The Effects of Summer Dams on Salmon and Steelhead in California Coastal Watersheds and Recommendations for Mitigating Their Impacts

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1.0 Problem Statement

Habitat loss is a major factor contributing to the decline of salmon and steelhead populations in California (NMFS 1996, NMFS 1996a, Myers et al.1998). Activities contributing to the loss of salmonid habitat include forestry practices, agriculture, urbanization, water diversion, and the construction of on-stream projects such as electrical and gas transmission corridors, highway crossings, and dams. There are numerous kinds of on-stream dams, and they are almost always detrimental to migratory fishes. Large dams for flood protection, water supply and hydroelectric production have appreciably reduced available habitat for salmon and steelhead. The proliferation of small, permanent on-stream dams and reservoirs for irrigation purposes have also taken their toll. In addition to these permanent structures are the often overlooked seasonal dams constructed each summer for the purpose of recreation, irrigation, groundwater recharge, fire suppression, and livestock watering. The effects of summer dams on salmon and steelhead, are not well documented; however, they have the potential to adversely affect salmonid habitat by 1) reducing stream habitat diversity, 2) diminishing stream water quality, 3) enhancing the habitats of salmonid predators, and 4) blocking or restricting fish movements.

The purpose of this paper is to examine the potential site-specific and cumulative effects of "summer dams" on anadromous salmonid species in California coastal streams. These species include coho salmon (*Oncorhynchus kisutch*), chinook salmon (*O.tshawytscha)*, and steelhead trout (*O. mykiss*). Specific objectives of this paper are to: (1) identify the likely effects of summer dams on these species; (2) review scientific literature pertaining to stream ecology and the ecology of anadromous salmonids as they relate to possible effects of summer dams on salmonid populations; and (3) propose scientifically-sound recommendations that would facilitate environmental regulatory review of summer dams in California.

2.0 Existing Regulatory Review and Construction of Summer Dams

In California, summer dams are regulated under Fish and Game Code Chapter 1600, through which the California Department of Fish & Game (CDFG) works with public and private entities to develop mutually agreeable proposals for projects that may adversely affect fish and wildlife resources. Projects addressed under Chapter 1600 are ones, "... that divert or obstruct the natural flow or substantially change the bed, channel, or bank of any river or stream or lake designated by the department, or use any material from the streambeds...." Summer dams operated by municipalities or other public agencies are addressed by Fish & Game Code, Section 1601. Public facilities are usually relatively large and located on mainstem rivers or larger tributaries. Summer dams constructed by landowners or other private entities are addressed under Fish & Game Code, Section 1603. These latter "private ponds" are the more common type of summer dam, which are typically constructed on relatively small streams. Most summer dams are not reviewed or permitted, because those constructing the dams are either not aware of the existing regulations or they choose to ignore those regulations.

Summer dams are constructed using a number of techniques. Many are crude structures created by the manual placement of cobbles across the stream. Such structures tend to be low and temporary and are typically washed out by fall freshets. More substantial structures are formed using earthen-berms or flashboard structures. Flashboard dams generally have a permanent foundation with wing-walls, a spillway apron, and a set of removable wooden flashboard planks. Earthen-berm summer dams are constructed with heavy machinery that push up berms of riverbed or bank material – hence, their other name, the "push-up dam". Push-up dams can be taller and more substantial than simple, manually constructed dams; therefore, they are often resistant to being washed out by fall freshets. Flashboard and large, earthen-berm dams generally need to be dismantled at the end of summer.

Summer dams are commonplace throughout coastal watersheds in California. In the

Russian River basin alone, it has been estimated that several hundred summer dams are installed annually (Chase et al. 2000). These dams are mostly constructed by private parties; few are constructed in coordination or consultation with CDFG (SEC 1996). A 1995 aerial flight revealed that most tributaries to the West Fork Russian River contained summer dams, and many drainages had a series of dams (SEC 1996). The Austin Creek watershed, a principal tributary of the Russian River, had as many as 38 summer dams on it in the 1960's and 1970's (CDFG 1999). Efforts to reduce the number of summer dams on Austin Creek by the U.S. Army Corps of Engineers and CDFG were largely successful, yet are ongoing. Enforcement personnel from NMFS and CDFG continued to find unpermitted summer dams on Austin Creek during summer 2000 and 2001.

Another river heavily affected by summer dams is the San Lorenzo River in Santa Cruz County, a watershed with approximately 60 miles of stream available to anadromous fish. The San Lorenzo supports annual runs of steelhead, and until 1988 it supported runs of coho salmon. During summer months, much of the mainstem of this river is converted to pond habitat by summer dams. Available information indicates up to 35 flash board recreational summer dams are present within this watershed.

3.0 Effects of Summer Dams on Listed Salmon and Steelhead

Salmonids require cool, clear, running water to support their freshwater life history stages (Bjornn and Reiser 1991). Incubating salmon eggs require clean gravel substrates. Juvenile fish typically rear in free-flowing streams providing a complex of alternating shallow, swift riffles and low-velocity pools with abundant cover in the form of woody debris, boulders, and undercut banks (see Appendix A for additional information on the life history and habitat requirements of California salmonids). Summer dams convert such natural stream habitats to artificial ponds that may extend a few hundred feet up to several thousand yards in length. The precise length of a summer pond is dependent upon both the height of the dam and the stream's gradient.

The degree to which a summer dam impacts anadromous salmonids depends, in part, upon the relationship between the timing of various life history stages of steelhead and salmon and the timing of the dam's installation and removal. Installations in late June are less likely to adversely affect the reproductive stages of these species. Whereas, installations in April or early May have the greatest potential adverse effect. April installations can impede steelhead migration to spawning grounds in some years, and in most years it would probably affect egg incubation of steelhead. In some years, installations in April may even affect incubating coho salmon eggs or fry that have yet to emerge from the gravel. April and May are also the months when juvenile salmonids typically outmigrate to the ocean in their "smolt" stage (Brown et al. 1994; Weitkamp et al 1995; Busby et al. 1996), and therefore, summer dams can obstruct those seaward movements if they are installed too early. Although less likely to impact salmonid reproduction than earlier installation dates, the construction of summer dams as late as June can also potentially affect the outmigration of late running steelhead and coho salmon smolts (Weitkamp et al 1995; SEC 1996; Titus et al. 1999). In most cases summer dams are removed at the end of summer or in early fall, before increased river flows cause local flooding and damage to the dam's infrastructure. However, if they are not removed by early fall, they can affect the upstream migration of adult chinook salmon. Those left in place during late fall and winter can impede the migrations of adult steelhead and coho salmon.

During the months of June, July, August and September, summer dams can diminish the quality of summer rearing habitat for juvenile salmon and steelhead by changing stream flow patterns, reducing habitat diversity, diminishing water quality, and creating barriers to the natural instream movements of juvenile stages. Summer dams can also enhance the quality of habitats for species that are predators of juvenile salmon and steelhead.

