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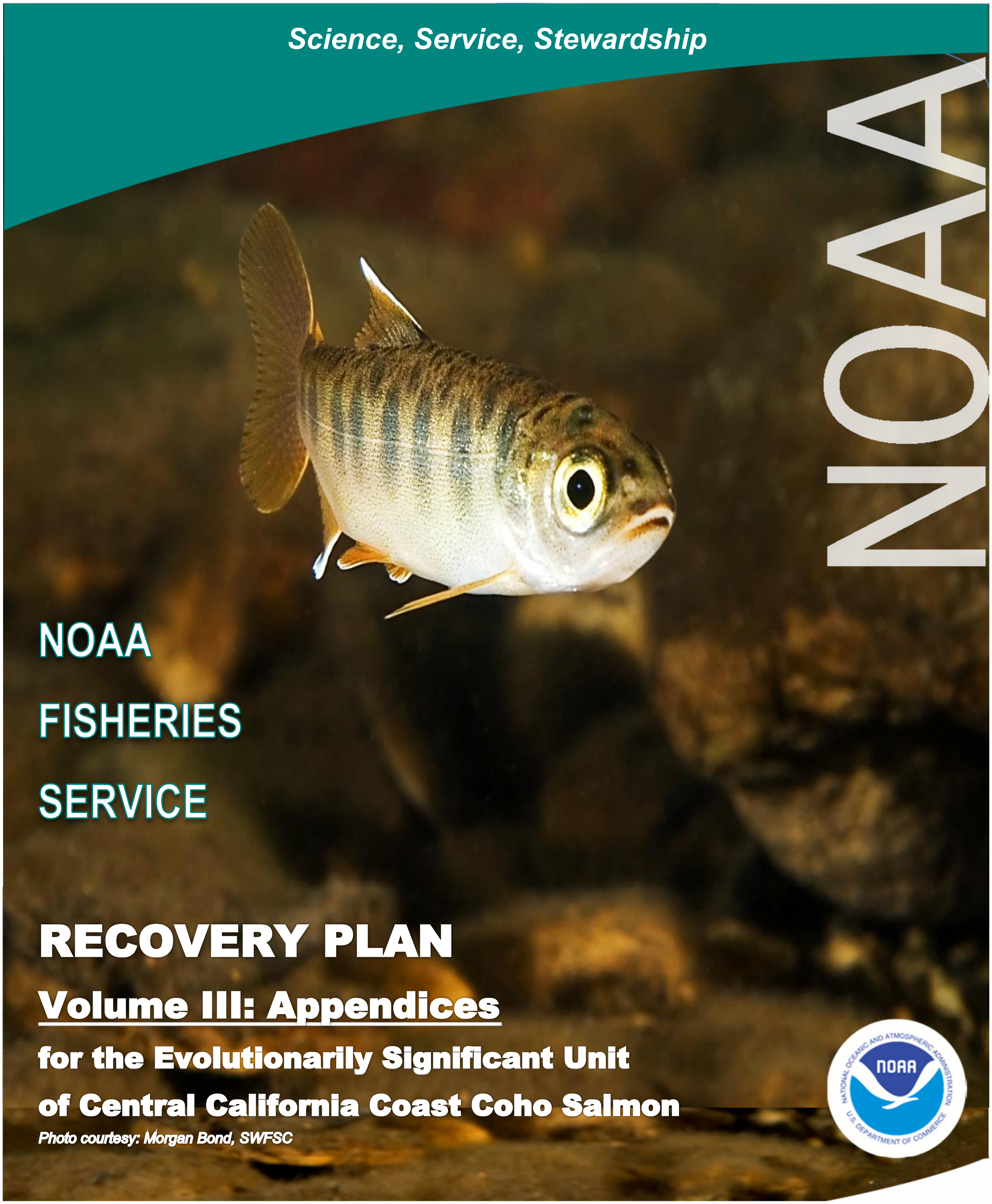

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NOAA FISHERIES SERVICE

RECOVERY PLAN

Volume III: Appendices

for the Evolutionarily Significant Unit of Central California Coast Coho Salmon

Photo courtesy: Morgan Bond, SWFSC

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First page photo courtesy of CCC coho salmon juvenile, Scott Creek, Santa Cruz County, Morgan Bond, Southwest Fisheries Science Center.

APPENDIX A

MARINE & CLIMATE

North Central California Coast Recovery Domain
CCC Coho ESU Recovery Plan

Marine and Climate

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Marine Habitat

"Thus blaming "ocean conditions" for salmon declines is a lot like blaming the iceberg for sinking the Titanic, while ignoring the many human errors that put the ship on course for the fatal collision. Managers have optimistically thought that salmon populations were unsinkable, needing only occasional course corrections such as hatcheries or removal of small dams, to continue to go forward. The listings as endangered species of the winter and spring runs of Central Valley Chinook were warnings of approaching disaster on an even larger scale. "Ocean conditions" may be the potential icebergs for salmon populations but the ship is being steered by us humans. Salmon populations can be managed to avoid an irreversible crash, but continuing on our present course could result in loss of a valuable and iconic fishery."

Peter B. Moyle, Professor of Fish Biology, and University of California

Marine Distribution of CCC coho salmon

CCC coho salmon spend the majority of their lives at sea, therefore evaluating marine distribution and associated stresses and threats is a necessary component for recovery planning. The evaluation is challenging because migration patterns and ecology of coho salmon in the marine environment are highly variable and incompletely understood.

Coho salmon occur in the epipelagic zone (top layer of the water column) in the open ocean, at observed depths of from about 10 to 25 meters (summarized by Quinn 2005). Information from hatchery releases in the range of the CCC coho salmon ESU, found that most individuals were recovered in northern California, followed by southern Oregon, with a small number found in Washington state waters (<1 percent). Based on these data, and assuming a correlation in migration patterns between hatchery and wild populations, it appears the majority of adult CCC coho salmon are located off of California and Oregon. Weitkamp and Neely (2002) found a high diversity of ocean migration patterns which suggests individuals within a population may be widely distributed in the coastal ocean areas.

Marine Phase of the coho salmon life cycle

Two life stages of coho salmon occur in the eastern Pacific Ocean; sub-adults and adults. These life stages occupy different environments and are exposed to different associated stresses and threats encountered within those areas. The sub-adult life stage is defined as individuals inhabiting nearshore marine areas, generally near the continental shelf. The adult life stage is defined as individuals occupying the larger offshore marine environment. Coho salmon utilize nearshore areas of the ocean for a number of months before they enter the open ocean, where they remain for eighteen months or more before they return to their natal streams as spawners. Some coho salmon never move offshore to the open ocean, but instead move north along the continental shelf and grow to adulthood in nearshore areas before returning to spawn (Sandercock 1991). Coho salmon survival in the marine environment is largely affected by individual attributes, such as body size, growth rate, and ocean entry date; as well as environmental conditions, predation and competition (Quinn 2005).

Sub-Adult Life Stage

CCC coho salmon appear to remain in nearshore habitats close to their watershed of origin for the first few months of ocean residency. A life history study by Shapovalov and Taft (1954) on coho salmon in Waddell Creek on the central California coast, showed coho stayed within 150 kilometers of shore for a few months following ocean entry. Other studies using recoveries of coded-wire tags (CWTs) also indicate coho salmon remain in the region of their natal stream during their first summer in the ocean (Fisher and Pearcy 1988). Residency in natal nearshore areas may be linked to smolt density and feeding conditions in those areas and likely varies from year to year (Healey 1980).

The first summer and fall at sea critically influences the likelihood of survival to adulthood (Hartt 1980; Beamish *et al.* 2004). Van Doornik *et al* (2007) and Beamish and Mahnken (2001) correlated the abundance of juveniles caught in September, with adult abundance the following year and determined the success of each year-class was largely set during the first summer in the ocean. The close correlation between jack (two-year old male) abundance and adult abundance further indicates the early ocean period is critical to adult salmon abundance, and that most mortality occurs after the first summer of ocean residency (Quinn 2005). Juvenile salmon that fail to reach a critical size by the end of their first marine summer do not survive the following winter, suggesting that attaining a large size in a short period of time is necessary for survival. Beamish *et al.* (2004) and Holtby *et al.* (1990) found a strong link between growth and survival, with faster growing coho salmon being more likely to survive the winter than slower growing fish, especially in years of low ocean productivity. Increased growth rates are influenced by both genetic disposition (Beamish *et al.* 2004) and feeding opportunities. Upon ocean entry, juvenile coho primarily feed on marine invertebrates, but transition to larger prey (predominantly fish) as they increase in size (Groot and Margolis 1991). Beamish and Mahnken (2001) also found within the first six months of ocean entry, early mortality is influenced by predation, and to a lesser degree a physiologically-based mortality.

Adult Life Stage

Once coho salmon enter the open ocean, they are subject to different food availability, environmental conditions, and stressors than present in the nearshore environment. The growth and survival of adult coho is closely linked to marine productivity, which is controlled by complex physical and biological processes that are dynamic and vary over space and time. Shifts in salmon abundance due to climatic variation can be large and sudden (Beamish *et al.* 1999). Short and long-term cycles in climate (*e.g.*, El Niño/La Niña and the Pacific Decadal Oscillation (PDO)) affect adult size, abundance, and distribution at sea, as does inherent year-to-year variation in environmental conditions not associated with climatic cycles.

Several studies have related ocean conditions specifically to coho salmon production (Cole 2000), ocean survival (Ryding and Skalski 1999; Koslow *et al.* 2002), and spatial and temporal patterns of survival and body size (Hobday and Boehlert 2001; Wells *et al.* 2006). The association between survival and climate operate via the availability of nutrients regulating the food supply and competition for food (Beamish and Mahnken 2001). For example, the 1983 El Niño resulted in increased adult mortality and decreased average size for Oregon's returning coho and Chinook salmon. Juvenile coho salmon entering the ocean in the spring of 1983 had low survival rates, resulting in low adult returns in 1985 (Johnson 1988). Larger-scale decadal to multi-decadal events also have been shown to affect ocean productivity and coho salmon abundance (Pearcy 1992; Lawson 1993; Hare and Francis 1995; Beamish *et al.* 1997; Mantua *et al.* 1997; Beamish *et al.* 1999). Although salmon evolved in this variable environment and are well suited to withstand climactic changes, the resiliency of the adult population has been reduced by the loss of life history diversity, low population abundance, cohort loss, and fragmentation of the spatial population structure. Changes in the freshwater environment have further adversely affected the ability of coho salmon to respond to the natural variability in ocean conditions.

Marine Survival

As noted above, marine survival and successful return as adults to spawn in natal streams is critically dependent on the first few months at sea (Peterman 1992; Unwin and Glova 1997; Ryding and Skalski 1999; Koslow *et al.* 2002). In a detailed study of Puget Sound hatchery coho salmon, Matthews and Buckley (1976), estimated 13 percent survival during the first six months at sea; and after twelve months survival was estimated at nine percent. The survival rate during the second year at sea was 99 percent.

Marine environmental conditions are also a major determinant in adult returns (Bradford 1995; Logerwell *et al*. 2003; Quinn 2005). In general, coho salmon marine survival is about 10 percent (Bradford 1995), although there is a wide range in survival rates (from <1 percent to about 21 percent) depending upon population location and ocean conditions (Beamish *et al*. 2000; Quinn 2005)[1]. Changes in marine survival rates often have large impacts on adult returns (Beamish et al. 2000; Logerwell *et al*. 2003). Recent data from across the range of coho salmon on the coast of California and Oregon reveal a 73 percent decline in returning adults in 2007/08 compared to

[1] Few data exist for coho salmon from California. Most marine survival data reported above are from Oregon, Washington, and Canadian coho populations. NMFS assumes marine survival rates for CCC coho salmon will be similar.

the same cohort in 2004/05 (MacFarlane *et al.* 2008). The Wells Ocean Productivity Index, a measure of Central California ocean productivity, predicted poor conditions during the spring and summer of 2006, when juvenile coho from the 2004/05 cohort entered the ocean (MacFarlane *et al.* 2008). However, strong upwelling in the spring of 2007 may have resulted in better ocean conditions for the 2007 coho salmon cohort.

Stresses

Major stresses identified which potentially affect coho salmon marine survival include: (1) reduced quantity and/or quality of food resources; and (2) reduced genetic and life history diversity. Although poorly understood, the complex physical and biological processes determining feeding opportunities have a large influence on the growth and survival of coho at sea, especially in the first six months of ocean residency. What we do know is that the life history plasticity and genetic diversity of coho salmon entering the ocean environment has been dramatically decreased. The loss of diversity has reduced the growth opportunities, the survival of populations, and the overall resiliency of the ESU. Predation and competition can also influence the size of the population in certain circumstances. An analysis of stresses affecting coho salmon at sea is summarized by life stage below.

Reduced quantity or quality of food

Oceanographic condition (*e.g.*, upwelling rates, sea-surface temperatures, *etc.*) is the major factor influencing salmonid food quantity and quality in the marine environment. The first few months in the ocean are critical for sub-adult coho salmon survival. As previously discussed, sub-adult fish must quickly grow to a large size prior to their first winter in the ocean or be subject to high mortality, thus survival is highly correlated with the amount and type of food available.

The availability and type of food resources in the nearshore environment is dependent upon the location and magnitude of upwelling and its influences on ocean productivity. Upwelling is caused by northerly winds that dominate from spring to early fall along the coastal region of the Pacific Northwest within the California Current marine ecosystem. These winds transport offshore surface water southward, while also transporting surface water away from the coastline (westward). This offshore, southward transport of surface waters is balanced by onshore northward transport (upwelling) of deep, cool, high-salinity, nutrient-rich water (Peterson *et al.* 2006). The shifting of this highly productive water to the surface of the nearshore environment triggers the formation of large phytoplankton blooms. Phytoplankton (minute aquatic plants) form the base of the marine food chain and are eaten by zooplankton (microscopic animals, such as copepods, that move passively with ocean currents). Zooplankton in turn, are preyed upon heavily by forage fish species and sub-adult coho salmon.

Coastal upwelling therefore, is a critical process affecting plankton production, and corresponding food availability. Moreover, the strength and timing of the upwelling event effects salmon survival by influencing the overall abundance and spatial distribution of plankton within the nearshore marine environment. Many studies have demonstrated this direct relationship. For example, Gunsolus (1978) and Nickelson (1986) correlated salmonid marine survival and the strength and/or timing of marine upwelling. Holtby *et al.* (1990) examined the scales of returning adult coho salmon in order to determine growth rates, and found that rapid ocean growth was “positively correlated with ocean conditions indicative of strong upwelling.” Better ecosystem productivity is also related to earlier seasonal upwelling events (Peterson *et al.* 2006). Additionally, Cury and Roy (1989) demonstrated a relationship between upwelling and recruitment of several pelagic forage fishes in the Pacific.

The cooler water temperatures resulting from upwelling currents along the eastern Pacific Ocean originating from the subarctic region support high plankton productivity and salmon survival. Marine productivity and salmon survival are typically much lower when warmer, less-saline water upwells from sub-tropic marine regions. Survival is also likely influenced by the species of zooplankton occupying the two water types (cooler subarctic waters, and warmer subtropical waters); sub-arctic copepods are larger and have more fat than sub-tropical ones, promoting better support growth and survival of salmon which prey on them, and on forage species which eat them (Peterson *et al.* 2006). Peterson *et al.* (2006) developed an index to predict salmonid year-class strength based on the species of copepods present over the continental shelf and the inferred source of the water transport.

Unfavorable oceanographic conditions also affect adult coho salmon through their impacts on forage fishes, the primary food of adult coho salmon. For example, Pacific herring recruitment in the Bering Sea and northeast Pacific was accurately forecast based on the air and sea surface temperatures when spawning occurred (Williams and Quinn II 2000), and many Pacific herring starved during a winter of low zooplankton abundance in Prince William Sound, Alaska (Cooney *et al*. 2001).

Reduced genetic and life history diversity

A number of life history and genetic traits also influence coho salmon growth and survival. For sub-adults these include timing of ocean entry, size and age at entry, growth characteristics, migration pathways, feeding behaviors, straying, and age and size at maturity (Quinn 2005). The influence of each of these traits on growth and survival is dependent on ocean conditions, and salmon have a diversity of life history and genetic traits to take advantage of the full range of variability which maximizes their resiliency. Overall, coho salmon have experienced a net loss of diversity and may not be able to exploit the full range of ocean conditions, which may place them at a greater risk of extinction.

As noted above, the timing of ocean entry can affect likelihood of survival. Ryding and Skalski (1999) documented a relationship between the marine survival rate of coded-wire tagged coho salmon released from Washington state and the ocean conditions when released. The authors concluded there are optimal environmental conditions for coho marine survival, and thus optimal dates for ocean-entry, for any given year. Similar patterns have been observed with pink salmon in Alaska (Cooney *et al*. 1995). Research by Mortensen *et al.* (2000) also suggests an indirect relationship between time of ocean entry and growth and vulnerability to predators of sub-adult coho salmon.

Although the date of ocean entry is critical to coho survival, the timing of peak ocean upwelling and productivity is quite variable and cannot be reliably predicted. Between 1967 and 2005, the date of spring transition (the start of upwelling), at 39 degrees North latitude, has varied from January 1 to early April (Bograd *et al.* 2009). Coho salmon migrate to sea over a number of months, which may increase salmonid year class strength because, although the timing of the upwelling event is variable, at least some coho should enter the ocean when conditions were favorable. Size and age variation during outmigration is an important mechanism to improve a population's ability to track environmental change and persist in the marine system[2].

The relationship between size and survival of sub-adult coho salmon has been documented in a number of studies (*e.g.,* Quinn 2005). Size-selective mortality in the ocean (mainly through predation) suggests larger individuals likely experience higher survival rates than smaller individuals (Holtby *et al.* 1990). Some individuals may also have a size advantage due to their genetic disposition, and this, in turn, may translate to increased growth and survival at sea (Beamish *et al.* 2004).

Once coho salmon reach the ocean they are thought to display a range of different migratory pathways depending on their behavior, life history, and genetic makeup (Weitkamp and Neely 2002). A wide distribution allows populations and the ESU to take advantage of numerous feeding opportunities and spreads the risk of isolated mortality events (such as predation,

[2] In Redwood Creek, California, some coho remain in freshwater for one year before outmigration to the ocean, while a small number remain for an additional year and smolt as two year-olds (Bell and Duffy 2007). In Pudding Creek, California, 12 percent of the smolts were two year-olds (Wright pers. comm. 2009). Two year-old coho salmon migrate at a larger size and may experience higher marine survival than smaller, one year-old fish, but are consequently exposed to an additional year of stresses unique to the freshwater environment. Depending on both ocean conditions and conditions in the freshwater environment, one or both life histories will likely succeed and contribute to the persistence of the population.

fisheries impacts, or ocean conditions). In turn, a wide distribution decreases the risk of any one population being extirpated in concentrated mortality events.

As adults, some coho salmon display a limited range of life history strategies. They either return to their natal streams to spawn after only a few months at sea as two year-olds (called jacks or grilse) or, more typically, after a year at sea as three year olds. Maintaining a healthy abundance of jacks in any population ensures some genetic overlap between brood years and is thought to increase the overall productivity of the population. Also important to the overall health and resilience of the ESU is the presence of strays, which do not return to their natal spawning grounds and consequently help to colonize new spawning areas and re-establish diminished populations.

A diverse array of behaviors and environmental sensitivities, such as those seen in salmon populations, are evolutionary responses to successful adaptation in uncertain environments (*e.g.*, see Independent Science Group 2000). At the metapopulation level, each species of Pacific salmon exhibits many such risk-spreading behaviors via a broad diversity of time-space habitat use by different stocks and substocks of the same species. Through reduced population size, lost connectivity between remaining populations, and the genetic dilution resulting from (past) hatchery use of non-native stock (Weitkamp *et al.* 1995), the CCC ESU has lost much of its historical life history and genetic diversity. The remnant life history characteristics likely limit extant populations from taking full advantage of the range of ocean conditions, diminishing overall productivity. In the marine environment, the impact from lost phenotypic diversity is probably most pronounced at the sub-adult life stage, since success at that life stage is closely correlated with ocean conditions. Because of the importance of maintaining a diverse set of life history strategies and genetic pool to the survival and growth of coho salmon at sea, the loss of these traits is considered a medium to high stress.

Threats

Overview of Threats

Major threats potentially affecting CCC coho salmon in the marine environment include incidental take from commercial and recreational fisheries, aquaculture, predation, harvest of kelp, wave energy generation, management of prey and competitors, hazardous spills, and introduction of non-native species. The threat of climate change also influences ocean productivity, but is discussed separately in the Climate Scenarios section of this appendix.

Commercial and recreational fishery bycatch

Directed commercial and sport fishing take

In 1993, the retention of coho salmon in ocean commercial fisheries was prohibited from Cape Falcon, Oregon south to the U.S.-Mexico border. The following year, coho salmon retention was prohibited in ocean recreational fisheries from Cape Falcon, Oregon to Horse Mountain, California, and expanded to include all California waters in 1995. These prohibitions prohibit direct sport and commercial harvest of coho salmon off the California and Southern Oregon coast, the sole exception being a mark-selective recreational coho salmon fishery that has taken place in recent years in Oregon waters. While the number of CCC coho harvested within the Oregon mark-selective fishery is difficult to determine, the percentage is likely lower than the projected 3.3 percent non-retention exploitation rate for Rogue/Klamath coho salmon (PFMC 2007) due to the more southern marine distribution of CCC coho versus Southern-Oregon Northern California Coast ESU (NMFS 1999a)[3]. Therefore, the primary harvest-related impact on CCC coho salmon likely arises from incidental take through other fisheries. This impact is likely largely restricted to adult fish and has little effect on the sub-adult life stage, which is likely too small to be efficiently captured in this fishery.

[3] NMFS (1999a) suggests exploitation rates for CCC coho salmon may be higher than SONCC coho salmon due to the overwhelming effect of the central and northern California sport and commercial Chinook fishery. However, due to recent declines in Klamath and Sacramento River Chinook salmon populations, Chinook salmon fishing off the California coast has been severely restricted in 2007, 2008, and 2009, and the size and extent of future seasons is uncertain.

The State of California has recently begun implementing a series of underwater parks and reserves along the California coast as part of the Marine Life Protection Act (MLPA) of 1999. The goal of the MLPA is to "protect habitat and ecosystems, conserve biological diversity, provide a sanctuary for fish and other sea life, enhance recreational and educational opportunities, provide a reference point against which scientists can measure changes elsewhere in the marine environment, and may help rebuild depleted fisheries (CDFG 2008)". Fishing will be closed or severely restricted in most protected areas, which will ultimately account for approximately 20 percent of state coastal waters (out to three miles off-shore). However, many of the restricted areas coincide with rocky benthic habitat which salmon may inhabit only sporadically, and many of the more popular salmon fishing areas are not expected to be part of the MLPA program. Furthermore, some MLPA areas where fishing is restricted make exceptions with regard to salmon fishing. For these reasons, NMFS does not expect a significant reduction in ocean salmon harvest resulting from the MLPA program.

Bycatch in Federal salmon fisheries

The Pacific Fishery Management Council (PFMC) manages salmonid fisheries in Federal waters. The CCC coho salmon ESU is one component of the Oregon Production Index (OPI) area coho stocks. Because there are insufficient hatchery releases from within the CCC coho ESU to support an estimate of fishery bycatch in the Chinook salmon fishery (CDFG 2002), the projected marine fishery impacts on Rogue/Klamath River (R/K) hatchery coho were used as a surrogate.[4] Coho are intercepted in Chinook-directed fisheries and must be immediately released. However, some will die, as reflected by the 13 percent marine fishery mortality rate allowed for R/K hatchery coho salmon (NMFS 1999a). Given that the estimated discard mortality rate for R/K hatchery coho salmon has been the 13 percent maximum for at least the last three years (PFMC 2007), and prohibitions on take of OPI area coho stocks have not changed, the Federal salmon fishery was determined to pose a low threat to the CCC coho salmon ESU.

[4] The assumption is that exploitation rates of hatchery and wild coho salmon stocks are similar.

Bycatch in State salmon fisheries

All marine fishing occurring within three miles of the California shore is managed by CDFG. Chinook salmon harvest is allowed in California waters and is subject to area restrictions, gear restrictions, seasonal closures, and bag limits (CDFG 2011). Harvest of coho salmon is prohibited in California waters (except Lake Oroville), and any incidentally hooked coho salmon must be immediately released unharmed (CDFG 2011).

The impacts of state-regulated Chinook salmon and steelhead fisheries on CCC coho salmon have not been evaluated but could be significant. Listed salmon and steelhead are likely to occur within the marine environment at the same time, and in the same locations, as non-listed salmonids, and are likely to be captured by the same gear and fishing methods. Bycatch mortality may be enough to hinder recovery due to the extremely low size of the population. In parts of California, ocean fishers use a "drift mooching" method of capturing salmonids, where bait is suspended in the water column and moved by the ocean currents as the boat drifts. Salmon are more likely to swallow the hook when caught using drift mooching than when caught while trolling, and are less likely to survive when released. The survival of Chinook salmon caught and released off Northern California from drift mooching was monitored for four days and compared to a control group (Grover *et al*. 2002). The overall hook-and-release mortality rate for the study was estimated at 42 percent, significantly greater than the 13 percent mortality cap in Federal ocean fisheries. While the study did not evaluate impacts to coho salmon (due to the statewide prohibition on harvest of this species) the impacts between species are likely similar. Given coho occur higher in the water column than Chinook salmon, fishers targeting Chinook salmon may not encounter coho salmon. However, since most of the lifetime mortality suffered by a coho salmon occurs before they reach adulthood (Quinn 2005), an adult coho salmon that has survived at least a year of ocean life and is not far from spawning age is particularly valuable for recovery. The PFMC salmon FMP includes the 42 percent bycatch mortality rate from mooching as part of its recreational bycatch mortality rate for the area south of Point Arena. However, as coho recover, this mortality rate could have a proportionately greater impact on the ESU than it does now, as the rate CCC coho are encountered increases. This fishing method could hinder recovery. Given the impact the state salmonid fishery on CCC coho salmon is unknown but potentially significant; this fishery was determined to pose a medium threat to the recovery of this ESU.

Federal non-salmon fisheries

The PFMC manages four stocks (*aka* stock complexes) in Federal waters potentially affecting CCC coho salmon through fishery bycatch: groundfish, coastal pelagic species (CPS), highly migratory species (HMS), and Pacific halibut. NMFS evaluated the impacts of the groundfish fishery on listed salmon and steelhead and concluded it was not likely to adversely affect salmon or adversely modify critical habitat (NMFS 1999b; NMFS 2005). Salmonids could be accidentally captured in fisheries targeting CPS, but NMFS determined, although some ESUs of coho salmon are captured in CPS fisheries, CCC coho are not captured (PFMC 2005). The HMS fishery targets various species of tunas, sharks, and billfishes as well as mahi-mahi. A 2004 Biological Opinion stated, although all listed salmonid ESUs could occur in the area where HMS fishing occurs, there are no records indicating any instance of take of listed salmon in any HMS fisheries.

Pacific halibut occur on the continental shelf from California to the Bering Sea. Harvest of this species is managed by the International Pacific Halibut Commission (IPHC), which determines allowable catch. Although fishing for this species is allowed in California, in the past ten years only one Pacific halibut was commercially landed in waters off California (Leaman, Executive Director, International Pacific Halibut Commission, personal communication, 2007). Based on surveys from 1200 stations off of Washington and Oregon, an average of less than one salmon is captured per year survey wide (Dykstra, Survey Manager, International Pacific Halibut Commission, personal communication, 2007). The number of salmon caught in the recreational halibut fishery off California appears very small (Palmer-Zwahlen, CDFG, personal communication, 2007).

Marine aquaculture

Concerns have been raised over environmental impacts of salmonid culture activities in nearshore or open ocean areas. Potential impacts include disease and parasite transmission, water quality impairment, and genetic interactions. The recovery of CCC coho salmon is unlikely to be hindered by current marine aquaculture activities because, aside from the shellfish farming (*e.g.*, oysters and abalone) occurring in estuaries, marine aquaculture is largely absent from the waters off the California coast where CCC coho salmon spend most of their ocean residency. Further, marine culture of salmonids cannot occur in California's jurisdictional waters, which extend three miles into the Pacific Ocean (see State of California's 2006 Sustainable Oceans Act). In Federal waters (between three and 200 miles from the west coast), the process for obtaining a permit to carry out aquaculture is unwieldy, time consuming, and unattractive to investors (NOAA 2007). A bill to establish Federal guidelines for offshore aquaculture and improve the permitting process was recently considered by congressional committees. This legislation would retain NMFS' review of permit applications to ensure they do not jeopardize the continued existence of CCC coho salmon. Given the low likelihood of any additional aquaculture operations off the California coast in the next five plus years, and the expected close evaluation of any proposals by NMFS, EPA, and other agencies, the threat to listed salmonids from the culture of animals in nearshore and offshore marine areas is rated as low.

Marine mammal predation

Predation by marine mammals (principally seals and sea lions) is of concern in areas experiencing dwindling run sizes of salmon (69 FR 33102). However, salmonids appear to be minor component of the diet of marine mammals (Scheffer and Sperry 1931; Brown and Mate 1983; Hanson 1993; Goley and Gemmer 2000; Williamson and Hillemeier 2001). Harbor seal and California sea lion numbers have increased along the Pacific Coast since passage of the Marine Mammal Protection Act of 1972, but available information indicates salmon are not a principal food source for pinnipeds (Quinn 2005). At the mouth of the Russian River in western

Sonoma County, Hanson (1993) reported foraging behavior of California sea lions and harbor seals with respect to anadromous salmonids was minimal. Hanson (1993) found predation on salmonids coincidental with the salmonid migrations, but the harbor seal population at the mouth of the Russian River was not dependent upon them. Nevertheless, this type of predation may, in some cases, kill a significant fraction of a run and local depletion might occur (NMFS 1997; Quinn 2005). At the ESU level, NMFS considers the threat of marine mammal predation low.

Avian predation

Avian predation is not expected to constitute a significant threat to adult CCC coho salmon because of their relatively large size once in the ocean. All documented incidences of significant effects of avian predation on juvenile salmonids have occurred in estuarine areas near large nesting colonies with high avian densities. While birds are also known to feed on schools of fish in the open ocean (Scheel and Hough 1997), indirect evidence shows salmonids do not generally occur in tight schools. Many salmon probably do not swim in sight of other salmon, and when they have been observed together it is usually in groups of less than four (Quinn 2005). Avian predation is not expected to constitute a significant threat to sub-adult coho salmon when they occur in nearshore oceanic areas used by CCC coho salmon.

Management actions affecting nearshore marine habitat

<u>Harvest of kelp from nearshore marine areas</u>

Both bull and giant kelp are currently harvested from California waters (Spinger *et al.* 2006). Small quantities of each species are currently harvested, due to limited commercial demand. The upper four feet of canopy and leaves of giant kelp are harvested, allowing the plant to continue to grow and reproduce (Spinger *et al.* 2006); therefore, giant kelp are essentially a renewing crop. However, when bull kelp are harvested, the pneumatocyst and associated fronds are removed, which eventually kills the plant. Harvest of bull kelp before it reproduces may destroy beds of this species and reduce the amount of habitat available to juvenile CCC coho salmon. The extent CCC coho salmon utilize kelp is unknown.

Surveys of the fish communities in kelp beds off California south of the CCC coho salmon ESU range are focused on rockfishes and do not mention salmon (*e.g.*, Paddack and Estes 2000). No salmon were found in studies of beds of bull kelp off South-central Alaska (Hamilton and Konar 2007), but salmon were found in beds of brown kelp off Southeastern Alaska (Johnson *et al*. 2003). In Washington's Strait of Juan de Fuca, juvenile Chinook and chum salmon appeared to preferentially use kelp beds (which included both bull kelp and giant kelp) over unvegetated habitats (Shaffer 2004).

The above studies suggest coho salmon could use kelp beds, and some of these kelp beds may be negatively affected by harvest. But at this time, there is no evidence CCC coho salmon rely on kelp beds for shelter in the nearshore marine environment, and no harvest of the kelp beds occurs within the CCC coho salmon ESU range. The threat to CCC coho salmon from the harvest of kelp from nearshore marine waters was rated as Low.

<u>Wave energy generation in the nearshore environment</u>

Wave energy can be harnessed to provide electricity, and there are three proposals to do so in the marine range of the CCC coho salmon ESU (Boehlert *et al.* 2008). The production has a potential to impact CCC coho salmon and their marine habitat. According to the proceedings of a recent workshop on the ecological effects of wave energy generation in the Pacific Northwest (Boehlert *et al.* 2008), the electromagnetic fields and noise associated with wave energy's underwater structures have the most potential of all wave energy efforts to negatively affect salmon. Salmon may avoid the structures due to electromagnetic fields and/or noise, and such avoidance could interfere with the migration of juveniles along the coast, and disrupt adult spawning migrations. The generation of electricity from waves reduces wave energy, changing nearshore wave processes and potentially altering benthic communities where juvenile salmon feed. The harnessing of wave energy may affect transport of zooplankton (Boehlert *et al.* 2008), and so could impact CCC coho salmon's food supply. The workshop participants acknowledged a high degree of uncertainty regarding the actual effects of wave energy generation on salmon, because little data documenting effects exists. Currently, wave energy poses a low threat to sub-adult and adult CCC coho salmon since no operational projects exist at this time. However, thorough research investigating potential adverse impacts on salmon and nearshore habitat should be required before future wave energy projects are permitted.

Management of coho prey and competitors

As coho grow in the ocean, their diet becomes more and more reliant on other fish species. Some concern has been raised over the possibility human harvest of salmon prey species may disrupt the aquatic ecosystem. If enough forage fish were harvested, there may not be enough prey items for higher level predators such as salmon and marine mammals. The effects of forage fish availability on salmonid predator behavior was recognized as a factor influencing the species when CCC coho were listed (69 FR 33102):

> *"The federally-managed fishery with the most potential to impact prey availability for CCC coho salmon is the coastal pelagic species (CPS) fishery. This group includes northern anchovy, market squid, Pacific bonito, Pacific saury, Pacific herring, Pacific sardine, Pacific (chub or blue) mackerel, and jack (Spanish) mackerel. Anchovy and sardine are known as important forage species for predators including salmon and steelhead (PFMC 2005; Quinn 2005). CPS are extremely important links in the marine food chain, and disruptions in their distribution and abundance may impact salmon population dynamics (PFMC 2003)."*

CPS harvest could indirectly affect salmon if it resulted in an inadequate amount of prey species for foraging salmon. The PFMC has adopted a conservative, risk-averse approach to management of CPS that reduces the likelihood of such negative effects. The need to "provide adequate forage for dependent species" is recognized as a goal and objective of the CPS FMP (PFMC 1998). A control rule is a simple formula used by the PFMC in evaluating allowable harvest levels for each of the CPS. The CPS control rules contain measures to prevent excessive harvest, including a continual reduction in the fishing rate if biomass declines. In addition, the control rule adopted for species with significant catch levels explicitly leaves thousands of tons of CPS biomass unharvested and available to predators. No ecosystem model currently exists to calculate the caloric needs of all predators in the ecosystem, so the amount of unharvested CPS biomass is an estimate which may be modified if new information becomes available. Ocean temperature is a factor in the control rule for Pacific sardine, in recognition of the effects of varying ocean conditions on fish production rates. Allowable harvest rates are automatically reduced in years of poor production.

The impacts of these fisheries on Federally-listed ESUs of salmon and steelhead were not evaluated by NMFS. However, due to the conservative control rules used to manage CPS and the preservation of a portion of the biomass for predator consumption, the CPS fishery poses a Low threat to CCC coho salmon recovery.

Transportation-related hazardous spills

Oil spills can have significant, catastrophic effects on aquatic ecosystems (National Research Council 2003), including acute mortality of fishes. The effects of crude oil on pink salmon were studied extensively since the Exxon Valdez oil spill in Prince William Sound, Alaska. Although some researchers found the oil spill affected growth rates of juvenile pink salmon (Moles and Rice 1983; Willette 1996), a review of all research on this topic showed the spill posed a low risk to this species (Brannon and Maki 1996). The relatively low depth of the oil entering the water column and the short time it remained in important natal gravel beds (Brannon and Maki 1996) may account for this effect. Oil spills appear to have the greatest effect on aquatic birds and marine mammals and benthic (bottom-dwelling aquatic) organisms (Boesch *et al.* 1987). The egg, alevin, and fry life stages of salmonids utilize benthic habitat in freshwater and brackish areas, and indeed toxic effects of crude oil were documented on the embryos and larvae of herring on oil-affected beaches (Hose *et al.* 1996). However, none of these salmonid life stages occur in nearshore marine areas or the open ocean, and direct effects of oil spills on salmon occurring in these areas is likely low. Indirect effects could include degradation of submerged aquatic vegetation such as kelp and eelgrass used by some juvenile salmonids in nearshore areas (Thorpe 1994). Disruption of the food web could also be detrimental to these fishes. Although in some circumstances crude oil may inhibit photosynthesis of natural phytoplankton communities, in inland areas of Nova Scotia, Canada, researchers determined that in open marine waters oil did not negatively affect photosynthesis (Gordon and Prouse 1973).

Introduction of non-native species

Some invasive species are detrimental to salmonids, particularly in the freshwater or estuarine environments. Conditions in the open ocean are less hospitable to many invasive species than estuaries[5], and non-marine fish do not tend to survive when released into marine waters. Of 22 fish species successfully introduced into marine waters, all of them came from marine waters, indicating introductions of freshwater or brackish fish species into marine waters were unsuccessful (Hare and Whitfield 2003). All but one of these 22 marine fish species was released from an aquarium or accidentally or intentionally stocked (Hare and Whitfield 2003). Since the sub-adult and adult life stages of CCC coho salmon occur in the ocean, introduction of non-native species is unlikely to affect them because the introduced species are unlikely to survive. Proposed national offshore aquaculture legislation would usually only allow marine culture of native species in Federal waters (NOAA 2007), making it is unlikely further stocking of potentially harmful non-native species will occur in marine waters off California. The threat to sub-adult and adult CCC coho salmon from introduction of additional non-native species was therefore rated low.

