1984, Hanson 1993). Principal food sources of marine mammals include lampreys (Jameson and Kenyon 1977, Roffe and mate 1984), benthic and epibenthic species (Brown and Mate 1983) and flatfish (Scheffer and Sperry 1931, Graybill 1981).

Predation may significantly influence salmonid abundance in some local populations when other prey are absent and physical habitat conditions lead to the concentration of adult and juvenile salmonids in small areas (Cooper and Johnson 1992). Pearcy (1992) reviewed several studies of salmonids off of the Pacific Northwest coastline and concluded that salmonid survival was influenced by the factional responses of the predators to salmonids and alternative prey. Pfeifer (1987) estimated that 43 percent of the steelhead run into the Lake Washington system, where there is a major fish passage problem, was lost due to sea lion predation during the 1986-87 season. Low flow conditions in streams can also enhance predation opportunities, particularly in central and southern California streams, where adult steelhead may congregate at the mouths of streams waiting for high flows for access (California Department of Fish and Game 1995). Also, warmer water temperatures due to water diversions, water development and habitat modification may affect steelhead mortality from predation directly or indirectly through stress and disease associated with wounds inflicted by pinnipeds or piscivorous predators. Several studies have indicated that piscivorous predators may control the abundance and survival of salmonids. Holtby et al. (1990) hypothesized that temperature-mediated arrival and predation by Pacific hake may be an important source of mortality for coho salmon off of the west coast Vancouver Island. Beamish et al. (1992) documented predation of hatchery-reared chinook and coho salmon by spiny dogfish (*Squalus acanthias*).

The relative impacts of marine predation on anadromous salmonids are not well understood, but most investigators believe it is a minor factor in steelhead declines. However, it is evident that anadromous salmonids have historically coexisted with both marine and freshwater predators and based on catch data, some of the best catches of coho, chinook, and steelhead along the West Coast of the United States occurred after marine mammals, kingfishers, and cormorants were fully protected by law (Cooper and Johnson 1992). Based on this, it would seem unlikely that in the absence of man's intervention, freshwater or marine predators would extirpate anadromous salmonids. Predators play an important role in the ecosystem, culling out unfit individuals thereby strengthening the species as a whole. As indicated above, the increased abundance of certain predators is due primarily to ecosystem modification. Therefore, it would seem more likely that increased predation is but a symptom of a much larger problem, namely, habitat modification and a decrease in water quantity and quality.

IV. *Inadequacy of Existing Regulatory Mechanisms.*

A variety of Federal and state laws and measures affect the abundance and survival of west coast steelhead and the quality of their habitat. The NMFS has prepared a separate report entitled *West Coast Steelhead Conservation Measures, A Supplement to the Notice of Determination for West Coast Steelhead Under the Endangered Species Act* which summarizes Federal, state, tribal and local steelhead conservation measures. This report is available from the addresses listed under Summary of Events Leading to the Status Review.

V. *Other Natural and Manmade Factors.*

A. Natural Factors

1. Drought

Drought conditions reduce the amount of water available for all salmonids. Further, during periods of drought, water diversions for agriculture and urban use may result in substantial reduction (or elimination) of flows needed for adult salmonid passage, egg incubation, and juvenile rearing and migration. A reduction in population size can also result in adverse genetic effects, such as inbreeding and a reduction in future adaptation potential.

In the Northwest, annual mean streamflows for the 1977 water year (October to September) were the lowest recorded for many streams since the late nineteenth century (Columbia River Water Management Group 1978). Precipitation levels in the Snake River Basin above Ice Harbor Dam also were below the 25-year average (1961-1985) in the 1979, 1981, 1985, 1987, 1988, and 1990 water years. The 1990 water year became a fourth consecutive year of drought condition (Columbia River Water Management Group 1991). Drought conditions have persisted in the Columbia River basin during the period of 1990 to 1994. However, recent weather patterns in 1995 have resulted in above average rainfall for much of the remaining West Coast. California's reservoirs are full for the first time in years, following severe droughts throughout the state in 1976 to 1978 and again from 1986 to 1994.