Proponents of summer dams often contend that seasonal impoundments benefit salmonids, because they enlarge and deepen stream habitats. Proponents often report that their historic summer dams provide excellent fish habitat and that their ponds support large numbers of fish. During the early 1980's, CDFG investigated such claims by conducting an experiment in a Sonoma County stream supporting coho salmon and steelhead. In mid-June 1984, CDFG planted 468 marked steelhead in a 75 meter-long seasonal impoundment; these fish were allowed to remain in the impoundment over the summer, and then the number of steelhead remaining in the impoundment were carefully enumerated in late September.   The numbers of fish in the impoundment were contrasted with those from nearby downstream reaches on the same creek. That study found that only one of the 468 marked steelhead remained in the summer impoundment; however, the biologists recovered 164 roach *(Hesperoleucus symmetricus*), 110 suckers *(Catostomus* spp), 47 sculpin (*Cottus* spp.), 4 tule perch (*Hysterocarpus traski*), and 1 pikeminnow (*Ptychocheilus grandis*) from the summer pond in September. The 62 meters of free-flowing stream that were surveyed downstream supported 14 times as many steelhead per unit length of stream and 93 times as many steelhead per unit of stream surface area. The CDFG memorandum that presented the results of that 1984 study is attached as Appendix B.
The following sections address specific concerns about the effects of summer dams on anadromous salmonids in coastal California streams.

3.1 Effects of the Installation and Removal of Summer Dams on Salmonids

During the installation and removal process, stream-dwelling salmonids may be impacted by both direct physical disturbance and by the temporary, but substantial changes in stream flow that may occur while the impoundment is being filled or drained.

Disturbance from Construction Activities

When summer dams are constructed at spawning sites, developing embryos or alevins (*i.e.*, recently hatched fry that have not yet emerged from the gravel) are vulnerable to crushing by heavy machinery traffic. They can also be dug up during the construction of push-up dams. Eggs and pre-emergent fry are also vulnerable to fine sediments that become suspended during construction and then settle downstream over redds (*i.e.*, salmonid egg "nests"). Sediments that clog the interstitial spaces within gravel substrates can smother developing embryos and pre-emergent fry (Hausle and Coble 1976; Alexander and Hansen 1986; Bjornn and Reiser 1991). The deposition of silt and fine sediments (<0.84mm) is especially damaging to later stages of incubating eggs and alevins (Reiser and White1988).

The impoundment of water over gravels containing incubating eggs or fry also poses a threat to egg development, because the newly formed pond environment impairs subsurface currents (*i.e.*, hyporheic flow) needed to 1) transport high levels of dissolved oxygen to developing eggs and fry and 2) remove metabolic waste products from those lifestages. It is the untimely installation of dams that poses the greatest threat to incubating eggs, given that fry emergence can extend into the month of May on many streams in many years.

Juvenile stages may also be injured or killed during construction activities. However, because of their mobility, juveniles are less susceptible to physical damage during summer dam installation and removal.

Effects of Stream Flow Perturbations

In addition to direct physical injury or mortality, the process of constructing and removing summer dams can adversely affect salmonid habitat by substantially altering stream flows. During the filling period, stream reaches below summer dams can lose all or most of their flow, and some reaches may become dewatered. Such reductions in downstream flow have the potential to strand fishes along stream banks, or they may cause fishes to become isolated in small pools or other marginal habitats. Risk of stranding is generally highest in areas where the channel slope is gentle or irregular. Fishes stranded in dewatered areas are vulnerable to both dessication and increased predation from birds or mammals. Those stranded in isolated pools are also more vulnerable to predation, and they may be subjected to higher rates of mortality due to the effects of deteriorating water quality (*e.g.*, elevated temperatures).

In practice, the risk of capturing all of a stream's flow during filling is remote for small, manual constructions and push-up dams because of the manner in which they are constructed. However, flashboard dams can substantially reduce stream flows during pond filling, because the installation of flashboards can be rapid and the barrier to flow is nearly instantaneous and complete. The duration of time that a downstream reach is exposed to artificial flow reductions depends upon the stream's discharge and the reservoir volume.

The removal of summer dams during fall can also affect stream flows to the detriment of salmon and steelhead. Removing a seasonal dam can cause a sudden decrease in the water surface elevation in the impoundment with a concomitant surge in downstream flows, followed by a quick reduction in downstream flows after the pond is drained. These sudden changes in flow and water surface elevations have the potential to cause stranding of juvenile salmonids both upstream and downstream of the dam. During the dam removal process, the risk of stranding is generally greater in cases in which the reservoir volume is relatively great and the rate of water release is high. The adverse effects of rapid, artificial fluctuations in stream flows on fisheries resources are well documented (Cushman 1985).

3.2 Summer Dam Effects on Habitat Diversity

The installation of summer dams can reduce the production of juvenile salmonids by degrading the quality of rearing habitat for these species. As previously noted, juvenile salmonids prefer heterogeneous stream environments comprised of free-flowing riffle-pool complexes containing a mix of pools with ample cover and shallow, swift reaches that support high production of freshwater invertebrates (Bjornn and Reiser 1991; Groot and Margolis 1991). The production of juvenile salmonids can be directly related to stream channel complexity (Fausch and Northcote 1992; Horan et al. 2000). More structurally complex streams containing boulders, logs, and bushes support larger numbers of coho salmon fry than simpler stream sections (Scrivener and Andersen 1982). Summer dams diminish habitat complexity by reducing stream sections to large, shallow pools. In some cases, heavy machinery is used to re-contour stream beds and banks to create areas for recreational swimming. Such actions flatten and widen stream beds and further reduce habitat complexity. Woody debris providing needed shelter for juvenile fish is often cleared from summer ponds to enhance recreational values. This removal of cover objects exposes juvenile fish to greater risks of predation, and it can eliminate low-velocity refuges in main channel areas.

In addition to simplifying stream habitats, artificial impoundments can degrade stream habitats by damaging riparian vegetation. Many riparian plant species, such as most stream side trees in California, cannot survive for long periods in flooded environments. Riparian vegetarian may be flooded during high winter flows; however, extended inundation in summer ponds can weaken and eliminate some riparian species. This can lead to decreasing bank stability, erosion and channel widening. The functional values and benefits of riparian vegetation to aquatic systems and stream fish populations are well documented (Hall and Lantz 1969; Karr and Schlosser 1978; Lowrance et al. 1985; Wesche et al. 1987; Gregory et al. 1991; Platts 1991; Welsch 1991; Castelle et al. 1994; Wang et al. 1997). Loss of riparian vegetation due to summer dams can also contribute to stream warming, reduced recruitment of woody debris, and loss of allochthonous invertebrate production.

The installation of summer dams also causes the benthic fauna to change from species adapted to flowing environments to those adapted to lacustrine conditions. Benthos living in the sediment deposits of seasonal ponds are less productive and available as food for rearing juvenile salmonids (Wayne Fields, Hydrozoology personal communication). Therefore, summer dams can impair the volume of benthic drift, thereby adversely affecting the growth and condition of salmonids in downstream reaches.

3.3 Summer Dam Effects on Stream Water Temperature

Summer dams have the potential to adversely affect juvenile salmonids by raising stream temperatures above optimal levels for their growth and survival. The temperature requirements of coho salmon, chinook salmon, and steelhead are reviewed in Appendix A. In general, these species prefer coldwater habitats with water temperatures less than about 15EC. Upper lethal limits generally range in the vicinity of about 23 - 25E C, although many salmonid species can survive short-term exposures to temperatures as high as 27 or 28EC (Lee and Rinne 1980). Fluctuating diurnal water temperatures with night-time cooling facilitate the survival of salmonids exposed to high daily temperatures. Large, thermally stratified pools, springs, and cool tributary inflow can also provide coldwater refuges that help juveniles survive hot summer temperatures (Nielsen et al. 1994).