Recovery Strategy for CCC coho salmon in the eastern pacific

Marine factors will strongly influence CCC coho salmon recovery, but not solely due to obvious threats such as pollution or over-harvest. Rather, freshwater and marine impacts have reduced CCC coho salmon genetic and life history diversity, leaving the species less equipped to deal

[5] This has led to a requirement to replace ballast water in the ocean before entry into California state waters if the vessel intends to dock at any California port (State of California 2003).

with variable, unpredictable, and often hostile oceanic conditions. The best means to improve CCC coho salmon survival in the marine environment is to preserve and strengthen the existing genetic and life history diversity in the ESU, which will likely improve population abundance over the long-term. In addition, a better understanding of the ocean conditions each year is necessary so that managers could account for periods of poor ocean productivity and high marine mortality when estimating population abundance, harvest levels, and ultimately the progress toward ESU recovery.

Improve the quantity and/or quality of food resources

This is the top-ranked stressor for sub-adult and adult CCC coho salmon, because it results from unfavorable ocean conditions. As ocean conditions are not under human control in the time frame relevant to CCC coho salmon recovery (*e.g.*, 50 years), there are no recovery strategies which could "improve" them. However, strategies which improve genetic and life history diversity in the CCC coho salmon ESU would effectively equip the salmon to better survive an unpredictable ocean environment. Further research is necessary to discern possible connections between global climate change and cyclic patterns of ocean productivity. If a link is found, actions identified to alleviate or diminish global climate change may have value in moderating marine productivity patterns and improving salmon survival.

Increase genetic and life history diversity

Before anthropogenic stressors within the freshwater, estuarine, and marine environment depressed the CCC coho salmon population to a level requiring protection under the ESA, abundant, genetically diverse juvenile salmon entered the ocean each year over a wide range of dates, seasons, and ages from approximately 76 CCC coho salmon populations (Bjorkstedt *et al.* 2005). It is necessary to restore this lost diversity and life-history adaptation to allow CCC coho salmon populations to adapt and persist within the variable ocean environment. To foster greater life history and genetic diversity, recovery actions must be undertaken to improve the various habitats supportive of diverse life history strategies. Management and recovery strategies must adapt to address and conserve the full range of life history potential of a given populations, and hatchery practices must be managed to avoid degrading the genetic diversity of wild stocks.

Increase population size

Federal fisheries have been evaluated and appear to pose a low threat to CCC coho salmon, likely due to coho salmon harvest prohibitions in California and a low allowable CCC coho salmon bycatch mortality rate for Federally-managed ocean fisheries. The harvest prohibition extends into ocean waters managed by the state of California. All existing prohibitions and bycatch mortality rates should be retained or made more conservative. Salmonid fisheries in state waters have the potential to negatively impact the ESU and the extent of such impact has not been evaluated. Development of a Fishery Management Evaluation Plan (FMEP) is necessary for NMFS to determine what risk, if any, these fisheries pose to the CCC coho salmon ESU. The effects of drift mooching on CCC coho salmon should be minimized through educating anglers on the use of drift mooch methods that lessen the probability of gut hooking, as suggested in Grover *et al.* (2002).

CLIMATE CHANGE

"There are two key sources of greenhouse gas emissions: fossil fuels and forest change. Any successful climate strategy must address both."

Laurie Wayburn, Pacific Forest Trust

Overview: Climate Change and Pacific Salmon

The best available scientific information indicates the climate is warming, driven by the accumulation of greenhouse gasses (GHGs) in the atmosphere (IPCC 2007). The Intergovernmental Panel on Climate Change (IPCC) concluded in 2007, warming of the climate system is "unequivocal," based on observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level. In a recent 2011, report on the Global Climate Change Impacts in the U.S. it was noted, "…salmon in the Northwest are under threat from a variety of human activities, but global warming is a growing source of stress." Salmon and steelhead from northern California to the Pacific Northwest are challenged by a global warming induced alteration of habitat conditions throughout their complex life cycles (Mantua and Francis 2004; Glick 2005; ISAB 2007; Martin and Glick 2008; Glick et al. 2009). Salmon productivity in the Pacific Northwest is sensitive to climate-related changes in stream, estuary, and ocean conditions. Specific characteristics of a population vulnerable to climate change include temperature requirements, reliance on snowpack, suitability of available habitat, and the genetic diversity of the ESU. These changes could alter freshwater habitat conditions and affect the recovery and survival of Pacific salmon stocks.

Climate shifts can affect fisheries, with profound socio-economic and ecological consequences (Osgood 2008). Climate change introduces additional, uncertain impacts to California's ecosystems and species, ranging from changes in the timing of bird migrations in spring, to large-scale movement of species, to increased frequency of forest fires. These are other impacts threaten to disrupt existing current natural communities, and may push many species toward extinction. In addition, climate change will interact with other stressors, such as habitat destruction, that are already threatening species and ecosystems, making it more difficult to achieve conservation goals.

In the Pacific Region, global climate change will lead to major alterations in freshwater environments. The biological implications of physical habitat changes on Pacific salmon are significant. Changes in timing/magnitude of flow and thermal regimes can affect the behavior and physiological responses of salmon during their freshwater life stages. Human activities can affect biophysical changes by imposing additional stressors such as unsustainable exploitation rates on vulnerable populations, and reduced water availability in stressed areas. Threat minimization actions may include adjustment of harvest rates and improved management of freshwater supplies.

Climate variability is an important factor controlling the distribution and abundance of organisms and determining the ecosystem structure. Changes in seasonal temperature regimes affect fish and wildlife (Quinn and Adams 1996; Schneider and Root 2002; Walther et al. 2002). These effects manifest themselves differently in different organisms, some undergo changes in the timing of spring activities, including earlier migration and breeding in birds, butterflies and amphibians, and flowering of plants (Walther *et al.* 2002). In response to warmer water temperatures, a number of fish species shift their distribution to deeper, cooler water, or move pole ward (Osgood 2008). Along with the increase in global temperatures, smaller scale geographic changes in temperature, wind, and precipitation are anticipated (CEPA 2006; Osgood 2008) . Freshwater streams (a key habitat for coho salmon), may experience increased frequencies of floods, droughts, lower summer flows and higher temperatures (Luers *et al*. 2006; Lindley *et al.* 2007; Schneider 2007; Osgood 2008). Estuarine and lagoon habitats are likely to experience a sea level rise and changes in entering stream flow (Scavia *et al.* 2002). The marine environment is important to sub-adult and adult salmonids and is likely to experience changes in temperature, circulation, chemistry, and food supplies (Brewer and Barry 2008; Turley 2008; O'Donnell *et al.* 2009). Because coho salmon depend on freshwater streams and oceans during different stages of their life history cycle, their populations are likely to be affected by many of the climate induced changes shown below in **Figure 1.**

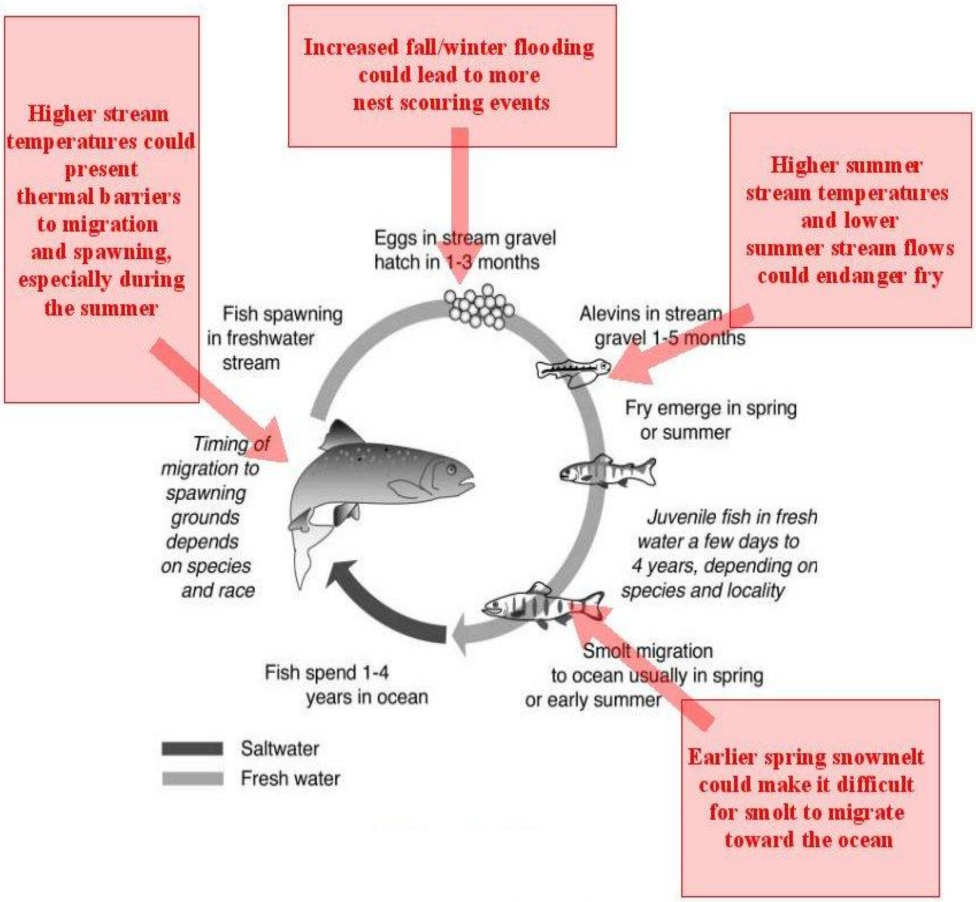

Figure 1: Salmon life history and the impacts of climate change.

Pacific salmon are affected by climate change across a hierarchy of coarse and fine spatial and temporal scales and each of these scales has distinct requirements in the development of policy that will cover climate change effects (Schindler *et al.* 2008). Efforts to minimize the impacts of climate change will take national and international actions beyond the scope of this recovery plan. Although at a local scale, identification and mitigation of impacts from global climate change can help alleviate its effects at (Osgood 2008). Effective management is important and adaptive strategies must consider climate variability. Nearly 75 percent of California's anadromous salmonids are vulnerable to climate change, and future climate change will affect the ability to influence their recovery in most or all of their watersheds (Moyle *et al.* 2008). The following sections describe key issues for consideration regarding impacts of climate change to coho salmon in the CCC ESU.

Climate Change in California

Recent studies call for improved legal and planning protection explicitly accounting for the impacts of climate change in California (Luers and Mastrandrea 2008; Mastrandrea and Luers 2012). A number of climate models evaluate climate change uncertainties and forecast future climate conditions at global and regional scales. Although, studies were conducted to examine the projected impacts of climate change on salmon habitat restoration, specifically Chinook salmon (Battin *et al.* 2007), few studies examine projected impacts to coho salmon.

Integral to understanding climate change effects on salmon is an understanding of how variations in salmon abundance corresponds to climate-related ecosystem regime shifts (Irvine and Fukuwaka 2011). The IPCC-AR4 global climate models (GCMs) do not resolve certain parameters at a fine enough resolution and/or sufficient detail to produce a true forecast, and higher resolution regional climate models (RCMs) are under development (King et al. 2011). Available model predictions show a range of relatively low to high impacts depending on which model is used and the greenhouse gas emissions scenario considered. Even the low impact predictions show changes in California's temperatures, rainfall, snowpack, vegetation, as well as potential changes in ocean conditions likely to have negative impacts on salmonid population numbers, distribution, and reproduction. It is likely, one of the greatest near-term climate challenges California will face are more intense and/or frequent extreme weather events (Meehl *et al.* 2007; Mastrandrea and Luers 2012).

Impacts on Freshwater Streams

Climate change impacts in California suggests average summer air temperatures will increase(Lindley et al. 2007). Heat waves are expected to occur more often, and temperatures peaks are likely to increase (Hayhoe *et al.* 2004). Total precipitation in California may decline and the frequency of critically dry years may increase (Lindley *et al.* 2007; Schneider 2007) which under unimpaired condition would result in decreased stream flow. Wildfires are expected to increase in frequency and magnitude, by as much as 55 percent under the medium emissions scenarios modeled (Luers *et al.* 2006). Vegetative cover may also change, with decreases in evergreen conifer forest and increases in grasslands and mixed evergreen forests. Impacts on forest productivity are less clear. Tree growth may increase under higher CO_2 emissions, but as temperatures increase, the risk of fires and pathogens also increases (CEPA 2006).

Air temperature

According to NOAA's 2008, State of the Climate Report and NASA's 2008, Surface Temperature Analysis, the average surface temperature has warmed about 1° F since the mid-1970's. The Earth's surface is currently warming at a rate of about 0.29° F/decade or 2.9° F/century, and the eight warmest years on record (since 1880) have all occurred since 2001, with the warmest year occurring in 2005. The range of surface water temperatures are likely to shift, resulting in higher high temperatures as well as higher low temperatures in streams. A recent study of the Rogue River basin in Oregon determined annual average temperatures are likely to increase from 1° to 3° F (0.5° to 1.6° C) by around 2040 and 4° to 8° F (2.2° to 4.4° C) by around 2080. Summer temperatures may increase 7° to 15° F (3.8° to 8.3° C) above baseline by 2080, while winter temperatures may increase 3° to 8° F (1.6° to 3.3° C) (Doppelt *et al.* 2008). Temperature changes throughout the NCCC Domains are likely to be similar. A study by Littell *et al.* (2009) suggested one third of the current habitat for listed Pacific salmon species may be unsuitable by the end of this century when temperature thresholds are exceeded.

Increasing air temperatures have the potential to limit the quality and availability of summer rearing habitat for juvenile CCC coho salmon by increasing water temperatures. Increases in fall and winter temperature regimes might shorten incubation and emergence for developing eggs, which Burger *et al*., (1985) predicted would lead to lower survival rates. Increases in summer temperatures will lead to thermal stress, decreased growth and affect survival of out migrating juveniles. For example, modeling results reported by Lindley *et al*. (2007) show, as warming increases, the geographic area experiencing mean August air temperature exceeding 25° C moves further into coastal drainages and closer to the Pacific Ocean. This increase in temperature will likely lead to an increase in stream temperatures in these areas, many of which are areas with focus populations. Many stream temperatures in the CCC coho salmon ESU are at or near the high temperature limit of coho salmon and increasing water temperatures may limit habitat suitability in an unknown number of stream reaches.

Precipitation

Annual precipitation could increase by up to 20% in northern California. Most precipitation will occur during the mid-winter months as intense rainfall events. These weather patterns will likely result in a higher numbers of landslides and greater and more severe floods (Doppelt *et al.* 2008; Luers *et al.* 2006). For the California's North Coast (including the northern part of the NCCC Domain), some models show large increases (75% to 200 %), while other models show decreases of 15 to 30% (Hayhoe 2004) in rainfall events. Increases in rainfall during the winter have the potential to increase the loss of salmon redds via streambed scour from more frequent high stream flows. Reductions in precipitation will likely lower flows in streams during the spring and summer, reducing the availability of flows to support smolt migration to the ocean as well as the availability of summer rearing habitat.

Sea Level Rise

According to the 2002, report released by the U.S. Global Climate Research Program (USGCRP), sea level is expected to rise exponentially over the next 100 years, and is estimated to rise 50-80 cm by the end of the 21st century. Additional research on sea level rise estimates the high end of possible sea level rise by 2200, to be 1.5 m to 3.5 m Vellinga *et al.* (2008). It is predicted that low lying coastal areas will eventually be inundated by seawater or periodically over-washed by waves and storm surges. Coastal wetlands will become increasingly brackish as seawater inundates freshwater wetlands. As a result, new brackish and freshwater wetland areas will be created (Pfeffer *et al.* 2008). Sea level rise will also alter estuarine habitat; which may provide increased opportunity for feeding and growth of salmon, but in some cases sea level rise will lead to the loss of estuarine habitat and a decreased potential for estuarine rearing.

In 2009, The Pacific Institute released a study on the impacts of sea-level rise on the California Coast. The study included a detailed analysis of the current population, infrastructure, and property at risk from projected sea-level rise if no actions are taken to protect the coast, and the cost of building structural measures to reduce that risk. Findings from the report conclude; (1) a sea-level rise of 1.4 m would flood approximately 150 square miles of land immediately adjacent to current wetlands, potentially creating new wetland habitat if those lands are protected from further development; (2) approximately 1,100 miles of new or modified coastal protection structures are needed on the Pacific Coast and San Francisco Bay to protect against coastal flooding, and (3) continued development in vulnerable areas will put additional areas at risk and raise protection costs (Heberger *et al*. 2009). San Francisco Bay is of particular concern, with increased risk to; existing wetlands, unprotected developed areas, and existing levees (Knowles 2010; Cloern *et al.* 2011).

NOAA is developing a strategic approach to integrate its coastal activities, with a specific focus on improving risk assessment and adaptation to climate change in coastal areas. Significant efforts are underway to improve the design, development, and delivery of effective climate services to NOAA and stakeholders through a National Climate Service as part of the National Climate Service Act of 2009. To aid understanding of the impacts of sea level rise on coastal communities, NOAA's Coastal Services Center provides a number of new mapping tools and techniques illustrating the impacts of sea level rise and coastal flooding. One of these tools is the *Sea-level Rise and Coastal Flooding Impacts Viewer* that; (1) displays future sea level rise, (2) provides simulations of sea level rise at local landmarks, (3) communicates the spatial uncertainty of mapped sea level rise, (4) models potential marsh migration, (5) overlays social and economic data on potential sea level rise and (6) examines how tidal flooding will become more frequent with sea level rise. These tools/techniques will increase understanding of the impacts of sea level rise on salmonid habitats and should aid in an adaptive management strategy for coho salmon recovery.

Wildfire

The frequency and magnitude of wildfires are expected to increase in California (Luers *et al.* 2006; Westerling and Bryant 2006). The link between fires and sediment delivery to streams is well known (Wells 1987; Spittler 2005). Fires increase the incidence of erosion by removing vegetative cover from steep slopes. Subsequent rainstorms produce debris flows that carry sediments to streams. Increases in stream sediment can reduce egg to emergence survival and stream invertebrate production, an important food source for rearing salmon and steelhead juveniles (Bjornn and Reiser 1991; Waters 1995).

Vegetative cover

Changes in vegetative cover can impact coho salmon habitat in California by reducing stream shade (thereby promoting higher stream temperatures), and changing the amount and characteristics of woody debris in streams. High quality habitat for most CCC coho salmon streams with extant populations is dependent upon the recruitment of large conifer trees to streams. Once trees fall into streams, their trunks and root balls provide hiding cover for salmonids. In streams, large conifer trees can also interact with stream flows and stream beds and banks, creating deep stream pools needed by salmonids to escape summer high water temperatures. These pools are essential for coho salmon feeding and rearing.

Impacts on the Marine Environment

Marine ecosystems will change as a result of global climate change; many of these changes will likely have deleterious effects on salmon growth and survival while at sea. There is uncertainty about the effects of changing climate on marine ecosystems given the degree of complexity and overlapping climatic shifts currently exist (*e.g.*, El Niño, La Niña, and Pacific Decadal Oscillation). El Niño events and periods of unfavorable ocean conditions threaten the survival of salmonid populations (at low abundance) due to degradation of estuarine habitats and reduced food availability (NMFS 1996). Scientists studying the impacts of global warming on the marine environment predict the coastal waters, estuaries, and lagoons of the West Coast of the will experience increased climate variability, changes in the timing and strength of the spring transition (onset of upwelling), warming and stratification, and changes in ocean circulation and chemistry (Scavia et al. 2002; Diffenbaugh et al. 2003; Feely 2004; Osgood 2008).

Current and projected changes in the North Pacific include: rising sea surface temperatures that increase the stratification of the upper ocean; changes in surface wind patterns impacting the timing and intensity of upwelling of nutrient-rich subsurface water; and increasing ocean acidification which will change plankton community compositions with bottom-up impacts on marine food webs (ISAB 2007). Ocean acidification also has the potential to dramatically change the phytoplankton community due to the likely loss of most calcareous shell-forming species such as pteropods. Recent surveys show ocean acidification is increasing in surface waters off the west coast, and particularly the northern California coast at a more rapid rate than previously estimated (Feely *et al.* 2008). Shifts in prey abundance, composition, and distribution are the indirect effects of these changes.

Direct effects to marine organisms include decreased growth rates due to ocean acidification and increased metabolic costs as sea surface temperatures increase (Portner and Knust 2007). Northwest salmon populations have fared best in periods having high precipitation, cool air and water temperatures, cool coastal ocean temperatures, and abundant north-to-south "upwelling" winds in spring and summer. If conditions are warmer, upwelling may be delayed, and salmon may encounter less food or may have to travel further from to find satisfactory habitat, increasing energy demands, and slowing growth and delaying maturity (ISAB 2007).

Climate Variability and the Spring Transition

Global warming may change the frequency and magnitude of natural climate events that affect the Pacific Ocean (Osgood 2008). For instance, intense winter storms may become more frequent and severe. El Niño events may occur more often and be more severe. The Pacific Decadal Oscillation (PDO) is expected to remain in in warmer ocean conditions in the California current, which may result in reduced marine productivity and salmonid numbers off the coast of California (Mantua *et al*. 1997; Osgood 2008). In addition, the plankton production fueled by coastal upwelling may become more variable than in the past, both in magnitude and timing. While the winds that drive upwelling are likely to increase in magnitude, greater ocean stratification may reduce their effect (Osgood 2008). The strongest upwelling conditions may also occur later in the year (Diffenbaugh *et al.* 2003; Osgood 2008). The length of the winter storm season may also affect coastal upwelling. For example, if the storm season decreases in length, upwelling may start earlier and last longer (Osgood 2008).

Weak early season upwelling can have serious consequences for the marine food web, affecting invertebrates, birds, and potentially other biota (Barth *et al.* 2007). Weak upwelling results in low plankton production early in the spring, when salmonid smolts are entering the ocean. Plankton is the base of the food web off the California Coast, and low levels of plankton reduce food levels throughout the coastal environment. Variations in coho salmon survival and growth in the ocean are similar to copepod (salmonid prey) biomass fluctuations, which are also linked to climate variations (Mackas *et al.* 2007). Salmon smolts entering California coastal waters could be impacted by reduced foraging opportunities, which could lead to lower marine survival rates during the critical first months of their ocean rearing phase (Osgood 2008).

Ocean Warming

Ocean warming has the potential to shift coho salmon ranges northward. Warming of the atmosphere is anticipated to warm the surface layers of the oceans, leading to increased stratification. Many species may move toward the Earth's poles, seeking waters meeting temperature preferences (Osgood 2008; Cheung *et al.* 2009). Salmonid distribution in the ocean is defined by thermal limits and salmonids may move their range in response to changes in temperatures and prey availability (Welch *et al.* 1998). The precise magnitude of species response to ocean warming is unknown, although recent modeling suggests high latitude regions are likely to experience the most species invasions, while local extinctions may be the most common in the tropics; Southern Ocean, North Atlantic, the Northeast Pacific Coast, and enclosed seas (such as the Mediterranean) (Cheung *et al.* 2009).

Ocean Circulation

The California Current brings prey items for salmonids south along the coast. This current, driven by the North Pacific subtropical gyre, starts near the northern tip of Vancouver Island, Canada, flows south near the coast of North America to southern Baja, Mexico (Osgood 2008). Coastal upwelling and the PDO influence both the strength of this current and the types of marine plankton it contains. If upwelling is weakened by climate change, and the PDO tends toward a warm condition, the quantity and quality of salmonid food supplies brought south by the current could decrease (Osgood 2008). However, if rising global temperatures increase the strength of coastal upwelling, cold water fish like salmonids may do well regardless of the PDO phase (Osgood 2008).

Ocean Acidification

Although impacts to coho salmon are difficult to predict, increases in ocean acidity are of concern because they may affect the ocean's food web. The increase in atmospheric CO_2 is changing the acidity of the oceans (Feely 2004; Turley 2008; O'Donnell *et al.* 2009). The world's oceans absorb CO_2 from the atmosphere, and rising levels of atmospheric CO_2 are increasing the amount of CO_2 in seawater (Feely 2004, Turley 2008). Chemical reactions fueled by CO_2 input are increasing ocean acidity at a rate matched only during ancient planet-wide extinction events (Sponberg 2007; Brewer and Barry 2008; Turley 2008). Shelled organisms in the ocean (some species of phytoplankton and zooplankton, and snails, urchins, clams, *etc.*) are likely to have difficulty maintaining and even forming shell material as CO_2 concentrations in the ocean increase (Feely 2004; The Royal Society 2005; Brewer and Barry 2008; O'Donnell et al. 2009). Under worst case scenarios, some shell forming organisms may experience serious impacts by the end of this century (The Royal Society 2005; Sponberg 2007; Turley 2008). In addition, increased CO_2 in the oceans is likely to impact the growth, egg and larval development, nutrient generation, photosynthesis, and other physiological processes of a wide range of ocean life (Turley 2008; O'Donnell *et al*. 2009). However, the magnitude and timing of these impacts on ocean ecosystems from these effects remains uncertain (Turley 2008).

Impacts on Estuarine Environments

Impacts to estuaries and lagoons from global climate change may have greater effects on CCC coho salmon in the northern portion of their range because coho salmon likely use northern estuaries for extended rearing. CCC coho salmon in the southern portion of their range are less dependent on estuaries for rearing. In southern lagoons, observations of coho salmon occurred in April and May (Smith 1990) suggesting these fish were smolts on their way to the ocean. In the northern portion of their range, coho salmon were observed in Albion River estuary from late May through late September, suggesting that some or all of these fish may spend more time rearing in this estuary prior to smolting (Maahs 1998).

Estuaries are likely to become increasingly vulnerable to eutrophication (excessive nutrient loading and subsequent depletion of oxygen) due to changes in precipitation and freshwater runoff patterns, temperatures, and sea level rise (Scavia *et al.* 2002). These changes may affect water residence time, dilution, vertical stratification, water temperature ranges, and salinity. For example, salinities in San Francisco Bay have already increased because increasing air temperatures have led to earlier snow melt in the Sierra's which reduces freshwater flows into Bay in spring. If this trend continues or strengthens, salinities in San Francisco Bay during the dry season will increase, contributing additional stress to an already altered and highly degraded ecosystem (Scavia *et al.* 2002). If these impacts occur elsewhere, the result may lead to reduced food supplies for coho salmon using estuaries for rearing before going to sea.

Scenarios for Recovery Planning

As described above, climate change is likely to further degrade salmonid habitats. Scientists have developed scenarios, based on reasonable assumptions, using the most up to date scientific data available. These scenarios describe how climate change may affect various aspects of the environment. NMFS has relied mainly on the scenario analysis conducted by the California Environmental Protection Agency (CEPA 2006)[6] to evaluate the impacts of climate change on CCC coho salmon and their habitats. CEPA considered three CO_2 emissions scenarios: high emissions, medium high emissions, and lower emissions. Details of the environmental, population, economic, resource use, and technological assumptions behind each scenario are described in CEPA (2006). These scenarios are among the most accurate predictions of how California will be affected by climate change. It is important to note the scenarios are rough estimates of changes by the end of this century using parameters such as temperature, rainfall, vegetation, *etc*., at a statewide, West Coast, and eco-region scale.

Modeling impacts of climate change is difficult to predict over shorter time scales (Cox and Stephenson 2007). Nonetheless, progress is being made to improve predictions from climate change at shorter time intervals, at the global and regional scales (Smith and Murphy 2007). Unfortunately, predicting impacts on local geographic areas in short time frames, such as the first decade of CCC coho salmon recovery plan implementation, still remains difficult. It is reasonable to assume, given California's complex topography and variety of micro climates, variation within the CCC coho salmon ESU to impacts from climate change[7] are likely.

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[6] These scenarios are being re-evaluated by CEPA based on current information (Franco 2008). When new scenario information becomes available, NMFS will incorporate it into this recovery plan.

[7] For example, a recent article in the Santa Rosa Press Democrat reported the incidence of high temperatures in the Ukiah Valley (which includes a large portion of the mainstem Russian River) has decreased during the last 50 years, while the incidence of high temperatures in Napa Valley have increased (Porter 2008). This information suggests climate change may actually be decreasing the incidence of high temperatures in the vicinity of the Russian River. Due to the absence of peer reviewed climate change models linking global temperature changes to the Russian River watershed, we cannot project cooler temperatures in the Ukiah Valley forward into the future without developing a series of additional scenarios. Ukiah Valley temperatures could continue to drop at the same rate or a different rate, stabilize at some point in time, stabilize and then begin to go up, *etc*.

NMFS considered potential effects of the three scenarios developed by the CEPA (2006) on future habitat conditions and threats for CCC coho salmon in the freshwater environment[8]. We used many of the same habitat attributes, indicators, and threats used to evaluate the current and future condition of coho salmon habitat in this plan. In many cases, scenarios available for California are not specific enough (*i.e.*, watershed scaled) to project changes in habitat indicators or threats with reasonable certainty. Nonetheless, we conclude from the information provided by CEPA (2006) there is a higher probability of greater negative changes to coho salmon habitat under higher CO_2 emissions.

In the following sections we have focused on attributes, indicators, and threats most likely affected by climate change. For example, we considered how passage flows (all life stages), passage at river mouths (adults and smolts) and base flows are impacted by droughts as well as water diversions, impoundments and fire and fuel management. For the threat of increased magnitude and frequency of storms and flooding, we considered how redd scour and pool habitat (shelter, LWD, *etc*.) would be affected. Finally, we also considered the impacts on temperature, riparian species composition, size, and canopy cover, as well as disease, predation, and competition.

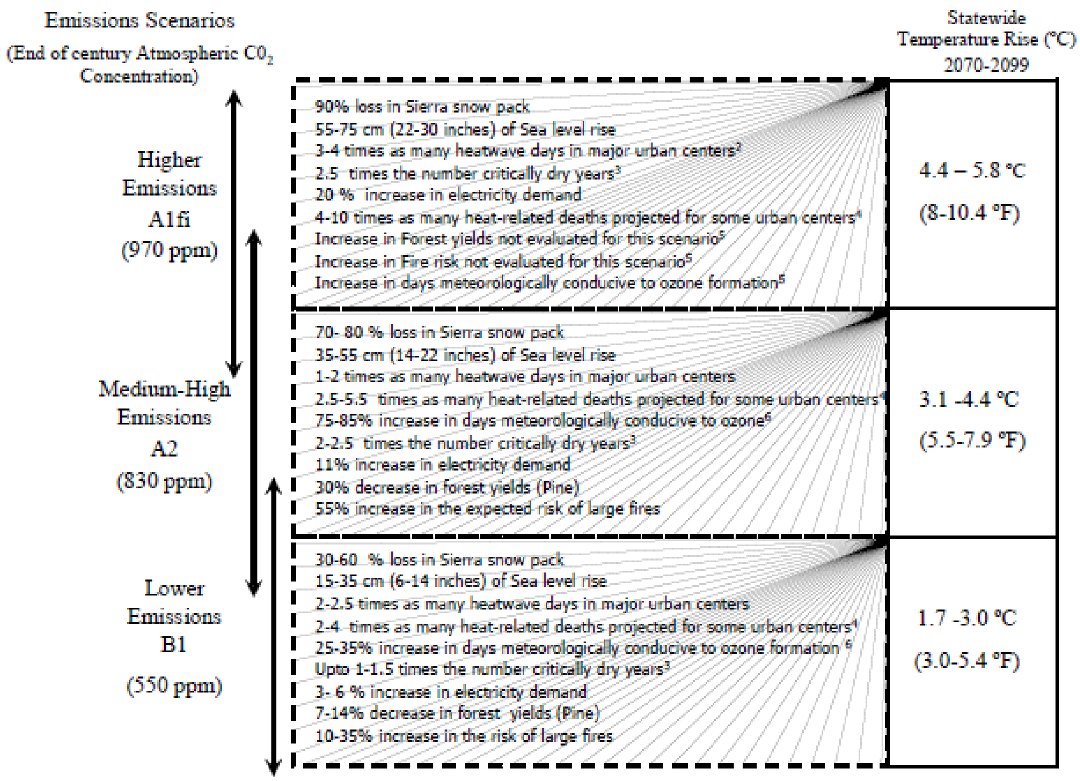
Other habitat attributes were not addressed for CCC coho salmon because: (1) they can be easily linked to changes in the above attributes, or (2) we are unable to make reasonable predictions regarding the impacts of global climate change on these attributes, indicators, or threats based on the available information. For example, agricultural practices, identified as a threat for some populations in the Recovery Plan, can result in sedimentation and turbidity. It is unclear how farmers will respond to increased droughts and changes in vegetation growth patterns, and what resulting impacts on sediment and turbidity would be. Farmers may respond by (1)

[8] We focused on the freshwater environment because more is known about habitat conditions, underlying processes that create and maintain habitat, and there is more information about what may happen due to climate change. Estuarine habitat was not analyzed because available information suggests CCC coho in the southern portion of their range use these habitats for a relatively brief interval as transitional habitat between fresh and saltwater rather than for protracted rearing as do steelhead. However, more studies are necessary from estuaries in the northern portion of the range to determine if this trend holds true throughout the ESU or if it is in response to available habitat conditions.

stopping farming and allowing the land to go fallow, (2) stopping farming and selling the land for residential or urban development, (3) changing or modifying crop rotations, (4) building additional reservoirs and/or, (5) conserving water resources, *etc*.

Emission and Temperature Scenario Overview

The CEPA model consisted of three emissions scenarios; high (970 ppm), medium-high (830 ppm), and low emissions (550 ppm) and predicted condition outcomes (CEPA 2006) (Figure 2). Modeling results indicated minor changes among the environmental impacts for different emissions scenarios between the years 2035-2050. After 2050, the environmental impacts of high emissions scenarios begin to show marked differences from lower emissions scenarios (CEPA 2006; IPCC 2007; Burgett 2009). Emissions and air temperature scenarios from Lindley *et al*. (2007) were used to access the impacts. The Lindley *et al*. (2007) modeling effort focused on Central Valley salmonids, however their analysis was illustrative because their temperature scenario maps included projections for coastal areas used by CCC coho salmon (Figure 3). NMFS recognizes such projections do not provide the level of precision and accuracy needed to determine when air temperatures may reach certain levels in particular streams.



1. Impacts presented relative to 1961–1990.
2. Los Angeles, San Bernardino, San Francisco, Sacramento, and Fresno.
3. Measures for the San Joaquin and Sacramento basins.
4. For Los Angeles, Riverside, and Sacramento.
5. Impacts expected to be more severe as temperatures rise. However, higher temperature scenarios were not assessed for the project.
6. Formation in Los Angeles and the San Joaquin Valley.

Figure 2: Emission scenarios for California for a 30-year period, identifying increased threats associated with average annual air temperature (Lindley *et al.* 2007).

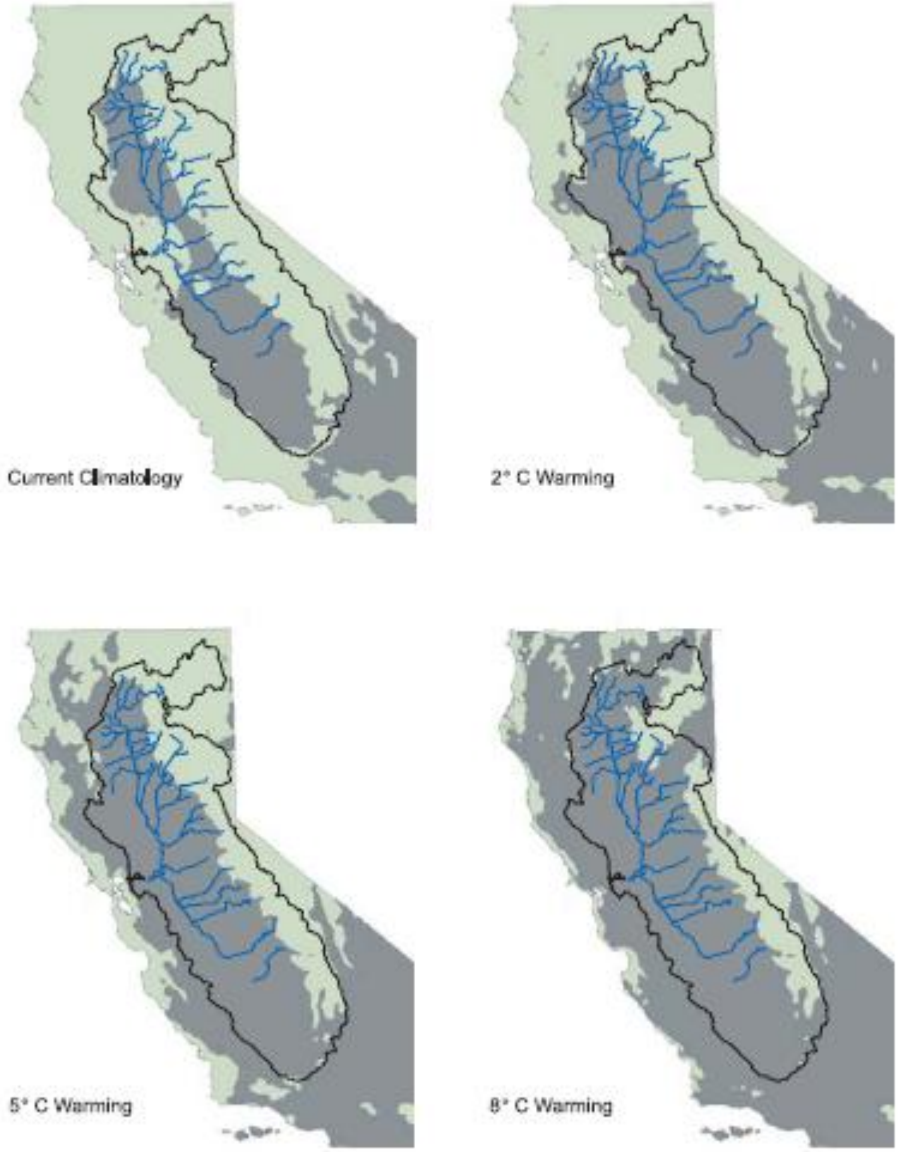


Figure 3: Geographic areas in California experiencing a mean August air temperature >25 °C by year 2100 under different warming scenarios (Lindley *et al.* 2007).