Steelhead populations have persisted in California and throughout the Pacific Northwest for many thousands of years (Behnke 1992) despite drastic and catastrophic climate changes, both long and short-term. There are indications that the six-year drought that California recently experienced may be a very mild event compared to past droughts (Stine 1994). The key to survival in this type of variable and rapidly changing environment is the evolution of behaviors and life history traits that allow steelhead to cope with a variety of environmental conditions. A wide tolerance range is manifested in a variety of steelhead behaviors and life history traits relative to other Pacific salmon. This suggests steelhead populations (coastal and inland forms) possess the ability to survive in, and adapt to, the unique environments throughout the species range

Although natural events may have harmful effects on species or populations, anthropogenic impacts on fish habitat may be the final factor that determines the ability of these populations to persist. Populations that are fragmented and/or reduced in size and range are more vulnerable to extirpation by natural events. At this time it is not clear whether recent climatic conditions represent a long-term change which will continue to affect salmonid stocks or whether these changes are short-term environmental fluctuations that can be expected to reverse in the near future. Many of the steelhead population declines began prior to these recent drought conditions. Humans have little control of the oceanic or atmospheric cycles; therefore, it is important to minimize anthropogenic effects on protecting anadromous salmonid habitat in streams, rivers, and estuaries in order to buffer salmonid populations against these natural cycles.

2. Floods

During flood events, land disturbances resulting from logging, road construction, mining, urbanization, livestock grazing, agriculture, fire, and other uses may contribute sediment directly to streams or exacerbate sedimentation from natural erosive processes (California Advisory Committee on Salmon and Steelhead Trout 1988, California State Lands Commission 1993, Forest Ecosystem Management Assessment Team 1993). Sedimentation of stream beds has been implicated as a principle cause of declining salmonid populations through-out their range. Judsen and Ritter (1964), the California Department of Water Resources (1982b), and the California States Lands Commission (1993) have stated that northwestern and central coastal California have some the most erodible terrain in the world. Many of the parent soil materials present in this area are extremely steepened and are prone to flooding and landslides (California Department of Water Resources 1982a). Several studies have indicated that in this region, catastrophic erosion and subsequent stream sedimentation (such as during the 1955 and 1964 floods) resulted from areas which had been clearcut or which had roads constructed on unstable soils (Janda et al. 1975, Wahrhaftig 1976, Kelsey 1980, Lisle 1982, Hagans et al. 1986). In addition to problems associated with sedimentation, flooding can cause scour and redeposition of spawning gravels in typically inaccessible areas.

As streams and pools fill in with sediment, flood flow capacity is reduced. Such changes cause decreased stream stability and increased bank erosion, and subsequently exacerbate existing sedimentation problems (Lisle 1982). All of these sources contribute to the sedimentation of spawning gravels and filling of pools and estuaries used by all anadromous salmonids. Channel widening and loss of pool-riffle sequence due to aggradation has damaged spawning and rearing habitat of all salmonids. By 1980, the pool-riffle sequence and pool quality in some northern California and southern Oregon streams still had not fully recovered from the 1964 regional flood. In fact, Lisle (1982) found that many north coast streams continue to show signs of harboring debris flow. Such streams have remained shallow, wide, warm, and unstable since these floods.

3. Ocean Conditions

When steelhead smolts from North America enter the Pacific Ocean they begin a directed movement into offshore waters of the Gulf of Alaska (Light et al. 1988). Steelhead seem to generally follow a counter clockwise migration pattern in epipelagic waters east of 167° East longitude (Light et al. 1988), primarily within 27 feet (8.2 meters) of the water surface, but have been captured at depths of 62 feet (18.9 meters) (Light et al. 1989, Burgner et al. 1992). The northern limit of steelhead migration in the ocean extends slightly north of the Aleutian islands and is closely associated with the 41° F (5° C) sea surface isotherm, while the southern limit of steelhead migration and rearing is approximately 39° North latitude and is closely associated with the 59° F (15° C) sea surface isotherm (Light et al. 1989, Pearcy 1992, Burgner et al. 1992). Burgner et al. (1992) reported that coastal Oregon and California steelhead stocks may have more restricted westward migrations than do more northerly stocks.