Summer dams widen stream reaches, thereby exposing a stream's water mass to increased solar radiation. Heated surface waters are then spilled over the dams to downstream reaches. Because coldwater habitats are generally limited during summer months, juvenile salmon and steelhead often contend with stream temperatures that are already less than optimal. Artificial structures that exacerbate stream warming can turn good quality habitat into marginal habitat, and turn marginal habitats into lethal conditions.

The extent to which small on-stream dams affect stream temperatures depends upon the degree of shading, groundwater inflow, and stream flow, and therefore it is difficult to make generalizations (McRae and Edwards 1994). However, Winkle et al. (1990) reference three studies documenting the warming effect of beaver dams on small creeks. In California, stream sections with large unshaded ponds had water temperatures higher than unaffected (control) stream reaches in the Guadalupe River, Los Gatos and Coyote Creeks (SCVWD 1995). That study indicated that due to the already high water temperatures found in those creeks, the additional stream warming resulting from these dams would likely adversely affect steelhead.

3.4 Summer Dam Effects on Fish Passage

Any dam is a fish migratory impediment; some are sufficiently difficult to be migration barriers. A dam may retard or stop adults moving upstream to spawn or smolts moving downstream to the sea. Summer dams may also be barriers to the dispersal and movements of juveniles during summer months. Given their variability in design, size, and level of permanence, it is difficult to generalize about fish passage problems at summer dams. However, very few have special structures for fish passage.

Timing of summer dam installation is important. Along the North-Central California Coast, the smolt stages of coho salmon, steelhead, and chinook salmon typically emigrate from February through early June. As they migrate, smolts transform their osmotic regulatory system from a freshwater mode in anticipation of life in saltwater (Dickhoff et al. 1982). Summer dams installed prior to the end of the emigration period may either delay or trap smolts. A dam or series of dams that create sufficient delays will adversely affect smolts, because this life stage is physiologically, more vulnerable to stress (Fagerlund et al. 1995), and because smolts will gradually revert back to a pre-smolt condition (parr-reversion), thereby rendering them ill-suited for life in the marine environment (Folmar et al. 1982). Post-smolts stranded in lower rivers are then vulnerable to losses when the river's habitat quality becomes diminished with the onset of elevated summer water temperatures and low summer stream flows.

To successfully pass a dam during low flows, downstream migrating smolts require a confined spillway that drops water to a pool that is sufficiently deep to cushion their fall. Small dams constructed by manual placement of materials and push-up dams are not designed to provide downstream passage. Some dams use plastic sheeting as chinking to more effectively trap streamflow. Streamflow generally spills over the top of the dam as sheetflow and onto the sloping dam walls and not into a pool. Outmigrating smolts attempting to cross dam crests are vulnerable to bird predation. Summer dams formed from gravel berms can be even greater barriers to smolts, because stream flow often percolates through the berms with no spill and no provision for passage. Flashboard dams can have V-notches cut into the flashboards to help facilitate passage, but they are usually constructed without notches and without downstream pools.

Summer dams can also block the upstream migration of adults, if they are not removed prior to the spawning migration season. In California, coho salmon normally migrate to spawning areas from October to March, with most activity during November through January (SEC 1996; Brown et al. 1994). Steelhead may migrate upstream as early as November to as late as June in years with greater water availability, but typical migration occurs between December and April (SEC 1996; Titus et al. 1999). Chinook salmon may migrate from August through January, with most activity occurring between about October and December (Myers 1998). To successfully pass over a dam, upstream migrating adults require a sounding pool of sufficient depth just downstream of the barrier, and a horizontal and vertical distance that is sufficiently short. Lacking these, a fish ladder is necessary. Summer dams rarely, if ever, provide these features. Flashboard dams may have concrete aprons that lengthen the horizontal distance from pool to barrier, making them even more difficult to pass.

In addition to blocking migrations to and from the ocean, summer dams can obstruct the natural movements of juveniles during their freshwater residence. Juvenile salmonids often migrate relatively long distances (*i.e.*, several km) in response to 1) changes in their environment (*e.g.*, summer warming, pollution events), 2) changes in habitat needs as they grow, and 3) competition with other individuals. The movements of stream-dwelling salmonids have been the subject of extensive research (Chapman 1962, Edmundson et al. 1968; Fausch and White 1981; Gowan et al. 1994). Although many juvenile salmonids are territorial or exhibit limited movement, many undergo extensive migrations (Gowan et al. 1994; Fausch and Young 1995). For example, salmonid fry often disperse downstream from headwater spawning sites. In California, hot summer temperatures can promote the movement and aggregation of juvenile steelhead into thermally stratified pools that provide coldwater refugia (Nielsen and Lisle 1994). Additional movements can occur as intraspecific competition for resources causes the dispersal of subordinate individuals (Chapman 1966; Everest and Chapman 1972; Hearn 1986). Downstream movements of juveniles may also occur in response to growth or simply because environmental conditions such as water depth or velocity are no longer suitable (Edmundson et al. 1968; Leider et al. 1986).

3.5 Summer Dam Effects on Salmonid Predation

Summer dams enhance the suitability of streams as habitat for a variety of species that prey upon juvenile salmonids. Temporary summer ponds increase stream depths, reduce current velocities, and increase a stream section's turnover rate (*i.e.*, the time required for a volume of water to pass through a stream section or impoundment). As a result, summer ponds can convert coldwater stream habitat into coolwater or warmwater habitats that are optimal for smallmouth bass (*Micropterus dolomieui*), largemouth bass *(M. salmoides*), and Sacramento pikeminnow (*Ptychocheilus grandis*). Green sunfish (*Lepomis cyanellus*), another potential predator, has wide environmental tolerances (Stuber et al. 1982) and likely benefits from the construction of summer dams. Preferring temperatures of 21-27^{o}C (Clancey 1980), smallmouth bass is a highly adaptable predator that lives in both stream and lake habitats (Edwards, Gebhart, and Maughan 1983). Sacramento pikeminnow, another cool-water piscivore, is a known predator of juvenile salmonids (Brown and Moyle 1981). Pikeminnow prefer cool water with almost no velocity (Dettman 1978); however, they are also highly tolerant of warm water. In the mainstem of California's Eel River, Sacramento pikeminnow are abundant in sections that routinely reach 26-28EC (CDFG, Region 3, unpublished data). The combination of deeper water, warmer temperatures and slower current velocities afforded by summer dams provides more suitable habitat for these species than the natural conditions of many small rivers and streams. Largemouth bass, a warmwater predator tolerant of temperatures over 30EC, may also benefit by the lacustrine habitat and warmer water temperatures created by summer dams (McCormick et al. 1981; Stuber et al. 1982).

Summer dams may also attract wading birds such as egrets and herons, which are highly effective fish predators. The ponded waters of summer dams generally have much less surface turbulence than free-flowing streams, and thus the shallow ponding of water makes rearing salmonids more visible targets to these predators. The belted kingfisher (*Ceryle alcyon*) and common merganser (*Mergus merganser*), other avian piscivores, also benefit from the clear placid waters of summer impoundments.

3.6 Cumulative Effects of Summer Dams

The largest threat of summer dams is their abundance. Each summer dam generates its own turbidity and sediment load; each may close the stream to fish movement; each may degrade juvenile salmon and steelhead rearing habitat; each changes the benthic community and interrupts energy flow, and each may kill some number of embryos, alevins or juveniles. Individually, these effects may be minor. However, each summer dam also fragments stream continuity. Habitat fragmentation results when a large area is subdivided into smaller, isolated patches (Wilcove et al. 1986) and is perhaps the most important problem threatening the survival of many species (Wilcox and Murphy 1985).