High Emissions Scenario

Under the high emissions scenario, statewide average annual temperature is expected to rise between 4.4° and 5.8° C (Luers *et al.* 2006). The temperature rise is predicted to cause loss of nearly all of the Sierra snowpack (the CCC ESU is not affected by Sierra snowpack), increase in droughts and heat waves, increased fire risk, and changes in vegetation. The North Coast is expected to experience similar effects, although the model appears to differ regarding the incidence of large storms.

Droughts

Natural climate variations such as droughts can dramatically affect habitat conditions for CCC coho salmon. In the high emission scenario, model output from droughts in California, show 2.5 times more critically dry years are possible than have occurred over the recent period (Luers *et al.* 2006). On the North Coast, various modeling efforts have produced varying results for rainfall patterns. Variations in rainfall patterns may produce various effects on CCC coho salmon and their habitat. Nonetheless, due to the uncertainties associated with rainfall on the North Coast, NMFS assumed a "worst case" reduction in precipitation similar to the statewide prediction (*i.e.*, a 2.5 increase in the number of critically dry years). Based on the overall threats ratings for droughts, and water diversions and impoundments outlined in the plan, it is reasonable to assume increases in the level of droughts will dramatically reduce total available freshwater habitat and alter the remaining habitat.

Reductions in freshwater habitat are expected to reduce freshwater survival for CCC coho across their range. The greatest impacts are expected to occur in the Coastal and Santa Cruz Mountains Diversity Strata, where droughts are rated as very high threats in many of the targeted watersheds with focus populations. In these diversity strata, NMFS anticipates severe reductions or elimination of summer rearing habitat due to limited or depleted summer base flows, leading to increased instream temperatures or dewatering. Not only are CCC coho salmon affected during baseflow conditions under this scenario, but migration flows for adults are expected to be severely curtailed, delayed, and/or absent in some years. Adults may experience increased energetic costs during migration because of low flow impediments that are more prevalent during drought than normal water years. NMFS anticipates the greatest negative impacts will be during smolt outmigration because spring flows will decline sooner under drought conditions, reducing migration opportunities. In Northern Coastal watersheds, NMFS expects, under this scenario impacts from increased droughts would be less severe, although some watersheds will exhibit large reductions in the availability of summer rearing habitat due to lack of stream flows.

Key habitat attributes at risk from climate effects were also analyzed. The current condition indicators most likely to worsen due to climate change for each watershed are discussed below. NMFS assumed vulnerability of individual CCC coho salmon populations to increased drought frequency mostly relates to the current condition of specific habitat indicators. For example, San Lorenzo River, Gazos Creek, Pescadero Creeks, Russian River, Gualala River, and Navarro Rivers are likely to be the most vulnerable to reduced adult passage flows due to drought conditions under any emissions scenario.

Fires

Increases in fire frequency or areas affected by fire were not modeled by CEPA (2006) for this scenario; however, the prevalence of fire is expected to increase under higher emission scenarios. NMFS assumes fire frequency and areas affected will be greater than the modeled results for the medium-high emissions scenario described below. Impacts from increased fires are likely to include additional sedimentation to streams. Sedimentation may fill in pools in some areas, decreasing or eliminating the value of in stream restoration efforts to increase the amount of complex habitats available for salmonids.

Storms and Flooding

A worse-case high emissions scenario was assumed which predicts storms and flooding will dramatically increase during the winter months. Increased frequency and magnitude of flows from storms and flooding are likely to increase redd scour and may affect the quantity and quality of spawning gravels, and the amount and quality of pool habitat in many watersheds. Winter rearing populations, without access to velocity refugia, are vulnerable due to increases in flood flows.

In addition, the compounding effects of roads are also a high threat for all targeted populations in the ESU. Therefore, increased magnitudes and frequency of storm and flood events are likely to cause greater sediment output and turbidity due to existing roads. Consequently, these heightened events will overwhelm the drainage capacity of many road crossings, especially under the high emission scenario. Populations most vulnerable to these impacts include the Russian River and San Lorenzo River. Based on the information in the plan, coho populations in the Santa Cruz Mountains Diversity Stratum are the most vulnerable to storms and flooding events.

Temperature

Fish, including salmonids, are sensitive to water temperature changes. Previous sections of this plan explain coho salmon temperature requirements how current stream temperature conditions in the ESU were evaluated. NMFS used, in part, the current condition ratings for temperature to identify populations most susceptible to increases in water temperatures due to climate change. Under the high emissions scenario, a 4.4° C to 5.8° C warming of statewide average annual air temperature was assumed. Figure 4 from Lindley *et al.* (2007) shows areas that may experience August mean air temperature over 25° C. These higher air temperatures are likely to cause an increase in water stream temperatures, unless other factors, such as adequate quantities of cold groundwater input are present. Figure 4 also illustrates where CCC coho salmon may be vulnerable to air temperature increases. According to this map, the interior watershed areas used by the Navarro River, Big River, Garcia River, Gualala River, and Russian River populations may experience high air and water temperatures that dramatically reduce the amount of stream habitat available to coho juveniles during the summers. This impact appears most pronounced in the Russian River, where most of the watershed, except for tributaries near the coast, may experience high temperatures. However, and as noted above, the Ukiah Valley (which contains much of the interior Russian River watershed) currently appears to be cooling, which adds to the degree of uncertainty regarding the impacts of the high temperature scenario for the coast of California.

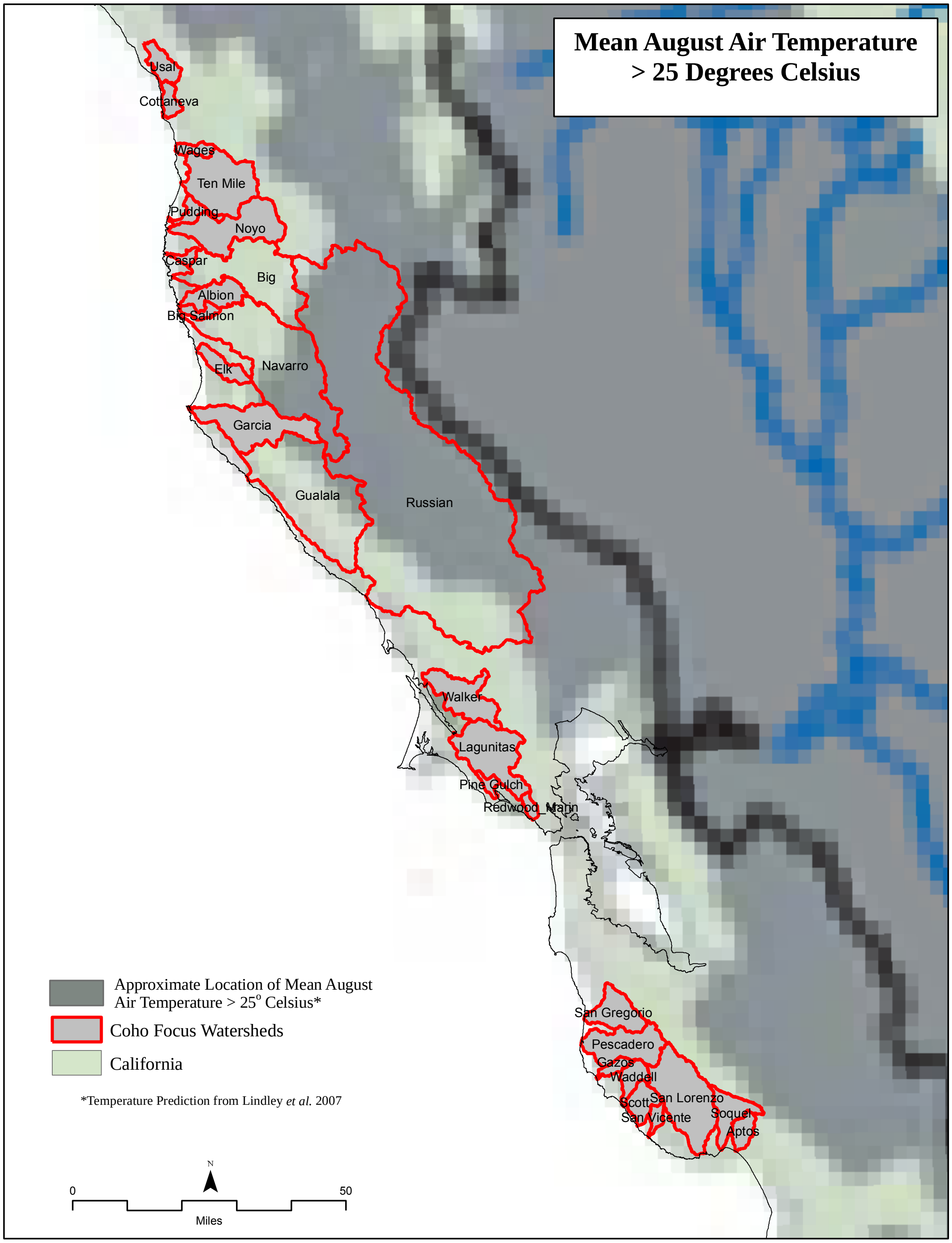


Figure 4: Approximate location of mean August air temperatures greater than 25°C in relation to coho salmon focus populations, under a 5° C warming scenario (modified from (Lindley *et al.* 2007).

Riparian Species Composition, Size, and Canopy Cover

Vegetation near streams provides shade for cooler water temperatures, bank stability, large woody debris to stream channels, and habitat for salmonids prey. Climate change is likely to affect vegetation in California, favoring some vegetation types over others, based on potential changes to air temperatures and rainfall. Scenarios developed for CEPA (2006) concerning vegetation did not include a high emissions scenario. NMFS assumed changes in vegetative cover will be more pronounced than those described under the moderate high emissions scenario. There is uncertainty regarding current information on potential changes in forest productivity. Some studies indicate the potential for increased forest productivity, while others suggest a decline (CEPA 2006). Due to this uncertainty, scenarios for tree size and canopy cover are not included in this discussion[9].

Disease, Predation, and Competition

CEPA (2006) scenarios did not include disease, predation, or competition information directly related to salmonids. However, CEPA and others (Harvell *et al.* 2002) noted that increasing instream temperatures can allow pathogens to spread into areas where they are currently absent because temperature limits their range. In some cases, increasing temperatures may limit or restrict diseases (Harvell *et al.* 2002). However, increasing temperatures likely have a greater potential to increase the susceptibility of coho salmon to disease (coho salmon prefer cooler water temperatures). Given the potential for increasing droughts, disease outbreaks will likely increase if coho salmon are crowded together in areas of low stream flow and higher water temperatures.

[9]Linking tree productivity scenarios to changes in instream habitat will be difficult in this and other scenario exercises. For example, if forest productivity decreases, LWD sizes might decline over time. However, droughts and higher temperatures are likely to raise vulnerability to pests and pathogens, which could increase tree death and thus the contribution of LWD to streams.

Moderate High Emissions Scenario

Under the moderate-high emissions scenario, statewide average annual temperature is expected to rise between 3.1° C and 4.4° C (Luers *et al*. 2006). Statewide, impacts to California's climate are similar to the high emission scenarios and include loss of most of the Sierra snowpack, increase in droughts and heat waves, increase in fire risk, and changes in vegetation.

Droughts

Statewide, there is a 2-2.5 times greater probability of a critical dry year during the medium-high emission scenario (Luers *et al*. 2006). Impacts to CCC coho salmon and their freshwater habitat are likely to be similar to those described in the high emissions scenario.

Fires

Fires are also expected to increase under this scenario. The model predicts an overall 55% increase in the risk of large fires in California (Luers *et al*. 2006). In particular, Northern California modeling results predict an overall 90% increased risk of fires (Westerling and Bryant 2006). By the end of the century the risk of fire occurrences will likely increase, even in some coastal areas that currently experience fog and cool temperatures in the summers (Westerling and Bryant 2006). Similar to the high emission scenario, impacts from increased fires are likely to include additional sedimentation in streams potentially decreasing or eliminating the amount of complex habitat for coho salmon.

Storms and Flooding

Scenarios for increased magnitudes and frequencies for storm and flood events were not modeled for Northern California. A worse-case moderate-high emissions scenario was assumed where storms and flooding dramatically increase during the winter months. Impacts under this scenario are likely similar to those expected for the high emissions scenario, although the magnitude and frequency of storm flows may be less. Similar to the high-emission scenarios, coho populations in the Santa Cruz Mountains Diversity Stratum are the most vulnerable to storms and flooding events.

Temperature

As with the high emissions scenario, NMFS used the 5° C warming-map from Lindley *et al.* (2007), which shows areas that may experience August mean air temperature over 25° C (Figure 4) as a predictor of potential change in the ESU. The higher air temperatures are likely to increase stream temperatures (unless other factors, such as cold groundwater input, are present). Impacts to coho salmon and their freshwater habitats are likely to be similar, while somewhat less than, the impacts described under the high emissions scenario.

Riparian Species Composition, Size, and Canopy Cover

Climate change will likely affect vegetation patterns in California by favoring some vegetation types over others based on potential changes to air temperatures and rainfall. Based on the maps produced by CEPA for the California moderate high emissions scenario for tree species distribution (Lenihan *et al.* 2006), NMFS inferred mixed evergreen forest (Douglas-fir, tanoak, madrone, oak) may expand toward the coast and into areas currently dominated by evergreen conifer forest (coastal redwoods) by the end of the century. Increases in tanoak, a hardwood, in coastal riparian areas could ultimately decrease the value of future LWD (although this would likely take a considerable time to actually occur due to the longevity of redwood). Streams in riparian forests composed of hardwood species generally have less LWD volume than streams in conifer riparian forests (Gurnell 2003). LWD is an important component of pool formation in some streams, and large decreases in conifer LWD could reduce the number, depths, and longevity of pools in IP-km, ultimately reducing the amount of high quality rearing and over wintering habitat available for CCC coho salmon.

Disease, Predation, and Competition

Similar to the high emission scenario, CEPA scenarios do not include disease, predation, or competition information regarding salmonids. NMFS assumed increasing temperatures may increase exposure risk, given the potential for increasing frequency of droughts. If drought frequency increases, disease outbreaks will likely increase if coho salmon are crowded together in smaller amounts of wetted habitats as well as increased competition for food and rearing resources. Potential impacts are expected to be somewhat less in severity for the moderate high emissions scenario than in the high emissions scenario.

Low Emissions Scenario

Under a low emissions scenario, statewide average annual temperature is expected to rise between 1.7° C and 3.0° C (Luers *et al.* 2006). Statewide, one-third to one-half of the Sierra snowpack is expected to be lost (although this will have little impact to the CCC ESU); there will be an increase in droughts and heat waves, increase fire risk, and changes in vegetation type and composition. Changes for the North Coast are likely to be similar, although model results appear to differ regarding the incidence of large storms, as described above in the high scenario.

Droughts

Statewide the probability of critically dry years increases 1-1.5 times for the low emission scenario (Luers *et al.* 2006). Due to the uncertainties associated with rainfall on the North Coast, a worse-case reduction in precipitation (similar to the statewide prediction) was assumed; yielding a 1-1.5 increase in the number of critically dry years. In comparison to the high and medium emission scenarios, CCC coho salmon and their freshwater habitat are less likely to be adversely affected. Impacts will most likely affect the Coastal and Santa Cruz Mountains Diversity Strata under this scenario

Fires

Fires are expected to increase under this scenario with an overall 10% to 35% increase in the risk of large fires in California (Luers *et al.* 2006). For northern California, modeling results predicted an overall 40% increase in fire risk (Westerling and Bryant 2006). By the end of the century, based upon the fire risk maps provided by Westerling and Bryant (2006), the risk of fire near the coast may increase, although the magnitude of the increase appears limited. Impacts from increased fires are likely to include additional sedimentation in streams and increased turbidity. Sedimentation may fill in pools in some areas, decreasing or eliminating the value of instream restoration efforts to increase the amount of complex habitats available.

Storms and Flooding

Scenarios for increases in storms and flooding are not available because variation in model results for climate change impacts on precipitation in Northern California. For storms and flooding, a worse case lower emissions scenario was assumed where storms and flooding increase during the winter months. Based on threat rankings, Santa Cruz Mountain Diversity Stratum coho populations are likely, the most vulnerable to storms and flooding. Impacts under this scenario are likely to be less than those expected for the moderate high and medium emissions scenarios described above.

Temperature

Current condition ratings for temperature were used to identify populations susceptible to increases in water temperatures from climate change. Under low emissions scenario, a 1.7° to 3.0° C warming of statewide average annual air temperature was assumed likely to occur. The 2° C warming-map from Lindley *et al.* (2007), was used to predict potential changes to the CCC ESU (Figure 4). According to results presented on the map, the interior Russian River and Navarro River are the areas affected by air temperature increases. However, fewer subbasins within these watersheds are more affected than in the other emission scenarios.

Riparian Species Composition, Size, and Canopy cover

See discussion in moderate high emissions scenario. These potential impacts are likely to be less than those in the moderate high emissions and high emissions scenarios.

Disease, Predation, and Competition

See discussion in the moderate high emissions scenario. These potential impacts are likely to be less than those in the moderate high emissions and high emissions scenarios.

Most Vulnerable Populations

Using the best available scientific data and information compiled in the Plan, NMFS found the following populations to be a high or very high risk of threat from climate: Pudding, Big River, Navarro River, Russian River, Lagunitas Creek, San Lorenzo River and Soquel Creek.

Recovery Planning and Climate Change

The effects of climate variability on Pacific salmon abundance are uncertain because historical records are short and abundance estimates are complicated by commercial harvesting and habitat alternation. We cannot currently predict the precise magnitude, timing, and location of impacts from climate change on coho salmon populations or their habitat. Some CCC coho salmon populations are likely to be more vulnerable than others, and these populations are identified in the plan. Monitoring and evaluating changes across the CCC coho salmon ESU on a long-term scale is critical for devising better scenarios and adjusting recovery strategies.

Survival and recovery of CCC coho salmon under any climate change scenario depends on securing and expanding viable CCC coho salmon populations. Viable populations have a better chance of surviving loss of habitat, and can likely persist in the advent of range contraction, if habitat conditions in inland and at the southern extent of the range become more tenuous. Major differences in environmental impacts of high, medium, and low emissions scenarios may not become evident until about mid-century.

A number of federal, state and local adaptive/action plans have been developed for the U.S. and the State of California. For example, in 2010, NOAA released the *Adapting to Climate Change: A Planning Guide for State Coastal Managers* document and sea level inundation toolkit, to help U.S. state and territorial (states) coastal managers develop and implement adaptation plans to reduce the risks associated with climate change impacts (NOAA 2010). In 2008, under the Executive Order S-13-08 signed by the Governor of California, the State of California began to develop state-wide and local climate adaption/action plans that focus on topics such as: the economy, ecosystem/natural resources, human health, infrastructure, society and water resources. In 2009, the California Natural Resources Agency released the *California Climate Adaptation Strategy* document. Many of the issues discussed in this document address the impacts of sea level rise, drought, flooding, air temperature and precipitation on the topics mentioned above. In the NCCC Recovery Domain, climate adaption/action plans have been developed for the San Francisco Bay (SPUR 2011); the City of San Rafael (City of San Rafael Climate Change Action Plan (City of San Rafael 2009)); and the City of Berkeley (Berkeley Climate Action Plan (City of Berkeley 2009)). At present, the state of California is the only state in U.S. to develop a cap-and-trade program on GHGs. The program is a central element of California's Global Warming Solutions Act (AB 32) and covers major sources of GHG emissions in the State such as refineries, power plants, industrial facilities, and transportation fuels. Implementation of the cap-and-trade program will be an essential component in minimizing the impacts describe above to CCC coho salmon ESU.

In the future, climate change will likely surpass habitat loss as the primary threat to the conservation of most salmonid species (Thomas *et al.* 2004). Climate change will continue to pose a continued threat to salmonids in the foreseeable future throughout the Pacific Northwest (Battin *et al.* 2007). Overall, climate change is believed to represent a growing threat to CCC coho ESU. Understanding and successfully adapting to these changes will require additional knowledge of the likely consequences and the types of actions required.

Recommended Actions and Options for Adaptive Management:

Information from federal, state, private, and public entities was used to compile specific recommended actions and options for management for climate change which include but are not limited to:

- 2010 Interagency Climate Change Adaptation Task Force Progress Report to the President;
- 2010 National Park Service's Climate Change Response Strategy;
- 2010 U.S. Fish and Wildlife Service's Strategic Plan for Responding to Accelerating Climate Change;
- 2009 U.S. Global Climate Research Program Change (USGCRP) Climate Change Impacts in the United States Report;
- 2008 U.S. Forest Service's Strategic Framework for Responding to Climate Change; and
- 2007 IPCC Fourth Assessment Report Summary.

Although options for resource managers to minimize the harm to aquatic and terrestrial resources from climate change are limited, there are several management options that can help protect and recovery coho salmon.

Stewardship and Outreach

- Actively engage stakeholders and the public regarding climate change impacts to coho salmon recovery. The website **http://www.ipcc.ch** summarizes of climate change issues for North America and the suite of actions from the IPCC to be considered for ecosystem and human health.
- Work with staff, and other entities to encourage and incorporate climate change vulnerability assessments and climate change scenarios in consultations, permitting, and restoration projects to access the impacts on coho salmon.

Research and Monitoring

- Expand research and monitoring to improve climate change predictions and effects to salmon recovery. For example, investing in marine climate change research will facilitate improved decision making by resource managers and society. Improved predictions will help ensure the future utility, protection, and enjoyment of coastal and marine ecosystems. See Appendix K for specific research needs and strategies.
- Use existing models, tools and techniques (*i.e.*, Regional Climate System Model, Sea level Rise and Coastal Flooding Impacts Viewer) to improve accuracy of ecological forecasting in order to anticipate and offset impacts related to global human population growth and development, to salmon viability and habitat.
- Support development and application of GCMs and RCMs to support research and monitoring activities listed in the recovery plan.

- Model stream flows (ranging from critical dry to wet years) to identify, prioritize, and protect areas of cool water input vulnerable to ongoing and future increases in diversion.

Protection, Minimization, Mitigation and Restoration

- Minimize increases in water temperatures by maintaining well-shaded riparian areas.
- Ensure road drainages are disconnected from the stream network to reduce the effects of discharge peaks during intense rain events.
- Protect springs and large groundwater seeps from development and water diversion. Subterranean water sources that provide cool water inflow will be increasing important in watersheds with ongoing water diversions.
- Ensure fish have access to seasonal habitats such as off-channel wintering areas and summer thermal refugia.
- Promote and maintain forest stand structures promoting fog drip.
- Promote and support policies that (a) explicitly maintain instream flow by limiting water withdrawals, (b) enhance flood-plain connectivity by opening historically flooded areas where possible, (c) remove anthropogenic barriers for fish passage, and (d) expand riparian forests to increase habitat resilience.
- Encourage and increase voluntary carbon accounting in the forest sector through certification with the California Climate Action Registry and their Forest Protocols.
- Promote land management practices that enhance carbon storage. For example, promote biological carbon sequestration best management practices (BMPs). Focus on forestlands to store carbon and reduce greenhouse gasses (See also Logging and Wood Harvesting Strategies) by working with appropriate entities to prevent forest loss, conserve and manage for older forest, and restore forests where converted to other land uses.

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APPENDIX B

CONSERVATION ACTION PLANNING ATTRIBUTES, STRESSES & THREATS REPORT

North Central California Coast Recovery Domain
CCC Coho ESU Recovery Plan

Conservation Action Planning
Key Attributes, Stresses and Threats Report

Prepared by:

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INTRODUCTION

As described in Chapter 7 (Methods) of the Plan, NOAA's National Marine Fisheries Service (NMFS) assessed instream and watershed conditions and threats using a method developed by The Nature Conservancy (TNC) in collaboration with the World Wildlife Fund, Conservation International, Wildlife Conservation Society and others called Conservation Action Planning (CAP). The CAP protocols and standards were developed by the Conservation Measures Partnership, a partnership of ten different non-governmental biodiversity organizations (www.conservationmeasures.org). The method is a "structured approach to assessing threats, sources of threats, and their relative importance to the species' status." The CAP process was adopted as the recovery planning assessment tool for the North Central California Coast (NCCC) Recovery Domain in 2006. CAP is a sophisticated Microsoft Excel-based tool adaptable to the needs of the user. The NMFS application of the CAP protocol included (1) defining current conditions for habitat attributes across freshwater life stages believed essential for the long term survival of Central California Coast (CCC) coho salmon, and (2) identifying activities reasonably expected to continue, or occur, into the future that will have a direct, indirect, or negative effect on life stages, populations and the ESU (*e.g.*, threats). The results of this assessment provided an indication of watershed health and likely threats to coho salmon survival and recovery. These results are used to formulate recovery actions designed to improve current conditions (restoration strategies) and abate future threats (threats strategies). The CAP can also track and summarize large amounts of information for each population over time, and can be adapted and iterative as new information becomes available.

CONSERVATION ACTION PLANNING OVERVIEW

CAP was developed in collaboration with the World Wildlife Fund, Conservation International, Wildlife Conservation Society and others. CAP is a planning tool used to evaluate, prioritize, and address threats to ecosystems and species. CAP is aligned with a set of open standards[1] that were developed by the Conservation Measures Partnership; a partnership of 10 different biodiversity non-governmental organizations. CAP has been applied to more than 400 landscapes in 25 countries, and TNC has officially adopted CAP as its standard conservation planning tool. CAP is also recommended in the NMFS Interim Endangered and Threatened Species Recovery Planning Guidance (Crawford and Rumsey 2011) as a preferred method to assess threats and develop recovery strategies for federally-listed marine and anadromous species.

In 2006, NMFS Southwest Region, Protected Resources Division, North Central Coast Office, partnered with TNC for their assistance and support in applying the CAP framework (*e.g.*, CAP workbook) to NCCC recovery plans. The hands-on training and interactions with TNC staff facilitated development of a customized CAP workbook template used initially for coho salmon, and expanded and modified for the other salmonid species in the NCCC Recovery Domain. Other NMFS recovery domains in California are also using the CAP workbook, or a modified version of the process, to develop their recovery plans.

A CAP workbook was created for each of the 28 focus populations and each workbook has two assessment components: viability (evaluating current conditions) and threats (evaluating future stresses and source of stress). The CAP workbooks provided a foundation to analyze key habitat, landscape and watershed factors relative to specific life stage requirements of salmonids. The CAP workbooks were

[1] More information about the open standards is available at "conservationmeasures.org."

used to identify and analyze current conditions and, ongoing and future stresses and threats to each population. Key attributes define current conditions for each targeted salmonid population, while stresses and threats define current conditions and conditions in the future. The analysis of key attributes is a distinct and separate analysis from the analysis of stresses and threats. The CAP workbooks also provided rationale and transparency in development of specific recovery actions, and prioritization of recovery actions designed to improve habitat attributes ranked as "poor", and reduce stresses and threats ranked as "high" or "very high."

This report provides the rationale, analysis steps, and references behind habitat, landscape and watershed attributes and indicator results and ratings within the CAP workbook viability table. The viability table was used to assess the status of current conditions for CCC coho salmon. This report also provides similar rationale, analysis steps, and references for the stress and threat analysis portion of the CAP workbook.

Assessing Current Conditions: The Viability Table

Viability describes the status or health of a population of a specific plant or animal species (TNC 2007). More generally, viability indicates the ability of a conservation target to withstand or recover from most natural or anthropogenic disturbances and thereby persist for many generations or over long time periods. The viability table within each CAP workbook provides an objective, consistent framework for defining the current status and the desired future condition of a conservation target, while tracking changes in the status of a conservation target over time. The viability table defines specific life stages for each species as "conservation targets", and provides the structure for an assessment of current conditions supported by data from NMFS, other agencies, recovery partners, and the scientific literature.

Conservation Targets

Because salmonid habitat use varies substantially by species and life stage, targets for specific life stages and an additional target to evaluate watershed processes were defined. Discrete life stages were used to assess habitat attributes during critical time frames of the species life history. The targets used in the workbooks and their definitions are described below:

- Spawning Adults – Includes adult fish from the time they enter freshwater, hold or migrate to spawning areas, and complete spawning (September 1 to March 1);
- Eggs – Includes fertilized eggs deposited into redds and the incubation of these eggs through the time of emergence from the gravel (December 1 to April 1);
- Summer Rearing Juveniles – Includes juvenile rearing in streams and estuaries (when applicable) during summer and fall (June-October) prior to the onset of winter rains;
- Winter Rearing Juveniles – Includes rearing of juveniles from onset of winter rains through the winter months up to the initiation of smolt outmigration (November 1 to March 1);
- Smolts – Includes juvenile migration from natal rearing areas until they enter the ocean (March 1 to June 1); and
- Watershed processes - Includes instream habitat, riparian, upslope watershed conditions and landscape scale patterns related to landuse.

Key Attributes

Key attributes are defined as critical components of a conservation target's biology or ecology (TNC 2007). Viable populations result when key attributes function and support transitions between life history stages. By this definition, if attributes are missing, altered, or degraded then it is likely the species will experience more difficulty moving from one life stage to the next. Factors with the greatest potential to impair survival across life stages and limit salmonid production at the population scale were defined as key attributes.

Two categories of attributes describe aspects of the aquatic habitat and watershed processes that affect aquatic and riparian habitats (habitat condition and landscape context attributes), while a third (population size) describes viability parameters (*e.g.*, abundance and distribution) for salmonids. Each attribute is described below.

Indicators and Indicator Ratings

Indicators are a specific habitat, watershed process or population parameter providing a method to assess the status of a key attribute. An attribute may have one or more indicators, and each indicator is an objective, measurable aspect of an attribute (Table 1). Each indicator has a rating which is a reference value describing the conditions of the key attribute as it relates to life stage survival. These conditions are rated as poor, fair, good or very good. Most reference values or indicator ratings were developed using established values from published scientific literature. Measurable quantitative indicators were used for most indicators; however, the formulation of other more qualitative decision making structures were used when data were limited. Qualitative decision structures were used to rate three attributes: instream flow conditions, estuary conditions, and toxicity.

Table 1. CCC coho salmon CAP attributes and indicators by

CCC Coho Population Conditions By Target Life Stage		
Target	Attribute	Indicator
Adults	Habitat Complexity	Large Wood Frequency (BFW 0-10 meters)
		Large Wood Frequency (BFW 10-100 meters)
		Pool/Riffle/Flatwater Ratio
		Shelter Rating
	Hydrology	Passage Flows
	Passage/Migration	Passage at Mouth or Confluence
		Physical Barriers
	Riparian Vegetation	Tree Diameter (North of SF Bay)
		Tree Diameter (South of SF Bay)
	Sediment	Quantity & Distribution of Spawning Gravels
	Velocity Refuge	Floodplain Connectivity
	Water Quality	Toxicity
		Turbidity
	Viability	Density
Eggs	Hydrology	Flow Conditions (Instantaneous Condition)
		Redd Scour
	Sediment	Gravel Quality (Bulk)
		Gravel Quality (Embeddedness)
Summer Rearing Juveniles	Estuary/Lagoon	Quality & Extent
	Habitat Complexity	Large Wood Frequency (Bankfull Width 0-10 meters)
		Large Wood Frequency (Bankfull Width 10-100 meters)
		Percent Primary Pools
		Pool/Riffle/Flatwater Ratio
		Shelter Rating
	Hydrology	Flow Conditions (Baseflow)
		Flow Conditions (Instantaneous Condition)
		Number, Condition and/or Magnitude of Diversions
	Passage/Migration	Passage at Mouth or Confluence
		Physical Barriers
	Riparian Vegetation	Canopy Cover
		Tree Diameter (North of SF Bay)
		Tree Diameter (South of SF Bay)
	Sediment (Food Productivity)	Gravel Quality (Embeddedness)
	Water Quality	Temperature (MWMT)
		Toxicity
		Turbidity
	Viability	Density
		Spatial Structure
Winter Rearing Juveniles	Habitat Complexity	Large Wood Frequency (Bankfull Width 0-10 meters)
		Large Wood Frequency (Bankfull Width 10-100 meters)
		Pool/Riffle/Flatwater Ratio
		Shelter Rating
	Passage/Migration	Physical Barriers
		Tree Diameter (North of SF Bay)
		Tree Diameter (South of SF Bay)
	Sediment (Food Productivity)	Gravel Quality (Embeddedness)
	Velocity Refuge	Floodplain Connectivity
	Water Quality	Toxicity
		Turbidity
Smolts	Estuary/Lagoon	Quality & Extent
	Habitat Complexity	Shelter Rating
	Hydrology	Number, Condition and/or Magnitude of Diversions
		Passage Flows
	Passage/Migration	Passage at Mouth or Confluence
	Smoltification	Temperature
	Water Quality	Toxicity
		Turbidity
	Viability	Abundance
Watershed Processes	Hydrology	Impervious Surfaces
	Landscape Patterns	Agriculture
		Timber Harvest
		Urbanization
	Riparian Vegetation	Species Composition
	Sediment Transport	Road Density
		Streamside Road Density (100 m)

Each indicator has a set of indicator rating criteria representing quantitative or qualitative reference values describing the conditions of the key attribute as it relates to life stage survival and transition. These indicator rating criteria provide an assessment of the current health of each attribute across a population expressed through the most recent measurement for the indicator (TNC 2007). Any given attribute will vary naturally over time, and is considered within an acceptable range when meeting defined critical thresholds (TNC 2007). The status of the attribute can then be expressed in context (when the measurement is compared to indicator rating criteria) which are defined by quantitative thresholds to describe the range of variation. These conditions are rated as poor, fair, good or very good according to the following criteria:

Very Good	The indicator is in an ecologically desirable status, requiring little intervention for maintenance. Very good values were considered fully functional to allow complete life stage function and life stage transition.
Good	The indicator is within an acceptable range of variation, with some intervention required for maintenance. Good values were considered functional but slightly impaired.
Fair	The indicator is outside acceptable range of variation, requiring human intervention. Fair values were considered functional but significantly impaired.
Poor	Restoration is increasingly difficult, and may result in extirpation of the target. Poor values are inadequate for life stage transitions.

In watersheds where the majority of indicators were rated as good or very good, overall conditions were likely to be functional and support transitions between life stages within the historical range of variability.

The quantitative indicator rating criteria boundaries and thresholds vary by indicator and attribute type (*e.g*., condition, landscape or size). NMFS utilized references from the scientific literature and other sources to establish the quantitative ranges and thresholds for each of the rating categories for each indicator. In some cases, only the upward (*e.g*., good) and lower (*e.g*., poor) limits of each indicators' range were available from the scientific literature, so that fair and very good rating boundaries were established via interpolation, or left undefined. Measurable quantitative indicators were used for most indicators; however, the formulation of other more qualitative decision making structures were used when data were limited. Qualitative decision structures were used to rate three attributes: instream flow conditions, estuary conditions, and toxicity. In watersheds where the majority of indicators were rated as good or very good, overall conditions were likely to represent the historical range of variability and supporting transition between life stages.

The scale of available data used for rating an indicator varied by attribute type (*e.g*., condition, landscape and size). For example, landscape attribute data (*e.g*., most land cover data) are available via GIS datasets at the watershed level (*i.e*., population scale), or can be aggregated to a watershed scale. Condition and size attribute data however, are typically collected at much finer scales (*e.g*., site, reach or stream). These data require aggregation at multiple scales to arrive at a population rating. For example, data for many indicators (*e.g*., percent of primary pools) were available at the stream reach (or summarized habitat unit) level and these data must first be aggregated to obtain a stream level rating, then scaled across multiple streams to attain a population or watershed level rating.

Scaled Population Rating Strategy

A scaled population rating strategy was developed within the framework of TNC's CAP process and the intrinsic potential habitat (IP-km) model developed by the Bjorkstedt *et al*. (2005) and Spence *et al*. (2008). The IP-km model used criteria for stream gradient, valley width, and mean annual discharge, to provide quantitative estimates of potential habitat for each population in kilometers (km), with qualitative estimates of the intrinsic potential (IP) weighted (between 0 and 1). These values provided an estimate of the value of each km segment for each species (coho salmon, Chinook salmon, and steelhead) inhabiting a particular watershed. Historical and current IP-km estimates were used to determine historical and current population abundance targets. Known migration barriers were used to evaluate the current extent of IP. In many cases the current IP extent was modified based on the current condition and likely irretrievability of some stream reaches to achieve properly functioning conditions.

Scaled population ratings were based on the relevant contribution each site, reach, and stream makes to the population as a whole. Where data were collected at finer scales, data were aggregated up to arrive at a single rating for a given population. A typical rating scenario involved two to three steps; 1) a rating at the site or reach levels, 2) rating at the stream level, and 3) a rating at the population level, which aggregated multiple stream ratings. Reach and stream level ratings were incorporated into the CAP Workbook analysis for each population.

CDFG stream habitat-typing data, known as the HAB 8 dataset, informed many of the attribute indicators in the CAP Workbook. Data from multiple stream reaches were aggregated to rank each stream based on the criteria for each indicator, and its ability to support a particular life stage or stages. As an example, CDFG considers a primary pool frequency of 50 percent desirable for salmonids (Bleier *et al*. 2003). Primary pool frequency varies by channel depth and stream order[2] therefore, to extrapolate reach scale data upward to the stream scale, rating criteria were established which used a 25 percent boundary from the 50 percent threshold to describe good conditions (*i.e*. the indicator was within acceptable range of variation). Criteria for poor, fair and very good ratings followed the same procedure to establish numeric boundaries for each qualitative category at the stream level scale:

<u>Stream level percent primary pool</u>
Poor = < 25% primary pools;
Fair = 25% to 49% primary pools;
Good = 50% to 74% primary pools; and
Very Good = > 75% primary pools.

Because ratings were ultimately applied at the watershed or population scale, and a population could include multiple streams, stream level ratings were aggregated to obtain a population level rating, and characterize the contribution of each stream/watershed to the population. Good conditions were defined as the level which described an acceptable limit of the variation inherent to each indicator constituting the minimum conditions for persistence of the target. If the indicator measurement lies below this acceptable range, it was considered to be in degraded condition. Specifically, a "good" stream rating was considered the minimum value necessary to complete life stage function and transition. However, all streams cannot be expected to achieve optimal criteria within the entire population, at all places, at all times. To account for natural variation at the population scale, quartile ranges (< 50%, 50-75%, 75-90%, >

[2] Stream order is a hierarchal measure of stream size. First order streams drain into second order streams, and so on. The presence of higher order streams suggests a larger, more complex watershed.