Steelhead stocks are widely dispersed from California to Alaska and are extensively intermingled (Light et al. 1989, Pearcy 1992). The North Pacific Fisheries Commission (INPFC) trapped, disc- tagged, and released 1,722 adult steelhead in the open ocean from 1956 to 1988 during their high seas tagging experiments. Of these 1,722 tagged steelhead, 77 were recovered in North American coastal areas or spawning rivers (Burgner et al. 1992). Of the 77 North American steelhead returns, 22 were recovered in British Columbia streams, 15 were recovered in coastal Washington and Puget Sound streams, 19 were recovered in the Columbia River and tributaries, 12 were recovered from coastal Oregon streams and nine were recovered in California streams from the Carmel River in central California north to Crescent City (Burgner et al. 1992). Ocean migration and distribution of southern California steelhead populations are unknown. There have not been any tagging studies conducted on populations in southern streams to evaluate ocean distribution. Further, no steelhead tagged on the high seas have been recovered in these California streams (McEwan and Jackson 1996).

Large fluctuations in Pacific salmon catch have occurred during the past century. Annual world harvest of Pacific salmon has varied from 347 million pounds (lbs) (772 million kg) in the 1930s to about 184 million lbs (409 million kg) in 1977 and back to 368 million lbs (818 million kg) by 1989 (Hare and Francis 1993). Mechanisms linking atmospheric and oceanic conditions to fish population survivorship and production have been suggested for stocks in general (Shepherd et al. 1984) and for Pacific salmon specifically (Rogers 1984, Nickelson 1986, Johnson 1988, Brodeur and Ware 1992, Francis et al. 1992, Francis 1993, Hare and Francis 1993, Ward 1993). Vernon (1958), Holtby and Scrivener (1989), and Holtby et al. (1990) have reported associations between salmon survival during the first few months at sea and ocean conditions such as sea surface temperature and salinity. Some studies have tried to link salmon production to oceanic and atmospheric climate change. For example, Beamish and Bouillon (1993) and Ward (1993) found that trends in Pacific salmon catches were similar to trends in winter atmospheric circulation in the North Pacific.

The Subarctic Front, the most prominent feature of the North Pacific Transitional Region, plays a role in the definition of the major physical and biological domains in the Northeast Pacific Ocean. It is possible that changes in the location or structure of the Subarctic Front may affect any of the physical and biological gradients in this area (Pearcy 1991). McGowan (1986) reported that Subarctic Frontal dynamics influence forage aggregations and lead to higher biological productivity which impacts salmonid species at higher trophic levels. Furthermore, variability in the Subarctic Front may affect other physical features which alter productivity, both in the Central Subarctic Domain and downstream in the coastal domains (Reid 1962, Wickett 1967, Eber 1971, Favorite and McLain 1973, Colebrook 1977, Chelton et al. 1982a and 1982b, Fulton and LeBrasseur 1985, Ware and McFarlane 1989). Although the Subarctic Front can be analytically defined, its structure changes in both space (White 1982, Levine and White 1983) and time (White et al. 1980). It moves, intensifies, decays, and undergoes seasonal changes (Roden 1977).

The influence of Subarctic Frontal dynamics on salmonids is probably caused by indirect trophic interactions rather than a direct cause-effect relationship (Pearcy 1992). The interaction or population control might be "bottom-up" by lower trophic levels, or "top-down" by predators. This is especially true for prey organisms including phytoplankton, zooplankton, cephalopods, and fish (Pearcy et al. 1988), as well as predatory organisms including marine mammals and sea birds (Rogers 1984). Pearcy (1992) suggests that predatorial response to coho smolt and alternative prey availability could influence prey survival rates. This is especially important during years of high upwelling resulting in greater smolt dispersal and alternative prey availability. Several studies have examined the possibility that salmonid production or survival is indirectly related to primary production. For example, Pearcy and Fisher (1988) linked salmon abundance with coastal chlorophyll concentrations, primary production, and upwelling.