Summer dams affect habitat complexity. Habitat complexity has various components: structural (McMahon and Hartman 1989), hydraulic variation (Pearsons et al. 1992), and variations in depth, velocity and substrate (Gorman and Karr 1978; Angermeier and Schlosser 1989). With a series of summer dams there is less riffle and much more pool habitat. More pools mean less variation in depth and velocity, less benthic food production, and the reduction in stream currents for “food delivery”. With woody debris removal, a common activity associated with summer dams, there is less cover. A series of summer dams will break the upstream-downstream connectivity that is a natural feature of rivers (Allan 1995). Riffles are flooded and aquatic benthos are altered and become less available to juvenile salmonids. Each summer dam is a migratory challenge. Each summer dam contributes to changing water quality, especially stream warming. Each summer dam attracts a variety of fish predators. Together the impacts accumulate and the cumulative effect is significant. The habitats of anadromous salmonids extend from the ocean to spawning habitats that may be as far upstream as the headwaters of the watershed (Li et al. 1995). Threatened and endangered populations may not be able to recover when habitat is so splintered and migratory corridors are blocked.

A critical problem with summer dams is their effect on coho salmon, a species occurring in areas where many summer dams are constructed. In general, stream habitat loss is the single biggest cause of declines of coho salmon in the Pacific Northwest (Brown et al. 1994). Coho salmon are generally less tolerant to environmental degradation than steelhead with whom they generally share their habitat (Baker and Reynolds 1986). Most coho salmon mortality naturally occurs during the freshwater stages, as the result of destructive high flow events, winter freezing, summer droughts, or simply a lack of rearing space (Sandercock 1995). Summer dams worsen these conditions.

4.0 Recommendations

To recover populations of listed salmonids in California, we must limit further reductions in the quality and quantity of habitat for these species, and where possible we must restore those habitats. The annual installation of hundreds of summer dams in coastal rivers and tributaries has undoubtedly reduced available habitat for steelhead, coho salmon, and chinook salmon. These dams can also create barriers to the movements of juvenile stages of these species.

As a first step towards limiting these impacts, the following recommendations are proposed for application in coastal streams within the counties of Mendocino, Sonoma, Napa, Marin, Contra Costa, Alameda, Santa Clara, San Mateo, Santa Cruz, and Monterey. These areas have historically had extensive development of summer dams on streams supporting listed anadromous species. Coastal streams in these counties are jointly managed by California Department of Fish & Game, Region 3 and the National Marine Fisheries Service Santa Rosa field office.

1) Before being installed, each summer dam must undergo necessary environmental regulatory review. That review should entail the evaluation of potential direct and cumulative impacts of individual proposed summer dams on anadromous salmonids and their habitats. At a minimum, the dam owner must obtain a Fish & Game Code 1600 Streambed Alteration Agreement, and as necessary a Clean Water Act 404 permit from the Army Corps of Engineers (ACOE). Summer dams contributing to the degradation of juvenile rearing habitat should not be permitted.

2) Applicants seeking Stream Alteration Agreements for summer dams must demonstrate that their proposed project will not adversely affect juvenile salmonid habitat. Summer dams should not be approved or installed on stream reaches that provide viable juvenile rearing habitat for anadromous salmonids

during summer months. Reasons for inferring that a stream section does not provide habitat for juvenile anadromous salmonids include 1) documented evidence that the stream section is not within the historical range of anadromy (*e.g.*, upstream of a natural impassible falls), 2) excessively warm summer water temperatures throughout the reach (*e.g.*, on seven consecutive days, average maximum daily temperatures greater than 24EC; or sustained minimum daily [early morning] temperatures equal or greater than 20EC during seven consecutive days), or 3) other factual, site-specific information that demonstrates that the section to be impounded by the dam and the reach immediately downstream from the dam do not have the potential to support production of juvenile salmonids during summer months.

3) Summer dams should not be installed on seasonal stream reaches (*i.e.*, streams with intermittent flow), if they are an impediment to the movements of juvenile salmonids. Impoundments on seasonal stream reaches generally have no outflow during summer. Impoundments on seasonal reaches may impede both upstream and downstream movements of juvenile salmonids when spring flows subside.

4) For projects that do not adversely affect juvenile salmonid habitat and receive regulatory approval from CDFG and (as necessary) ACOE, summer dams should be constructed and operated consistent with the following guidelines:

   a) Annual construction or installation must not occur prior to June 15.

   b) Each year, the summer dam must be removed by the date specified in the Stream Alteration Agreement. The date of the removal will be contingent upon 1) the timing of upstream runs of anadromous species in the watershed, and 2) the presence on the summer dam of upstream passage facilities for adult salmonids. On the mainstem Russian River, which supports runs of chinook salmon, summer dams must be removed by August 15, unless they are equipped with NMFS/CDFG-approved upstream passage facilities for adult chinook salmon. All seasonal dams must be removed before commencement of the fall run of coho salmon and steelhead in the project stream. The timing of fish runs and dates for the removal of permitted dams will be determined by CDFG in consultation with NMFS.

c) All summer dams must be constructed in a manner that facilitates successful downstream passage of juvenile salmonids. At a minimum summer dams must have a notched spillway crest that concentrates and spills surface flows to a plunge-pool connected to the downstream channel. If sluices are necessary to convey fish to the plunge-pool, sluiceways must be sufficiently smooth to avoid abrasion of the fish (e.g., comparable to smoothed concrete or finished plywood). A V-notch is preferred for streams with average summer flows less than 5 cfs. For larger streams, notches should be sized to concentrate summer flows and provide minimum depths of 0.5 ft.

d) To avoid the stranding of fishes during the filling and emptying of the reservoir, summer dams must be equipped with a gate valve or other mechanism for maintaining downstream flow. A gate valve or other mechanism will facilitate the gradual emptying of the pond at season's end.

e) During the period of filling, average wetted width of the stream channel in the reach immediately downstream from the summer dam must not be reduced by more than 25% at any time.

f) At the time of dam removal, water in the impoundment must be lowered at a rate of not more than 2 inches per hour in order to minimize the risk of stranding fishes.

g) If water is diverted from the impoundment, the operators must use a CDFG and NMFS approved fish screen to avoid impingement or entrainment of fish.

h) Seasonal dams must not be stocked with fish of any species.

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Appendix A

Descriptions of the Life History and Status of coho salmon, chinook salmon, and steelhead in California

The ESA defines a “species” to include any “distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.” NMFS published a policy describing how it would apply the ESA definition of a “species” to anadromous salmonid species (56 FR 58612). More recently, NMFS and FWS published a joint policy, consistent with NMFS’ policy, regarding the definition of distinct population segments (61 FR 4722). To be considered an Evolutionarily Significant Unit (ESU), a population must satisfy two criteria: (1) It must be reproductively isolated from other population units of the same species, and (2) it must represent an important component in the evolutionary legacy of the biological species. The first criterion, reproductive isolation, need not be absolute, but must have been strong enough to permit evolutionarily important differences to occur in different population units. The second criterion is met if the population contributes substantially to the ecological/genetic diversity of the species as a whole. Section 3 of the ESA defines the term “endangered species” as “any species which is in danger of extinction throughout all or a significant portion of its range.” The term “threatened species” is defined as “any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.”