90%) were used for population level rankings to extrapolate stream level data upward to the population scale:

<u>Population level percent primary pool rating criteria</u>
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75% to 90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Represented schematically, Figure 1 illustrates this stepwise aggregation of data to arrive at a watershed level rating for each attribute.

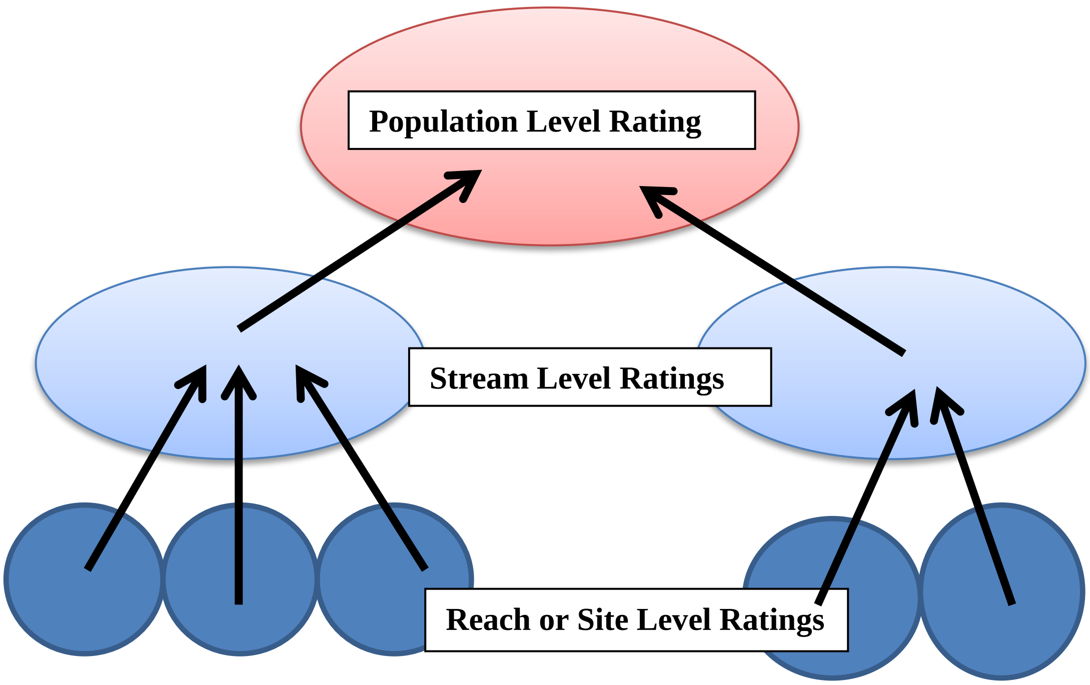

Figure 1. Schematic representation of stepwise aggregation of data, beginning with site or reach specific data, to arrive at a single population or watershed level attribute rating.

Stream attributes are unlikely to meet good conditions across 100 percent of a watershed/population, given the natural variability in geomorphic variables such as reach type, stream order, stream width and gradient, hydrologic variables such as rainfall, biologic factors such as vegetation, and the varying degree of natural disturbances such as fire, flood or drought.

Spatial Analysis

In situations where the percent-of-streams metric deviated from the percent IP-km metric or where the rating criteria is not consistent (*e.g.*, poor vs. good in different streams within the same watershed), the percent IP-km rating criteria was used as the default. In these cases, map based (GIS and Google Earth) analysis tools were used to visually evaluate each streams' contribution to the universe of good quality habitat for each population. Where quantitative measurements were lacking, a qualitative estimate was used based on best available literature, spatial data and IP-km extent and ranges (discussed below). Population level ratings are presented within each population profile (see Volume II) to summarize conditions and for comparative purposes across the ESU.

NMFS GIS staff mapped IP-km extent and value utilizing Google Earth (.kml files) to provide spatial representation of the historical intrinsic potential in for various data layers and analysis. These data were used in combination with the HAB 8 layer (#4 below), to compare the current condition of a given habitat segment to its historical expectation/performance/contribution. The following criteria were used:

1. IP extent and value per Calwater/sub-watershed unit GIS map for each recovery population/watershed provided spatial representation of each streams/sub-watersheds highest percentage IP-km values. IP-km valued habitats were color coded within each Calwater/sub-watershed unit;
2. IP numeric extent and rank per Calwater/sub-watershed unit Excel spreadsheet for each recovery population/watershed provided the numeric information corresponding to the Calwater/sub-watershed highest percentage maps. This spreadsheet included a breakdown of the ratio of IP-km valued habitat within each Calwater/sub-watershed unit; the extent (km) of each IP-km valued habitat within each Calwater/sub-watershed unit; and the total (km) of IP-km valued habitat within a given Calwater/sub-watershed unit;
3. CDFG surveyed reaches (HAB 8 data) were overlaid on Google Earth providing spatial representation of the extent of HAB 8 data. This was utilized in combination with the IP-km layer (#1) to aid the viewer in making a determination of the extent in which a given populations IP-modeled habitat had been surveyed; and
4. Reach scale HAB 8 survey extent overlaid on IP-km modeled habitat on maps to evaluate discrepancies between percent of stream and percent of IP-km rating criteria for a particular indicator. Maps also displayed IP-km modeled habitat color coded by value (high, medium, low) and specific HAB 8 surveyed reach locations.

Confidence Ratings

The assessment of watershed conditions for the indicators defined below relied heavily on CDFG's stream habitat-typing data (HAB 8 dataset[3]). While this dataset provided the best available coverage throughout the NCCC Recovery Domain, it did not cover all IP-km or all watersheds, and in some cases covered only small portions of a watershed.

We analyzed the variable coverage of HAB 8 data across watersheds to measure the confidence in our conclusions at the population scale. Two measures were investigated; 1) the percent of IP-km covered by HAB 8 surveys, and 2) the relative distribution of IP-km values within the surveyed areas compared to the population as a whole.

The percent of IP-km covered gave a measure of sample size. For example, confidence might be low if less than 20 percent of all IP-km in the population were surveyed, which could be significant if this indicator alone characterized the population as a whole. Table 2 shows how confidence increased as a function of increased coverage.

Table 2. Confidence ratings for HAB 8 data as a function of percent of IP-km surveyed.

Confidence	Low	Fair	High	Very High
% Coverage	< 20	20-50	50-80	> 80

[3]Methods for Hab-8 surveys are described in Flosi *et al*. (2004).

To determine whether surveyed areas were representative of habitat throughout the population, we the distribution of IP-km values (between 0 and 1) were compared within the surveyed reaches to the overall distribution of IP-km values in the population. For both sets the average IP-km value and standard deviations (SD) was calculated. The Albion River population for example, had an average IP-km value of 0.58 (SD 0.28). This Albion River comparison provides a relative indication of total surveyed areas compared to other watersheds (0.71 (SD 0.39)).

Putting it all together: Attributes, Indicators and Ratings

This section details all key attributes, indicators, and ratings used in the CAP workbooks and describes methods used to inform those ratings.

Attribute: Estuary/Lagoon

Estuaries and lagoons provide important habitat for the physiological changes young salmonids undergo as they prepare to enter the ocean (smoltification), and provides important habitat for some rearing salmonids.

Condition Indicator: Estuary/Lagoon Quality & Extent for Sumer Rearing and Smolt Targets

Many estuaries and lagoons across the NCCC Domain have been degraded by management actions such as channelization, artificial breeching, encroachment of infrastructure such as highways, bridges, residential and commercial development, and sediment deposition. These and other anthropogenic effects have reduced estuary and lagoon habitat quality and extent.

Ratings:

An estuary protocol was developed using a variety of components of estuary/lagoon habitat using a qualitative decision structure. Rating thresholds were defined in the following manner:

Poor = Impaired/nonfunctional;
Fair = Impaired but functioning;
Good = Properly functioning conditions; and
Very good = Unimpaired conditions.

Methods:

Because data were lacking in many populations a qualitative decision structure was developed to derive ratings for the estuary/lagoon indicator. The protocol provided a structured process to capture and evaluate diverse types of data where it was available, and to apply qualitative assessments where data were lacking. It included three major components:

- General rating parameters applied to all estuaries and lagoons to evaluate the current extent and adverse alterations to the river mouth, hydrodynamics (wetland and freshwater inflow), and artificial breeching;
- Rating parameters for estuaries functioning or managed as open systems from March 15 to November 15 (to include the pre-smolt timing of the summer rearing period); and
- Rating parameters for lagoons currently functioning or managed as close systems from March 15 to November 15 (to include the pre-smolt timing of the summer rearing period).

I. General Rating Parameters for Estuaries and Lagoons

***Includes the pre-smolt timing of the summer rearing period.**

Criteria	Population Name	Confidence/Source
1. Current Extent: Fraction of the Estuary/Lagoon in Natural Conditions		
2. Alteration to River Mouth Dynamics (Estuary Opening Patterns)		
3. Alterations to Hydrodynamics: Inner Estuary/Lagoon Wetlands		
4. Frequency of Artificial Breaching (Seasonal)		
5. Alterations to Freshwater Inflow (refer to Instream Flow Protocol)		
Overall ranking		

1. **Current Extent: Fraction of the estuary and/or lagoon in natural conditions (prior to European settlement); including tracts of salt and freshwater marshes, sloughs, tidal channels, including all other tidal and lagoon inundated areas:**

Very Good	Good	Fair	Poor
≥ 95%	95-67%	66-33%	< 33%

2. **Alteration to river mouth dynamics leading to changes in estuary opening patterns due to jetties, tide gates, roads/railroads, bridge abutments, dredging, and artificial breaching, *etc*.:**

Very Good	Good	Fair	Poor
No modification	Slight modification to estuary entrance, but still properly functioning	Some modification altering the estuary entrance from naturally functioning	Major modification restricting the estuary entrance from properly functioning

3. **Alterations to *INNER* estuary/lagoon hydrodynamics (upstream of the river mouth) due to construction of barriers (dikes, culverts, tide gates, roads/railroads, *etc*.):**

Very Good	Good	Fair	Poor
No impairments	Some impairments; 95-67% of the estuary/lagoon remains hydrologically connected	Impairments, but 66-33% of the estuary/lagoon remains hydrologically connected	Extensive impairments, with <33% of the estuary/lagoon hydrologically connected

4. Frequency of artificial breaching events:

Very Good	Good	Fair	Poor
No artificial breaching occurs: natural variability	<1 artificial breaching event immediately following a rain event; no artificial breaching during the rearing season (March 15 – November 15)	Artificial breaching events only occur prior to significant storm events	Winter and summer breaching events independent of rain events

5. Alterations to freshwater inflow (refer to Instream Flow Protocol for guidance):

Very Good	Good	Fair	Poor
No impoundments within the watershed	Total impoundment volume <20% median annual flow	Total impoundment volume 20-50% median annual flow	Total impoundment volume 51-100% median annual flow

II. Estuary: ***Currently Functioning or Managed as an Open System (*Rearing Season: March 15 – November 15)***
***Includes the pre-smolt timing of the summer rearing period.**

Criteria	Population Name	Confidence/Source
Tidal Prism: Estuarine Habitat Zones		
Tidal Range (Flushing Rate)		
Temperature (C): Estuarine Habitat Zones		
Dissolved Oxygen (mg/L): Estuarine Habitat Zones		
Macro-Invertebrates Abundance and Taxa Richness: Estuarine Habitat Zones		
Habitat Elements and Complexity		
Toxicity (Metal, Pesticides, Pollution, *etc*.)		
Exotic Pest Species		
Overall ranking		

1. **Estuarine Habitats Zones: Marine salinity zone (33 to 18 ppt); mixing/transitional zone (18 to 5 ppt); and riverine/freshwater tidal zone (5 to 0 ppt):**

Very Good	Good	Fair	Poor
All zones are present and are relatively equal in total area - natural tidal prism (33.3% ea.)	Any approximate percentage ratio with a 40/40/20 combination (example: 20% MSZ; 40% MZ; 40% RTZ)	Any approximate percentage ratio with a 45/45/10 combination	Any approximate percentage ratio with <10% of any one zone represented

2. **Tidal Range (flushing rate):**

Very Good	Good	Fair	Poor
Estuary reach very well flushed (macro-tidal); excellent vertical mixing	Estuary reach moderately well flushed (meso-tidal); good vertical mixing	Estuary reach is moderately flushed (micro-tidal); some vertical mixing occurs, but some areas remain stagnant (not mixed or flushed)	Estuary reach very poorly flushed (ultra micro-tidal); poor vertical mixing resulting in reduced water quality (low DO)

3. **Relative temperature within each Estuarine Habitat Zones (marine salinity zone, mixing/transitional zone, and riverine tidal zone):**

   a. **Temperature: Marine Salinity Zone (33 to 18 ppt) -** Immediately inside the mouth of the estuary to the start of the mixing/transitional zone:

Very Good	Good	Fair	Poor
< 14.0° C	14.1-16.5° C	16.6-18.0° C	> 18.0° C

   b. **Temperature: Mixing/Transitional Zone (18 – 5 ppt) –** Area where the salinity within the Estuarine Habitat Zone ranges from 18 to 5 ppt:

Very Good	Good	Fair	Poor
< 16.0° C	16.1°-18.0° C	18.1°-20.0° C	> 20.1° C

   c. **Temperature: Riverine or Freshwater Tidal Zone (<5 ppt) –** Area from the mixing/transitional zone to the head-of-tide:

Very Good	Good	Fair	Poor
< 17° C	17.1°-19.0° C	19.1°-21.5° C	> 21.6° C

4. **Relative Dissolved Oxygen (mg/L) for a given duration within each Estuarine Habitat Zones (marine salinity zone, mixing/transitional zone, and riverine tidal zone):**

   a. **Dissolved Oxygen (mg/L): Marine Salinity Zone -** Immediately inside the mouth of the estuary to the beginning of the mixing/transitional zone:

Very Good	Good	Fair	Poor
>7.75 mg/L at all times	7.74-6.5 mg/L at all times	Fall below 6.4 mg/L, but stays above 5.0 mg/L for < 24hrs	Falls below 5.0 mg/L for periods > 24 hours

b. **Dissolved Oxygen (mg/L): Mixing/Transitional Zone –** Area where the Estuarine Habitat Zone ranges from 18 to 5 ppt:

Very Good	Good	Fair	Poor
>7.75 mg/L at all times	7.74-6.5 mg/L at all times	Fall below 6.4 mg/L, but stays above 5.0 mg/L for < 24hrs	Falls below 5.0 mg/L for periods > 24 hours

c. **Dissolved Oxygen (mg/L): Riverine or Freshwater Tidal Zone –** Area from the mixing/transitional zone to the head-of-tide:

Very Good	Good	Fair	Poor
> 7.75 mg/L at all times	7.74-6.5 mg/L at all times	Fall below 6.4 mg/L, but stays above 5.0 mg/L for < 24hrs	Falls below 5.0 mg/L for periods > 24 hours

5. **Relative Macro- Invertebrate Abundance and Taxa Richness within each Estuary Habitat Zone –** Macro-invertebrates that are known or would be considered to be available prey items for juvenile salmonids:

a. **Relative Macro- Invertebrate Abundance and Taxa Richness): Marine Salinity Zone -** Immediately inside the mouth of the estuary to the start of the mixing zone:

Very Good	Good	Fair	Poor
Abundance and taxa richness are considered to be high	Abundance of prey items is high, but taxa richness is relatively low	Abundance is of prey items and/or taxa richness are moderate	Abundance of prey items and/or taxa richness are low

b. **Relative Macro- Invertebrate Abundance and Taxa Richness Mixing/Transitional Zone –** Area where the salinity zone ranges from 18 to 5 ppt:

Very Good	Good	Fair	Poor
Abundance and taxa richness are considered to be high	Abundance of prey items is high, but taxa richness is relatively low	Abundance is of prey items and/or taxa richness are moderate	Abundance of prey items and/or taxa richness is low

c. **Relative Macro- Invertebrate Abundance and Taxa Richness: Riverine or Freshwater Tidal Zone –** Area from the mixing/transitional zone to the head-of-tide**:**

Very Good	Good	Fair	Poor
Abundance and taxa richness are considered to be high	Abundance of prey items is high, but taxa richness is relatively low	Abundance is of prey items and/or taxa richness are moderate	Abundance of prey items and/or taxa richness is low

6. **Habitat Elements and Complexity -** % area containing SAV, large or small WD, emergent and/or riparian vegetation, marshes, sloughs, tidal wetlands, pools > 2 meters, *etc*.:

Very Good	Good	Fair	Poor
> 70%	70-45%	45-20%	<20%

7. **Toxicity - Toxicity -** % of area where containments are detected (metals, pesticides, and pollution that are impacting the estuary ecosystem, *etc*.):

Very Good	Good	Fair	Poor
Not detected	< 2%	2.1-5%	> 5%

8. **Exotic Pest Species -** Number of exotic pest species that alter the estuary ecosystem and significantly impact salmonids (please note how exotic pest species impacts salmonids - *i.e*., stripers - predation):

Very Good	Good	Fair	Poor
No exotic pest species known to be present	One or more pest species present but there are **no major** impacts to salmonids and the estuary ecosystem	One or more pest species present and at least one is having a **moderate** impact to salmonids and the estuary ecosystem	One or more pest species present and at least one is having a **major** impact to salmonids and the estuary ecosystem

9. **Quantity of Rearing Habitat (Life Stage and Species) = OVERALL**

    a. **Quantity of rearing habitat for young-of-year coho and/or *NON-osmoregulating* salmonids (refer to rating listed above for guidance – Estuarine Habitat Zones, water quality parameters, etc.):**

Very Good	Good	Fair	Poor

    b. **Quantity of rearing habitat for osmoregulating salmonids (refer to rating listed above for guidance – Estuarine Habitat Zones, water quality parameters, *etc*.):**

Very Good	Good	Fair	Poor

III. **Lagoon:** ***Currently Functioning or Managed as a Closed System (*Rearing Season: March 15 – November 15)***

***Includes the pre-smolt timing of the summer rearing period.**

Criteria	**Population Name**	**Confidence/Source**
Seasonal Closure (date/month)		
Freshwater Conversion (d)		
Lagoon Elevation – NGVD (ft.)		
Temperature (C): Lagoon Habitat Zones		
Dissolved Oxygen (mg/L): Lagoon Habitat Zones		
Macro-Invertebrates Abundance and Taxa Richness: Lagoon Habitat Zones		
Habitat Elements and Complexity		
Toxicity (Metal, Pesticides, Pollution, etc.)		
Exotic Pest Species		
Overall ranking		

1. **Seasonal Closure –** Timing of sandbar formation creating a summer rearing lagoon (date/month):

Very Good	**Good**	**Fair**	**Poor**
April 15 – May 7	**May 7 – June 1**	**June 1 – June 21**	**Later than June 21st**

2. **Freshwater Conversion –** number of days required to complete freshwater transformation:

Very Good	**Good**	**Fair**	**Poor**
1 to 3	**3 to 7**	**7 to 14**	**>14**

3. **Freshwater Lagoon Elevation during seasonal closure (NGVD):**

Very Good	**Good**	**Fair**	**Poor**
> 5 feet	**> 4 feet**	**> 3 feet**	**< 3 feet**

4. **Relative temperature within each Lagoon Habitat Zone (Lower, Middle, Upper):**

   a. **Temperature: Lower Lagoon Habitat Zone -** Immediately inside the sandbar to approximately the middle reach of the lagoon:

Very Good	**Good**	**Fair**	**Poor**
< 16.0° C	16.1°-18.0° C	18.1°-20.0° C	> 20.1° C

b. Temperature: Middle Lagoon Habitat Zone:

Very Good	Good	Fair	Poor
< 17° C	17.1°-19.0° C	19.1°-21.5° C	> 21.6° C

c. Temperature: Upper Lagoon Habitat Zone:

Very Good	Good	Fair	Poor
< 17° C	17.1°-19.0° C	19.1°-21.5° C	> 21.6° C

5. Relative Dissolved Oxygen (mg/L) for a given duration within each of the Lagoon Habitat Zones (Lower, Middle, Upper):

a. Dissolved Oxygen (mg/L): Lower Lagoon Habitat Zone - Immediately inside the mouth of the estuary to the start of the mixing/transitional zone:

Very Good	Good	Fair	Poor
> 7.75 mg/L at all times	7.74-6.5 mg/L at all times	Fall below 6.4 mg/L, but stays above 5.0 mg/L for <24hrs	Falls below 5.0 mg/L for periods > 24 hours

b. Dissolved Oxygen (mg/L): Middle Habitat Zone:

Very Good	Good	Fair	Poor
> 7.75 mg/L at all times	7.74-6.5 mg/L at all times	Fall below 6.4 mg/L, but stays above 5.0 mg/L for < 24hrs	Falls below 5.0 mg/L for periods > 24 hours

c. Dissolved Oxygen (mg/L): Upper Lagoon Habitat Zone:

Very Good	Good	Fair	Poor
> 7.75 mg/L at all times	7.74-6.5 mg/L at all times	Fall below 6.4 mg/L, but stays above 5.0 mg/L for < 24hrs	Falls below 5.0 mg/L for periods > 24 hours

6. Relative Macro- Invertebrate Abundance and Taxa Richness within each Lagoon Habitat Zone – Macro-invertebrates that are known or would be considered to be available prey items for juvenile salmonids:

a. Relative Macro- Invertebrate Abundance and Taxa Richness: Lower Lagoon Habitat Zone:

Very Good	Good	Fair	Poor
Abundance and taxa richness are considered to be high	Abundance of prey items is high, but taxa richness is relatively low	Abundance is of prey items and/or taxa richness are moderate	Abundance of prey items and/or taxa richness are low

b. Relative Macro- Invertebrate Abundance and Taxa Richness: Middle Lagoon Habitat Zone:

Very Good	Good	Fair	Poor
Abundance and taxa richness are considered to be high	Abundance of prey items is high, but taxa richness is relatively low	Abundance is of prey items and/or taxa richness are moderate	Abundance of prey items and/or taxa richness is low

c. Relative Macro- Invertebrate Abundance and Taxa Richness: Upper Lagoon Habitat Zone:

Very Good	Good	Fair	Poor
Abundance and taxa richness are considered to be high	Abundance of prey items is high, but taxa richness is relatively low	Abundance is of prey items and/or taxa richness are moderate	Abundance of prey items and/or taxa richness is low

7. **Habitat Elements and Complexity -** % area containing SAV, large or small WD, emergent and/or riparian vegetation, marshes, sloughs, tidal wetlands, pools > 2 meters, etc.:

Very Good	Good	Fair	Poor
> 70%	70-45%	45-20%	< 20%

8. **Toxicity -** % of area where containments are detected (metals, pesticides, and pollution that are impacting the estuary ecosystem, etc.):

Very Good	Good	Fair	Poor
Not detected	< 2%	2.1-5%	> 5%

9. **Exotic Pest Species -** Number of exotic pest species that alter the estuary ecosystem and significantly impact salmonids (please note how exotic pest species impacts salmonids - *i.e.*, stripers - predation):

Very Good	Good	Fair	Poor
No exotic pest species known to be present	One or more pest species present but there are **no major** impacts to salmonids and the estuary ecosystem	One or more pest species present and at least one is having a **moderate** impact to salmonids and the estuary ecosystem	One or more pest species present and at least one is having a **major** impact to salmonids and the estuary ecosystem

10. Quantity of Rearing Habitat (Life Stage and Species) = OVERALL

a. Quantity of rearing habitat for young-of-year coho and/or *NON-osmoregulating* salmonids (refer to rating listed above for guidance – Lagoon Habitat Zones, water quality parameters, *etc*.):

Very Good	Good	Fair	Poor

b. Quantity of rearing habitat for osmoregulating salmonids (refer to rating listed above for guidance – Lagoon Habitat Zones, water quality parameters, *etc*.):

Very Good	Good	Fair	Poor

Attribute: Habitat Complexity

Habitat complexity is critically important for salmonids because complex habitats are typically highly productive, offer velocity refuges, places to hide, and lower temperatures. This attribute encompasses specific elements, such as large woody debris (LWD), and multi-faceted features such as shelter rating and the ratio of pools to riffles and flatwater. To capture the diversity and importance of this attribute, NMFS identified five different indicators for habitat complexity.

Condition Indicator: Large Woody Debris (LWD) BFW 0-10 and LWD BFW 10-100 for Adult, Summer and Winter Rearing Targets

Instream large wood has been linked to overall salmonid production in streams with positive correlations between large wood and salmonid abundance, distribution, and survival (Sharma and Hilborn 2001). Salmonids appear to have a strong preference for pools created by LWD (Bisson *et al*. 1982) and their populations are typically larger in streams with abundant wood (Naimen and Bilby 1998). Decreases in fish abundance occur following wood removal (Lestelle 1978; Bryant 1983; Bisson and Sedell 1984; Lestelle and Cederholm 1984; Dolloff 1986; Elliott 1986; Murphy et al. 1986; Hicks *et al*. 1991a) while increases in fish abundance have been found following deliberate additions of LWD (Ward and Slaney 1979; House and Boehne 1986; Crispin *et al*. 1993; Reeves *et al*. 1993; Naimen and Bilby 1998; Roni and Quinn 2001).

The LWD indicator is defined as the number of key pieces of large wood per 100 meters of stream. Separate rating criteria were developed for channels with bankfull width (BFW) less than 10 meters and greater than 10 meters. Key pieces are logs or rootwads that: (1) are independently stable within the bankfull width and not functionally held by another factor, and (2) can retain other pieces of organic debris (WFPB 1997). Key pieces also meet the following size criteria: (1) for bankfull channels 10 meters wide or less, a minimum diameter 0.55 meters and length of 10 meters, or a volume 2.5 cubic meter or greater, (2) for channels between 10 and 100 meters, a minimum diameter of 0.65 meters and length of 19 meters, or a volume six cubic meters or greater (Schuett-Hames *et al.* 1999). Key pieces in channels with a bankfull width of > 30 meters pieces only qualify if they have a rootwad associated with them (Fox and Bolton 2007).

Ratings: Number of LWD key pieces per 100 meters of stream length (BFW 0-10 and BFW 10-100)

The frequency of key pieces of LWD influences development and maintenance of pool habitat for multiple life stages of salmonids. LWD is the number of pieces (frequency) per stream length (100 meters) within each reach. Rating criteria were based on the observed distribution of key pieces of LWD in unmanaged forests in the Western Washington eco-region developed by Fox and Bolton (2007). Fox and Bolton's (2007) recommendations were followed using the top 75 percentile to represent a very good condition for LWD frequency. The California North Coast Regional Water Quality Control Board (NCRWQCB 2006) used similar information to develop indices for LWD associated with freshwater salmonid habitat conditions. Rating thresholds are as follows:

For smaller channels (0-10 meters BFW):

Poor = < 4 key pieces/100 meters;
Fair = 4 to 6 key pieces/100 meters;
Good = 6 to 11 key pieces/100 meters; and
Very Good = > 11 key pieces/100 meters.

For larger channels (10-100 meters BFW):

Poor = < 1 key pieces/100 meters;
Fair = 1 to 1.3 key pieces/100 meters;
Good = 1.3 to 4 key pieces/100 meters; and
Very Good = > 4 key pieces/100 meters.

Methods:
Assessing population condition with these criteria proved problematic due to the paucity of absence of adequate LWD surveys in most areas in the CCC ESU. For those populations without LWD survey data, SEC queried the percent LWD Dominant Pools attribute from HAB 8 data. SEC also queried percent pools with LWD and percent shelter that is LWD from the HAB 8 data, but percent LWD dominant pools produced discernible breaks in the distribution of observed values consistent with expected results. Therefore, the percent of LWD dominated pools was used as a proxy to evaluate LWD key piece frequency.

CDFG (2004) habitat typing survey methods follow a random sampling protocol stratified by stream reach (*i.e.*, Rosgen Channel type) used to assess stream habitat conditions from the mouth to the end of anadromy. Habitat data can be used to characterize each reach of stream, and these data were averaged over the surveyed reaches to characterize the stream. LWD is counted in shelter value rating as one of the components of shelter.

Assigning rating to LWD was complicated due to variability in assessment techniques, descriptions, and timing. It is possible that pieces of LWD recorded on some streams would not meet our criteria set for key pieces by this analysis. For example, in some cases, the criteria were not included in the stream inventories; in others, size classifications did not correlate well with our rating system (for example, 1-2 foot diameter and more than 20 foot long versus 0.55 meters in diameter and 10 meters long).

Reach distances and bankfull widths were converted to meters. Some dataset documented LWD per 100 feet and was provided for the habitat elements of riffles, pools, and flat water. In this case the percentage and length of each element given for a particular reach, was back calculated to estimate LWD density in that reach (

Table 3). SEC queried the stream summary database for LWD counts for each stream reach and extrapolated the data to characterize each population stream, for all populations where the data existed. Where HAB 8 data was lacking, a qualitative approach was used and based on the best available information (watershed assessments, *etc*.), spatial data and IP-Km habitat potential to inform Best Professional Judgment ratings.

Table 3. Categories used as rough equivalencies to key pieces of LWD.

TERM	**POTENTIAL ERROR** and/or Comment	**LOCATION(S)** (unless noted, includes subbasins)
"Debris Jams"	Underestimates # key pieces of LWD. Uncertainty was too high, so no rating was given.	Ten Mile River.
"Key LWD"	Criteria may not match	Noyo River Albion River
"Key pieces"	Criteria may not match	San Gregorio Creek
"LGWDDEB_NO" (Number of large woody debris)	Criteria may not match	Lagunitas Creek San Geronimo Creek
"LWD Forced Pool"	underestimates # of key pieces of LWD	Russian River subbasins: Willow Creek (Russian River) Freezeout Creek (Russian River) Unnamed tributaries (Russian River) Cottaneva Creek
"LWD per 100 ft" for: "Riffles," "Pools," and "Flat."	(1)Where percent of each element was recorded, LWD per 100m was calculated.	Pudding Creek Big Salmon Creek Walker Creek
"Number of pieces per 100 linear feet of stream within the bankfull channel"	Criteria may not match. Live trees included in total were subtracted before calculating	Caspar Creek
"Pieces of large wood"	Criteria may not match	Soquel Creek Gazos Creek
"Total # LWD"	Different criteria for LWD than for key pieces of LWD	Pescadero Creek
"Total Logs w/Estimates from LDA's (# per mile)"	Criteria may not match	Aptos Creek
"Key LWD Pieces/328 ft. w/Debris Jams"	Criteria may not match.	Navarro River Big River Russian River subbasins: Ackerman Creek Alder Creek Jack Smith Creek

"Total # of Debris Jams" + "Key LWD Pieces/100m w/o Debris Jams	Criteria may not match. Two totals were added (see comment for Navarro) Debris jams only recorded for 3 out of 22 reaches. In only one case did it change the rating—from fair to good.	Garcia River

Condition Indicator: Percent Primary Pools for Summer Rearing Target

Pools provide hydraulic and other environmental conditions favoring presence of summer rearing juvenile salmonids (Bisson *et al.* 1988). During high flow events, pools are usually scoured, leaving a coarse gravel channel armor and depositing material on the riffles (Florsheim *et al.* 2001). The percentage of pools within a stream is a common indicator for estimating amount of rearing habitat available for juvenile salmonids. The pool:riffle:flatwater ratio indicator (described below) describes the frequency of all pool habitat types (mid-channel, scour and backwater pools) relative to other habitat types across each population. However, quantitative information on pool frequency without accompanying qualitative information such as depth or shelter indicators and criteria, can give a false impression of habitat conditions (if, for example, there are numerous, shallow, short simple pools which are a common occurrence in aggraded streams). This indicator describes pool quality by assessing primary pools. These are the larger deeper pools preferentially occupied by juveniles and adults respectively, have specific depth criteria, and are a subset of all pool habitat types.

Deeper larger pools have larger volume and as such have a larger juvenile rearing carrying capacity. The frequency of these larger deep pools provides a conservative measure of the quality of significant rearing habitat and staging habitat. CDFG combined measures of pool depth and frequency in their watershed assessments by reporting the frequency of primary pools stratified by stream order. Primary pools in first and second order streams are two feet deep or more, while primary pools in third and fourth order streams were are three feet deep or more (Bleier *et al*. 2003).

Ratings: Percent of primary pools at the reach, stream and population scale

Juvenile salmonids prefer well shaded pools at least three feet deep with dense overhead cover or abundant submerged cover composed of undercut banks, logs, roots, and other woody material. Pool depths of three feet are commonly used as a reference for fully functional salmonid habitat (Overton *et al*. 1993; Brown *et al*. 1994; Baker and Smith 1998; Bauer and Ralph 1999).

Maximum pool depth is partially a function of channel size, and is highly affected by the physical properties that affect stream energy such as gradient, entrenchment, width, and sediment load. The Washington State Fish and Wildlife Commission (1997) recommended the following pool frequencies by length: "(f)or streams less than 15 meters wide, the percent pools should be greater than 55 percent, greater than 40 percent and greater than 30 percent for streams with gradients less than 2 percent, 2-5 percent and more than 5 percent, respectively."

Pool depths and volume can be impaired by sediment over-supply related to land management (Knopp 1993). Reeves *et al*. (1993) found diminished pool frequency in intensively managed watersheds. Streams in Oregon coastal basins with low timber harvest rates (< 25 percent) had 10-47 percent more pools per 100 meters than streams in high harvest basins (> 25 percent). Peterson *et al*. (1992) used 50 percent pools as a reference for good salmonid habitat and recognized streams with less than 38 percent pools by length as impaired, though Alaska studies showed ranges of 39-67 percent pools by length (Murphy *et al*. 1984).

The CDFG Watershed Assessment Field Reference (CDFG 1999) states good salmonid streams have more than 50 percent of their total available fish habitat in adequately deep and complex pools, though CDFG considers a primary pool frequency of less than 40 percent inadequate for salmonids (Bleier *et al*. 2003). Knopp (1993) summarized pool frequency in disturbed streams in northern California, and found a pool frequency average of 42 percent. Due to the number of variables influencing pool depth (stream order, gradient, entrenchment, substrate) a quartile approach was established to extrapolate up to a stream scale (versus a reach scale). The quartile approach set a 25 percent boundary from a 50 percent threshold to describe good conditions for primary pools to account for bias due to stream order and the natural range of variability.

The resulting criteria for primary pools are:

Stream level percent primary pool rating criteria
Poor = < 25% primary pools;
Fair = 25% to 49% primary pools;
Good = 50% to 74% primary pools; and
Very Good = > 75% primary pools.

Population scale encompasses multiple streams (including mainstem channels which cannot always be expected to achieve optimal criteria across all stream orders). Therefore stream level data were evaluated according to the following criteria:

Population level percent primary pool rating criteria
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75-90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Methods:
The CDFG habitat typing procedure evaluates pools by classifying 100 percent of the wetted channel by habitat type from the mouth to the end of anadromy (Flosi *et al*. 2004). The method is used in wadeable streams (stream orders 1-4). CDFG follows a random sampling protocol stratified by stream reach (*i.e*., Rosgen Channel type) to measure conditions within habitat types for variables such as width and depth. Typically, depth is recorded for every third habitat unit in addition to every fully-described unit. This provides an approximate 30 percent sub-sample for all habitat units. Habitat data can be used to characterize each reach of stream, and data can be averaged over the collection of reaches to characterize the stream. Habitat typing surveys (Flosi *et al.* 2004) provide a measure of pool frequency defined as the percentage of stream reaches in pools. This sub-sample is expressed as an average for each stream reach. SEC queried the stream summary database for the mean of each variable for each stream reach and then extrapolated the data to characterize each stream, for all streams within each population where the data existed. Rating each population for this variable required two steps; calculation of the mean values at the stream scale from reach scale data, then calculating the percentage of streams/IP-km meeting optimal criteria, at the population scale.

The CDFG reach summary output summarizes the frequency of primary pool indicator for the proportion of pools two feet deep or greater in first and second order streams, and three feet deep or greater in third and fourth order streams. For populations where SEC had access to the stream summary database (Russian River, Salmon Creek, Lagunitas Creek), the amount of primary pool from stream habitat data was calculated. Where data were lacking, other datasets and best professional judgment were utilized.

Condition Indicator: Frequency of Pools, Riffles, and Flatwater for Adult, Summer and Winter Rearing Targets

Pools provide hydraulic and other environmental conditions necessary for summer rearing of juvenile salmonids, and resting cover for adults; riffles provide hydraulic and environmental conditions critical for spawning adults and incubating eggs; while adjoining flatwater provide habitats for a diversity of life stages. In general, winter habitat is lacking where flatwater habitats dominate the channel, because they lack elements (velocity refuge, scour elements, cover and shelter) for fish to maintain residency under high flow conditions. The average frequency of pools:riffles:flatwater across all IP-km provides an indication of the habitat diversity available for various species and life stages.

Developing or enhancing pools habitats for rearing and riffle habitats for spawning are a common focus of restoration activities. When pools lacking depth or shelter, actions are typically recommended to deepen pools by adding instream complexity. This ultimately shortens adjoining flatwater types, or converts flatwater habitat types to pools. Conversely, when spawning gravels are lacking, actions are typically recommended to add instream structures as a technique to flatten the gradient and retain gravels. This ultimately shortens adjoining flatwaters or converts flatwater habitat types to riffles. In this case, the length or frequency of flatwater types are decreased in favor of increasing the percent length of pools/riffles or the frequency of pools/riffles respectively.

Ratings: Frequency of pools:riffles:flatwater at the reach, stream and population scale

As noted above, Reeves *et al*. (1993) found pools diminished in frequency in intensively managed watersheds. Streams in Oregon coastal basins with low timber harvest rates (< 25 percent) had 10-47 percent more pools per 100 m than did streams in high harvest basins (> 25 percent). The CDFG Watershed Assessment Field Reference (CDFG 1999) states good salmonid streams have more than 50 percent of their total available fish habitat in adequately deep and complex pools; and have at least 30 percent in riffles. Knopp (1993) summarized pool frequency in disturbed streams in Northern California, and found pool frequency averaged 42 percent.