A feature common to many studies of biological production is the identification of high or low periods of abundance for the study organism. Shifts in abundance for many organisms appear to have coincided with the shift in abundance of salmon in the late 1970s (Rogers 1984). Hare and Francis (1993) identified two interventions (statically significant changes in the mean of a time series) in the abundance of Alaskan pink and sockeye salmon between 1919 and 1988: one occurring in the late 1970s and the other occurring in the early 1950s. The intervention (increase) in the late 1970s was more pronounced than the earlier intervention (decrease) and matches well with the shift noted by Rogers (1984) and Ward (1993). Also, the timing of the 1970s intervention has often been correlated to change in the abundance of other organisms. Brodeur and Ware (1992) found that the abundance of zooplankton, several species of fish, and cephalopods in the central Subarctic Gyre changed significantly from the period of 1956-1962 to 1980-1989. These changes corresponded to a 1.7 fold increase in the estimated biomass of all North American salmon combined between the periods of 1956-1962 and 1980-1984 (Rodgers 1987).

Francis and Sibley (1991) and Francis et al. (1992) have developed a model linking decadal-scale atmospheric variability to salmon production hypotheses developed by Hollowed and Wooster (1991) and Wickett (1967), as well as evidence presented in many other studies. This model describes a time series of biological and physical variables from the Northeast Pacific which appear to share decadal-scale patterns; most notably synchronous shifts in mean conditions during the late 1970s and out-of-phase relationship between variables in the Coastal Upwelling and Coastal Downwelling domains. Biological and physical variables which appear to have undergone shifts during the late 1970s include the following: salmon (Rogers 1984, 1987, Hare and Francis 1993) and other pelagic fish, cephalopods, and zooplankton (Brodeur and Ware 1992); oceanographic properties such as current transport (Royer 1989), sea surface temperature and upwelling (Holowed and Wooster 1991); and atmospheric phenomena such as atmospheric circulation patterns, sea-surface pressure patterns, and sea-surface wind-stress (Trenberth 1990, Trenberth et al. 1993). Variables from the Coastal domains which appear to fluctuate out-of-phase include salmon (Francis and Sibley 1991), current transport (Wickett 1967, Chelton 1983), sea surface temperature and upwelling (Tabata 1984, Hollowed and Wooster 1991), and zooplankton (Wickett 1967).

Finally, Scarnecchia (1981) reported that near-shore conditions during the spring and summer months along the California coast may dramatically affect year-class strength of salmonids. Bottom et al. (1986) believed that coho salmon along the Oregon and California coasts may be especially sensitive to upwelling patterns because these regions lack extensive bays, straits, and estuaries which could buffer adverse oceanographic effects such as those found along the Washington, British Columbia, and Alaskan coasts. The paucity of high quality, near-shore habitat coupled with variable ocean conditions have served to make freshwater habitat more crucial for the survival and persistence of many steelhead populations.

a. El Niño

"El Niño" an environmental condition often cited as a cause for the decline of west coast salmonids. El Niño is an unusual warming of the Pacific Ocean off South America and is caused by atmospheric changes in the tropical Pacific Ocean (Southern Oscillation-ENSO). El Niño events occur when there is a decrease in the surface atmospheric pressure gradient from the normal-steady trade winds that blow across the ocean from east to west on both sides of the equator. There is a drop in pressure in the east off South America and a rise in the pressure in the western Pacific. The resulting decrease in the pressure gradient across the Pacific Ocean causes the easterly trade winds to relax, and even reverse in some years. When the trade winds weaken, sea level in the western Pacific Ocean drops, and a plume of warm sea water flows from west to east toward South America, eventually reaching the coast where it is reflected south and north along the continents.

El Niño ocean conditions are characterized by anomalous warm sea surface temperatures and changes coastal currents and upwelling. Principal ecosystem alterations include decreased primary and secondary productivity and changes in prey and predator species distributions. Several recent El Niño events have been recorded during the last several decades, including those of 1940-41, 1957-58, 1982-83, 1986-87, 1991-1992, and 93-94.

Anadromous salmonids have managed to persist in the face of numerous climatic events and changes. The long-term persistence of steelhead populations depends on their ability to withstand fluctuations in environmental conditions. It is apparent that the tremendous loss of freshwater habitat combined with extremely small population levels cause salmonid populations to be more vulnerable to extirpation from natural events. Until recently, when salmonid population levels have reached critical levels, these environmental conditions have largely gone unnoticed (Lawson 1993). Therefore, it would seem that environmental events and their impacts on remaining populations, serve more as an indication of unstable population levels rather than a direct cause of a decline.