There are ten federally listed salmonid ESU’s within California (Table 1). Of these ten listed ESU’s, two are coho salmon (*Oncorhynchus kisutch*), three are chinook salmon (*O. tshawytscha*) and five are steelhead (*O. mykiss*). Please refer to Table 1 for a summary of the federally listed ESU’s within California. This table includes the geographic range, listing status, listing date and Federal Register notice for each ESU. In addition to the ten listed ESU’s, there are two other ESU’s that are either candidate species or do not warrant listing at this time. These two other ESU’s are the Central Valley Fall/Late-Fall chinook salmon which are candidate/not warranted and the Southern Oregon/Northern California Coastal chinook salmon ESU which is not warranted for listing at this time.

Table 1. Federally listed ESU's within California including the geographic range, listing status, listing date and Federal Register Notice.

Species	ESU	Geographic Range	Listing Status	Listing Date (FR Notice)
Coho	Southern Oregon/Northern California Coast	Cape Blanco, Oregon south to Mattole River	Threatened	06 May 1997 (62 FR 24588)
Coho	Central California Coast	From Mattole River south to San Lorenzo River	Threatened	31 Oct 1996 (61 FR 56138)
Chinook	California Coastal	South of Klamath River to the Russian River	Threatened	16 Sept1999 (64 FR 50394)
Chinook	Sacramento River winter-run	Sacramento River and tributaries	Endangered	04 Jan 1994 (59 FR 440)
Chinook	Central Valley Spring-run	Sacramento River and tributaries	Threatened	16 Sep1999 (64 FR 50394)
Steelhead	Northern California	Redwood Creek south to Gualala River	Threatened	07 June 2000 (65 FR 3607)
Steelhead	Central California Coast	Russian River south to Aptos Creek, and drainages of San Francisco and San Pablo Bay	Threatened	18 Aug 1997 (65 FR 43937)
Steelhead	Central Valley	Sacramento and San Joaquin Rivers and their tributaries	Threatened	19 March 1998 (63 FR 13347)
Steelhead	South-Central California Coast	Pajaro River south to, but not including the Santa Maria River	Threatened	18 Aug 1997 (62 FR 43937)
Steelhead	Southern California	Santa Maria River, south	Endangered	18 Aug 1997 (62 FR 43937)

Salmonid Species General Habitat Requirements

Habitat requirements of salmon and steelhead generally depend on the life history stage (Cederholm and Martin 1983; Bjornn and Reiser 1991). Generally, stream flow, water temperature, and water chemistry must be adequate for adult immigration, juvenile rearing, and juvenile emigration. Low stream flow, high water temperature, low dissolved oxygen, high turbidity and/or physical barriers can affect both juvenile and adult salmonids by delaying or halting upstream migration of adults or downstream migration of juveniles. Suitable water depth, velocity, and substrate composition are the primary requirements for spawning. Dissolved oxygen concentration, pH, and water temperature are factors affecting survival of incubating embryos. Small particles, sand, and fine sediment can fill interstitial spaces between substrate thereby reducing water-flow and dissolved oxygen levels within a redd (i.e., salmonid nest). Juvenile salmon and steelhead require living space (different combinations of water depth and velocity), shelter from predators and harsh environmental conditions, food resources, and suitable water quality and quantity for growth and survival during summer and winter (Bjornn and Reiser 1991).

The following information describes life history and biological requirements of coho salmon, chinook salmon and steelhead. Many of the habitat requirements detailed below are adversely affected by summer dams.

Coho Salmon (*Oncorhynchus kisutch*)

Coho salmon are native to the north Pacific Ocean. The historic distribution of coho salmon in North America included coastal streams from Alaska south to northwestern Mexico (Moyle 1976; Weitkamp *et al*. 1995). Currently the San Lorenzo River in Santa Cruz County, California is thought to have the southern-most persistent population of coho salmon in North America (Weitkamp *et al*. 1995). Coho salmon are also found in Asia from the Anadyr River, Russia, south to Hokkaido, Japan and tributaries of Peter the Great Bay on the Sea of Japan (Hart 1973; Sandercock 1991).

Life History and Biological Requirements

Coho salmon are typically associated with small to moderately-sized coastal streams characterized by heavily forested watersheds; perennially-flowing reaches of cool, high-quality water; dense riparian canopy; deep pools with abundant overhead cover; instream cover consisting of large, stable woody debris and undercut banks; and gravel or cobble substrates.

The life history of the coho salmon in California has been well documented by Shapovalov and Taft (1954) and Hassler (1987). In contrast to the life history patterns of other anadromous salmonids, coho salmon in California generally exhibit a relatively simple 3-year life cycle. Adult salmon typically begin the freshwater migration from the ocean to their natal streams after heavy late-fall or winter rains breach the sand bars at the mouths of coastal streams (Sandercock 1991). Delays in river entry of over a month are not unusual (Salo and Bayliff 1958; Eames *et al*. 1981). Migration continues to March, generally peaking in December and January, with spawning occurring shortly after reaching the spawning ground (Shapovalov and Taft 1954).

Female coho salmon choose spawning sites usually near the head of a riffle, just below a pool, where water changes from a laminar to a turbulent flow and there is small to medium gravel substrate. The flow characteristics of the location of the redd usually ensure good aeration of eggs and embryos, and the flushing of waste products. The water circulation in these areas also facilitates fry emergence from the gravel. Preferred spawning grounds have nearby overhead and submerged cover for holding adults; water depth of 10-54 cm; water velocities of 20-80 cm/s; clean, loosely compacted gravel (1.3-12.7 cm diameter) with less than 20 percent fine silt or sand content; cool water (4-10EC) with high dissolved oxygen (8 mg/l); and an intergravel flow sufficient to aerate the eggs. The lack of suitable gravel limits successful spawning in many streams.

Each female builds a series of redds, moving upstream as she does so, and deposits a few hundred eggs in each. Fecundity of coho salmon is directly proportional to female size; coho salmon may deposit from 1,000-7,600 eggs (reviewed in Sandercock 1991). Coho salmon may spawn in more than one redd and with more than one partner (Sandercock 1991). The female may guard a nest for up to two weeks (Briggs 1953). Coho salmon are semelparous, they die after their first spawning season.

The eggs generally hatch between 4 to 8 weeks, depending on water temperature. Survival and development rates depend on temperature and dissolved oxygen levels within the redd.

According to Baker and Reynolds (1986), under optimum conditions, mortality during this period can be as low as 10 percent; under adverse conditions of high scouring flows or heavy siltation, mortality may be close to 100 percent. McMahon (1983) found that egg and fry survival drops sharply when fines make up 15 percent or more of the substrate. The newly-hatched fry remain in the gravel from two to seven weeks until emergence from the gravels (Shapovalov and Taft 1954).

Low summer flows reduce potential rearing areas, may cause stranding in isolated pools, and increase vulnerability to predators (Sandercock 1991). Also the combination of reduced flows and high ambient air temperatures can raise the water temperature to the upper lethal limit of 25EC for juvenile coho (Brett 1952). As they grow, they often occupy habitat at the heads of pools, which generally provide an optimum mix of high food availability and good cover with low swimming cost (Nielsen 1992). As the fish continue to grow, they move into deeper water and expand their territories until, by July and August, they are in the deep pools. Juvenile coho salmon prefer well shaded pools at least 1 m deep with dense overhead cover; abundant submerged cover composed of undercut banks, logs, roots, and other woody debris; preferred water temperatures of 12-15EC, but not exceeding 22-25EC for extended time periods; dissolved oxygen levels of 4-9 mg/l; and water velocities of 9-24 cm/sec in pools and 31-46 cm/sec in riffles. Water temperatures for good survival and growth of juvenile coho salmon range from 10-15EC (Bell 1973; McMahon 1983). Growth is slowed considerably at 18EC and ceases at 20EC (Stein *et al*. 1972; Bell 1973).