CDFG considers a primary pool frequency of less than 40 percent, and riffle frequency less than 30 percent inadequate for salmonids (Bleier *et al.* 2003). Based on this consideration NMFS established rating criteria (discussed previously) using a 10 percent boundary from the target threshold for subsequent ratings for pools and riffles.

The resulting criteria are:

Stream level pool:riffle:flatwater frequency rating
Poor = < 20% pools and < 10% riffles;
Fair = 20% to 29% pools and > 10% to 19% riffles;
Good = > 30% to 39% pools and = >20% to 29% riffles; and
Very Good = > 40% pools and = > 30% riffles.

To extrapolate stream level data upward to the population scale, we then rated each population on the following criteria.

Population level pool:riffle:flatwater frequency rating
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75% to 90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Methods:
CDFG habitat typing is a standardized method that physically classifies 100 percent of the wetted channel by habitat type from the mouth to the end of anadromy (Flosi *et al.* 2004). The attributes distinguishing the various habitat types include stream order, over-all channel gradient, velocity, depth, substrate, and the channel type features responsible for the unit's formation. Level I categorizes habitat into riffles or pools. Level II categorizes riffles into riffle or flatwater habitat types, for a total of three types (riffle, pool, and flatwater). Level III further differentiates riffle types on the basis of water surface gradient, and pool types according to location in the stream channel. At Level IV, pools are categorized by the cause of formation; riffles are categorized by gradient; and flatwaters are categorized by depth and velocity. Typically, habitats are described according to location, orientation, and water flow at the Level IV scale. However, habitat can be summarized at any habitat scale and used to characterize each reach of stream, as well as the stream as a whole.

The length and frequencies of a habitat type depends on stream size and order. Generally a stream will not contain all habitat types, as the mix of habitat types reflects the overall channel gradient, flow regime, cross-sectional profile, and substrate particle size. Therefore collapsing the habitat types at the Level II scale provides a reasonable measure of diversity to describe the complexity of habitats that occur across watersheds, which also describes the critical habitat needs across species in a population. SEC calculated the calculated the frequency of Level II habitats (pools, riffles and flatwater) from the database of streams where surveys are available.

SEC queried the stream summary database for pool:riffle:flatwater frequency for each stream reach and extrapolated the data to characterize each stream, for all streams within each population where the data existed. As with other data collected at smaller scales, rating each population required two steps; calculation of the mean at the stream scale from reach scale data, then determining the percentage of streams/IP-km meeting optimal criteria, at the population scale.

Condition Indicator: Shelter Ratings for Adult, Summer and Winter Rearing, and Smolt Targets
Depending on spring flow conditions, salmonids require pool habitats with adequate complexity and cover for multiple life stages, including rearing and smolt outmigration. Winter habitat is considered impaired in habitats lacking velocity refuge, cover and shelter during period of high stream flow. Pool shelter rating was used to evaluate the ability of pool habitat to provide adequate cover for salmonid survival throughout the population.

Shelter rating is a measure of the amount, and diversity, of cover elements in pools. Shelter rating is used by CDFG in their stream habitat-typing protocol (Flosi *et al.* 2004). It is an useful indicator of pool complexity. Shelter/cover elements include undercut bank, large and small woody debris, root mass, terrestrial vegetation, aquatic vegetation, bubble curtain, boulders, and bedrock ledges (Bleier *et al*. 2003).

Ratings: Pool shelter averaged at the reach, stream and population scales
Bleier *et al*. (2003) identified a shelter rating value of < 60 as being inadequate, and > 80-100 as good for salmonids. Average shelter value below 80 was rated fair; average shelter value above 100 was rated to identify high value refugia areas. The stream level criteria are:

Stream level shelter rating
Poor = < 60 average shelter value;
Fair = 60 to 79 average shelter value;
Good = 80 to 100 average shelter value; and
Very Good = > 100 average shelter value.

Given that the population scale encompasses multiple streams, the following ratings were used to extrapolate shelter conditions for each population:

Population level shelter rating
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75% to 90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Methods:
The CDFG (2004) habitat typing survey method estimates shelter ratings in all pool habitats measured. Typically, pool habitats are described in every third habitat unit in addition to every fully-described unit which provides an approximate 30 percent sub-sample. Habitat data were used to characterize each reach of stream, and data were averaged over the collection of reaches to characterize the entire stream.

Shelter rating values were generated by multiplying instream shelter complexity values by estimated percent area of pool covered. Scores were obtained by assigning an integer value between 0 and 3 to characterize type and diversity of cover elements and multiplying that value by the percent cover (Table 4). A shelter rating between 0 and 300 is derived, with 300 being equal to 100% cover with maximum diversity (Flosi *et al.* 2004).

SEC calculated average shelter rating across all reaches using HAB 8 reach summation information. This sub-sample is expressed as an average for each stream reach. SEC queried the stream summary database for mean percent shelter ratings for each stream reach and extrapolated the data to characterize each stream, within each population (where data were available). As with other reach level data, deriving ratings for the each population required two steps; calculation of shelter value at the stream scale from reach scale data, then determining the percentage of streams/IP-km meeting optimal criteria at the population scale. A bias analysis was also conducted for the population shelter rating value reflecting the percent of potential IP-km evaluated.

Table 4. Values and examples of instream shelter complexity. Values represent a relative measure of the quality and composition of the instream shelter. Adapted from Flosi *et al.*, 2004.

Value	Instream Shelter Complexity
0	No Shelter
1	1-5 boulders
	Bare undercut bank or bedrock ledge
	Single piece of LWD (>12" diameter and 6' long)
2	1-2 pieces of LWD associated with any amount of small woody debris (SWD) (<12" diameter)
	6 or more boulders per 50 feet
	Stable undercut bank with root mass, and less than 12" undercut
	A single root wad lacking complexity
	Branches in or near the water
	Limited submersed vegetative fish cover
	Bubble curtain
3 (Combinations of at least 2 cover types)	LWD/boulders/root wads
	3 or more pieces of LWD combined with SWD
	3 or more boulders combined with LWD/SWD
	Bubble curtain combined with LWD or boulders
	Stable undercut bank with greater than 12" undercut, with root mass or LWD
	Extensive submerged vegetative fish cover

Attribute: Hydrology

Hydrology, as a key attribute, includes all aspects of the hydrologic cycle relevant to the spawning, incubation, rearing and migration of salmonids. The magnitude, timing, and seasonality of local precipitation and geology determine a watershed's historical discharge patterns. These patterns however, can be modified by individual and cumulative water use practices to interfere with a salmonids' ability to complete their life cycle. Because stream flow is rarely measured throughout a watershed (*i.e.*, in tributaries), flow requirements for fish in individual watersheds are rarely specified. However, since these species evolved under unimpaired flow regimes, it is reasonable to assume that approximating these conditions will likely foster favorable conditions. Hydrology was assessed using six different indicators.

Condition Indicator: Passage Flows for Adult and Smolt Targets

This indicator considered the effect of flow impairments on smolt and adult passage. Considerations included; (1) impairment precluding passage over critical riffles, and (2) the degree flow impairments reduce pulse-flows necessary for adult and smolt migration (including considerations on the magnitude, duration, and timing of freshets).

Ratings: Four life stages (egg, summer rearing, smolt and adult) are rated on four instream flow criteria: 1) summer rearing baseflows, 2) instantaneous flow reductions affecting eggs and summer rearing, 3) adult and smolt passage flows, and 4) redd scour affecting eggs. For most populations, there is generally little information about the suitability of flows to support these habitat attributes, although there may be sufficient data for some individual sub-populations, and for others there may be data for only one or two of the five indicators.

Assessment of the suitability of instream flows for CCC coho salmon relied in part on information developed via input from 15 fisheries researchers and aquatic resource managers familiar with stream flow issues in north-central coastal California. To further evaluate instream flow habitat attributes, a qualitative decision structure was created (a.k.a., the instream flow protocol) to develop ratings for each flow indicators.

The distribution and differences in seasonality of each target life stage were considered so as to accurately assess flow-related impacts. Watershed flow conditions were rated by reviewing relevant published information and seeking unbiased input from resource managers and researchers familiar with instream flows on a watershed by watershed basis. Each of the four flow related habitat attributes were scored using a instream flow protocol. The protocol analyzed three risk factors: setting, exposure and intensity, as defined below.

Setting rated the degree of aridity of a watershed given the natural setting of climate, precipitation, *etc*. in an undisturbed state. Four classes of setting were identified: xeric, mixed, mesic, and coastal (Table 5). Xeric watersheds are dominated by arid environments such as oak savannah, grassland, or chaparral. Mixed watersheds have a combination of xeric, mesic, and/or coastal habitats within them. Mixed watersheds are typically larger watersheds with inland regions. Mesic settings have moderate amounts of precipitation; examples include mixed coniferous/hardwood forest and hardwood-dominated forest (*e.g*., oak woodland, tanoak, *etc.*). Coastal settings are watersheds dominated by the coastal climate regime with cool moist areas. Coastal watersheds typically have high levels of precipitation, are heavily forested, and are predominantly within the redwood forest zone. Maps of vegetation types and average precipitation were provided to resource manager during the review.

Exposure rated the extent of stream likely impaired relative to each flow attribute. Specifically, exposure is the estimated proportion of historical IP-km habitat (by length) appreciably affected by reduced flows (Table 5). A stream reach may be appreciably affected, for example, if the value of summer rearing habitat is degraded by water diversions that reduce space, degrade water quality, reduce food availability, or restrict movement. NMFS reviewed maps of each watershed showing the spatial relationship between relevant habitat areas and high-risk land uses, such as agriculture. Exposure war rated (percent IP-km habitat by length) as > 15%, 5% to 15%, < 5%, or none, based on existing information and best professional judgment.

Intensity rated the likelihood that the land uses within the area of exposure divert substantial amounts of water during critical time periods. High intensity (Table 5) land use activities regularly require substantial water diversions from the stream at levels that impair the habitat attribute. Moderate intensity activities typically require irrigation, or have regular demand, but satisfy that demand often by means other than direct pumping of surface or subterranean stream flows. Low land use activities require diversions in small amounts. The intensity of water diversion impacts in the population was rated as high, moderate, low, or none, using existing information and knowledge of local land uses.

Table 5. Rating matrix for assessing flow conditions for four hydrology indicators.

	Poor	Fair	Good	Very Good
Setting	Xeric	Mixed	Mesic	Coastal
Exposure	> 15%	5-15%	< 5%	None
Intensity	High	Moderate	Low	None

Overall scores for each of the flow habitat attributes for each applicable life stage was determined by two steps. For a given habitat attribute, each risk-factor rating was assigned a value (Table 6). Then, the three risk factor rating scores were averaged to determine the overall rating. For example, to determine the rating for baseflow on summer rearing: the setting in the watershed is mixed (75), the exposure (of historical potential rearing habitat) to impacts of impaired summer base flows was > 15% (100), and the intensity was high (100), the average score of these three risk factors is 92, which results in an attribute rating of poor for summer rearing base flows in that watershed.

Table 6. Risk factor scores and the criteria defining poor, fair, good or very good ratings for a combined average risk score for each life stage and flow indicator.

	Poor	Fair	Good	Very Good
Setting	Xeric	Mixed	Mesic	Coastal
Score	100	75	50	25
Exposure	> 15%	5-15%	<5%	None
Score	100	75	50	25
Intensity	High	Moderate	Low	None
Score	100	75	50	25
Attribute Rating	**Poor**	**Fair**	**Good**	**Very Good**
Score Class	>75	51-75	35-50	<35

Recognizing that, for some populations, data may be very limited or non-existent for exposure and intensity ratings for individual flow related habitat attributes. Every reasonable effort was made to provide reliable sources for these ratings. Ratings were not solely based on professional judgment and/or personal communications. At least one quality reference (published document, agency report, *etc.*) was used and supplemented with one or two "personal communications" if possible. In cases where flow conditions (exposure and/or intensity) related to a particular habitat attribute could not be determined, the indicator was scored as unknown. Such ratings resulted in recovery plan recommendations for further investigation of the suitability of flow conditions for that attribute.

Condition Indicator: Flow Conditions (Instantaneous Condition) for Eggs and Summer Rearing Targets

This indicator provided an indication of the degree short-term artificial streamflow reductions impact juveniles or the survival-to-emergence of incubating embryos. This condition is often associated with instream diversions (*e.g.*, diversions for frost protection irrigation) and can be exacerbated in more arid conditions or smaller tributaries.

Ratings: As described above, all flow related indicators were assessed using the instream flow protocol conducted by a team of experts.

Condition Indicator: Redd Scour for Eggs Target

Redd scour refers to mobilization of streambed gravels at spawning sites that result in dislodging of embryos from their redds and subsequent mortality. This process is not strictly a function of stream flow but is a combination that is influenced by channel configuration, sediment dynamics, and channel roughness and stability largely control the stability of spawning substrates.

Ratings: As described above, all flow related indicators were assessed using the instream flow protocol conducted by a team of experts.

Condition Indicator: Flow Conditions (Baseflow) for Summer Rearing Target

This indicator measures the degree a watershed currently supports surface flows within historical rearing areas. Surface flows provide rearing space, allow for movement between habitats, maintain water quality, and facilitate delivery of food for juvenile salmonids. Inadequate surface flow may result from cumulative water diversions and/or significant physical changes in the watershed. Water diversions are withdrawals from stream surface waters and/or from subterranean stream flows that are likely hydrologically connected to the stream (*e.g.*, pumping from wells in alluvial aquifers that are in close proximity to the stream).

Ratings: As described above, all flow related indicators were assessed using the instream flow protocol conducted by a team of experts.

Condition Indicator: Number, Conditions, and/or Magnitude of Diversions for Summer Rearing and Smolts

Diversions are structures or sites having potential to entrain or impinge of smolts. The indicator is the frequency of diversions along the IP-km smolt outmigration route. The diversion structure or sites analyzed were unscreened diversions located along the stream channel. Diversions without an actual structure in the stream were not included in the analysis.

Ratings: Frequency of diversions across IP-km

SEC assessed the density of diversions in each population across all IP-km, regardless if those areas are currently accessible by salmonids. This allowed assessment of conditions throughout all areas of potential importance to recovery, not just within the species' current distribution. Due to data limitations this rating only applied to the number of diversions and did not identify whether existing diversions are fish passage compliant (screened).

Once the data were analyzed, the following rating criteria were established to define good, fair, poor, based on the observed distributions (*i.e.*, *a posteriori*):

Poor = > 5 diversions/10 IP-km;
Fair = 1.1 to 5 diversions/10 IP-km;
Good = 0.01 to 1 diversions/10 IP-km; and
Very Good = 0 diversions/10 IP-km.

Methods:

SEC queried the CDFG 2006 Passage Assessment Database to identify diversions and estimate the number of diversions in a watershed. SEC also reviewed the California State Water Resources Control Board (SWRCB) Division of Water Rights Point of Diversion (POD) database but found it of limited use at the time of analysis because it could not be downloaded for geographic analysis to associate it with appropriate IP-km. Although this database was complete, SEC was unable to determine the quantity of water diverted from each diversion. We therefore based the diversion indicator on the density of diversions, regardless of volume. The diversion density was calculated as the number of diversions per 10 IP-km.

Landscape Indicator: Impervious Surfaces for Watershed Processes Target

Modifications of the land surface (usually from urbanization) produce changes in both magnitude and type of runoff processes (Booth *et al*. 2002). Manifestation of these changes include increased frequency of flooding and peak flow volumes, decreased base flow, increased sediment loadings, changes in stream morphology, increased organic and inorganic loadings, increased stream temperature, and loss of aquatic/riparian habitat (May *et al*. 1996). The magnitude of peak flow and pollution increases with total impervious area (TIA) (*e.g.*, rooftops, streets, parking lots, sidewalks, *etc*.).

Spence *et al*. (1996) recognized channel damage from urbanization is clearly recognizable when TIA exceeds 10 percent. Reduced fish abundance, fish habitat quality and macroinvertebrate diversity was observed with TIA levels from 7.01-12 percent (Klein 1979; Shaver *et al.* 1995). May *et al*. (1996) showed almost a complete simplification of stream channels as TIA approached 30 percent and measured substantially increased levels of toxic storm water runoff in watersheds with greater than 40 percent TIA.

Ratings: Percentage of impervious surfaces in a watershed as:

Poor = > 10% of the total watershed;
Fair = 7% to 10% of the total watershed;
Good = 3% to 6% of the total watershed; and
Very Good = < 3% of the total watershed,

Methods:

The primary assessment tool used was the National Land Cover Database (Edition 1.0) which was produced by the Multi-Resolution Land Characteristics Consortium[4]. The rating thresholds apply to the TIA across all 28 focus populations. Statistics for percent coverage of each land cover type with an associated imperviousness rating were calculated using GIS thresholds for TIA from Booth (2000), May *et al*. (1996) and Spence *et al.* (1996).

Attribute: Landscape Patterns

We defined landscape patterns as disturbance resulting from land uses that cause perturbations resulting in direct or indirect effects to watershed processes. These are typically the result of land uses such as agriculture, timber harvest, and urbanization. These landuses were used as indicators to describe the degree of disturbance in a population.

Landscape Context Indicator: Agriculture for Watershed Processes Target

Agriculture is defined as the planting, growing, and harvesting of annual and perennial non-timber crops for food, fuel, or fiber.

Ratings: Percent of population area used for agricultural activities

[4] http://www.mrlc.gov/nlcd2006.php

Irrigated agriculture can negatively impact salmonid habitat (Nehlsen *et al*. 1991) due to insufficient riparian buffers, high rates of sedimentation, water diversions, and chemical application and pest control practices (Spence *et al*. 1996). On level ground, agricultural activities near streams are typically assumed to have more negative effects on streams than agriculture further away from streams due to the potential for stream channelization, clearing of riparian vegetation, and increased erosion. However, vineyards are often planted on steep terrain and may contribute to instream sedimentation even when located a substantial distance from stream channels.

Specific methods for conserving salmonid habitats on agricultural lands are not well developed but the principles for protecting streams on agricultural lands are similar to those for forest and grazing practices (Spence *et al*. 1996).

We defined ratings *a posteriori* based on the observed distribution of results. The following rating classes were thus formed:

Poor = >30% of population area used for agricultural activities;
Fair = 20% to 30% of population area used for agricultural activities;
Good = 10% to 19% of population area used for agricultural activities; and
Very Good = < 10% of population area used for agricultural activities.

Methods:
Assessments of agriculture were conducted via GIS interpretation of digital data layers. The California Department of Conservation, Division of Land Resource Protection, Farmland Mapping and Monitoring Program (FMMP) was the primary method used to measure the extent of agriculture in a population. Where these data were not available, USGS National Land Cover Database Zone 06 Land Cover Layer (Edition 1.0) was used. The FMMP data are presented by county, therefore where a population extended into more than one county the layers were merged to create a single dataset. The area represented by farmland polygons for each population was calculated using GIS.

<u>Landscape Context Indicator: Timber Harvest for Watershed Processes Target</u>
Rate of timber harvest was used to define the percent of a population exposed to timber harvest activities within the most recent 10 year period.

Ratings: Average rate of timber harvesting in population over last 10 years
Adverse changes to salmonid habitat resulting from timber harvest are well documented in the scientific literature (Hall and Lantz 1969; Burns 1972; Holtby 1988; Hartman and Scrivener 1990; Chamberlin *et al*. 1991; Hicks *et al.* 1991a). The cumulative effects of these practices include changes to hydrology (including water temperature, water quality, water balance, and soil structure, rates of erosion and sedimentation, channel forms and geomorphic processes (Chamberlin *et al*. 1991) which adversely affect salmonid habitats. These processes operate over varying time scales, ranging from a few hours for coastal streamflow response, to decades or centuries for geomorphic channel change and hill-slope evolution (Chamberlin *et al*. 1991).

Reeves *et al.* (1993) found that pools diminished in frequency in intensively managed watersheds. Streams in Oregon coastal basins with low timber harvest rates (< 25 percent) had 10 to 47 percent more pools per 100 meters than did streams in high harvest basins. Additionally, Reeves *et al.* (1993) correlated reduced salmonid assemblage diversity to rate of timber harvest.

Ligon *et al*. (1999) recommend a harvest limitation of 30-50 percent of the watershed area harvested per decade as a "red flag" for a higher level of review. Recent work in the Mattole River suggests a harvest threshold of 10 to 20 percent (Welsh, Redwood Sciences Laboratory, personal communication). Harvest areas of 15 percent of watersheds are considered excessive for some timberlands (Reid 1999). Based on these findings we defined these ratings for rate of timber harvesting per population:

Poor = >35% of population area harvested in the past 10 years;
Fair = 26% to 35% of population area harvested in the past 10 years;
Good = 15% to 25% of population area harvested in the past 10 years; and
Very Good = <15% of population area harvested in the past 10 years.

Methods:
Cal Fire's timber harvest history information was used to determine the aerial extent of approved timber harvest plans, by population. However, we only included the aerial footprint once in this analysis regardless of the number of times an area was harvested in the 10 year period.

The 25 categories of harvest associated with timber harvest in California were initially condensed in the following general categories; even aged harvest, uneven aged harvest, conversion, no harvest, and transition. However, due to the relatively short ten year period, it was determined that the only areas excluded from the rate-of-harvest analysis would be those where "no harvest" was included in the timber harvest plan. We acknowledge the different effects of the various silvicultural techniques (*i.e*., even aged versus uneven aged harvest) but decided to combine all these harvest methods in order to capture all the potential cumulative effects of timber harvest within a population.

<u>Landscape Context Indicator: Urbanization for Watershed Processes Target</u>
Urbanization was defined as the growth and expansion of the human landscape (characterized by cities, towns, suburbs, and outlying areas which are typically commercial, residential, and industrial) such that the land is no longer in a relatively natural state.

Urbanization has affected only two percent of the land area of the Pacific Northwest, but the consequences of urbanization to aquatic ecosystems are severe and long-lasting. The land surface, soil, vegetation, and hydrology are all significantly altered in urban areas (Spence *et al*. 1996). Urban land use is commonly a low percentage of total catchment area, yet it exerts a disproportionately large influence, both proximately and over distance (Paul and Meyer 2001). Despite the many factors potentially limiting Pacific salmon populations, the percentage of urban land alone explained more than 60% of the variation in Chinook salmon recruitment in the interior Columbia River Basin (Regetz 2003; Allan 2004).

Major changes associated with increased urban land area include increases in the amounts and variety of pollutants in runoff, more erratic hydrology due to increased impervious surface area and runoff conveyance, increased water temperatures due to loss of riparian vegetation and warming of surface runoff on exposed surfaces, and reduction in channel and habitat structure due to sediment inputs, bank destabilization, channelization, and restricted interactions between the river and its land margin (Paul and Meyer 2001; Allan 2004). Enhanced runoff from impervious surfaces and stormwater conveyance systems can degrade streams and displace organisms simply because of greater frequency and intensity of floods, erosion of streambeds, and displacement of sediments (Lenat and Crawford 1994).

The degree of impervious surfaces, as discussed earlier (see hydrology attribute above), influences storm flow quantity and timing, and results in a concomitant decrease in baseflow. However, other impacts related to urban development such as runoff which contains a variety of pollutants that degrade water quality (Wang *et al*. 2001), and reductions in overall biological diversity and integrity have been shown to be negatively correlated with the percentage of urban land cover (Klein 1979; Steedman 1988; Limburg and Schmidt 1990; Lenat and Crawford 1994; Weaver and Garman 1994; Wang *et al*. 1997; Klauda *et al*. 1998), human population density (Jones and Clark 1987; Schueler 1997), and house density (Benke *et al.* 1981). These more general impacts, independent of the degree of impervious surfaces, require additional attention. For example, Yates and Bailey (2010) reported declining numbers of benthic macroinvertebrate taxa, and replacement of intolerant taxa with more tolerant (often warm water) taxa, due to increasing density of human development.

While agricultural and timber land uses have best management land-use practices that, if properly implemented, can minimize adverse impacts to watershed process, the impacts of urbanization are generally permanent. Wang *et al.* (1997; 2000; 2001) found that relatively low levels of population urbanization inevitably lead to serious degradation of the fish community. Additionally, while conservation measures exist for reversing or mitigating the degree of impervious surfaces (expanding riparian corridors, developing settling basins, storm water treatment, *etc.*), the other effects of urbanization can permanently alter natural watershed processes, and in some cases, little may be done to mitigate these effects.

Uncertainty exists as to the most appropriate predictor of disturbance to watershed process and subsequent biological response. Two assessment methods were considered; the total extent of urban land and impervious surface. Biological response measures have been predicted by impervious area in several landscape studies of stream urbanization (Walsh *et al*. 2001; Wang *et al*. 2001; Ourso and Frenzel 2003) and by urban land area in others (Morley and Karr 2002), suggesting hydrologic influences are primary in some studies, but the broader range of influences represented by urban area may be more important in others (Allan 2004); (Boyer et al. 2002).

Anadromous fish have been shown to be adversely affected by urbanization. Wang *et al*. (2001) found the impacts of urbanization occur to stream habitat and fish, across multiple spatial scales, and that relatively small amounts of urban land use in a watershed can lead to major changes in biota. There also appears to be threshold values of urbanization beyond which degradation of biotic communities is rapid and dramatic (May *et al*. 1997; Wang *et al*. 2000).

Limburg and Schmidt (1990) demonstrated a measurable decrease in spawning success of anadromous species (primarily alewives) for Hudson River tributaries from streams with 15 percent or more of the watershed area in urban land use. Stream condition almost invariably responds nonlinearly to a gradient of increasing urban land or impervious area (IA). A marked decline in species diversity and in the index of biological integrity scores with increasing urbanization has been reported from streams in Wisconsin around 8–12 percent IA *(Wang et al*. 2000; Stepenuck *et al.* 2002), Delaware, 8–15 percent IA, (Paul and Meyer 2001), Maryland, greater than 12 percent IA, (Klein 1979), and Georgia, 15 percent urban land (Roy *et al*. 2003). Additional studies reviewed in Paul and Meyer (2001) and Stepenuck *et al.* (2002) provide evidence of marked changes in discharge, bank and channel erosion, and biotic condition at greater than 10 percent imperviousness. Also, the supply of contaminants in urban storm runoff may vary independent of impervious area Allan (2004). Although considerable evidence supports a threshold in stream health in the range of 10 to 20 percent IA or urban land, others disagree (Karr and Chu 2000; Bledsoe and Watson 2001), and the relationship is likely too complex for a single threshold to apply.

Ratings: Percent of population area developed for urban activities

Criteria were developed for five density classes of urbanization and condensed into for rating criteria:

> Poor = > 20% of watershed area in urban > 1 unit/20 acres;
> Fair = 12% to 20% of watershed area in urban > 1 unit/20 acres;
> Good = 8% to 11% of watershed area in urban > 1 unit/20 acres; and
> Very Good = < 8% of watershed area in urban > 1 unit/20 acres.

Methods:
Efforts to estimate impacts from urbanization in managed watersheds, require quantitative and predictive models describing the relationship between urbanization and the biological integrity of the community (Wang *et al.* 1997; Wang *et al.* 2000). One challenge in constructing such models is the identification of appropriate indicators reading the amount and extent of urbanization in statistical analysis and modeling. Urban land use encompasses a wide range of interrelated human activities that can be difficult to summarize numerically. Moreover, not only the type, but also the intensity and the location of the land use within the watershed are likely to determine its impact on the biological community of the stream (Booth and Jackson 1997; May *et al*. 1997). Proximity to the stream and width of riparian corridors also appear to be an important consideration in estimating the impact of urban land uses on stream biological communities, though accounting for this variability across the large scale of the NCCC Domain is problematic. In addition, adverse impacts of urban land use are clearly experienced at considerably lower percentages of catchment area than is true for agricultural land use, and most studies report a nonlinear response of stream condition to increasing urbanization.

The primary method used to measure the extent of urban development in a watershed (population) was to query data from the California Department of Forestry and Fire Protection, Fire and Resource Assessment Program (FRAP), and from the GIS layer of DENCLASS10. This GIS layer provided year 2000 census block data merged, with county Topologically Integrated Geographic Encoding and Referencing (TIGER) files, into a single statewide data layer. These data sources provided a detailed depiction of spatial demographics, primarily in sparsely populated rural areas. The data were collapsed from ten classification of housing density into five classes represented by urban polygons to summarize and describe the intensity of urban development for each population area.

Total areas of the populations were then calculated in GIS from population boundary polygons, and these areas used to describe the percentage of urban development over five classes of housing density within each population (density classes range from lowest to highest):

> 0 to less than 1 housing unit /160 acres;
> 1 unit/160 acres to 1 unit/20 acres;
> 1 unit/20 acres to 1 unit/5 acres;
> 1 unit/5 acres to 2 units/acre; and
> 2 units/acre to greater than or equal to 5 units/acre.

Attribute: Passage/Migration

Passage was defined as the absence of physical barriers that prevent or impede the up- or downstream passage of migrating adult, smolts, and juvenile salmonids. Excluding spawning salmonids from portions of their IP-km can increase the likelihood of extirpation by reducing the amount of available spawning and rearing habitat and thereby lower the carrying capacity of the watershed (Boughton *et al*.

2005). Assessment of the percentage of IP affected by barriers should include all IP-km (including upstream of impassable dams if they are proposed for remediation). Passage requirements were evaluated individually for each target, according to the time period specific to each life stage. Passage was assessed using two indicators.

Condition Indicator: Physical Barriers for Adult, Summer and Winter Rearing Targets
Physical barriers are structures or sites preventing or impeding up- or downstream passage of migrating adult and juvenile salmonids.

The indicator was defined as the proportion of IP-km free of known barriers and thereby accessible to migrating salmonids. The physical barriers attribute included only total barriers which are complete barriers to fish passage for all anadromous species at all life stages at all times of year. Passage was evaluated individually for each target, according to the time period specific to the life stage.

Ratings: Accessible proportion of IP-km
Rating thresholds were defined according to the following criteria:

Poor = < 50% or < 32 IP-km of historical IP-km accessible;
Fair = 50% to 74% historical IP-km habitat accessible;
Good = 75% to 90% of historical IP-km accessible; and
Very Good = > 90% of historical IP-km accessible.

Ratings for poor conditions addressed accessible proportions of the watershed, and the minimum threshold of potential habitat (expressed as IP-km) required for the population to be considered viable -in-isolation (32 IP-km for coho salmon, 20 IP-km for Chinook salmon, and 16 IP-km for steelhead). These thresholds assume populations historically operated close to the natural carrying capacity of the watershed.

Methods:
SEC queried the CDFG Passage Assessment Database (PAD)[5] to calculate the proportion of IP-km blocked to anadromy by impassable barriers. The PAD contains data and point file coverage for all known fish passage barriers. Each barrier in the database was identified as a full, partial or natural barrier. SEC evaluated only total or complete barriers to avoid overestimating actual impediments to migration.

In each population, the furthest downstream barrier was identified and listed in a Microsoft Excel spreadsheet. SEC calculated the total IP-km lost per barrier. All lost IP-km were summed, and divided by the watershed IP-km for each population to yield the percent inaccessible IP-km.

Other passage impediments were also considered; such as estuary mouths and flow-related barriers (*e.g.*, at critical riffles). These passage impediments were separated into their own attributes due to substantial differences in assessment methods. Natural barriers were not included in this attribute because they are already taken into consideration in the development of the IP networks. IP-km inadvertently indicated above natural barriers was removed from the IP-km network..

[5] http://nrm.dfg.ca.gov/PAD/Default.aspx

Large dams were evaluated as barriers because any IP reaches upstream of these barriers may have value to recovery. Spence *et al*. (2008) presented viable population targets both with and without IP km above large dams. For some watersheds it may be possible in to attain recovery goals without passage over these dams.

Condition Indicator: Passage at Mouth or Confluence for Adult, Summer Rearing, and Smolt Targets
Passage into and out of tributaries from the mainstem migratory reaches or estuaries is critical for spawning adults and emigrating smolts. Juvenile salmonids also move between stream reaches during the summer rearing phase.

Flow variability and channel conditions may limit salmonid migration into and out of tributaries and mainstem channels. Depending upon rainfall year, low flows may disconnected tributary confluences due to aggradation, or channel incision. Inaccessible tributaries may preclude the adult spawning population from accessing historical habitats, limiting overall carrying capacity and diversity in the population. Spawners waiting for flows to rise in order to access natal streams are susceptible to predation and other forms of mortality such as recreational fishing. Impacts to smolt outmigration and summer movement could also limit carrying capacity.

Ratings: Accessible proportion of IP-km
Thresholds are defined as follows:

> Poor = <50% or <32 IP-Km of historical IP-Km accessible;
> Fair = 50% to 74% of historical IP-Km habitat accessible;
> Good = 75% to 90% of historical IP-Km accessible; and
> Very Good = >90% of historical IP-Km accessible.

Methods:
Ratings were determined based on reviews of watershed reports, co-manager feedback, literature reviews, and best professional judgment. Conditions considered include:

- ❑ Annual variability in passage;
- ❑ Seasonality of passage conditions;
- ❑ Severity of condition; and
- ❑ Geographic scope of problem.

Attribute: Riparian Vegetation

Riparian vegetation is all vegetation in proximity to perennial and intermittent watercourses potentially influencing salmonid habitat conditions. Riparian vegetation mediates a variety of biotic and abiotic factors interacting and influence the stream environment. An adequately sized riparian zone with healthy riparian vegetation filters nutrients and pollutants, create a cool microclimate over a stream, provide food for aquatic organisms, maintain bank stability and provide hard points around which pools are scoured (Spence *et al.* 1996). NMFS (1996a) noted that “studies indicate that in Western states, about 80 to 90 percent of the historic(al) riparian habitat has been eliminated.” Four indicators were developed to evaluate this attribute.

Condition Indicator: Canopy Cover for Summer Rearing Target

Canopy cover is the percentage of stream area shaded by overhead foliage. Riparian vegetation forms a protective canopy, particularly over small streams by: (1) maintaining cool stream temperature in summer and insulating the stream from heat loss in the winter, (2) contributing leaf detritus, and (3) facilitating insect fall into the stream which supplements salmonid diets (Murphy and Meehan 1991). Reduction in canopy cover can change the stream environment and adversely affect salmonids by; (1) elevating temperature beyond the range preferred for rearing, (2) inhibiting upstream migration of adults, (3) increasing susceptibility to disease, (4) reducing metabolic efficiency, and (5) shifting of the competitive advantage of salmonids to non salmonid species (Hicks *et al*. 1991b).

Ratings: Average canopy closure at the reach, stream and population scale
CDFG (2004) recognized 80 percent canopy as optimal for salmonid habitat at a reach scale. Given canopy closure varies inversely with stream order (as a function of channel width), an average canopy closure of 70 percent was used to describe good conditions. This accounts for the natural range of variability, and acknowledged bias in riparian shading estimates. Average stream canopy closure below 70 percent was rated progressively lower; average stream canopy above 80 percent was rated to identify refugia areas.

Stream level rating criteria
Poor = < 50% average stream canopy;
Fair = 50% to 69% average stream canopy;
Good = 70% to 80% average stream canopy; and
Very Good = > 80% average stream canopy.

Each population rating according to the following criteria:

Population level rating
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75% to 90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Methods:
CDFG (2004) habitat typing survey methods use a spherical densitometer to estimate relative vegetative canopy closure or canopy density to provides an index of stream shading. Four measurements are taken from the middle of the stream, in four quadrants from the middle of a habitat unit (downstream, right bank, upstream, left bank). Typically, canopy is recorded in approximately every third habitat unit in addition to every fully-described unit. This provides an approximate 30 percent sub-sample for all habitat units. The sub-sample is expressed as an average for each stream reach. SEC queried the stream summary database for mean percent canopy cover for each stream reach and extrapolated these data to characterize each stream, for all streams within each population (where survey data existed). Canopy closure at the stream scale was calculated from reach scale data, and aggregated by determining the percentage of streams/IP-km meeting optimal criterion at the population scale.

<u>Condition Indicator: Diameter at Breast Height (DBH) for Adult, Summer and Winter Rearing Targets</u>
Intact riparian zones, often characterized by an adequate buffer of mature hardwood and/or coniferous forests, are an important component of a properly functioning habitat conditions for salmonids. Buffers mediate upslope processes such as sediment delivery.

Spence *et al*. (1996) recognized the distance equal to the potential height of riparian trees (one site potential tree height[6]) as a minimum buffer to allow for recruitment of large wood to Pacific salmon streams. The Forest Ecosystem Management Assessment Team (1993) extended the zone of influence to two site potential tree heights or to the top of any inner gorge areas. The 100 meter buffer used for this indicator is approximately equivalent to two site potential tree heights in old growth Douglas-fir or forests or 1½ site potential tree heights in mature redwoods. Spence *et al.* (1996) suggested 200-240 feet as an appropriate site potential tree height for redwoods. Beardsley *et al*. (1999) used a diameter of 40 inches as indicative of old growth forests in the Sierra Nevada. The diameter of coastal riparian redwoods before disturbance may often have been several feet in diameter (Noss 2000). Due to data limitations south of San Francisco, two ratings for this indicator were developed.

Rating 1: Tree Diameter (North of the Golden Gate), percent of riparian zones (100 meters from centerline of the active channel) in CWHR class 5 and 6
Tree diameter was used as an indicator of riparian function based on the average DBH of a stand of trees within a buffer that extends 100 meters back from the edge of the active channel.

The California Wildlife Habitat Relationships (CWHR) model[7] was used to determine predominant vegetation patterns and corresponding size class categories to estimate average tree size diameters within 100 meters of all IP-km. CWHR is an information system and predictive model for terrestrial species in California. The information in CWHR is based on current published and unpublished biological information and professional judgment by recognized experts on California's wildlife communities. Using CWHR information obtained from CalFire, GIS was used to evaluate riparian conditions across all IP-km in independent populations and all anadromous blue-line streams in dependent populations. Data on tree size classifications were available only for the populations north of the Golden Gate. Classes 5 and 6 are typically older, larger trees expected to contribute to good conditions and were rated as follows:

Poor = ≤ 39% CWHR size class 5 and 6 across IP-km;
Fair = 40% to 54% CHWR size class 5 and 6 across IP-km;
Good = 55% to 69% CWHR size class 5 and 6 across IP-km; and
Very Good = > 69% CWHR size class 5 and 6 across IP-km.