4. Other Natural Occurrences

The eruption of Mount St. Helens in 1980 resulted in devastating impacts to nearby rivers and streams. Impacts have included high sedimentation in spawning and rearing areas, resulting in stock displacement to various tributaries (Leider 1989), increased water temperature, and decreased fish foraging efficiency (Palmisano et al. 1993).

Wildfires are also a factor which can contribute to an increase in short-term sediment runoff (Wells 1987). In addition, water quality degradation can result from chemical agents used to control forest fires (U.S. Forest Service 1993).

B. Manmade Factors

1. Artificial Propagation

In 1993, over 81 million juvenile salmonid hatchery fish were released into the Snake and Columbia River system above Bonneville Dam (Columbia Basin Fish and Wildlife Authority 1994). Juvenile steelhead have accounted for about 15 percent of these releases, or about 12 million juveniles (Columbia Basin Fish and Wildlife Authority 1994). Juvenile steelhead hatchery releases average about 2.5 million below Bonneville Dam since 1987 (Columbia Basin Fish and Wildlife Authority 1994). The Snake River system has accounted for the majority of steelhead hatchery production in areas above Bonneville Dam, averaging about 5 million juveniles per year since 1980 (Columbia Basin Fish and Wildlife Authority 1994). During the period from 1978 to 1987, the following steelhead hatchery production occurred in the western United States: Washington at 6,782,000, Idaho at 5,372,000, Oregon at 4,537,000, California at 2,304,000, and Alaska at 62,000 fish per year (Light 1987). A total of 24,605,000 steelhead smolts on average are produced each year in these four western states.

Non-native steelhead stocks have historically been introduced as broodstock in hatcheries and widely transplanted in many coastal rivers and streams. Altukhov and Salmenkova (1986) have shown that anadromous salmonids transferred to other watersheds rarely persist for more than two generations without repeated artificial propagation. Withler (1982) showed that there has been no successful case of establishing a new run of anadromous salmonids by transplanting stocks anywhere along the Pacific Coast. Fisheries agencies within the states of Washington, Oregon, Idaho and California, along with other organizations, have transplanted non-native steelhead stocks throughout their respective states within this past century (Bryant 1994, Busby et al. 1996).

Many concerns exist regarding the impacts of artificial propagation on wild stocks of salmon.

Competition which can occur among hatchery and native adults for spawning sites and food, may lead to decreased production. Hatchery may outnumber wild fish and monopolize available spawning habitat when wild stocks are small and hatchery supplementation occurs. Fleming and Gross (1992) stated that the negative effect of such competition can be magnified by the fact that naturally spawning hatchery stocks have lower spawning success than do wild fish. Steward and Bjornn (1990) found that hatchery stocks may also produce fewer smolts and returning adults. Nielsen (1994) found the introduction of hatchery reared coho salmon into the Noyo River, California, led to displacement of wild cohorts from their usual microhabitats and shifts in their foraging behavior. Stempel (1988) concluded that competition might be occurring in the mainstem of the Klamath and Trinity rivers among hatchery and wild salmonids, resulting in low survival of both.

Juvenile steelhead which have been derived from non-native, hatchery broodstock may stray and interact with native populations. Altukhov and Salmenkova (1986) reported that when non-native hatchery strays spawn in the wild, young fish with some non-native genes may result. Studies conducted in areas of the Pacific Coast have found that juvenile salmonids produced from stray hatchery fish and hatchery-wild hybrids have relatively low survival rates compared to native fish (Chilcote et al. 1986, Riesenbichler and McIntyre 1977). Waples (1991), Hindar et al. (1991), and Steward and Bjornn (1990) found that the impact of stock transfers increases dramatically if non-native salmonids are planted on top of wild populations for several generations. When this method of transfer occurs, Altukhov and Salmenkova (1986) found a loss of local adaptations which may lead to extirpation of that local stock.