Preferred rearing habitat has little or no turbidity and high sustained invertebrate forage production. Juvenile coho salmon feed primarily on drifting terrestrial insects, much of which are produced in the riparian canopy, and on aquatic invertebrates growing in the interstices of the substrate and in the leaf litter in the pools. As water temperatures decrease in the fall and winter months, fish stop or reduce feeding due to lack of food or in response to the colder water, and growth rates slow down. During December-February, winter rains cause increased stream flows and by March, following peak flows, fish again feed heavily on insects and crustaceans and grow rapidly.

In the spring, as yearlings, juvenile coho salmon undergo a physiological process, called smoltification, which prepares them for living in the marine environment. They begin to migrate downstream to the ocean during late March and early April, and out migration usually peaks in mid-May, if conditions are favorable. At this point, the smolts are about 10-13 cm in length. After entering the ocean, the immature salmon initially remain in nearshore waters close to their parent stream. They gradually move northward, staying over the continental shelf (Brown *et al*. 1994). Although it is thought that they range widely in the north Pacific, movements of coho salmon from California are poorly known.

Status of California Coho salmon stocks

A comprehensive review of estimates of historic abundance, decline and present status of coho salmon in California is provided by Brown *et al*. (1994). They estimated that coho salmon annual spawning population in California ranged between 200,000 and 500,000 fish in the 1940s, which declined to about 100,000 fish by the 1960s, followed by a further decline to about 31,000 fish by 1991, of which 57 percent were artificially propagated. The other 43 percent (13,240) were natural spawners, which included naturally-produced, wild fish and naturalized (hatchery-influenced) fish. Brown *et al*. (1994) cautioned that this estimate could be overstated by 50 percent or more. Of the 13,240, only about 5,000 were naturally-produced, wild coho salmon without hatchery influence, and many of these were in individual stream populations of fewer than 100 fish each. In summary, Brown *et al*. (1994) concluded that the California coho salmon population had declined more than 94 percent since the 1940s, with the greatest decline occurring since the 1960s. Both coho salmon ESU's within California are federally listed as threatened species.

Chinook Salmon (*Oncorhynchus tshawytscha*)

Chinook salmon historically ranged from the Ventura River in southern California north to Point Hope, Alaska, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). Myers *et al*. (1998) reports no viable populations of chinook salmon south of San Francisco, California. Although chinook salmon is a wide-ranging species, it is the least abundant Pacific salmon in North America (Moyle 1976; Page and Burr 1991).

Life History and Biological Requirements

Chinook salmon is anadromous and the largest member of *Oncorhynchus*, with adults weighing more than 120 pounds having been reported from North American waters (Scott and Crossman 1973; Page and Burr 1991). Chinook salmon exhibit two main life history strategies: ocean-type fish and river-type fish (Healy 1991). Ocean-type fish typically are fall- or winter-run fish that spawn shortly after entering freshwater and their offspring emigrate shortly after emergence from the redd. River-type fish are typically spring- or summer-run fish that have a protracted adult freshwater residency, sometimes spawning several months after entering freshwater. Progeny of river-type fish frequently spend one or more years in freshwater before emigrating.

Chinook salmon generally remain in the ocean for two to five years (Healey 1991), and tend to stay along the California and Oregon coasts. Some chinook salmon return from the ocean to spawn one or more years before full-sized adults return, and are referred to as jacks (males) and jills (females). Fall-run chinook salmon upstream migration occurs from June through December with a peak in September and October. Spawning occurs from late-September through December with a peak in late-October. These fish typically enter freshwater at an advanced stage of maturity, move rapidly to their spawning areas on the mainstem or lower tributaries of rivers, and spawn within a few weeks of freshwater entry (Healey 1991). Run timing is also, in part, a response to stream flow characteristics.

Egg deposition must be timed to ensure that fry emerge during the following spring at a time when the river or estuary productivity is sufficient for juvenile survival and growth. Adult female chinook salmon prepare redds in stream areas with suitable gravel composition, water depth, and velocity. Spawning generally occurs in swift, relatively shallow riffles or along the edges of fast runs at depths greater than 24 cm. Optimal spawning temperatures range between 5.6-13.9EC. Redds vary widely in size and location within the river. Preferred spawning substrate is clean, loose gravel, mostly sized between 1.3-10.2 cm, with no more than 5 percent fines. Gravels are unsuitable when they have been cemented with clay or fines or when sediments settle out onto redds, reducing intergravel percolation (62 FR 24588). Minimum intragravel percolation rate depends on flow rate, water depth, and water quality. The percolation rate must be adequate to maintain oxygen delivery to the eggs and remove metabolic wastes. The chinook salmon's need for a strong, constant level of subsurface flow may indicate that suitable spawning habitat is more limited in most rivers than superficial observation would suggest. After depositing eggs in a redd, adult chinook salmon guard the redd from 4 to 25 days before dying.

Chinook salmon eggs incubate for about 30 to 150 days, depending on water temperature. Successful incubation depends on several factors including dissolved oxygen levels, temperature, substrate size, amount of fine sediment, and water velocity. Maximum survival of incubating eggs and pre-emergent fry occurs at water temperatures between 5.6-13.3EC with a preferred temperature of 11.1EC. Fry emergence begins in December and continues into mid-April (Leidy and Leidy 1984). Emergence can be hindered if the interstitial spaces in the redd are not large enough to permit passage of the fry. In laboratory studies, Bjornn and Reiser (1991) observed that chinook salmon and steelhead fry had difficulty emerging from gravel when fine sediments (6.4 mm or less) exceeded 30-40 percent by volume.

After emergence, chinook salmon fry seek out areas behind fallen trees, back eddies, undercut banks and other areas of bank cover (Everest and Chapman 1972). As they grow larger, their habitat preferences change. Juveniles move away from stream margins and begin to use deeper water areas with slightly faster water velocities, but continue to use available cover to minimize the risk of predation and reduce energy expenditure. Fish size appears to be positively correlated with water velocity and depth (Chapman and Bjornn 1969; Everest and Chapman 1972). Optimal temperatures for both chinook salmon fry and fingerlings range from 12-14EC, with maximum growth rates at 12.8EC (Boles 1988). Chinook feed on small terrestrial and aquatic insects and aquatic crustaceans. Cover, in the form of rocks, submerged aquatic vegetation, logs, riparian vegetation, and undercut banks provide food, shade, and protect juveniles from predation.

The low flows, high temperatures, and sand bars that develop in smaller coastal rivers during the summer months favor an ocean-type life history (Kostow 1995). With this life history, smolts typically outmigrate as subyearlings during April through July (Myers *et al.* 1998). The ocean-type chinook salmon in California tend to use estuaries and coastal areas for rearing more extensively than stream-type chinook salmon. The brackish water areas in estuaries moderate the physiological stress that occurs during parr-smolt transitions.