Rating 2: Tree Diameter (South of the Golden Gate), WHR density classes across blue line streams in population
For the Santa Cruz diversity stratum (stream south of the Golden Gate), no comprehensive CWHR classification of the various size classes was available. WHR data were compiled into CWHR density classes of conifer, conifer-hardwood, and hardwood woodland categories. Because these data lack a structural element, it was necessary to default to the WHR density criteria as a proxy of riparian structure while acknowledging these data are not as robust as the diversity stratum north of the Golden Gate[8]. We

[6] Site potential tree height is the expected height a tree would attain under properly functioning conditions and varies by tree species, local climate, soils, *etc*.

[7] For more information on the CWHR model, go to: http://ceic.resources.ca.gov/catalog/FishAndGame/WildlifeHabitatRelationshipsWHRSystem.html

[8] Recovery staff were familiar with riparian stand conditions in the Santa Cruz diversity stratum and those north of San Francisco Bay and overall tree species structure and composition in these areas. Staff determined Santa Cruz

compared the high density categories (conifer, conifer-hardwood, hardwood woodland) of the Santa Cruz diversity stratum to the equivalent high density categories from the northern diversity strata and determined conditions were good if ≥ 80 percent of the population had high density categories of conifer, conifer-hardwood, and/or hardwood woodland, on average in the riparian buffer for the watershed (population). This condition was described as 60 to 100 percent canopy closure; CWHR class D. For the Santa Cruz Diversity Stratum, this indicator was rated using the percentages of size classes under density rating D to obtain the following total percentage for the size classes:

Poor = ≤ 69% CWHR density rating D across IP-km;
Fair = 70% to 79% CHWR density rating D across IP-km;
Good = ≥ 80% CWHR density rating D across IP-km; and
Very Good = no rating.

Methods:
CWHR vegetation characterization exists for three of the four coho salmon diversity strata targeted for recovery actions. Unlike data available for the northern diversity strata, to date no wide scale CWHR categorization data was available for the Santa Cruz diversity stratum. Typically, the most current and detailed data were collected for various regions of the state or for unique mapping efforts (farmland, wetlands, riparian vegetation). Various sources were compiled into the CWHR system classification. The dates for the source data vary from 1970's (urban areas) to 2000. The bulk of the forest and rangeland data were collected by CalFire/USFS 1994-1997.

Alternative tree size criteria were initially considered when evaluating riparian stand condition. This alternative considered 100 meter wide riparian stands, where more than 80 percent of the stand was comprised of trees with average DBH of 20 inches or greater, was indicative of very good conditions. However, the 20-inch DBH criteria could not be used because the corresponding CWHR size class (size class 4), encompasses a wide range of tree diameters (11-23.9 QMD (quadratic mean diameter)) (Table 7). The large range rendered size class 4 an unsuitable proxy for the 20 inch indicator. The difference in size and ecological function in a tree with an 11 inch DBH versus a 24-inch DBH is substantial, where an 11 inch tree (depending on site conditions) is almost always younger (unless it is suppressed and/or located on poor soil types) and smaller (in height as well as diameter than a 24 inch tree). Therefore, we applied size class 5 and 6 when evaluating riparian condition. Overall, we believe CWHR is the best available GIS tool to characterize riparian condition across large landscapes due to it wide-spread application, ease of use via GIS, and its standardization as an assessment tool.

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structure and composition generally comports to that in the northern diversity strata and was not comprised of inordinate proportions of dense stands of CWHR size class 1-3 trees.

Table 7. CWHR Size Class Criteria.

CWHR Code	CWHR Size Classes	DBH
1	Seedling tree	< 1.0″
2	Sapling tree	1.0″ – 5.9″
3	Pole tree	6.0 – 10.9″
4	Small tree	11.0″ – 23.9″
5	Medium/large tree	≥ 24.0″
6	Multi-layered stand	A distinct layer of size class 5 trees over a distinct layer of size class 4 and/or 3 trees, and total tree canopy of the layers > 60% (layers must have > 10.0% canopy cover and distinctive height separation).

CWHR size classes were reviewed for watersheds considered to maintain properly functioning riparian condition in four locations: Smith River at Jedidiah Smith State Park, Redwood Creek in Redwood National Park, Prairie Creek, and the South Fork Eel at Humboldt Redwoods State Park. In total, we reviewed CWHR size classes in the riparian zones of 95 miles of blue line streams and used this information to establish criteria for reference conditions. These data indicated at least 70 percent of the 100 meter wide riparian zones were comprised on CWHR size class 5 and 6 forest. From these results we determined a 100 meter wide riparian buffer consisting, on average, of ≥ 69 percent CWHR size class 5 and 6 tree represented very good conditions in the three northern diversity strata.

Landscape Context Indicator: Riparian Species Composition for Watershed Processes Target

Changes to the historical riparian vegetative community due to introduction of non-native plants or domination of early seral communities can adversely affect salmonid habitat. Invasive non-native plants such as *Arundo donax* can out-compete native plants and even form barriers to migration. Early seral species such as alder can suppress long lived conifers and significantly delay future large woody debris recruitment of these conifers. Hardwoods like alder do not form long lived woody debris elements as do conifers such as redwood and Douglas-fir.

Ratings: Current departure of riparian vegetation (within 100 meters of streams across IP-km) from historical conditions

Ecological status relates the degree of similarity between current vegetation and potential vegetation for a site or population. It can be measured on the basis of species composition within a particular community type or on the basis of community type composition within a riparian complex. Ratings were derived from Winward (1989) who developed criteria for potential natural communities.

Species composition is the presence and persistence (composition and structure) of the historical vegetative community within 100 meters of a watercourse within all IP-km of a population. Rating criteria were defined as follows:

Poor = < 25% historical riparian vegetation species composition;
Fair = 25% to 50% historical riparian vegetation species composition;
Good = 51% to 74% historical riparian vegetation species composition; and
Very Good = ≥ 75% historical riparian species composition.

Methods:
Historical vegetation status per population was difficult to obtain. We reviewed CalFire's database on major vegetation communities and determined major differences in historical vegetation species composition based on the percent of population in urban, agriculture, and herbaceous categories. Some inaccuracy likely exists with this approach because some urban areas and agricultural areas may have some riparian areas within the range of historical vegetation species composition. However, based on the widths of the riparian buffers used in this assessment we believe the majority of the areas in these categories do not maintain the historical vegetation patterns.

Attribute: Sediment

Sediment provides several important habitat functions for salmonids, including supporting spawning redds, delivering intergravel flows capable of delivering oxygen to incubating eggs, and supporting food production for rearing juveniles.

Condition Indicator: Gravel Quality Bulk samples and Embeddedness for Eggs Target
Sediment, relative to its function as a key habitat attribute for the egg life stage, was defined as streambed gravels with particle size distribution of sufficient quality to allow successful spawning and incubation of eggs. These substrates must be located within spawning habitat defined by the IP-km model.
Gravel quality was defined using two evaluation methods: bulk sampling (Valentine 1995) and embeddedness (Flosi *et al*. 2004). When bulk sampling data is available, the indicator is the portion of the sampled substrate consisting of > 0.85 millimeters and/or < 6.4 millimeters (NCRWQCB 2006). For HAB 8 data, gravel quality was defined as the distribution of embeddedness values.

Rating 1: Percent pool-tail outs sampled with embeddedness values of 1 and 2
SEC calculated the percentage of pool tail-outs within all IP km with embeddedness values of 1, 2, 3, 4, or 5 and presented them as frequency distributions at the stream scale. A bias analysis was used to determine our degree of confidence in the data and to extrapolate the data to characterize each stream. Ratings were based on frequency distributions because embeddedness scores (1-5) are ordinal numbers; and cannot be averaged and used in the simple rating of poor = > 2, fair = 1 -2, and good = < 1. Also, embeddedness estimates are visual and involve some subjectivity. Embeddedness estimates are not as rigorous as bulk gravel samples in describing spawning and incubation habitat conditions (KRIS Gualala[9]).

As described in Flosi *et al*.(2004), a score of 1 indicates substrate is less than 25 percent embedded; this is considered optimal salmonid spawning habitat. A score of 2 indicates 25-50 percent embedded and moderately impaired. A score of 3 indicates 50-75 percent embedded and highly impaired, 4 indicates 75-100 percent embedded and severely impaired, a 5 indicates the substrate is unsuitable for spawning. The embeddedness ratings used by Bleier *et al.* (2003) states the best spawning substrate is 0-50 percent embedded. CDFG's target value is 50 percent or greater of sampled pool tail-outs are within this range. Streams with less than 50 percent of their length in embeddedness values of 50 percent or less, are considered inadequate for spawning and incubation.

Typically, embeddedness ratings are recorded in every pool habitat unit, in addition to every fully-described unit which provides an approximate 30 percent sub-sample for all habitat units. This sub-

[9] http://www.krisweb.com/krisgualala/krisdb/html/krisweb/index.htm

sample is expressed as an average for each stream reach. Embeddedness rating criteria is based on criteria developed in the North Coast Watershed Assessment Program (Bleier *et al*. 2003):

Stream level embeddedness
Poor = <25% of the scores were 1s and 2s;
Fair = 25% to 50% of the scores were 1s and 2s;
Good = >50% of the scores were 1s and 2s; and
Very Good = Not defined.

The representative nature of the datasets were extrapolated to the overall population, for all streams within each population (where data were available). Rating each population required two steps; calculation of the average at the stream scale from the reach scale data, and determining the percentage of streams/IP-Km meeting optimal criteria, at the population scale.

Each population was rated according to the following criteria:

Population level embeddedness
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75% to 90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Rating 2: Percent of fines in low flow bulk samples from potential spawning sites
Ratings criteria for bulk sampling data were developed from a variety of sources, including the regional sediment reduction plans by the USEPA (1998; 1999) and the North Coast Regional Water Quality Control Board (2000; 2006) who developed a threshold of 0.85 mm for fine sediment with a target of less than 14 percent. NMFS (1996b) Guidelines for Salmon Conservation also used fines less than 0.85 millimeters as a reference and recognized less than 12 percent as properly functioning condition, 12-17 percent as at risk, and greater than 17 percent as not properly functioning. Fine sediments less than 11 percent are fully suitable, 11-15.5 percent somewhat suitable, 15.5-17 percent somewhat unsuitable and over 17 percent fully unsuitable. McMahon (1983) found that egg and fry survival drops sharply when fines make up 15 percent or more of the substrate.

Rating criteria for bulk samples are:

Poor = > 17% 0.85mm and/ or > 30% 6.3mm;
Fair = 15% to 17% 0.85mm;
Good = 12% to 14% 0.85mm and/or <30% 6.3mm; and
Very Good = < 12% 0.85mm.

Methods:
SEC queried regional data sources for bulk sediment core sample (McNeil) surveys as the preferred method for evaluating spawning gravel quality. However, few watersheds had data sufficient for a comprehensive analysis. In these circumstances, SEC used HAB 8 data from CDFG.

<u>**Condition Indicator: Quantity and Distribution of Spawning Gravels for Adult Target**</u>

The quantity and distribution of spawning substrate is the amount of spawning habitat available to the spawning population. Distribution indicates the degree of dispersion of habitat across IP-km in a population.

Ratings: Amount of optimal spawning habitat available
Female salmonids usually spawn near the head of a riffle, just below a pool, where water changes from a laminar to a turbulent flow and where there is small to medium gravel substrate. The flow characteristics at the redd location usually ensures good aeration of eggs and embryos, and flushing of waste products. Water circulation in these areas facilitates fry emergence from the gravel. Optimal conditions for spawning have nearby overhead and submerged cover for holding adults and emerging juveniles; water depth of 10 to 54 centimeters (cm); water velocities of 20 to 80 cm per second; clean, loosely compacted gravel (1.3 to 12.7 cm in diameter) with less than 20 percent fine silt or sand content; cool water (4° to 10° C) with high DO (8 mg/l); and an intergravel flow sufficient to aerate the eggs. The lack of suitable gravel often limits successful spawning in many streams.

Ratings for were developed to spatially estimate the percentage of streams within each population meeting optimal conditions. Optimal conditions are based on scientific literature, and defined according to the following criteria:

Poor = < 50% IP-km meet optimal conditions;
Fair = 50% to 74% of IP-km meet optimal conditions;
Good = 75% to 90% of IP-km meet optimal conditions; and
Very Good = > 90% of IP-km meet optimal conditions.

Methods:
To assess population conditions relative to these criteria, watershed reports, co-manager documentation and knowledge, and literature reviews to obtain quantitative data or estimates were used. Where quantitative data were lacking, a qualitative approach was used based upon best available information, spatial data and IP-km habitat potential to inform best professional judgment ratings.

<u>Condition Indicator: Gravel Quality (Embeddedness) for Summer and Winter Rearing Targets</u>
We defined food productivity, relative to its function as a key habitat attribute for summer survival, as streambed gravels with particle size distribution of sufficient quality to facilitate productive macro-invertebrate communities. These substrates must be located within spawning habitat as defined by the IP-km model. Gravel quality was defined using the distribution of embeddedness values from HAB 8.

Suttle *et al*. (2004) examined degraded salmonid spawning habitat, and its effects on rearing juveniles due to fine bed sediment in a northern California river. Responses of juvenile salmonids, and the food webs supporting them, showed increasing concentrations of deposited fine sediment decreased growth and survival. Declines were associated with a shift favorable in invertebrates toward unfavorable invertebrates (burrowing taxa unavailable as prey). Fine sediment can transform the topography and porosity of the gravel riverbed and profoundly affect the emergent ecosystem, particularly during biologically active periods of seasonal low flow. Salmonid growth decreased steeply and roughly linearly with increasing fine sediment concentration. This result was consistent with the effects of sedimentation on the food supply available to salmonids.

Ratings: Embeddedness scores
Rating criteria for embeddedness are:

Stream level embeddedness
Poor = < 25% of the embeddedness scores were 1s and 2s;
Fair = 25% to 50% of the embededdness scores were 1s and 2s;
Good = > 50% of the embededdness scores were 1s and 2s; and
Very Good = Not defined.

The representative nature of the datasets were extrapolated to the overall population, for all streams within each population where the data existed to rate each population by determining the percentage of streams/IP-km met optimal criteria, at the population scale. Each population was rated according to the following criteria:

Population level rating criteria
Poor = < 50% of streams/IP-km rating good or better;
Fair = 50% to 74% of streams/IP-km rating good or better;
Good = 75% to 90% of streams/IP-km rating good or better; and
Very Good = > 90% of streams/IP-km rating good or better.

Methods:
SEC queried CDFG HAB 8 data to rate this indicator. As described in Flosi *et al*. (2004), a score of 1 indicates substrate is less than 25 percent embedded; this is considered optimal salmonid spawning habitat. A score of 2 indicates 25-50 percent embedded and moderately impaired. A score of 3 indicates 50-75 percent embedded and highly impaired, 4 indicates 75-100 percent embedded and severely impaired, a 5 indicates the substrate is unsuitable. The percentage of pool tail-outs within all IP-km was calculated for embeddedness values, as discussed above, as a surrogate indicator for productive food availability for rearing juveniles.

Attribute: Sediment Transport

Sediment transport is the rate, timing, and quantity of sediment delivered to a watercourse. Because of their significant contribution to increased sediment in streams, two road related indicators were developed for this attribute.

<u>Landscape Context: Road Density for Watershed Processes Target</u>
Road density is the number of miles of roads per square mile of population. A series of data layers were used to calculate road density within each dependent and independent population.

Construction of a road network can lead to greatly accelerated erosion rates in a watershed (Haupt 1959; Swanson and Dryness 1975; Swanson *et al*. 1976; Beschta 1978; Gardner 1979; Reid and Dunne 1984). Increased sedimentation in streams following road construction can be dramatic and long lasting. The sediment contribution per unit area from roads is often much greater than that from all other land management activities combined, including log skidding and yarding (Gibbons and Salo 1973). Sediment entering streams is delivered chiefly by mass soil movements and surface erosion processes (Swanston 1991). Failure of stream crossings, diversions of streams by roads, washout of road fills, and accelerated scour at culvert outlets are also important sources of sedimentation in streams within (Furniss *et al.* 1991). Sharma and Hilborn (2001) found lower road densities (as well as valley slopes and stream gradients) were correlated with higher coho smolt density.

According to Furniss *et al*. (1991) "…roads modify natural drainage networks and accelerate erosion processes. These changes can alter physical processes in streams, leading to changes in streamflow regimes, sediment transport and storage, channel bank and bed configuration, substrate composition, and stability of slopes adjacent to streams. These changes can have important biological consequences, and they can affect all stream ecosystem components. Salmonids require stream habitats for food, shelter, spawning substrate, suitable water quality, and access for migration upstream and downstream during their life cycles. Roads can cause direct and indirect changes to streams that affect each of these components."

Ratings: Number of road miles per square mile in population
Cederholm *et al.* (1980) found fine sediment in salmon spawning gravels increased by 2.6 - 4.3 times in watersheds with more than 4.1 miles of roads per square mile of land area. Graham Matthews and Associates (1999) linked increased road densities to increased sediment yield in the Noyo River in Mendocino County, California. King and Tennyson (1984) found the hydrologic behaviors of small forested watersheds were altered when as little as 3.9 percent of the watershed was occupied by roads. NMFS (1996b) guidelines for salmon habitat characterize watersheds with road densities greater than three miles of road per square mile of watershed area (mi/sq. mi) as "not properly functioning" while "properly functioning condition" was defined as less than or equal to two miles per square mile, with few or no streamside roads.

Armentrout *et al.* (1998) used a reference of 2.5 mi./sq. mi. of roads as a watershed management objective to maintain hydrologic integrity in Lassen National Forest watersheds harboring anadromous fish. Regional studies from the interior Columbia River basin (USFS 1996) show that bull trout do not occur in watersheds with more than 1.7 miles of road per square mile. The road density ranking system shown in Figure 2 was developed based on the Columbia basin findings (USFS 1996).

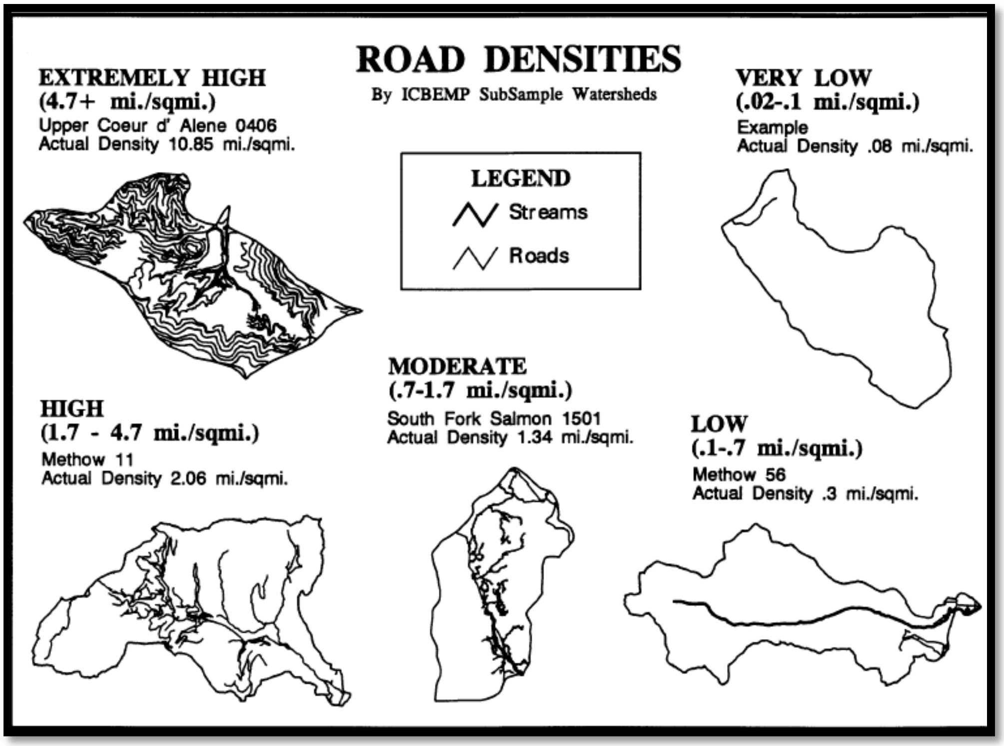

Figure 2. Graphic from the Interior Columbia Basin Management Plan, showing classes of road densities for sample watersheds (USFS, 1996).

The most inclusive datasets available for each population (see below) were used. The goal was to be as precise as possible for each population while acknowledging some inconsistency (due to the use of four datasets) may result from this approach.

Poor = > 3 miles/square mile of population
Fair = 2.5 to 3 miles/square mile of population
Good = 1.6 to 2.4 miles/square mile of population
Very Good = < 1.6 miles/square mile of population

Methods:
GIS analysis of the miles of road networks within a population made use of several data sources:

1. CalFire Timber Harvesting History. GIS vector dataset, 1:24,000. 2007. Watersheds between Cottaneva Creek (inclusive) and the Russian River (inclusive);
2. CalTrans, Tana_rds_d04. GIS vector dataset, 1:24,000. 2007. Marin County watersheds;
3. U.S. Census Bureau, Roads. GIS vector dataset., 1:24,000. 2000. San Mateo County watersheds; and
4. County of Santa Cruz – Roads; Streets. GIS vector dataset, 1:24,000. 1999. Santa Cruz County watersheds.

The resulting linear measurement (in miles) was compared against the total population area in square miles to derive watershed (population) road density.

__Landscape Context Indicator: Streamside Road Density for Watershed Processes Target__

Streamside road density is the density of roads, per square mile of a 200 meter riparian corridor (100 meters on either side of the stream centerline) within the population.

Roads frequently constitute the dominant source of sediments delivered to watercourses. Roads constructed within the riparian buffer zone pose many risks to salmonids habitat including the loss of shade, decreased large wood recruitment, and delivery of fine sediment and initiation of mass wasting (Spence *et al*. 1996). Rock revetments are often used to prevent streams from eroding road beds, resulting in channel confinement that can lead to incision of the stream bed. Roads in close proximity to watercourses may have a greater number of crossings which may act as: (1) impediments to migration, (2) flow restrictions which artificially change channel geometry, and (3) sources of substantial sediment input due to crossing failure.

Ratings: Number of road miles per square mile within 100 meters of the watercourse (centerline)
The USFS (2000) provides data for near stream roads in road miles per square mile and a frequency distribution was used to derive values showing very low relative risk as very good (<0.1 mi/sq. mi) and the opposite end of the frequency spectrum as posing high relative risk to adjacent coho habitat as poor (> 1 mi/sq. mi).

Poor = > 1 mile/square mile of riparian corridor;
Fair = 0.5 to 1 mile/square mile of riparian corridor;
Good = 0.1 to 0.4 mile/square mile of riparian corridor; and
Very Good = < 0.1 mile/square mile of riparian corridor.

Methods:
The most inclusive datasets available for each population were used. The goal was to be as precise as possible for each population while acknowledging some inconsistency (due to the use of four datasets) may result from this approach.

A series of GIS data layers were used to calculate the riparian buffer and road density within each dependent and independent population:

To create the riparian buffer these stream files were used:
1. Streams - CalFire, Hydrography watershed Assessment; Wahydro. GIS vector dataset, 1:24,000. 1998. Watersheds from Cottaneva Creek (inclusive) to the Russian River (inclusive); and
2. Streams - USGS National Hydrography Dataset; Flowline (1801, 1805), vector digital dataset, 1:24,000. 2004. Watersheds in Marin, San Mateo, and Santa Cruz counties.

To create the road layer these stream files were used:

1. CalFire Timber Harvesting History. GIS vector dataset, 1:24,000. 2007. Watersheds between Cottaneva (inclusive) and the Russian River (inclusive);
2. CalTrans, Tana_rds_d) 4. GIS vector dataset, 1:24,000. 2007. Marin County watersheds;
3. U.S. Census Bureau, Roads. GIS vector dataset., 1:24,000. 2000. San Mateo County watersheds; and
4. County of Santa Cruz – Roads; Streets. GIS vector dataset, 1:24,000. 1999. Santa Cruz County watersheds.

Attribute: Smoltification

This attribute focuses on temperature criteria required during the physiological changes young salmonids undergo in preparation to enter the ocean (smoltification) and potential anthropogenic sources which lead to alterations in stream water temperature. While the smoltification process can occur throughout the wet season, most salmonids smolt and emigrate to the ocean during the spring months (specific emigration periods vary between and among species and across the geographic range). Naturally occurring warmer water temperatures (such as those that may occur in streams within the southern extent of the NCCC Recovery Domain or where solar radiation occurs naturally) were distinguished from temperature impairments due to human induced alterations.

Condition Indicator: Smoltification Stream Temperature for Smolt Target

The extent and magnitude of spatial and temporal temperature variations within emigration routes was considered when evaluating potential impacts. For example, where access to cold water refugia is lost, the length of warm water exposure was considered with respect to behavior alteration and/or physiological impairment during smoltification.

Ratings:

In considering anthropogenically altered water temperature regimes and effects on smoltification and emigration, location, extent, magnitude (significance of temperature alteration), and duration of the effects were evaluated. The rating criteria considered the following factors:

- Magnitude of temperature alteration (*i.e.*, how much does the temperature deviate from natural stream water temperatures or from preferred criteria);
- Relative percent of rearing habitat, or relative percent of the emigrating population affected by anthropogenically altered temperature regimes;
- Relative location and extent of the affected reaches within the population (*i.e.*, the importance of the individual reach to the population); and
- The duration these effects persist (including effects on diel temperature fluctuations).

The basis for establishing the effect of temperature on smoltification and emigration was made where possible, it must ultimately be extrapolated to the population level. For example, a large anthropogenic temperature alteration low in the mainstem of a watershed could be considered fairly significant in affecting not only the reach in which the alteration occurs, but for the entire population, since emigrating smolts from the upstream reaches will have to pass through the downstream affected reach(s).

For rating the population, optimal conditions are described as > 6° C but < 16° C [Temperature expressed as maximum weekly maximum temperature (MWMT)], and/or anthropogenic thermal inputs/alterations do not affect smoltification or emigration.

Temperature ratings are:

Poor = < 50% IP-km (> 6° and < 16° C);
Fair = 50% to 74% IP-km (> 6° and < 16° C);
Good = 75% to 90% IP-km (> 6° and < 16° C); and
Very Good = > 90% IP-km (> 6° and < 16° C).

Methods:

A literature review was conducted to identify sources of temperature information, and evaluate temperature thresholds necessary to support and to avoid delays smoltification and emigration. Examples of anthropogenic sources of in-stream temperature alteration to be considered include, but are not limited to:

- ❑ Off channel pond discharges;
- ❑ On-channel pond complexes;
- ❑ Agricultural land discharges;
- ❑ Dams and reservoirs (USEPA 2003);
- ❑ Riparian clearing that reduces canopy cover and increases instream solar warming;
- ❑ Water withdrawals (USEPA 2003);
- ❑ Channeling, straightening or diking (USEPA 2003); and
- ❑ Removing upland vegetation or creating impervious surfaces (USEPA 2003).

Attribute: Velocity Refuge

Velocity refuge is habitat providing space and cover for adult and juvenile salmonids during high velocity flood flows. Refuge habitats may include main-channel pools with LWD (or other forms of complexity), or off-channel habitats such as alcoves, backwaters, or floodplains (Bustard and Narver 1975; Bell *et al*. 2001). Floodplains are geomorphic features frequently inundated by flood flows, and often appear as broad flat expanses of land adjacent to channel banks.

Condition Indicator: Floodplain Connectivity for Adult and Winter Rearing Targets

Floodplain connectivity is the frequency of floodplain inundation in unconfined reaches. Frequencies approximating those of an unaltered state retain the ability to support the emergent ecological properties associated with floodplain connectivity. Although this definition goes beyond an indication for velocity refuge, the broader concept was refined because it represents important habitat features for the target life stages.

Ratings: Percent of floodplain connectivity of flood-prone zones within IP-km

Periodic inundation of floodplains by storm flows provides several ecological functions beneficial to salmon, including: coarse sediment sorting, fine sediment storage, groundwater recharge, velocity refuge, formation and maintenance of off-channel habitats, and enhanced forage production (Stanford *et al*. 2004). Floodplain connectivity is associated with more diverse and productive food webs (Power *et al*. 1996). Channel incision can result in the reduction or elimination of access for biota to lateral floodplain habitats (Power *et al*. 1996).

Stream complexity that creates low velocity areas during high flow events, whether from LWD, off-channel habitats, or wetland areas, is an important component of winter rearing habitat. Bell (2001) documented increased fidelity and survival of winter rearing juvenile coho salmon in alcoves and backwaters in a Northern California stream. Others have documented increased densities of coho salmon in side-channel pools (Bjornn and Reiser 1991). In British Columbia, juveniles preferred stream flows < 15 cm/sec (Bustard and Narver 1975). Bisson *et al.* (1988) indicated a preferred velocity of < 20 cm/sec, and < 30 cm/sec was cited in a third study (Tschaplinski and Hartman 1983). Salmonids use off-channel habitats during winter for refuge during high flow events and floodplains for feeding during early spring and summer.

The United States Forest Service (USFS) (2000) Region 5 watershed condition rating system is aimed at maintaining "…the long-term integrity of watersheds and aquatic systems on lands the agency manages." Scores were based on best professional judgment, by staff familiar with instream conditions necessary of salmonid rearing using criteria are similar to regional standards (USDA 1995; Spence *et al.* 1996).

The USFS considers channel condition to be properly functioning when more than 80 percent of the low gradient response reaches have floodplain connectivity, while 50-80 percent was considered partially functional and less than 50 percent non-functional. Ratings are as follows:

Poor = < 50% response reach connectivity;
Fair = 50% to 80% response reach connectivity;
Good = > 80% response reach connectivity; and
Very Good = Not defined.

Methods:
This indicator was assessed by quantifying the degree of urbanization, channelization, incision and other factors affecting flood-prone areas for each population. Federal Emergency Management Agency's (FEMA) delineation of Zone A Flood Zone Designation maps assisted this interpretation in the definition of flood-prone areas. NMFS watershed characterization maps and statistics also assisted to describe the degree of urbanization and other land uses such as agriculture.

The ratings for this indicator were determined based on NMFS analysis of watershed reports, co-manager documentation, literature reviews, and best professional judgment. Where quantitative data was lacking, a qualitative approach was utilized using the best available literature, spatial data and IP-km habitat potential to inform best professional judgment ratings

Attribute: Viability

This attribute addresses a suite of demographic indicators defining population status and provides an indication of their extinction risk. The viability attribute is a population metric and, in conjunction with habitat attributes, provides a means to validate assumptions and conclusions. For example, if habitat quality was rated as good, and fish density or abundance was poor, it provided a basis to re-evaluate conclusions and examine assumptions about causative relationships between populations and habitat. In the specific context of a key attribute, viability is the suite of demographic indicators defining the population status (which relate directly to their extinction risk).

Size Indicator: Density for Adult Target
Density was used as an indicator for the spawner life-stage because it is one of the principle metrics used to define population viability in the biological viability report (Spence *et al*. 2008) developed by the Technical Recovery Team (TRT).

Ratings: Average spawner density per IP-km

The TRT established criteria of one spawning adult per IP-km as a reasonable threshold to indicate a population at high risk of depensation [10] (Spence *et al*. 2008). This threshold was used as an indicator for a poor spawner density.

The TRT also developed density criteria for population viability. For the smallest of independent populations (*i.e*., those with 32 IP-km), adult spawning densities should exceed 40 fish per IP-km. Densities may decrease to 20 fish per IP-km as the size of an independent population approaches ten times the minimum size (*i.e*., 32 IP-km). This formula represents the spawner density threshold for a low risk of extinction, and was used as our criteria for a good rating (Table 8). A fair rating was any density between poor and good. A criterion rating for very good was not established.

Table 8. Population specific density (# of adults/IP-km) criteria for spawning adult coho based on TRT density criteria (Spence *et al*. 2008).

Population	Poor	Fair	Good	Very Good
Usal Creek	≤1	Between	≥34.0	None
Cottaneva Creek	≤1	Between	≥34.0	None
Ten Mile River	≤1	Between	≥34.9	None
Wages Creek	≤1	Between	≥34.0	None
Pudding Creek	≤1	Between	≥34.0	None
Noyo River	≤1	Between	≥34.0	None
Caspar Creek	≤1	Between	≥34.0	None
Big River	≤1	Between	≥28.9	None
Albion River	≤1	Between	≥38.1	None
Big Salmon Creek	≤1	Between	≥34.0	None
Navarro River	≤1	Between	≥28.3	None
Garcia River	≤1	Between	≥34.9	None
Gualala River	≤1	Between	≥24.8	None
Russian River	≤1	Between	≥20.0	None
Salmon Creek	≤1	Between	≥34.0	None
Pine Gulch	≤1	Between	≥34.0	None
Walker Creek	≤1	Between	≥37.5	None
Lagunitas Creek	≤1	Between	≥37.3	None
Redwood Creek	≤1	Between	≥34.0	None
San Gregorio Creek	≤1	Between	≥34.0	None
Pescadero Creek	≤1	Between	≥38.0	None
Gazos Creek	≤1	Between	≥34.0	None
Waddell Creek	≤1	Between	≥34.0	None
Scott Creek	≤1	Between	≥34.0	None

[10] At very low densities, spawners may find it difficult to find mates, small populations may be unable to saturate predator populations, and group dynamics may be impaired, *etc*. Small populations may experience a reduction in per-capita growth rate with declining abundance, a phenomenon known as depensation (Spence *et al*. 2008).

San Vicente Creek	≤1	Between	≥34.0	None
San Lorenzo River	≤1	Between	≥34.6	None
Soquel Creek	≤1	Between	≥34.0	None
Aptos Creek	≤1	Between	≥34.0	None

Methods:
To assess the indicator by population, the estimated annual spawning population (N_a) divided by the amount of IP-Km available for spawning (N_a/IP-Km). N_a was measured as the geometric mean of annual spawner abundance for the most recent three to four generations (Spence *et al.,* 2008). The TRT evaluated current abundance for all independent populations in the ESU and found data availability was insufficient in most cases. We were therefore forced to make reasonable inferences based on what information was available. Data sources we used for this assessment included the NMFS Fisheries Science Center database, literature review, and previous status assessments (Good *et al.* 2005; Spence and Williams 2011).

<u>Size Indicator: Abundance for Smolt Target</u>
We use abundance as an indicator not only because it is a direct measure of population size, but because smolt populations can be estimated with various out-migrant trapping and mark and recapture methods.

Ratings
We used the following equation was used to calculate the number of smolts (at time *t*) needed to satisfy abundance criteria (S_t):

$$S_t = \frac{A_{t+i}}{0.01_i}$$

Where A_{t+1} is the adult abundance after time interval (*i*) divided by the assumed marine survival of 1 percent during time interval *i*. Therefore, to calculate smolt abundance criteria for each population: good criteria would be the low risk abundance (the low risk adult target in Spence *et al*. (2008) divided by 0.01); and poor criteria would be the "high risk abundance" (the high risk adult target in Spence *et al*. (1996) divided by 0.01). Fair criteria would be abundance levels between low risk and high risk. For example, for the Noyo River this calculation yields the following rating (Table 9).

Table 9. Example of smolt indicator criteria for smolt abundance Noyo River coho calculated from TRT adult abundance criteria.

Smolt Abundance	Poor	Fair	Good
	<High Risk	Moderate Risk	> Low Risk
Noyo River	<11,800	11,800- 400,000	>400,000

Methods:
To assess the status of smolt production for a given population we need to rely on available monitoring data, most of which is contained in data sources such as the NMFS Fisheries Science center database, NMFS recovery library, and previous status assessments (Good *et al.* 2005). When no population estimates are currently available for the smolt life stage (or any other), we reviewed the data sources and made reasonable inferences as to the probable status of smolts.

<u>Size Indicator: Density for Summer Rearing Target</u>

Assessing juvenile density provides an indication of species presence and relative carrying capacity. Consistently low density estimates within a population may suggest the population or habitat is not functioning properly. High density estimates suggest a population is properly functioning and can be used by fishery managers to prioritize threat abatement efforts.

Ratings: Average juvenile density in population

Although methods for estimating the population abundance of juvenile coho salmon have been developed (Hankin and Reeves 1988), there are few estimates for populations within the CCC coho salmon ESU using these techniques. Estimates of juvenile density however, are more common and provide some indication of life-stage-specific status. Density estimates may also be useful in indicating habitat quality if streams are adequately seeded.

Rating criteria for juvenile density were based on the assumption that approximately 1.0 fish per square meter is a reasonable benchmark for fully occupied, good habitat (Nickelson *et al.* 1992; Solazzi *et al.* 2000). Ratings are as follows:

Poor = < 0.2 fish/meter^2;
Fair = 0.2 to 0.5 fish/meter^2;
Good = 0.5 to1.0 fish/meter^2; and
Very Good = > 1.0 fish/meter^2

Methods:

The juvenile density indicator was informed through a review of the literature including CDFG reports, NMFS technical memorandums, watershed analyses, section 10 research reports, and fisheries management and assessment reports. Co-managers were also interviewed. The information was compiled and synthesized by NMFS biologists (with extensive field experience) who used best professional judgment to rate the density.

<u>Size Indicator: Spatial Structure for Summer Rearing Target</u>

Current distribution of the population occupying available habitat is one of the four key factors in determining salmonid population persistence (McElhany *et al*. 2000). Species occupying a larger proportion of their historical range have an increased likelihood of persistence (Williams *et al.* 2007). To evaluate current distribution the historical range (IP-km) was compared to the percentage of habitat currently occupied by the juvenile life stage in the population.

Ratings: Current versus historical juvenile distribution across IP-Km

The following indicator ratings developed by Williams *et al.* (2006) for a similar conservation assessment described in Williams *et al.* (2007)

Poor = < 50% of historical range;
Fair = 50% to 74% of historical range;
Good = 75% to 90% of historical range; and
Very Good = > 90% of historical range.

Methods

California Department of Fish and Game, NMFS, and other agency and organization surveys, data sources and reports were used in evaluating the percentage of historical habitat currently occupied by the species. Population characterization maps were compared with IP-km maps to provide a spatial representation to estimate the percentage of the historical range currently occupied.