Genetic changes in hatchery stocks of Pacific salmonids have been documented and models have recently been constructed by Waples (1990a,b) and Waples and Teel (1990) to aid in understanding the consequences of these changes. Steward and Bjornn (1990) noted that large differences in the genetic structure of wild and hatchery stocks may potentially lead to lower survival rates. Steward and Bjornn (1990) also reported that supplementation with hatchery stocks can have differing effects depending on the size of the wild population. Shapovalov and Taft (1954) noted an inverse correlation between the number of downstream migrants and adult returns, implying that low intraspecific competition increases oceanic survivorship of downstream migrants.

Crowded conditions in hatcheries can create favorable environments for many disease organisms. Introduction of exotic stocks can also introduce a new disease into a wild population. The ability of a wild stock to cope with an introduced disease is reduced if the stock's genetic variability has been reduced through selection or genetic drift (Allendorf and Phelps 1980).

The capture of broodstock may also adversely impact small or declining wild populations due to pre-spawning mortality during capture or transport, differential viability of gametes in artificial situations, disease, and artificial selection. Verspoor (1988) noted that wild broodstock typically contribute little genetic diversity to subsequent generations of hatchery fish. Taking more wild fish for broodstock in an attempt to overcome these problems in hatchery stocks may ultimately increase risks to wild populations.

The relatively low number of spawners needed to sustain a hatchery population can result in high harvest-to-escapement ratios in waters where regulations are set according to hatchery production. This practice can lead to over-exploitation and reduction in size of wild populations coexisting in the same system. For example, in a declared "hatchery management area" in British Columbia, harvest rates on coho salmon are as high as 95 percent (Hilborn 1992). This is sustainable only because of the most successful hatchery stocks, and, as a result, wild stocks have declined (Hilborn 1992).

Available research indicates that interactions between non-native and wild stocks may have contributed to the decline of this species across its range. More recent hatchery practices, such as utilizing native broodstocks and limiting native and hatchery fish interactions through temporal or geographic means, may reduce negative impacts to wild stocks. However, hatcheries may palliate the widespread loss and destruction of habitat, concealing the real problems facing anadromous resources (Goodman 1990, Hilborn 1992, Meffe 1992).

Summary

Steelhead on the west coast of the United States have experienced dramatic declines in abundance during the past several decades as a result of human-induced and natural factors. The scientific literature is replete with information documenting the decline of steelhead populations and anadromous salmonid habitats. There is no single factor solely responsible for this decline. Every factor identified in this report has contributed in varying degree to this decline. Given the complexity of this species' life history and the ecosystem in which it resides, the authors believe it is impossible to accurately quantify the relative contribution of any one factor to the decline of a given steelhead ESU. Rather, the authors have found it only possible to highlight those factors which have significantly affected the status of a particular ESU (Table 1). This list will expand and contract as more information becomes available. It is important to note in reviewing this list that recovery efforts must focus on those areas which are within human influence and control.

Water storage, withdrawal, conveyance, and diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible habitat. Modification of natural flow regimes have resulted in increased water temperatures, changes in fish community structures, depleted flows necessary for migration, spawning, rearing, flushing of sediments from spawning gravels, gravel recruitment and transport of large woody debris. Physical features of dams, such as turbines and sluiceways, have resulted in increased mortality of both adults and juvenile salmonids. Attempts to mitigate adverse impacts of these structures have to date met with limited success.

Natural resource use and extraction leading to habitat modification can have significant direct and indirect impacts to steelhead populations. Land use activities associated with logging, road construction, urban development, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality. Associated impacts of these activities include: alteration of streambank and channel morphology; alteration of ambient stream water temperatures; degradation of water quality; elimination of spawning and rearing habitat; fragmentation of available habitats; elimination of downstream recruitment of spawning gravels and large woody debris; removal of riparian vegetation resulting in increased stream bank erosion; and increased sedimentation input into spawning and rearing areas resulting in the loss of channel complexity, pool habitat, suitable gravel substrate, and large woody debris. Studies indicate that in most western states, about 80 to 90 percent of the historic riparian habitat has been eliminated. Further, it has been estimated that during the last 200 years, the lower 48 United States have lost approximately 53 percent of all wetlands. Washington and Oregon's wetlands have been estimated to have diminished by one third, while it is estimated that California has experienced a 91 percent loss of its wetland habitat.