Status of California Chinook Salmon Stocks

Although northern coastal California streams support small, sporadically monitored populations of fall-run chinook salmon and the Central Valley supports four runs of chinook salmon, estimates of absolute population abundance are not available (Myers *et al*. 1998). Data available to assess trends in abundance are limited. Recent trends have been mixed, with predominately strong negative trends in the Eel River Basin and in streams that are farther south along the California coast (Myers *et al*. 1998). Five chinook salmon ESU's have been identified in California. Three ESU's are federally listed at this time, one is listed as endangered and the other two are listed as threatened (Table 1).

Steelhead (*Oncorhynchus mykiss*)

Steelhead are native to the north Pacific Ocean and in North America are found in coastal streams from Alaska south to northwestern Mexico (Moyle 1976; Busby *et al*. 1996). At this time NMFS has listed only the anadromous life form of rainbow trout: steelhead.

<u>Life History and Biological Requirements</u>

Steelhead spend from one to five years in saltwater, however, two to three years are most common (Busby *et al.* 1996). Some return as "half-pounders" that over-winter one season in freshwater before returning to the ocean in the spring. The distribution of steelhead in the ocean is not well known. Coded-wire tag recoveries indicate that steelhead migrate both north and south along the continental shelf (Barnhart 1986).

The timing of upstream migration is correlated with higher flow events, such as freshets or sand bar breaches, and associated lower water temperatures. The minimum stream depth necessary for successful upstream migration is 13 cm (Thompson 1972). The preferred water velocity for upstream migration is in the range of 40-90 cm/s, with a maximum velocity, beyond which upstream migration is not likely to occur, of 240 cm/s (Thompson 1972; Smith 1973). There are two types of steelhead, summer steelhead and winter steelhead. Summer steelhead return to freshwater during June through September, migrate inland toward spawning areas, overwinter in the larger rivers, and then resume migration to natal streams and spawn (Meehan and Bjornn 1991). Winter steelhead return to freshwater in autumn or winter, migrate to spawning areas, and then spawn in late winter or spring. Upstream migration of winter steelhead occurs from September through May with the peak run occurring in February (CDFG 1997). Most spawning takes place from January through April. Steelhead may spawn more than once before dying (iteroparity), in contrast to other species of the *Oncorhynchus* genus. Repeat spawning rates typically range from 13-24 percent in California coastal streams.

Because rearing juvenile steelhead reside in freshwater all year, adequate flow and temperature are important to the population at all times (CDFG 1997). Generally, throughout their range in California, steelhead that are successful in surviving to adulthood spend at least two years in freshwater before emigrating downstream. Emigration appears to be more closely associated with size than age. In Waddell Creek, Shapovalov and Taft (1954) found steelhead juveniles migrating downstream at all times of the year with the largest numbers of age 0+ and yearling steelhead moving downstream during spring and summer. Smolts can range from 14-21 cm in length.

Steelhead spawn in cool, clear streams featuring suitable water depth, gravel size, and current velocity. Intermittent streams may be used for spawning (Barnhart 1986; Everest 1973). Reiser and Bjornn (1979) found that gravels of 1.3-11.7 cm in diameter and flows of approximately 123 cm/s were preferred by steelhead. The survival of embryos is reduced when fines of less than 6.4 mm comprise 20-25 percent of the substrate. Studies have shown a higher survival of embryos when intragravel velocities exceed 20 cm/hr (Phillips and Campbell 1961; Coble 1961). The number of days required for steelhead eggs to hatch is inversely proportional to water temperature and varies from about 19 days at 15.6^{o}C to about 80 days at 5.6^{o}C. Fry typically emerge from the gravel two to three weeks after hatching (Barnhart 1986).

Upon emerging from the gravel, fry rear in edgewater habitats and move gradually into pools and riffles as they grow larger. Older fry establish territories which they defend. Cover is extremely important in determining distribution and abundance, with more cover leading to more fish (Bjornn and Reiser 1991). Young steelhead feed on a wide variety of aquatic and terrestrial insects, and emerging fry are sometimes preyed upon by older juveniles. In winter, they become inactive and hide in any available cover, including gravel or woody debris.

Water temperature influences the growth rate, population density, swimming ability, ability to capture and metabolize food, and ability to withstand disease of these rearing juveniles (Barnhart 1986; Bjornn and Reiser 1991). Rearing steelhead juveniles prefer water temperatures of 7.2-14.4^{o}C and have an upper lethal limit of 23.9^{o}C. They can survive up to 27EC with saturated dissolved oxygen conditions and a plentiful food supply. Fluctuating diurnal water temperatures also aid in survivability of salmonids (Busby *et al.* 1996).

Dissolved oxygen (DO) levels of 6.5-7.0 mg/l affected the migration and swimming performance of steelhead juveniles at all temperatures (Davis *et al.* 1963). Reiser and Bjornn (1979) recommended that DO concentrations remain at or near saturation levels with temporary reductions no lower than 5.0 mg/l for successful rearing of juvenile steelhead. Low DO levels decrease the rate of metabolism, swimming speed, growth rate, food consumption rate, efficiency of food utilization, behavior, and ultimately the survival of the juveniles.

During rearing, suspended and deposited fine sediments can directly affect salmonids by abrading and clogging gills, and indirectly cause reduced feeding, avoidance reactions, destruction of food supplies, reduced egg and alevin survival, and changed rearing habitat (Reiser and Bjornn 1979). Bell (1973) found that silt loads of less than 25 mg/l permit good rearing conditions for juvenile salmonids.

Status of California Steelhead Stocks

Historically, steelhead likely inhabited most coastal and many inland streams along the west coast of the United States. During this century, however, over 23 indigenous, naturally reproducing stocks have been extirpated, and many more are at risk for extinction. The most recent data show that current summer and winter steelhead abundance is well below estimates from the 1980s, and is greatly reduced from levels in the 1960s (65 FR 6960).

Only two estimates of historical (pre-1960s) abundance are available: an average of about 500 adults in Waddell Creek in the 1930s and early 1940s (Shapovalov and Taft 1954), and 20,000 steelhead in the San Lorenzo River before 1965 (Johnson 1964). In the mid-1960s, 94,000 steelhead adults were estimated to spawn in central California rivers, including 50,000 and 19,000 fish in the Russian and San Lorenzo rivers, respectively (CDFG 1965). Recent estimates indicate an abundance of about 7,000 fish in the Russian River (including hatchery steelhead) and about 500 fish in the San Lorenzo River. These estimates suggest that recent total abundance of steelhead in these two rivers is less than 15 percent of their abundance 30 years ago. Recent estimates for several other streams (Lagunitas Creek, Waddell Creek, Scott Creek, San Vincente Creek, Soquel Creek, and Aptos Creek) indicate individual run sizes of 500 fish or fewer. Steelhead in most tributaries to San Francisco and San Pablo bays have been virtually extirpated (McEwan and Jackson 1996). Fair to good runs of steelhead apparently still occur in coastal Marin County tributaries. In a 1994 to 1997 survey of 30 San Francisco Bay watersheds, steelhead occurred in small numbers at 41 percent of the sites, including the Guadalupe River, San Lorenzo Creek, Corte Madera Creek, and Walnut Creek (Leidy 1997).

Six steelhead ESU's have been identified in California. Five are federally listed at this time. Of the five listed ESU's, one is listed as endangered and four are listed as threatened (Table 1).