Attribute: Water Quality

Water quality was assessment as an attribute to classify three indicators: water temperature, toxicity, turbidity.

Condition Indicator: Temperature (Mean Weekly Maximum Temperature (MWMT)) for Summer Rearing Target

Water temperature is an important indicator of water quality, particularly with respect to juvenile coho salmon, due to a close association with temperature conditions. Juvenile salmonids respond to stream temperatures through physiological and behavioral adjustments that depend on the magnitude and duration of temperature exposure. Acute temperature effects result in death after exposures ranging from minutes to days. Chronic temperature effects are associated with exposures ranging from weeks to months. Chronic effects are generally sub-lethal and may include reduced growth, disadvantageous competitive interactions, behavioral changes, and increased susceptibility to disease (Sullivan *et al*. 2000). A measure of chronic temperature was used because it is more typical of the type of stress experienced by summer rearing juveniles in the CCC coho ESU rather than acute temperature stress.

Ratings: Proportion of IP-km in each temperature threshold class

Juvenile salmonids prefer water temperatures of 12° C to 15° C (Brett 1952; Reiser and Bjornn 1979), but not exceeding 22° C to 25° C (Brungs and Jones 1977) for extended time periods. Chronic temperatures, expressed as the maximum weekly average temperature, in excess of 15° C to 18° C, are negatively correlated with coho salmon presence (Hines and Ambrose 2000; Welsh *et al*. 2001). Sullivan *et al.* (2000) recommended a chronic temperature threshold of 16.5° C for this species. Water temperatures for good survival and growth of juvenile coho salmon range from 10° to 15° C (Bell 1973; McMahon 1983). Growth slows considerably at 18° C and ceases at 20° C (Stein *et al*. 1972; Bell 1973). The likelihood of juvenile coho salmon occupying habitats with maximum weekly average temperatures exceeding 16.3° C declined significantly (Welsh *et al.* 2001) in the Mattole River watershed in southern Humboldt County, California.

Temperature thresholds for chronic exposure are typically based on the maximum weekly average temperature (MWAT) metric. Due to some confusion in the literature regarding the appropriate definition and application of MWAT, the seven day moving average of the daily maximum (7DMADM or MWMT) indicator was used, rather than the seven day moving average of daily average (7DMADA or MWAT), because it correlated more closely correlated with observed juvenile distribution (Hines and Ambrose 2000). However, where MWMT data was not available, MWAT was used. We established two sets of rating criteria where the calculation of for MWMT was two degrees Celsius higher than the MWAT.

Work by Hines and Ambrose (2000) and Welsh *et al*. (2001) in northwestern California found that coho salmon juveniles were absent in streams where the MWAT exceeded 16.8° C. Welsh *et al*. (2001) noted transitory water temperature peaks can be harmful to salmonids and are better reflected by the maximum floating weekly maximum water temperature (MWMT). The Oregon Department of Fish and Wildlife uses an MWMT value of 64° F as a criterion protective of water quality, which is similar to the finding of Welsh *et al*. (2001).

Population level temperature ratings are:

Poor = < 50% IP-km (< 16° C MWMT);
Fair = 50% to 74% IP-km(< 16° C MWMT);
Good = 75% to 90% IP-km(< 16° C MWMT); and
Very Good = > 90% IP-km (< 16° C MWMT).

Methods:
To assess conditions throughout each population, it was necessary to evaluate temperature conditions throughout all potential rearing areas (*i.e*. across all IP-km). A method for spatializing site-specific temperature data was established by plotting these data on a map of the IP-km network. Each data point was color coded to indicate the temperature threshold the site exceeded (*i.e.,* sites with MWMT > 16° C were colored red, *etc*.). For locations with multiple years of data, we averaged the MWMT or MWAT values and indicated the number of years of data and standard deviations. The temperatures were extrapolated to IP-km reaches based upon an understanding of typical spatial temperature patterns and staff knowledge of specific watershed conditions. Finally, where temperature data was limited or absent, best professional judgment was used and assigned a low confidence rating in the results.

<u>Condition Indicator: Toxicity for Adult, Summer and Winter Rearing, and Smolt Targets</u>
Optimal conditions for salmonids, their habitat and prey, include clean water free of toxins, contaminants, excessive suspended sediments, or deleterious temperatures. Toxins are substances (typically anthropogenic in origin) which may cause acute, sub-lethal, or chronic effects to salmonids or their habitat. These include (but are not limited to) toxins known to impair watersheds, such as copper, diazinon, nutrients, mercury, polyaromatic hydrocarbons (PAHs), pathogens, pesticides, and polychlorinated biphenyls (PCBs), herbicides and algae.

All target life stages of salmonids depend on good water quality, and the water quality attribute is impaired when toxins or other contaminants are present at levels adversely affecting one or more salmonid life stages, their habitat or prey. Salmonids are sensitive to toxic impairments, even at very low levels (Sandahl *et al.* 2004; Baldwin and Scholz 2005). For example, adult salmonids use olfactory cues to return to their natal streams to spawn, and low levels of copper has been show to impair this ability (Baldwin and Scholz 2005).

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). These same flows may carry toxins from a variety of point and non-point sources to the stream. The exposure of returning adults to toxins in portions of their IP-km can reduce the viability of the population by impairing migratory cues, or reducing the amount of available spawning and rearing habitat, thereby lowering the carrying capacity of the population. Each life stage was assessed according to the seasonality of effects produced by the toxin for each life stage across all IP- km.

Ratings: Risk of adverse effects to salmonids due to toxins
Ratings for toxicity are:

Poor = Acute effects to fish and their habitat (*e.g.,* mortality, injury, exclusion, mortality of prey items);

Fair = Sub lethal or chronic effects to fish and their habitat (*e.g.,* limited growth, periodic exclusion, contaminants elevated to levels where they may have chronic effects). Chronic effects could include suppression of olfactory abilities (affecting predator avoidance, homing, synchronization of mating sues, *etc*.), tumor development (*e.g*., PAHs). This could include populations without data but where land use is known to contribute pollutants (*e.g*., significantly urbanized or supporting intensive agriculture, particularly row crops, orchards, or confined animal production facilities);

Good = No acute or chronic effects from toxins are noted and/or population has little suspect land uses, and insufficient monitoring data are available to make a clear determination. Many Northern California populations (particularly those held in private timber lands) are likely to meet these criteria; and

Very Good = No evidence of toxins or contaminants. Sufficient monitoring conducted to make this determination, or areas without contributing suspect land uses (*e.g.*, many wild and scenic rivers, wilderness areas, *etc*.). Available data should support very good ratings.

Methods:
For this analysis, some constituents were excluded from consideration because they were assessed by other indicators (*i.e.,* Water Quality/Temperature). We reviewed a variety of materials to derive appropriate ratings, including data from the California Regional Water Quality Control Boards, the U.S. Environmental Protection Agency, and other local and regional sources to inform our ratings of water quality limited segments for any toxins known or suspected of causing impairment to fish. We also reviewed scientific literature, and available population specific water quality reports. Working with SEC and NMFS staff water quality specialists, a qualitative decision structure was developed (Figure 3) to rate each population where more specific data were lacking.

Decision Matrix for Each Life Stages/Water Quality/Toxicity for Key Independent/Dependent Populations

Each life stage must be assessed according to the seasonality of affects produced by the toxin for each life stage across all IP-km.

1. Are toxins/chemicals present in the watershed which could potentially (through direct discharge, incidental spills, chronic input, etc.) entering the water column?

   a. Yes: > 2
   b. No: Toxicity not a threat (assumed to be good)

2. Is the chemical/substance a known toxin to salmonids?

   a. Yes: >3
   b. No: Toxicity not a threat (assumed to be good)

3. Are salmonids spatially/temporally exposed to the toxin during any life stage or are the toxin present in a key subwatershed (where salmonids no longer occur) important for species viability.

   a. Yes: > 4
   b. No: Toxicity not a threat (assumed to be Good/Fair)

4. Potential salmonid presence to toxin established. Use best professional judgment to assign Fair/Poor rating. Consider toxicity of chemical compound, persistence of the compound, spatial extent/temporal exposure, future reintroduction efforts, and potential overlap of land use activities (*e.g.*, pesticide/herbicide intensive farming practices) to species viability/presence when assigning rating.

Figure 3. Qualitative decision structure for evaluating water quality/toxicity. The matrix was used to determine the likelihood of toxins being present and adversely affecting freshwater salmonid life history stages.

<u>**Condition Indicator: Turbidity for Adult, Summer and Winter Rearing, and Smolt Targets**</u>

Research has demonstrated highly turbid water can adversely affect salmonids, with harmful effects as a direct result of suspended sediment within the water column. The mechanisms by which turbidity impacts stream-dwelling salmonids are varied and numerous. Turbidity of excessive magnitude or duration reduces feeding efficiency, decrease food availability, impair respiratory function, lower disease tolerance, and can also directly cause fish mortality (Cordone and Kelley 1961; Berg and Northcote 1985; Gregory and Northcote 1993; Velagic 1995; Waters 1995; Harvey and White 2008). Mortality of very young salmonids due to increased turbidity has been reported by Sigler *et al.* (1984). Even small pulses of turbid water will cause salmonids to disperse from established territories (1995), which can displace fish into less suitable habitat and/or increase competition and predation, decreasing chances of survival.

Ratings:

Risks to each life stage were assessed according to the seasonality of affects produced by the turbidity for each life stage across all IP-km.

The ratings were based upon the percentage of IP-km habitat within a population maintaining a moderate or lower sub lethal effect in regard to turbidity dose (*i.e.*, based upon both concentration and exposure duration). Using Figure 4, turbid conditions that score a 4 SEV or higher during any time scale along the x-axis represent conditions likely limiting juvenile salmonid survival. Conversely, a score of 3 SEV or lower represent conditions favoring survival to the next life stage. The extent that favorable turbidity conditions exist across the spatial population scale determines the overall score for a given population.

Data regarding turbidity was unavailable for many populations. In the absence of turbidity data, information and data from reports regarding sediment input from roads, sediment contributions from landslides and other anthropogenic sources, and best professional judgment was used to assess turbidity risk at the population scale.

Each target life stage was assessed independently according to the seasonality of affects produced by the turbidity for adults, summer and winter juvenile rearing, and smolts across IP-km:

Poor = < 50% of IP-km maintains score of 3 SEV or lower;
Fair = 50% to 74% of IP-km maintains score of 3 SEV or lower;
Good = 75% to 90% of IP-km maintains score of 3 SEV or lower; and
Very Good = > 90% of IP-km maintains score of 3 SEV or lower.

Methods:
Turbidity indicators focused on suspended sediment concentration and duration of exposure. To document the relationship between dose (the product of turbidity and exposure time) and the resultant biological response of fish, Newcombe (2003) reviewed existing data to develop empirical equations to estimate behavioral effects from a given turbidity dose. For juvenile and adult salmonids, the expected behavioral response and severity of ill effects (SEV) is illustrated in Figure 4 (from Newcombe 2003).

Visual clarity of water (*y*BD) and related variables — *alternate*: NTU, *z*SD; *preferred*: BA, *y*BD

Duration of exposure to conditions of reduced VISUAL CLARITY ($\log_e$ hours)

Severity-of-ill-effect Scores (SEV) -- Potential

$$SEV = -4.49 + 0.92 (\log_e h) - 2.59 (\log_e yBD)$$

Fish reactive distance: *calibrated for trout*

NTU	*z*SD (m)	BA (m^{-1})	*y*BD (m)		0	1	2	3	4	5	6	7	8	9	10	ψ_{BD} (cm)	*x*RD (cm)	
				Δ_{15}	Δ_{16}	Δ_{17}	Δ_{18}	Δ_{19}	Δ_{20}	Δ_{21}	Δ_{22}	Δ_{23}	Δ_{24}	Δ_{25}				
1100	0.01	500	0.010	Δ_{14}	7	8	9	10	11	12	13	14				1	-	O
			0.014	Δ_{13}	7	7	8	9	10	11	12	13	14			1	-	N
400	0.03	225	0.02	Δ_{12}	$_{P}6^{\pi}$	7	7	8	9	10	11	12	13	14		2	-	M
			0.03	Δ_{11}	4	5	6	7	8	9	10	11	12	13	14	3	-	L
150	0.07	100	0.05	Δ_{10}	3	$_{P}4^{\pi}$	$_{P}5^{\pi}$	6	7	8	9	10	11	12	13	5	-	K
			0.07	Δ_{9}	2	3	4	5	6	7	8	9	10	11	11	7	-	J
55	0.15	45	0.11	Δ_{8}	$_{P}\underline{1}^{\pi}$	2	3	4	5	6	7	8	9	10	10	11	6	I
			0.16	Δ_{7}	$\underline{0}$	$\underline{1}$	2	3	4	$\underline{5}$	6	7	8	9	9	16	17	H
20	0.34	20	0.24	Δ_{6}	$\underline{0}$	$_{P}\underline{0}^{\pi}$	$_{P}\underline{1}^{\pi}$	2	3	4	5	6	7	$\underline{8}$	8	24	30	G
			0.36	Δ_{5}	$\underline{0}$	$\underline{0}$	$\underline{0}$	$\underline{1}$	2	3	4	5	6	6	7	36	42	F
7	0.77	9	0.55	Δ_{4}	$\underline{0}$	$_{P}\underline{0}^{\pi}$	$\underline{0}$	$\underline{0}$	$\underline{1}$	2	3	4	4	5	6	55	55	E
			0.77	Δ_{3}	$\underline{0}$	$_{P}\underline{0}^{\pi}$	$^{P}\underline{0}_{\pi}$	$\underline{0}$	$\underline{0}$	$\underline{1}$	2	3	4	4	5	77	66	D
3	1.53	4	1.09	Δ_{2}	$\underline{0}$	$_{P}\underline{0}^{\pi}$	$\underline{0}$	$\underline{0}$	$\underline{0}$	$\underline{0}$	1	2	3	4	5	109	77	C
			1.69	Δ_{1}	$\underline{0}$	$\underline{0}$	$\underline{0}$	$\underline{0}$	0	0	0	1	2	2	3	169	90	B
1	3.68	2	2.63		$^{P}\underline{0}_{\pi}$	$^{P}\underline{0}_{\pi}$	$^{P}\underline{0}_{\pi}$	0	0	0	0	0	0	1	2	263	104	A
						Δ_{1}	Δ_{2}	Δ_{3}	Δ_{4}	Δ_{5}	Δ_{6}	Δ_{7}	Δ_{8}	Δ_{9}	Δ_{10}			
					1	3	7	1	2	6	2	7	4	11	30			
					Hours			Days			Weeks		Months					
					a	b	c	d	e	f	g	h	i	j	k			

Figure 1. Impact Assessment Model for Clear Water Fishes Exposed to Conditions of Reduced Water Clarity. A model to estimate severity of impact on rearing success of clear water fish as a function of reduced visual clarity of water (m) and duration of exposure (h), for juvenile and adult life history phases; includes calibration for reactive distance of trout.

KEY:

*y*BD Black disk sighting range (m): horizontal measurement in water of any depth (reciprocal of beam attenuation).

ψBD Black disk sighting range (cm): a convenient calibration for measurements made in very cloudy water.

BA Beam attenuation (m^{-1}): measures absorption and scattering of light by "water constituents" – clay and color; reciprocal of black disk sighting range.

*z*SD Secchi disk sighting range (m): a vertical measurement, usually in deep water.

*x*RD Reactive distance of adult trout (pooled data for rainbow, lake and brook) to fish prey as a function of visual clarity. Alternate, proportional, calibrations can be inferred for largemouth bass and bluegill based on their maximum reaction distances (200 cm, and 30 cm, respectively).

NTU Nephelometric turbidity units: a measure of light scattering by suspended clay particles (0.2 to 5 µm diameter).

SEV Severity of Ill Effect Scale

a. Semi-Quantitative

$0 \le$ nil < 0.5; $0.5 \le$ minor < 3.5; $3.5 \le$ moderate < 8.5; $8.5 \le$ severe < 14.5. Impact assessment is based on net duration (less clear water intervals) and weighted average visual clarity data. Recurrent events sum when integrated over relevant intervals: for a year class (a life history phase, or a life cycle); a population ("year over year" events); habitat damage (hours < duration ≤ years); and restoration (year < time ≤ years). For events involving suspended sediment (may include clay as one of the particle sizes in a range of sizes) (see Newcombe and Jensen, 1996).

b. Qualitative

0: *Ideal.* Best for adult fishes that must live in a clear water environment most of the time.

1-3: *Slightly Impaired.* Feeding and other behaviors begin to change.

4-8: *Significantly Impaired.* Marked increase in water cloudiness could reduce fish growth rate, habitat size, or both.

9-14: *Severely Impaired.* Profound increases in water cloudiness could cause poor "condition" or habitat alienation.

c. Stipple – Areas with least available data (1 day to 30 months).

Predator Prey Dynamics

(a) $^{P}\underline{0}_{\pi}$: Some predatory fish (P) catch more prey fish (π) in clear water ($^{P}\pi$) than they do in cloudy water.

(b) $_{P}\underline{1}^{\pi}$, $_{P}\underline{5}^{\pi}$: Survival of some fishes is enhanced ($_{P}{}^{\pi}$) by natural, seasonal, cloudiness (two examples shown).

(c) *SEV*: Severity of ill effect data, underscored, are from published sources (see Literature Cited), or have the support of consensus within the discussion group, or both.

aA, kO Row labels (upper case) and column labels (lower case); paired, these serve as cell coordinates (two examples shown).

Figure 4. Impact Assessment Model for Clear Water Fishes Exposed to Conditions of Reduced Water Clarity (from Newcombe 2003).

Assessing Future Conditions: Stresses

Stresses and threats are the drivers and mechanisms leading to population decline. Stresses are defined as "the direct or indirect impairment of salmonid habitat from human or natural sources" (TNC 2007). Stresses represent altered or impaired key attributes for each population, such as impaired watershed hydrology or reduced habitat complexity. They are the inverse of the key attributes. For example, the attribute for passage would become the stress of impaired passage. These altered conditions, irrespective of their sources, are expected to reduce population viability. Stresses are initially evaluated as the inverse of the key attribute ranking (*e.g*., key attributes rated as poor may result in a stress ranking as very high or high). Ultimately the resulting stress ranking is determined using two metrics, the severity of damage and scope of damage. For each population and life stage, stresses were ranked using these metrics, which were combined using algorithms contained in CAP to generate a single rank for each stress identified. Stresses ranked very high or high are likely sources of significant future threats and may impair recovery.

Severity of damage is defined as the level of damage to the conservation target that can reasonably be expected within ten years under current circumstances (*i.e*., given the continuation of the existing situation). Severity is ranked from low to very high according to the following criteria:

Very High	The stress is likely to destroy or eliminate the conservation target over some portion of the target's occurrence at the site.
High	The stress is likely to seriously degrade the conservation target over some portion of the target's occurrence at the site.
Medium	The stress is likely to moderately degrade the conservation target over some portion of the target's occurrence at the site.
Low	The stress is likely to only slightly impair the conservation target over some portion of the target's occurrence at the site.

Scope of damage is defined as the geographic scope of impact on the conservation target at the site that can reasonably be expected within 10 years under current circumstances (*i.e*., given the continuation of the existing situation). Scope is ranked from low to very high according to the following criteria:

Very High	The stress is likely to be very widespread or pervasive in its scope, and affect the conservation target throughout the target's occurrences the site.
High	The stress is likely to be widespread in its scope, and affect the conservation target at many of its locations at the site.
Medium	The stress is likely to be localized in its scope, and affect the conservation target at some of the target's locations at the site.

Low	The stress is likely to be very localized in its scope, and affect the conservation target at a limited portion of the target's location at the site.

Fifteen stresses were identified and evaluated for specific conservation targets (life stages):

1. Altered Riparian Species Composition & Structure;
2. Altered Sediment Transport: Road Condition & Density;
3. Estuary: Impaired Quality & Extent;
4. Floodplain Connectivity: Impaired Quality & Extent;
5. Hydrology: Gravel Scouring Events;
6. Hydrology: Impaired Water Flow;
7. Impaired Passage & Migration;
8. Impaired Watershed Hydrology;
9. Instream Habitat Complexity: Altered Pool Complexity and/or Pool/Riffle Ratios;
10. Instream Habitat Complexity: Reduced Large Wood and/or Shelter;
11. Instream Substrate/Food Productivity: Impaired Gravel Quality & Quantity;
12. Landscape Disturbance;
13. Reduced Density, Abundance & Diversity;
14. Water Quality: Impaired Instream Temperatures; and
15. Water Quality: Increased Turbidity or Toxicity.

Stresses with a high level of severity and/or broad geographic scope are ranked as high or very high. For example, in Table 10, the stress of hydrology – impaired water flow was ranked as very high for impacts to the summer rearing life stage. This stress also ranked as high for smolts, because in low water years, flows are inadequate for out-migration. This stress was ranked medium for adults and eggs, indicating it was not as severe and/or more limited in scope and, therefore, not as detrimental to those life stages, because flows during adult migratory and egg development periods are typically adequate. Stresses to the population are compiled in a summary table to describe major stresses for each population by target life stage (Table 10).

Table 10. CAP stress summary table for Soquel Creek population.

Stress Matrix

Central California Coast Coho Salmon ~ Soquel Creek

	Stresses (Altered Key Ecological Attributes) Across Targets	Adults	Eggs	Summer Rearing Juveniles	Winter Rearing Juveniles	Smolts	Watershed Processes
		1	2	3	4	5	6
1	Reduced Density, Abundance & Diversity	Very High		Very High		Very High	
2	Instream Habitat Complexity: Reduced Large Wood and/or Shelter	High		Very High	High	Very High	
3	Hydrology: Impaired Water Flow	Medium	Medium	Very High		High	
4	Instream Substrate/Food Productivity: Impaired Gravel Quality & Quantity	Low	High	Medium	High		
5	Instream Habitat Complexity: Altered Pool Complexity and/or Pool/Riffle Ratios	High		Medium	High		
6	Floodplain Connectivity: Impaired Quality & Extent	Medium			High		
7	Water Quality: Impaired Instream Temperatures			High		Low	
8	Altered Sediment Transport: Road Condition & Density						High
9	Hydrology: Gravel Scouring Events		High				
10	Impaired Watershed Hydrology						High
11	Water Quality: Increased Turbidity or Toxicity	Medium		Medium	Medium	Medium	
12	Impaired Passage & Migration	Medium		Medium	Low	Low	
13	Estuary: Impaired Quality & Extent			Medium		Medium	
14	Landscape Disturbance						Medium
15	Altered Riparian Species Composition & Structure			Low			Low

Assessing Future Conditions: Sources of Stress (Threats)

Threats are termed the "sources of stress," and are defined as the "proximate activities or processes that have caused, are causing or may cause the stress" (TNC 2007). NMFS used the CAP common threat taxonomy as a basis to define the principal factors most relevant to the recovery of CCC coho salmon. CAP defines direct threats to the species as the sources of stress likely to limit viability into the future. Threats may result from currently active actions s such as ongoing land uses, or from actions likely to occur in the future (usually within ten years), such as increased water diversion or development. Threats contribute to stresses in ways likely to impair salmonid habitat into the future. Many threats are driven by human activities, however, naturally occurring events such as severe weather events may also threaten the species. For each population and life stage, threats were ranked using two metrics, contribution and irreversibility, which are combined by CAP algorithms to generate a single rank for each threat identified.

Contribution is defined as the expected contribution of the source of stress, acting alone, to the full expression of a stress under current circumstances (*i.e.*, given the continuation of the existing management/conservation situation). Threats ranked as very high for contribution are very large contributors to the particular stress and low ranks are applied to threats that contribute little to the particular stress. Contribution is ranked from low to very high according to the following criteria:

Very High	The source is a very large contributor of the particular stress.
High	The source is a large contributor of the particular stress.
Medium	The source is a moderate contributor of the particular stress.
Low	The source is a low contributor of the particular stress.

Irreversibility is defined as the degree to which the effects of a threat can be reversed. Irreversibility is ranked from low to very high according to the following criteria:

Very High	The source produces a stress that is not reversible, for all intents and purposes (*e.g.*, wetland converted to shopping center).
High	The source produces a stress that is reversible, but not practically affordable (*e.g.*, wetland converted to a agriculture).
Medium	The source produces a stress that is reversible with a reasonable commitment of additional resources (*e.g.*, ditching and draining of wetland).
Low	The source produces a stress that is easily reversible at relatively low cost (*e.g.*, ORVs trespassing in wetland).

Threats with a high level of contribution to a stress and/or high irreversibility are ranked as high or very high. For example, in Table 11 the threat of residential and commercial development was ranked as very high for its effects to two life stages, and high for three others, because residential development is a very high contributor to poor water quality and impaired riparian conditions in Soquel Creek (as an example).

The threat of development is also essentially irreversible. Summary tables of threats ranked for each population describe major threats for each target life stage (Table 11). The overall threat rank (last column) summarizes the aggregate threat rating and thereby identifies the most limiting threats to a population.

The threat status for each target (last row) summarizes the aggregate ranks applied across all life stages and illustrates the targets that are most vulnerable. Threats ranked as high or very high are more likely to contribute to a stress that in turn, reduces the viability of a target life stage. When multiple life stages of a population had high or very high threats, the viability of the population was diminished.

Table 11. CAP threat summary table for Soquel Creek population.

Summary of Threats
Central California Coast Coho Salmon ~ Soquel Creek

	Threats Across Targets	Adults	Eggs	Summer Rearing Juveniles	Winter Rearing Juveniles	Smolts	Watershed Processes	Overall Threat Rank
	Project-specific threats	1	2	3	4	5	6	
1	Residential and Commercial Development	High	Medium	Very High	High	Very High	High	Very High
2	Water Diversion and Impoundments	Medium	Medium	Very High	Medium	Very High	High	Very High
3	Severe Weather Patterns	Medium	High	Very High	High	High	High	Very High
4	Roads and Railroads	High	High	High	High	High	High	Very High
5	Fire, Fuel Management and Fire Suppression	Medium	Medium	High	Medium	High	Medium	High
6	Logging and Wood Harvesting	Medium	Medium	High	Medium	High	Medium	High
7	Channel Modification	Medium	Medium	High	High	Medium	Low	High
8	Fishing and Collecting	High	-	Medium	-	High	-	High
9	Mining	Medium	Medium	Medium	Medium	Medium	Medium	Medium
10	Agriculture	Medium	Medium	Medium	Medium	Medium	Low	Medium
11	Disease, Predation and Competition	Medium	-	Medium	Low	Medium	Low	Medium
12	Recreational Areas and Activities	Low	Low	Medium	Low	Medium	Low	Medium
13	Livestock Farming and Ranching	Low	Low	Low	Low	Medium	Low	Low
14	Hatcheries and Aquaculture	-	-	-	-	-	-	-
	Threat Status for Targets and Project	High	High	Very High	High	Very High	High	Very High

Threats evaluate future impediments likely to adversely affect recovery for each targeted salmonid population. The list of threats is based on their known impact to salmonid habitat, species viability, and the likelihood that the threat would continue into the future. Using the CAP common threat taxonomy as a basis, the following fourteen threats were evaluated in relation to each stress for a specific life stage:

1. Agriculture;
2. Channel Modification;
3. Disease/Predation/Competition;
4. Fire, Fuel Management and Fire Suppression;
5. Fishing/Collecting;
6. Hatcheries;
7. Livestock Farming and Ranching;
8. Logging and Wood Harvesting;
9. Mining;
10. Recreational Areas and Activities;
11. Residential and Commercial Development;
12. Roads and Railroads;
13. Severe Weather Patterns; and
14. Water Diversion and Impoundments.

Some threats occurred in all or most populations (*e.g.*, roads), while others were more limited in distribution (*e.g.*, mining). Where a threat did not occur in a given population, it was not evaluated and did not receive a rating. A matrix was developed illustrating which threats contribute to a particular stress (Table 12). This ensured a direct linkage between the threat and a particular stress. For example, the threat of fishing and collecting was only ranked against the population stress of reduced abundance, diversity, and competition. This approach reduced the potential for over estimating the effect of a stress across multiple threats. In this example, the threats of agriculture, livestock and recreation were not ranked against the stress of hydrology - impaired water flow. While these threats may contribute to impaired water flow, all impairments to water flow were evaluated only under the threat of water diversion and impoundments. Finally, the matrix facilitated the development of recovery actions with direct relationships to stresses or threats.

Very high or high threats are driven by social, economic, or political causes that then become the focus of conservation strategies. Conservation strategies are developed into recovery actions intended to reduce or abate the high or very high threats. In some cases recovery actions were developed for medium ranked threats based on knowledge or information that the threat could increase in the near future due to anticipated changes. The following section describes each threat and the information considered for ranking each major threat to CCC coho salmon recovery.

Table 12. Matrix showing which threats were evaluated against which stresses.

Stresses	Habitat Condition											Watershed Processes			Population
Threats	Estuary: Impaired Quality & Extent	Floodplain Connectivity: Impaired Quality & Exent	Hydrology: Gravel Scouring Events	Hydrology: Impaired Water Flow	Instream Habitat Complexity: Altered Pool	Instream Habitat Complexity: Reduced Large Wood	Instream Substrate/ Food Productivity: Impaired	Impaired Passage & Migration	Water Quality: Increased Turbidity or Toxicity	Water Quality: Impaired Instream Temperatures	Altered Riparian Species Composition & Structure	Impaired Watershed Hydrology	Landscape Distrubance	Altered Sediment Transport: Road Construction	Reduced Density, Abudance & Diversity
Agriculture				N/A											N/A
Channel Modification															N/A
Disease/Predation/ Competition(Invasive Animals and Plants)			N/A	N/A			N/A								
Fire				N/A											N/A
Fishing/Collecting	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	
Hatcheries	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A			N/A	N/A	N/A	N/A	
Livestock				N/A											N/A
Logging				N/A											N/A
Mining				N/A											N/A
Recreation				N/A											N/A
Residential Development				N/A											N/A
Roads				N/A											N/A
Severe Weather Patterns															N/A
Water Diversion and Impoundments															

Threat: Agriculture

Agriculture was defined as annual and perennial crop farming and associated operations and, for recovery planning analysis purposes, excludes grazing, ranching or timber harvest.

Impacts to Salmonids: Agricultural practices can adversely affect salmonid habitat by altering riparian vegetation and natural drainage patterns, introducing water-borne pollutants, and increasing the likelihood of channel simplification, and chronic input of fine sediment.

Application to the ESU: The major agricultural practices within the CCC coho salmon ESU are vineyards and orchards (apples and pears), generally located north of San Francisco Bay. Brussel sprouts, lettuce, and flower crops (greenhouse and row crops) are grown in the southern areas of the ESU.

Threat Context: Some agricultural activities and programs have made strides in improving riparian protections, implementing pollution and sediment discharge controls, and promoting instream habitat restoration (*e.g.*, Fish Friendly Farming, Code of Sustainable Winegrowing Practices, TMDL's and others). However, the overall impact to coho salmon and their habitat is generally vary substantial where these activities occur, and particular aspects of agriculture can have major direct and indirect impacts (*e.g.*, use of plethoris to control gypsy moth and removal of riparian vegetation from farming areas due to perceived threats regarding *e-coli* from wild animals).

Threats Evaluated and Ranked: The analysis included all practices and operations associated with agriculture, including land conversions, continuous or seasonal ground disturbances, maintenance, planting, harvesting, and fertilizing of row crops, orchards, vineyards, commercial greenhouses, nurseries, gardens, *etc.*

Threats were evaluated for their potential to:

1. Introduce water-borne pollutants, such as sediment and pesticides, into the aquatic environment, or adversely alter nutrient levels;
2. Alter riparian vegetation integrity, diversity, function, and composition;
3. Alter natural drainage channels and hydrology patterns; and
4. Simplify channel complexity and destabilize stream banks.

The final threat rankings were determined by the following:

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered. High or very high threats could include practices requiring large areas in cultivation and large quantities of pesticides and herbicides over significant proportions of the watershed.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered, but the effects could be reversed or ameliorated.

Low threat ranking results when ecosystem function and process are (or are expected to be) largely intact, slightly altered, and easily reversible. A low threat could include practices that have a low impact and use little or no herbicides and pesticides in the watershed and do not impact riparian vegetation.

Resources Utilized: GIS analysis of the total acres, and percentage of a watershed under cultivation, watershed specific assessments, NMFS staff knowledge of watersheds, and ongoing practices, *etc*.

Threat: Channel Modification

Channel modification was defined as directly and/or indirectly modifying and/or degrading natural channel forming processes and morphology of perennial, intermittent and ephemeral streams and estuarine habitats.

Impacts to Salmonids: Channel modifying structures such as rip rap and gabions reduce the occurrence and creation of undercut banks and side channels, limit or eliminate large woody debris (LWD) recruitment, and often result in the removal of riparian vegetation. These techniques are used extensively to line channel banks and beds. Bank stabilization structures eliminate or severely reduce streambed gravel recruitment necessary for salmonid spawning and macroinvertebrate habitat. Bank stabilization, levee construction for flood control, and filling in floodplains for land reclamation also disconnect rivers and streams from their floodplains. These activities prevent the creation of, or block access to, off-channel habitat used by salmonids as refuge from high stream flows, and impede stream geomorphic processes.

Application to the ESU: In the process of protecting public and private infrastructure and property, channel modification has reduced salmonid habitat suitability by permanently altering natural channel forming processes, particularly in the many urbanized watersheds within the CCC coho salmon ESU.

Threat Context: Permits from the U.S. Army Corps of Engineers (Corps) are required for most channel modifications. Issuance of a permit to alter streams (including channelization, removal of LWD, and placement of rock slope protection, *etc*.) utilized by listed salmonids requires an Endangered Species Act (ESA) Section 7 consultation with NMFS. Once channel modifying infrastructure is in place it is usually followed by increased development, which in turns leads to additional channel modification. For example, bank armoring at one site can cause erosion downstream, resulting in sequential armoring of a stream reach. Once infrastructure is in place it is often impractical, difficult, and expensive to remove. With a growing human population the pressure to modify natural stream channels is expected to continue.

Threats Evaluated and Ranked: The analysis included evaluation of estuarine management (*e.g*., lagoon breeching, dredging), flood control activities, large woody debris removal, levee construction, vegetation removal, herbicide application, stream channelization, bank stabilization (hardening that limits channel movement or meander), dredging and other forms of sediment removal. These actions typically occur within the two-year bankfull stage and adversely affect channel forming processes.

Threats were evaluated for their potential to:

1. Damage instream and near stream habitat and lower habitat complexity;

2. Precipitate riparian habitat loss, decrease channel roughness (decrease in Manning's N roughness coefficient);
3. Alter drainage channels and hydrologic patterns;
4. Alter riparian zone diversity, function, and composition;
5. Alter channel and stream bank stability;
6. Alter or destroy floodplain, estuarine, and wetland habitats;
7. Introduce water-borne pollutants, such as sediment and chemicals, into the aquatic environment, or adversely alter nutrient levels; and
8. Simplify channel morphology (*e.g.*, by increasing incision rate and decreasing floodplain connectivity).

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered. High or very high threats could include large levee projects within salmonid habitat that adversely modify sediment transport, impair salmonid migration, accelerate stream velocities, and alter riparian vegetation structure from historical conditions.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered but could be reversed or ameliorated.

Low threat ranking results when ecosystem function and process are (or are expected to be) largely intact, slightly altered, and easily reversible. A lower threat could include bank stabilization projects that use bioengineering techniques.

Resources Utilized: No central repository of channel modifying activities exists for watercourses in the CCC coho salmon ESU, and the quality and quantity of information varies significantly between watersheds. Information sources included watershed assessments, CDFG habitat typing information, personal communications with local experts, and staff knowledge of individual watersheds.

Threat: Disease, Predation and Competition

Disease, predation and competition includes diseases having, or predicted to have, significant harmful effects on salmonids and/or their habitat, as well as native (*e.g.*, sea lions, mergansers, *etc*.) and non-native predator species (*e.g.*, large mouth or striped bass). It also includes invasive non-native plants (*e.g.*, *Arundo donax*) that degrade riparian or aquatic habitats.

Impacts to Salmonids: Infectious disease can influence adult and juvenile coho salmon survival. Salmonids are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment. Specific diseases such as bacterial kidney disease, *ceratomyxosis*, *columnaris*, *furunculosis*, infectious hematopoietic necrosis virus, redmouth and black spot disease, erythrocytic inclusion body syndrome, and whirling disease, among others, are present and are known to affect coho salmon (Rucker *et al.* 1953; Wood 1979; Leek 1987; Foott *et al.* 1994). Diseases such as bacterial kidney disease have been identified as a limiting factor in some populations (*e.g.*, Noyo River), particularly those subject to artificial propagation.

Piscivorous predators may also affect the abundance and survival of salmonids. Cooper and Johnson (1992) and Botkin *et al*., (1995) reported marine mammal and avian predation may occur on some local salmonid populations, but it was a minor factor in the decline of coast wide salmonid populations. However, Moyle (2002), found that when fish populations are low, predation by seals and sea lions on returning spawners may prevent recovery. Predation by marine mammals (primarily harbor seals and California sea lions) is of concern in some areas experiencing dwindling run sizes of salmon. Predation by non-native striped bass (*Morone saxatilis*) may also impact some coho salmon populations. Although predation does occur from a number of sources, it is believed to be a minor factor in the overall decline of coastwide salmonid populations but may play a significant role in keeping small populations from increasing.

Principal competitors for the food and space of juvenile coho salmon are other salmonids, especially Chinook salmon and steelhead (Moyle 2002), both of which are listed species within the range of CCC coho salmon. Other sources of competition include invasive non-native riparian plant species (*e.g., Arundo donax*) which can completely disrupt riparian communities.

Application to the ESU: Disease, predation and competition may significantly influence salmonid abundance in some local populations when other prey species are absent and physical conditions lead to the concentration of salmonid adults and juveniles (Cooper and Johnson 1992). Also, altered stream flows can create unnatural riverine conditions that favor non-native species life histories over the native cold water species (Brown et al. 1994; California Department of Fish and Game 1994; McEwan and Jackson 1996; National Marine Fisheries Service 1996a).