The degree of spatial and temporal connectivity between and within watersheds is an important consideration for maintaining aquatic riparian ecosystem functions is. Loss of this connectivity and complexity, such as the loss of deep pool habitats, has contributed to the decline of steelhead. In Washington, the number of large, deep pools in National Forest streams has decreased by as much as 58 percent due to sedimentation and loss of pool-forming structures such as boulders and large wood. Similarly, in Oregon, the abundance of large, deep pools on private coastal lands has decreased by as much as 80 percent.

Steelhead have been, and continue to be, an important recreational fishery throughout their range. During periods of decreased habitat availability, the impacts of recreational fishing on native anadromous stocks may be heightened. While not generally targeted in commercial fisheries in the ocean, high seas driftnet fishing may have been partially responsible for declines in steelhead abundance. Research has estimated that unauthorized high seas driftnet fisheries may have harvested between 2 and 28 percent of the steelhead that were destined to return to the Pacific coast of North America. However, such fisheries cannot account for the total declines in steelhead abundance observed in North America.

Introduction of non-native species and modification of habitat have resulted in increased predator populations and salmonid predation in numerous river systems. Marine predation is also of concern in some areas given the dwindling steelhead run-size in recent years. In general, predation rates on steelhead are considered by most investigators to be an insignificant contribution to the large declines observed in west coast populations. However, predation may significantly influence salmonid abundance in some local populations when other prey are absent and physical habitat conditions lead to the concentration of adults and juveniles.

Natural environmental conditions have served to exacerbate the problems associated with degraded and altered riverine and estuarine habitats. Recent floods and persistent drought conditions have reduced already limited spawning, rearing, and migration habitat. Furthermore, climatic conditions appear to have resulted in decreased ocean productivity which may help offset degraded freshwater habitat conditions to some degree. Environmental conditions such as these have gone largely unnoticed until recently, when salmonid populations have reached critical low levels.

In an attempt to mitigate for lost habitat and reduced fisheries, extensive hatchery programs have been implemented throughout the range of steelhead on the West Coast. While some of these programs have been successful in providing fishing opportunities, the impacts of these programs on wild stocks are not well understood. Competition, genetic introgression, and disease transmission resulting from hatchery introductions may significantly impact the production and survival of wild steelhead. Furthermore, displacement of wild fish for broodstock purposes may result in additional negative impacts to small or dwindling natural populations. It is important to note however that the use of hatcheries will likely play an important role in reestablishing depressed stocks of Pacific salmonids. Alternative uses of supplementation, such as for the creation of terminal fisheries, must be fully explored to limit negative impacts to remaining wild populations. This use must be tempered with the understanding that protection of wild fish and their habitats is critical to maintaining healthy, fully-functioning ecosystems.

Authors

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Table 1. Summary of Factors Affecting Each Steelhead ESU

Name of ESU	Geographic Range of ESU	Factors Affecting ESU
1) Puget Sound	Strait of Juan De Fuca, Puget Sound, and Hood Canal, WA.	! Habitat blockages ! Hatchery introgression ! Urbanization ! Logging ! Hydropower development ! Harvest
2) Olympic Peninsula	West of Elwha River and south to, but not including, Grays Harbor drainage, WA.	! Hatchery introgression ! Logging ! Minor habitat blockages ! Hydropower development ! Harvest
3) Southwest Washington	Grays Harbor drainage, WA to Columbia River below Cowlitz River, WA and below Willamette River, OR.	! Hatchery introgression ! Logging ! Agriculture ! Harvest ! Hydropower ! Predation
4) Lower Columbia River	Columbia River and tributaries between Cowlitz and Wind Rivers in WA, Willamette and Hood Rivers in OR.	! Hatchery introgression ! Habitat blockages ! Logging ! Eruption of Mt. Saint Helens ! Hydropower development ! Predation ! Harvest
5) Upper Willamette River	Willamette River, OR upstream from Willamette Falls.	! Urbanization ! Logging ! Habitat blockages ! Predation ! Agriculture ! Harvest
6) Oregon Coast	Oregon coast north of Cape Blanco, OR excluding Columbia River tributaries.	! Logging ! Hatchery introgression ! Agriculture ! Minor habitat blockages ! Historic flooding ! Harvest