Appendix B

CDFG memorandum concerning investigation of use of a summer pond by steelhead in Austin Creek

State of California

The Resources Agency

SON - ST K. S. F. Am
Austin Creek

Memorandum

To : Files

Date: September 26, 1984

From : **Department of Fish and Game** -- Bill Cox

Subject: Austin Creek, Sonoma County

One issue which has been raised in regard to the summer dams on Austin Creek is the suitability of the impoundments for the production of steelhead. Many people have reached the logical conclusion that more water means more fish, and many have pointed out the large numbers of fish which could be seen in the impoundments. Department biologists agree that generally more water does mean more fish and that large numbers of fish can be seen in the impoundments; we have maintained, however, that more water in impoundments produce more roach, sucker, and squawfish but not more trout and that it is the roach, sucker, and squawfish that can be seen in the impoundments.

To determine the suitability of the impoundments for steelhead production, a known number of marked fish were planted in an impoundment, allowed to grow through the summer, then recaptured in the fall. The fish used in the experiment were locally caught by electro-fishing and represented the size and strain of fish which would be naturally in the stream sections impounded or which could be added to the impoundments through fish rescue. The impoundment used in the experiment was selected for its relatively small size to facilitate drainage during recapture, for having a dam forming its upstream limit so that we would be able to measure the production of the impoundment not of the free-flowing stream above the impoundment, and for the presence of a substantial amount of cover (logs, tree stumps, boulders) to provide the best chance to successfully produce steelhead. We hoped to be able to completely isolate the impoundment to prevent any fish from entering or leaving; we believe this goal was achieved at the upstream end but there was some spill over the flashboards downstream early in the summer which may have allowed some fish to leave the impoundment but would not have allowed fish to enter.

On June 19, 1984 480 steelhead rainbow trout with adipose fin clips were released into the impoundment behind the Wayland/Baron summer dam. Mortality on release was 12 fish. The marked fish were acquired from lower Kidd Creek, lower Ward Creek, Austin Creek at the confluence of Ward Creek, Austin Creek at King Flats, and Austin Creek at the Wayland/Baron dam (25 fish). The 25 fish captured above the Wayland/Baron dam represented an unknown, but probably large percentage of the fish initially in the impoundment behind the Wayland/Baron dam; all 25 fish were found in the outlet pipe of the Hyde dam which forms the upper limit of the Wayland/Baron impoundment. The marked fish planted in the impoundment were of various sizes representing the full range of steelhead lengths found in the area during the period in which they were captured.

Although the marked fish were not measured they were similar in size to steelhead which had been captured in downstream migrant traps in Austin Creek and Ward Creek; measured steelhead from the downstream traps for the period June 11 to June 15, 1984 had the following length distribution:

Forklength (mm)	# of steelhead
30-39	2
40-49	22
50-59	157
60-69	69
70-79	6
80-89	2
90-99	1
100-109	2

On September 24, 1984 the Wayland/Baron impoundment was drained and electro-fished to recapture the marked steelhead and assess their survival in the impoundment through the summer. First a bundle of tree branches was pulled through the outlet pipe of the Wayland/Baron dam to remove any fish living in the pipe, then a net was set below the outlet pipe and the flashboards pulled to drain the impoundment as far as possible; the net would capture any fish that were carried out of the impoundment as it drained. After the water level had dropped about 18 inches, the remaining impoundment was electro-fished with a Coffelt streamside electro-fisher powered by a gasoline engine driven generator. The pulsator and generator were moved several times to obtain full coverage of the impoundment. Maximum depth was about 4 feet which allowed the entire impoundment to be waded. The electro-fisher had sufficient power to stun fish as small as 30 mm in water 4 feet deep. Water clarity was excellent when the electro-fishing was started but degenerated rapidly due to the silt and dead algae on the bottom. As there was no current, turbidity remained localized so that each new area that was shocked was initially clear. Even when the water became turbid stunned fish could be seen at depths of at least 2 feet. The steelhead in the impoundment were known to be 50 mm or more in length, an effort was made to capture all fish of all species in this size range. Western roach were present in very large numbers in a length range of 30-40 mm; a few of these were captured but no attempt was made to capture them all. The fish could be readily identified while still in the water. The bundle of tree branches was used to chase any fish out of the outlet pipe in the Hyde dam at the upper end of the impoundment; none were found there. The impoundment was allowed to clear for about two hours and was electro-fished a second time.

During the time the impoundment was clearing two sections of free-flowing stream below the dam were electro-fished with a backpack electro-fisher. Each section was blocked off with small nets to prevent fish from entering or leaving the section while it was being sampled. As in the impoundment, very small roach were quite abundant and no effort was made to capture them all.

Only 2 steelhead were recovered from the Wayland/Baron impoundment, one was marked (98 mm FL) the other (83 mm FL), found in the net below the outlet pipe, was not. On the first electro-fishing pass we captured 1 steelhead (SH), 69 sucker (SKR), 113 roach (RCH), 2 tule perch (TP), and

31 sculpin (SCP). On the second electro-fishing pass we captured 41 sucker, 51 roach, 1 squawfish (SQ), 2 tule perch, and 16 sculpin.

The length frequency distribution for all the steelhead and a representative sample of the other species caught above and below the dam is shown on the attached table.

The Seber and LeCren two pass population estimate for steelhead is 2 + or - 0, and for marked steelhead 1 + or - 0, at the 95% confidence level. The survival rate of the marked steelhead in the impoundment was 1 out of 468. It may be logical to assume that the survival of steelhead already in the impoundment at the time of planting was the same but this would indicate that there were about 468 steelhead present, this is unlikely.

In the free-flowing stream just below the Wayland/Baron dam the first 25 meters yielded 3 steelhead, 4 sucker, 47 roach, 4 squawfish, and 2 sculpin. The next 37 meters was electro-fished with 2 passes; the first pass yielded 14 steelhead, 13 sucker, 19 roach, and 2 sculpin, the second pass yielded 4 steelhead, 4 sucker. 1 roach, and 1 squawfish. The population estimate for steelhead in the entire 62 meter section is 23 with a 95% confidence limit of 21 to 28.

The impoundment measured approximately 75 x 10 meters and produced only 2 steelhead, even with the planting in early summer of 468 fish, the free-flowing section of stream measured approximately 62 x 1.5 meters and produced 23 steelhead. The stream section produced 14 times as many steelhead per unit length of stream as the impoundment and 93 times as many per unit of surface area.

The results of this experiment support the Department's position that the Austin Creek summer dams are eliminating steelhead nursery habitat by changing a free-flowing stream into a series of ponds unsuited to producing steelhead.

Bill Cox

Bill Cox
Unit Fishery Biologist
Sonoma/Marin

FISH LENGTH FREQUENCY DISTRIBUTION

	IMPOUNDMENT ABOVE DAM						STREAM BELOW DAM	
Species code	SH	SKR	RCH	SQ	TP	SCP	SH	SQ
Forklength (mm)								
10-19								
20-29								
30-39			8					
40-49			1					
50-59		1	2					
60-69		5	1				1	1
70-79		4	3				8	3
80-89	1	3	4			3	5	1
90-99	1	2			1	1	6	
100-109					2	1	1	
110-119		1			1	3		
120-129						1		
130-139								
140-149		1						
150-159		2						
160-169		2						
170-179		3						
180-189		2						
190-199		1						
200-209		2						
210-219		3						
220-229								
230-239				1				
240-249								
250-259								
260-269								
270-279								
280-289								
290-299								
300+		1						