Table 1. Summary of Factors Affecting Each Steelhead ESU (continued)

Name of ESU	Geographic Range of ESU	Factors Affecting ESU
7) Klamath Mountains Province	Elk River, OR to Klamath and Trinity Rivers in CA.	! Hatchery introgression ! Logging ! Water diversion\extraction ! Habitat blockages ! Poaching ! Agriculture ! Hydropower development ! Historic flooding ! Mining
8) Northern California	Redwood Creek, Humboldt County, CA to Gualala River, CA.	! Historic Flooding ! Predation ! Water diversion\extraction ! Minor habitat blockages ! Poaching ! Logging ! Agriculture ! Mining
9) Central California Coast	Russian River, CA to Soquel Creek and the drainages of San Francisco and San Pablo Bays, CA; excluded is the Sacramento/San Joaquin River Basin.	! Water diversion\extraction ! Habitat blockages ! Agriculture ! Logging ! Historic flooding ! Hatchery introgression ! Poaching ! Mining ! Urban development ! Harvest
10) South/Central California Coast	Pajaro River, CA to north of the Santa Maria River, CA.	! Urbanization ! Water diversion\extraction ! Historic flooding ! Habitat blockages ! Agriculture ! Poaching ! Harvest
11) Southern California	Santa Maria River, CA to southern extent of species range.	! Water diversion\extraction ! Habitat blockages ! Urbanization ! Agriculture ! Harvest

Table 1. Summary of Factors Affecting Each Steelhead ESU (continued)

Name of ESU	Geographic Range of ESU	Factors Affecting ESU
12) Central Valley	Sacramento River, CA and San Joaquin River, CA.	! Water diversion\extraction ! Mining ! Agriculture ! Urbanization ! Habitat blockages ! Logging ! Harvest ! Hydropower development ! Hatchery introgression
13) Middle Columbia River Basin	Mosier Creek, OR to the Yakima River, WA inclusive.	! Water diversion\extraction ! Hydropower development ! Agriculture ! Hatchery introgression ! Predation ! Harvest
14) Upper Columbia River Basin	Columbia River upstream from Yakima River, WA.	! Hydropower development ! Water diversion\extraction ! Agriculture ! Hatchery introgression ! Predation ! Harvest
15) Snake River Basin	Snake River Basin, ID, upstream from confluence with Columbia River.	! Logging ! Agriculture ! Hydropower development ! Water diversion\extraction ! Hatchery introgression ! Habitat blockages ! Mining ! Harvest

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Appendix A. Impacts of sedimentation on aquatic ecosystems (Koski and Walter 1978)

I. Inorganic Sediment Input

A. Changes in salmonid spawning habitat
1. Suffocation of eggs and alevins
2. Blockage of fry emergence
3. Change in timing of fry emergence
4. Reduction of size of fry at emergence
5. Interference with homing ability of adults

B. Changes in salmonid rearing habitat
1. Reduction in living space and shelter
2. Change in food availability
3. Increase in emigration

C. Changes affecting macroinvertebrates
1. Reduction in living space around rocks
2. Reduction in food (periphyton)
3. Increase in drift rate
4. Prevention of larval development and emergence

D. Changes affecting aquatic plants and algae
1. Reduction in photosynthetic rate
2. Reduction in abundance by dislodgement and deposition

II. Input of Organic debris

A. Fine organic debris
1. Reduction of dissolved oxygen
2. Production of slime bacteria
3. Loss of habitat for riffle-dwelling benthic invertebrates and algae

B. Large organic debris
1. Debris jams
    a. Blockage or delay of juvenile and adult fish migration
    b. Cover spawning and rearing areas
2. Washout of debris jams
    a. Release of fine sediment
    b. Scour and destruction of benthic invertebrates and salmon eggs and alevins
3. Contribution to channel and streambank instability by diversion of streamflow and washout