Threat Context: Relative to other threats, disease and predation are not major factors contributing to the overall decline of coho salmon in the CCC ESU. However, they may compromise the ability of depressed populations to rebound. Competition in the context of habitat alteration leading to reduced survival is a serious limiting factor in some streams in the ESU.

Threats Evaluated and Ranked: The following threats were evaluated and ranked: introduction of non-native animal species that prey upon and/or (directly or indirectly) compete with native salmonids; introduction of non-native vegetation that competes with and/or replaces native vegetation; and creation of conditions favorable to increased populations and/or concentration of native predators.

Threats were evaluated for their potential to:

1. Simplify or modify instream or riparian habitat condition;
2. Reduce feeding opportunities;
3. Shift the natural balance between native/non-native biotic communities and salmonid abundance, resulting in disproportional predation and competition;
4. Increase opportunities for infectious disease;
5. Change water chemistry (*e.g*., inputs of acidic detritus from *Eucalyptus*, or low dissolved oxygen (DO) resulting from increased foreign biomass) and,
6. Impede instream movement and migration, or reduce riparian function (*e.g., Arundo donax*).

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered, or impacts to the population are severe. High or very high threats occur when amelioration of the consequences of this threat are largely irreversible.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered, but the effects could be reversed or ameliorated, or impacts to the population are moderate. Medium threats occur when the consequences of this threat are largely irreversible but could be ameliorated.

Low threat ranking results when ecosystem function and process are (or are expected to be) largely intact, slightly altered, and easily reversible

Resources Utilized: NMFS used a variety of resources to evaluate this threat, from region wide assessments of the impacts of predation to site specific watershed assessments and individual reports. In general, there was little site specific information to evaluate this threat, and in many cases NMFS staff solicited the opinions of local experts as well as utilizing best professional judgment after considering information on pinniped and bird predation and competition and predation by non-native species.

Threat: Fire and Fuel Management

Threats include fires (wildfires and prescriptive burns) and fire suppression actions (firefighting and fire prevention).

Impacts to Salmonids: Fire, particularly catastrophic wildfires, can impair salmonid habitat by reducing or eliminating riparian canopy, resulting in increased soil erosion that can render instream rearing habitat unsuitable for many decades. Hotter fires consume organic matter that binds soils, leading to an increase in erosion potential, and high intensity fires can volatilize minerals in the soil causing it to become hydrophobic. Fire retardants used in suppression may contain chemicals potentially harmful to the environment. Many retardants contain ammonia, which is toxic to fish, and its conversion products, including nitrates, increase oxygen demand in streams and stimulate algal growth. Use of water pumped directly from streams to suppress fires may degrade salmonid habitat.

Application to the ESU: The interior and southern areas of the ESU may have significant fire risk with potential for watershed disturbance and increased sediment yield. Coastal ecosystems have higher rainfall, more resilient vegetation (*e.g*., redwood forest), less extreme summer air temperatures and, therefore, less risk of catastrophic fire. Spence *et al*. (1996) recognized the extent of watershed damage and risk to salmonid habitat is directly related to burn intensity.

Threat Context: Fire management techniques such as prescriptive burns or timber thinning would not normally take place in riparian vegetation, so impacts to coho salmon are expected to be inadvertent, or resulting from severe fire conditions. Few areas within the range of CCC coho salmon are on Federal lands, so most firefighting activities are conducted by local fire districts and CalFire. Unlike federal lands, where NMFS has extensive interaction with the Forest Service to minimize adverse consequences from firefighting actions, NMFS has little interaction with local firefighting agencies in the CCC ESU. Consequently, impacts from firefighting (*e.g*., road building and construction of fire breaks, water diversion, aerial retardants) likely have considerable adverse impacts to CCC coho salmon and their habitats.

Threats Evaluated and Ranked: Construction of fire breaks, roads, application of fire retardants, water use planning, fuels management, and fire suppression.

Threats were evaluated for their potential to:

1. Increase erosion, sedimentation and landslide potential;
2. Elevate fuel loading leading to a higher potential of catastrophic burns;
3. Impair future large woody debris recruitment; and
4. Alter vegetative/riparian communities through invasive species/post-fire management.

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered. High threats may include high fuel loading over a large area, or extensive burns upstream of, or adjacent to, critical spawning and rearing areas.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered, but the effects could be reversed or ameliorated.

Low threat ranking results when ecosystem function and process are (or are expected to be) largely intact, slightly altered, and easily reversible. A mature redwood forest upstream or adjacent to salmonid habitat generally will rank as a low threat due to the fire resistant qualities of redwood.

Resources Used: The current prediction for regional effects from fire intensity, frequency and duration as well as fire and fuel management practices (fire suppression, prescribed burning and limited use of mechanical treatments to reduce fire fuel loads) were examined.

Threat: Fishing and Collecting

This threat includes harvesting salmonids for recreation, subsistence, in-situ research, or cultural purposes, and includes illegal and legal activities such as accidental mortality/bycatch.

Impacts to Salmonids: Commercial and sport-fishing for coho salmon is closed in California due to recognition of the dramatic species declines. However, coho salmon are incidentally caught as bycatch by both commercial and sport-fishers. These activities are most likely to impact the adult lifestage. The amount of bycatch is unknown, but it may have a significant adverse effect due to the extremely low population levels, where every individual is of greater significance to the population's persistence than when the population was large. Fish deaths caused by activities such as fishing could be more damaging to the population when populations are depleted due to natural conditions (such as changes in ocean productivity) (National Research Council 1996). Handling hooked fish before releasing them also contributes to mortality (Clark and Gibbons 1991).

Application to the ESU: Moyle (2002) states that the present populations are so low that moderate fishing pressure on wild coho may prevent recovery, even in places were stream habitats are adequate. In California, coho salmon caught incidentally must be immediately released, but the act of capture comes at a cost to the individual through energetic expenditure, injury, increased susceptibility to disease, or eventual predation (*i.e*. marine mammals eating the fish before it is landed).

Threat Context: The opening of freshwater the sport-fishing season (Table 13) as early as November 1 north of San Francisco Bay[11] and December 1 south of San Francisco Bay[12], likely preferentially targets coho salmon during the early portion of fishing season as this species migrates into freshwater earlier than steelhead (Shapovalov and Taft 1954). This early start likely places adult coho salmon at greater risk of capture than if the season were setback to a later date.

Table 13. Independent (I) and dependent (D) watersheds where winter freshwater fishing for hatchery steelhead is permitted by California 2012-2013 sport-fishing regulations. Note: sport-fishing regulations include additional possession limits and additional regulations may apply.

Watershed	Season	Daily Bag Limit
Albion (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Aptos (D)	Dec 1 – Mar 7	0
Big River (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Cottaneva (D)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Garcia (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Gualala (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Navarro (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Noyo (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Pescadero (I)	Dec 1 – Mar 7	0
Russian (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Salmon (D)	Nov 1 – Mar 31	0
San Gregorio (D)	Dec 1 – Mar 7	0
San Lorenzo (I)	Dec 1 – Mar 7	0
Scott (D)	Dec 1 – Mar 7	0
Soquel (D)	Dec 1 – Mar 7	0
Ten Mile (I)	Nov 1 – Mar 31	2 hatchery trout or hatchery steelhead
Waddell (D)	Dec 1 – Mar 7	0
Walker (~I)	Nov 1 – Mar 31	0

The bag limits set forth in the 2012-2013 California Freshwater Sport Fishing Regulations are likely a source of confusion for some fishers and should be amended to reflect actual fishery conditions. Eight independent watersheds and one dependent watershed have a bag limit for both hatchery trout or hatchery steelhead, when in reality only the Russian River has hatchery trout or steelhead plantings. The current stated bag limits may encourage fishers to unknowingly target specific streams where no stocking occurs and in turn, incidentally hook coho salmon.

Commercial and ocean sport-fishing near the mouths of a watershed when sandbars remain closed may inadvertently result in increased rates of adult coho salmon capture. Adult coho

[11] Minimum flow requirements (based on a minimum of 500 cfs at the gauging station on the mainstem Russian River near Guerneville (Sonoma County) and 15 cfs at the gauging station at the Oak Knoll Bridge on the mainstem Napa River (Napa County))

[12] Minimum flow requirements are determined (based on an undefined flow at the Big Sur and Carmel rivers in Monterey County) by DFG.

salmon congregating offshore while awaiting entry into the estuaries are likely at more risk of capture than those returning to watersheds without sandbars, or where sandbars have breached.

Most streams in the ESU do not have minimum flow requirements, which has resulted in some sport-fishing in streams at extremely low flows early in the season when coho are likely present. This may also result in increased risk to adults.

Threats Evaluated and Ranked: Incidental harvest for recreation and subsistence, authorized relocation, research and collection, incidental capture (*e.g*., hooking), and illegal activities such as poaching and unpermitted collection.

Threats were evaluated for their potential to:

1. Increase mortality/harm and displacement;
2. Increase competition when fish are relocated; and
3. Precipitate dispensatory effects at the population level.

High or very high threat rankings results when impacts to the population are (or are expected to be) severe. High or very high threats may occur in critical adult staging areas with extensive legal and illegal fishing pressure.

Medium threat ranking results when impacts to the population are (or are expected to be) moderate but could be reversed or ameliorated.

Low threat ranking results when impacts to the population are (or are expected to be) low and easily reversible. Low threat may occur in watersheds under large private (*i.e.,* commercial timberlands) ownership where public access is restricted or in areas with significant enforcement presence.

Resources Used: Recreational steelhead angling was the main activity considered for this indicator rating because it is the type of fishing most likely to impact adult salmonids. We ranked the impact of fishing and collecting by tallying the number of fishing trips reported in the CDFG Steelhead Fishing Report and Restoration Card during each species' adult migration period for the most recent year of record when available.

Threat: Hatcheries

Hatcheries are artificial propagation facilities designed to produce fish for harvest, or for escaping harvest to spawn. A conservation hatchery differs from a production hatchery since it specifically tries to supplement or restore naturally spawning salmon populations. Artificial propagation, especially the use of production hatcheries, has been a prominent feature of Pacific salmon fisheries enhancement efforts for several decades.

Impacts to Salmonids: Hatchery operations can affect salmonids in a number of ways, including adverse effects to the species through changes in their genetics, ecological and behavioral patterns, harvest rates (overfishing) and disease.

<u>Genetic Risks</u>

Genes determine the characteristics of living things. Human intervention in the rearing of wild animals has the potential to cause genetic change. These genetic changes impact salmon diversity and the health of salmon populations. Hatchery programs vary and therefore the risks identified below vary by hatchery. Genetic risks of artificial propagation to wild populations include:

1. *Inbreeding* - Inbreeding can occur when the population for a hatchery comes from a small percentage of the total wild and/or hatchery fish stock (*e.g*., 100 adults are used as broodstock out of a population of 1 million). If only a small number of individuals are used to create the new hatchery stock, genetic diversity within a population can be reduced. Inbreeding can affect the survival, growth and reproduction of salmon;
2. *Intentional or artificial selection for a desired trait (such as growth rate or adult body size)* - Although not common practice today, some hatchery programs intentionally select for larger fish (or other specific traits). This selection changes the genetic makeup of the hatchery stock, moving it further away from naturally reproducing salmon stocks;
3. *Selection resulting from nonrandom sampling of broodstock* - The makeup of a hatchery population comes from a selection of wild salmon and/or returning hatchery salmon that are taken into captivity (*i.e*., broodstock). If, for example, only early-returning adults are used as broodstock, instead of adults that are representative of the population as a whole (*i.e*., early, normal, and late-returning adults), there will be genetic selection for salmon that return early;
4. *Unintentional or natural selection that occurs in the hatchery environment* - Conditions in hatchery facilities differ greatly from those in natural environments. Hatcheries typically rear fish in vessels (*i.e*., circular tanks and production raceways) that are open and have lower and more constant water flow than occurs in natural streams and rivers. They also tend to hold fish at much higher densities than occurs in nature. This type of environment has the potential to alter selection pressures in favor of fish that best survive in hatchery rather than natural environments; and
5. *Temporary relaxation during the culture phase of selection that otherwise would occur in the wild* - Artificial mating disrupts natural patterns of sexual selection. In hatcheries, humans select the adult males and females to mate, not the salmon. Humans have no way of knowing which fish would make the best natural breeders. In addition, selection pressures that would normally be encountered in the wild, such as predation and foraging challenges, are relaxed until the time when juveniles are released from the hatchery. Fish raised in hatchery environments face very different pressures than those raised in the wild.

Ecological and Behavioral Risks

Hatchery-produced fish often differ from wild fish in their behavior, appearance, and/or physiology. Ecological risks of artificial propagation on wild populations include:

1. *Competition for food and territory* - Competition between wild and hatchery fish can occur. It is most likely to occur if the fish are of the same species (*e.g.,* between wild Chinook salmon and hatchery reared Chinook salmon), and if they share the same habitat (quiet, shallow water or deep fast water) and diet;
2. *Predation by larger hatchery fish* **-** If hatchery released salmon are larger than wild salmon, evidence suggests that, for certain species, hatchery released salmon can feed on wild salmon;

3. *Negative Social Interactions* - Juvenile salmon establish and defend foraging territories through aggressive contests. When large numbers of hatchery fish are released in streams where there are small numbers of wild fish, hatchery fish are more likely to be more aggressive, and disrupt natural social interactions;
4. *Carrying Capacity Issues* - Carrying capacity is a measure of the maximum population (*e.g.,* numbers of salmon) supported by a particular ecosystem. Carrying capacity changes over time with varying predator abundance and resources such as food and habitat. When hatchery fish are released into streams where there are wild fish, competition for food and space can arise. Many streams and watersheds are degraded due to contamination, development, *etc*., and have a reduced carrying capacity; and
5. *Behavioral* - Hatchery environments are different than stream environments. Hatcheries typically rear fish in vessels (*i.e*., circular tanks and production raceways) that produce sterile environments where there are no complex habitat features (*i.e*., sticks and wood), little or no overhead cover (such as cover from nearby trees and undercut stream banks), and a predictable food supply. Consequently, hatchery fish tend to have different foraging, social, and predator-avoidance behavior.

<u>Overfishing</u>
Large-scale releases of hatchery fish have supported commercial, Tribal, and sport fishing practices for many years. However, large-scale releases of hatchery fish in a mixed population fishery creates a risk of overfishing for wild populations. Because hatchery populations are typically abundant and have high survival rates, they can generally support higher harvest rates. Wild stocks, on the other hand, are typically less abundant, and their populations could be harmed by high harvest rates. NMFS and CDFG fisheries managers are currently evaluating opportunities to support selective harvest of hatchery fish (*i.e*., harvest that doesn't impact wild stocks). Selective harvest opportunities could be supported through catch and release programs and/or in places where hatchery stocks are isolated from wild stocks (*i.e*., where hatchery stocks use a different stream or enter the stream at a different time than wild stocks).

<u>Fish Health</u>
The effect of disease on hatchery fish and their interaction with wild fish is not well understood. However, hatcheries can have disease outbreaks, and once diseased fish are released, they can transmit disease to wild fish.

Application to the ESU: Historically, out of basin and out-of-ESU hatchery coho salmon were released in many watersheds in the ESU. Some fish originated from Baker Lake in Washington State in the early part of the last century and, until recently, coho salmon from the Noyo River Egg Collecting Station (ECS) were outplanted in many watersheds in the ESU. Most of the hatcheries in the ESU were smaller than the production hatcheries in other parts of California but the long history of outplanting has likely adversely affected genetic diversity of coho salmon in the ESU to some degree. Disease, particularly bacterial kidney disease, has been a source of concern in regards to the Noyo ECS (now closed). In addition, excluding grilse from the Noyo ECS spawning program may have decreased genetic diversity of the Noyo population.

Threat Context: Two hatcheries are currently operating in the ESU: the Corps' Don Clauson Hatchery at Warm Springs Dam in the Russian River watershed, and the King Fisher Flat facility on Scott Creek operated by Monterey Bay Salmon and Trout Project. Both facilities are operated as conservation hatcheries, and receive considerable oversight from NMFS and CDFG. Conservation hatcheries are not operated for maximum production but are operated with the goal of ensuring genetic integrity of the target population. See Spence *et al*. (2008) for additional information.

Threats Evaluated and Ranked: High or very high threat rankings result when impacts to the population are (or are expected to be) severe. High or very high threats may include a facility operated for the purpose of maximum production with no consideration for genetic impacts to the population.

Medium threat ranking results when impacts to the population are (or are expected to be) moderate but could be reversed or ameliorated. Medium threats might include a facility operated with minimal regulatory oversight or that takes a significant proportion of a spawning run but attempts to minimize genetic impacts.

Low threat ranking results when impacts to the population are (or are expected to be) low and easily reversible. An example of low threat would include a conservation broodstock facility operated with significant oversight by regulatory agencies and with backup rearing facilities.

Resources Used: Sources of information included, personal communications with local experts, hatchery managers, and NMFS and CDFG staff knowledgeable with the operations of the two existing broodstock facilities.

Threat: Livestock Farming and Ranching

This treat is considered as domestic terrestrial animals raised in one location, or domestic or semi-domesticated animals allowed to roam in the wild and supported by natural habitats (*e.g*., cattle feed lots, chicken farms, dairy farms, and cattle ranching).

Impacts to Salmonids: Livestock grazing is the most widespread land-management practice in the western North America, occurring over 70 percent of the western United States (Noss and Cooperrider cited in Donahue 1999). The impacts of livestock grazing in riparian areas have been widely studied. Direct effects include elevated levels of fecal coliform bacteria and sediment in streams, degraded stream banks and bottoms, altered channel morphology from livestock trampling, lowered ground water tables and reduced streamside vegetation leading to a deterioration of fish habitat (Duff *et al*. 1980; Armour et al. 1991; Kovalchik and Elmore 1992; Overton *et al*. 1994; Belsky et al. 1999; Donahue 1999).

Animal waste carried by runoff can contaminate water sources through the addition of oxygen-depleting organic matter (Knutson and Naef 1997). Runoff from concentrated fecal sources can degrade water quality, causing lethal conditions for fish. As the biochemical oxygen demand increases, dissolved oxygen within the water column decreases and ammonia is released, creating water quality conditions stressful to fish.

Application to the ESU: Behnke and Zarn (1976) and Armour *et al*., (1991) indicated that overgrazing is one of the major contributing factors in the decline of Pacific Northwest salmon. George *et al*., (2002) found that cattle trails in California produced 40-times more sediment than adjacent vegetated soil surfaces. In the CCC ESU, the adverse impacts from cattle grazing are believed to be less problematic than other areas of California, because it is limited in extent. Point source impacts from livestock facilities have impacts in some watersheds in the ESU.

Threat Context: To address potential environmental impacts of livestock operations, several programs have been developed. These programs assist landowners in developing best management practices for their respective land use. These include the Rangeland Water Quality Short-course, and the Dairy Quality Assurance Program. Livestock grazing and ranching is generally concentrated in just a few of the watersheds targeted for coho recovery.

Threats Evaluated and Ranked: NMFS evaluated grazing intensity and seasonality, stockyard proximity to the stream channel, damage to riparian zones, water quality impacts resulting from animal waste, and increased erosion.

Threats were evaluated for their potential to:

1. Elevate the concentration of water-borne pollutants such as sediment, toxic chemicals/substances (*i.e*., hormones), and nutrient levels;
2. Alter riparian zone diversity, function, and composition;
3. Alter drainage channels and hydrology (soil compaction); and
4. Simplify channel structure and alter stream bank stability.

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered but could be reversed or ameliorated.

Low threat ranking results when ecosystem function and process are largely intact, (or are expected to be) slightly altered, and easily reversible.

Resources Utilized: The quality and quantity of information varied significantly between watersheds. Sources of information included watershed assessments, CDFG stream survey notes, personal communications with local experts, and NMFS staff knowledge of individual watersheds.

Threat: Logging and Wood Harvesting

This threat includes the harvesting of trees and ancillary post-harvest effects of these activities; including changes to hydrologic patterns and increased contribution of water-borne pollutants, such as sediment and elevated nutrient levels. Additionally, this threat includes conversion of timberland (to vineyards, rural residential development, or other uses).

Impacts to Salmonids: Many watersheds in the CCC coho salmon ESU are heavily forested, and timber harvest is a major threat to coho salmon habitat. Spence *et al*., (1996) summarized the major effects of timber harvest on salmonids as follows: "Riparian logging depletes LWD, changes nutrient cycling and disrupts the stream channel. Loss of LWD, combined with alteration of hydrology and sediment transport, reduces complexity of stream micro- and macrohabitats and causes loss of pools and channel sinuosity. These alterations may persist for decades or centuries. Changes in habitat conditions may affect fish assemblages and diversity."

Spence *et al*., (1996) cited studies by McCammon (1993) and Satterland and Adams (1992) showing increased peak flows resulting from alteration of 15-30% of a watershed's vegetation, and concluded "that no more than 15-20% of a watershed should be in a hydrologically immature state at any given time." In many streams, reduced LWD as a result of past forestry practices has resulted in decreased cover and reduced gravel and organic debris storage. Reduced LWD has also decreased pool habitat volume and reduced overall hydraulic complexity (CDFG 2004). LWD also provides cover from predators and shelter from turbulent high flows. Heavy rainfall occurring after timber harvest operations can increase stream bank erosion, landslides, and mass wasting, resulting in higher sedimentation rates than historical amounts. This can reduce food supply, increase fine sediment concentrations which can reduce the quality of spawning gravels, and increase the severity of peak flows during heavy precipitation. Removing vegetative canopy cover increases solar radiation on the aquatic surface, which can increased water temperatures (Spence *et al.* 1996).

Application to the ESU: Timber harvest on non-federal land in California is regulated by the Z'berg-Nejedly Forest Practice Act of 1973 (Section 4511 of the Public Resources Code). NMFS believes that the current regulations are a qualitative improvement over historical practices; unfortunately, their effectiveness in protecting watershed processes that support salmonids has never been established (Dunne *et al.* 2001). The specific inadequacies of the Rules have been well-described by State organized committees, State and federal agencies and scientists(LSA Associates Inc. 1990; Little Hoover Commission 1994; CDFG 1995; CDF 1995; NMFS 1998a; Ligon et al. 1999; Dunne et al. 2001). Additionally, some timber harvest practices authorized in the ESU by CalFire (conversion) have been proven by NMFS Office of Law Enforcement to result in take of listed salmonids.

Threat Context:
Substantial timber harvesting has occurred in this ESU. Privately held forestlands currently support many of the remaining populations of CCC coho salmon, and the species is provided greater protection on forestlands than landscape subject to most other land use practices. The regulatory infrastructure and oversight represents an opportunity to meet recovery goals. NMFS analysis of this treat assumed that forest practices are being implemented at the minimum standard of the California Forest Practice Rules (CFPR).

Threats Evaluated and Ranked:
All operations associated with timber removal within the harvest unit, including skid trails, new road construction, opening of old road systems, and construction of landings and yarding corridors (does not include mainline transportation systems). Maintenance of road networks and erosion control devices following completion of harvest activities are also included.
Threats were evaluated for their potential to:

1. Introduce water-borne pollutants, such as sediment and toxic chemicals, into the aquatic environment, and adversely alter nutrient levels;
2. Alter riparian zone integrity, diversity, function (*i.e.*, LWD recruitment), and composition;
3. Alter drainage channels and hydrology;

4. Simplify channel complexity and lower stream bank stability; and
5. Compromise hillslope stability.

High or very high threat rankings results when (1) ecosystem function and process are (or are expected to be) severely altered or (2) impacts to the population are severe. High or very high threats occur when amelioration of the consequences of this threat are largely irreversible; or include activities that result in a permanent change to the landscape (*e.g*., conversion to agriculture, urban, or other uses or results in long-lived changes to vegetative communities).

Medium threat ranking results when (1) ecosystem function and process are (or are expected to be) moderately altered or (2) impacts to the population are moderate. Medium threats occur when the consequences of this threat are largely irreversible but could be ameliorated. Includes harvest activities meeting minimum requirements of the CFPRs.

Low threat ranking results when (1) ecosystem function and process remain largely intact or (2) are slightly altered, and easily reversible. This ranking includes, activities such as timber harvest that conforms to (or has higher standards beyond) CFPR (*e.g*., Pacific Forest Trust certified).

Resources Utilized:
NMFS used CalFire's Timber Harvest Plans in digital GIS format, which focused on land use over the last ten years, to analyze the percentage of land managed as timberlands. NMFS staff also used knowledge of watersheds assessments and ongoing practices for land use analysis.

Threat: Mining

This threat includes all types of mining and quarrying, including instream gravel mining.

Impacts to Salmonids:
Extraction of minerals and aggregate has affected fishery resources tremendously, and it continues to degrade salmonid habitat in many areas (Nelson *et al.* 1991). According to CDFG (2004), gravel extraction (the removal of sediment from the active channel) has various impacts on salmonid habitat by interrupting sediment transport and often causing channel incision and degradation (Kondolf 1993). The impacts from gravel extraction include; direct mortality, loss of spawning habitat, disruption of adult and juvenile migration and holding patterns, stranding of adults and juveniles, increases in water temperature and turbidity, degradation of juvenile rearing habitat, destruction or sedimentation of redds, increased channel instability and loss of natural channel geometry, bed coarsening, lowering of local groundwater level, and loss of LWD and riparian vegetation (Humboldt County Public Works 1992; Kondolf 1993; Jager 1994; Halligan 1997). Terrace mining (the removal of aggregate from pits isolated from the active channel) may have similar impacts on salmonids if a flood causes the channel to move into the gravel pits.

Application to the ESU:
Mining occurs within many watersheds in the ESU, including instream gravel mining on the mainstem Russian River. Upslope mining operations include barrow pits and mining operations in Soquel Creek and until recently, San Vicente Creek.

Threat Context:

According to CDFG (2004) while instream gravel extraction has had direct, indirect, and cumulative impacts on salmonids in the recent past, no direct impacts to coho salmon have been documented under the current (post-1995) mining monitoring. Reporting standards developed by CDFG and the mining industry were incorporated into the following regulatory efforts; County Conditional Use Permits, reclamation plans required by the Surface Mining and Reclamation Act and, the Corps Letters of Permission. Many rivers continue to suffer the effects of years of channel degradation from the millions of tons of aggregate removed from the systems over time (Collins and Dunne 1990). Most gravel mining operations occur in habitat that is currently considered migration habitat rather than current spawning and rearing. However, some of these instream operations occur in important areas for recovery of coho spawning and rearing habitat.

Threats Evaluated and Ranked:
Exploring for, developing, processing, storing, and producing minerals and rocks.

Threats were evaluated for their potential to:

1. Reduce the quantity and quality of stream gravel;
2. Reduce channel complexity;
3. Modify upstream channel sections (*e.g.*, headcuts);
4. Alter riparian zone integrity, diversity, function, and composition;
5. Alter channel geometry and hydrology;
6. Alter stream bank stability;
7. Simplify channels or cause incision and disconnection from its floodplain;
8. Alter or cause the loss of floodplain/estuarine habitats; and
9. Alter water quality by increasing sedimentation or turbidity, elevating water temperatures, and input of toxic metals.

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered. Activities that rank as high or very high threats may include instream gravel mining and mining activities within the 20-year bankfull channel.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered could be reversed or ameliorated. Activities ranking as a medium threat may include activities outside of the 20-year bankfull channel.

Low threat ranking results when ecosystem function and process are largely intact, (or are expected to be) slightly altered, and easily reversible. Activities that rank as low threats generally occur outside of the 100-year floodplain.

Resources Used:
No numeric values or categories were used to develop rankings. Instead NMFS utilized, watershed documentation, professional judgment, as well as consultations with knowledgeable individuals when ranking this threat after considering information and analyses from biological opinions on gravel mining operations through the CCC coho salmon ESU.

Threat: Recreational Areas and Activities

This threat addressed recreational activities (legal and illegal) that alter, destroy, and/or disturb habitats and species outside of established transport corridors.

Impacts to Salmonids:

The threat covers many types of activities that may directly and indirectly impact salmonids including: increased sedimentation to streams due to off road vehicle (ORV) use in the upper portion of a watershed; concentrated animal waste discharge from an equestrian facility that is directed into rearing habitat; loss of riparian vegetation due to construction and operation of on-stream recreational summer dams which leads to increased water temperature.

Application to the ESU:

Recreational areas and activities are numerous and diverse in the ESU. This threat category is often more likely to occur in areas with high human populations and includes legal and illegal activities and activities with temporary and permanent impacts.

Threat Context:

Since listing a number of actions have been undertaken to address some of the impacts related to recreational areas and activities. These actions include development of a white paper by NMFS regarding the impacts of recreational summer dams and increased enforcement and oversight by NMFS and CDFG regarding installation of these facilities. However, many of actions and their impacts remain unaddressed and impacts to salmonids and their habitat continue.

Threats Evaluated and Ranked:

Use of ORVs, mountain bikes, trail maintenance, equestrian uses, summer dams, amusement parks, and golf courses.

Stresses considered included the following:

1. Excessive erosion and sedimentation;
2. Stream crossings and effects of ORV or equestrian use in the channels;
3. Introduction of pollutants, garbage, toxic chemicals, and changes in nutrient levels;
4. Alteration in riparian zone integrity, diversity, function, and composition;
5. Alteration in streambank stability;
6. Diversion and/or impoundment of streams; and
7. Channel simplification, incision and disconnection from its floodplain.

High or very high threat rankings results when ecosystem function and process are (or are expected to be) severely altered. High or very high threat rankings may include heavy ORV use in riparian channels that results in the destruction or modification of stream banks and riparian vegetation or permanent alteration of high quality habitat due to construction of recreational facilities.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered but could be reversed or ameliorated. Medium threat ranking may include extensive mountain biking trails on steep slopes with substandard maintenance oversight.

Low threat ranking results when ecosystem function and process are largely intact, (or are expected to be) slightly altered, and easily reversible. Low threat ranking may include low impact activities such as hiking on designated and properly located and maintain trails.

Resources Used:
The category of Recreational Areas and Activities encompasses a diverse array of land and water uses and types of recreation. A centralized database was not available to adequately assesses this threat category. Staff used available watershed assessments and relied heavily upon their professional experience from working within the various watersheds to assess the degree of impact posed by this threat.

Threat: Residential and Commercial Development

This threat includes urban, industrial, suburban, recreational, or rural residential developments resulting in permanent alteration of the natural environment and encroachment onto floodplains and into riparian areas. Development includes military bases, factories, shopping centers, resorts, *etc*. This includes the physical and social (*e.g*., homeless encampments) consequences of development such as increased impervious surfaces, increased runoff, changes to the natural hydrograph (*e.g.,* flashy flows), household sewage, urban wastewater, increased sedimentation, industrial effluents, and garbage and other solid waste.

Impacts to Salmonids:
Urbanization can degrade habitat in obvious ways including; direct loss of habitat, channelization of streams, degradation of water quality, and dewatering of streams. It can also affect habitat in less obvious ways by altering and disrupting ecosystem processes that can have unintended impacts to aquatic ecosystems through increased flooding, channel erosion, landslides, and aquatic habitat destruction (Booth 1991).

According to CDFG (2004) the structure of the biological community and abundance and diversity of aquatic organisms are greatly altered by urban impacts on channel characteristics and water quality. Wang *et al*., (1997) found that high urban land use was strongly associated with poor biotic integrity and was associated with poor habitat quality. Fish populations are also adversely affected by urbanization. Limburg and Schmidt (1990, as cited in Spence *et al.* 1996) found a measurable decrease in spawning success of anadromous species in Hudson River tributaries that had 15 percent or more of the watershed in urban development. Wang *et al*. (2003) found a strong negative relation between urban land cover in the watershed and the quality of fish assemblages in coldwater streams in Wisconsin and Minnesota. In a study of urbanized Puget Sound streams in Washington State, Lucchetti and Fuerstenberg (1993, as cited in Spence *et al.* 1996) found that coho salmon appeared to be more sensitive than cutthroat trout (*Onchorynchus clarki*) to habitat alteration, increased nutrient loading, and degradation of the inter-gravel environment. They found, as impervious surfaces increased, coho salmon abundance declined, and concluded coho salmon are of particular concern in urbanized areas because of their specific habitat needs (smaller streams, relatively low velocity microhabitats and large pools). Other studies documented pollution associated with urban areas is causing impacts to juvenile Chinook salmon, including suppressed immune response due to bioaccumulation of PCBs and PAHs, increased mortality associated with disease, and suppressed growth (Spence *et al*. 1996).

Application to the ESU:
Historical records suggest coho salmon occurred in the Sacramento River system, but it was considered the rarest of the five salmon species known to inhabit the Central Valley (Hallock and Fry 1967; Brown *et al.* 1994). Though now extirpated, coho salmon did occur in streams that drained into the San Francisco Bay estuary. In fact, the earliest scientific specimen of coho salmon in California was collected by Professor Alexander Agassiz from Harvard University in San Mateo Creek, San Mateo County, in 1860 (Leidy 2004). Coho salmon are now extirpated from the Central Valley and the San Francisco Bay due to a variety of human caused factors – including urbanization. Watersheds where CCC coho salmon continue to persist have ongoing land management practices frequently cited as reasons for decline (dams, logging, roads, *etc.*) but in general have low rates of commercial and urban development. The adverse impacts of residential and commercial development are numerous, and these impacts are often closely interrelated with other activities evaluated separately in this document (*i.e.*, roads and channel modification).

Threat Context:
Within the California range of coho salmon, urban and suburban development occupy many of the watersheds targeted for recovery actions. Cities and towns with large developed areas within the range of CCC coho salmon include, from north to south, Fort Bragg, Ukiah, Healdsburg, Windsor, Sebastopol, Santa Rosa, Cotati, and Santa Cruz. Cities and towns with watersheds draining into the San Francisco Bay were not included in the recovery strategy.

Threats Evaluated and Ranked:
Threats were evaluated for their potential to:

1. Introduce pollutants, garbage (*e.g*., tires and common household trash), urban/industrial wastewater, sedimentation, toxic chemicals into the aquatic environment, and adversely alter nutrient levels (often as "shock pollution" occurring with the first flush of rains);
2. Alter riparian zone integrity, diversity, function, and composition;
3. Alter stream bank stability;
4. Simplify channels, or cause incision and disconnection from the floodplain;
5. Alter drainage channels and hydrology;
6. Increase stormwater runoff; and
7. Facilitate increased development and associated adverse consequences.

High or very high threat rankings result when (1) ecosystem function and process are (or are expected to be) severely altered or (2) impacts to the population are severe. High or very high threats occur when amelioration of the consequences of this threat is largely irreversible. High or very high threat rankings may occur in watersheds with extensive urban development resulting in extensive modification of riparian zones from historical conditions.

Medium threat ranking results when (1) ecosystem function and process are (or are expected to be) moderately altered or (2) impacts to the population are moderate. Medium threats occur when the consequences of this threat are largely irreversible but could be ameliorated.

Low threat ranking results when (1) ecosystem function and process remain largely intact or (2) are slightly altered, and easily reversible.

Resources Used:
GIS analysis of the percentage of watershed with impervious surfaces, watershed specific assessments, NMFS staff knowledge of watersheds and ongoing practices, *etc*., were examined.

Threat: Roads and Railroads

This threat includes roadways (highways, secondary roads, primitive roads, logging roads, bridges & causeways) and dedicated railroad tracks. It includes all roads (including mainline logging roads) not associated with the site-specific footprint of timber harvest activities.

Impacts to Salmonids:
Studies have documented the degradation that occurs to salmonid habitats as a result of forest, rangeland and other road networks (Furniss *et al.* 1991). Roads alter natural drainage patterns and accelerate erosion processes causing changes in streamflow regimes, sediment transport and storage, channel bed and bank configuration, substrate composition, and stability of slopes adjacent to roads systems (Furniss *et al.* 1991).

Application to the ESU:
Graham Matthews and Associates (1999) linked increased road densities to increased sediment yield in the Noyo River. NMFS (1996b) guidelines for salmon habitat characterize watersheds with road densities greater than three miles of road per square mile of watershed area (mi/mi2) as "not properly functioning" while "properly functioning condition" was defined as less than or equal to two miles per square mile, with few or no streamside roads.

Threat Context:
Since listing, a number of actions have been undertaken to address roads and road related threats. Through the Fishery Network of the Central California Coastal Counties (FishNet 4C) program, an evaluation of road related issues, including fish passage and ongoing maintenance practices has been conducted. Maintenance manuals and ongoing training programs were developed for roads staff in most counties in the ESU. The key focus of the FishNet 4C program is on implementing best management practices related to protecting water quality, aquatic habitat and salmonid fisheries. The guidelines outlined in the manuals address most routine and emergency road related maintenance activities undertaken by County Departments of Public Works, parks, and Open Space Districts, and other parties with responsibility for road maintenance. They address common facilities such as appropriate spoils storage sites and maintenance yards. The guidelines apply to activities related to county facilities, not to private development.

Restoration of problematic private and public roads is a large part of the CDFG restoration program and occurs in many of the targeted watersheds in the ESU. The magnitude of road related problems in the ESU is significant and it is anticipated that it will take many years to adequately address the most problematic roads. Additionally, many roads, particularly private non-timber roads are not subject to routine maintenance and chronic sediment input from these roads is a major problem in some watersheds.

Threats Evaluated and Ranked:
Threats were evaluated for their potential to affect:

1. Chronic and acute introduction of sediment from surface erosion and drainage;
2. Delivery of large quantities of sediment from road crossing or mass wasting associated with roads;
3. Passage impairment or blockage due to culverts, bridges, *etc*.;
4. Risks of spills;
5. Alteration of drainage channels, hydrology, infiltration and runoff;
6. Alteration in riparian zone diversity, function, and composition;
7. Channel simplification, incision and disconnection from its floodplain;
8. Alteration of channel and streambank stability;
9. Alteration or loss of floodplain or estuarine habitats;
10. Introduce water-borne pollutants, such as sediment and chemicals, into the aquatic environment, and adversely alter nutrient levels; and,
11. Facilitate increased development and associated consequences.

High or very high threat rankings result when (1) ecosystem function and process are (or are expected to be) severely altered or (2) impacts to the population are severe. High or very high threats occur when amelioration of the consequences of this threat is largely irreversible. A high or very high threat may occur in watersheds with high road densities, poor road maintenance practices, numerous stream crossings, and road placement on unstable areas and adjacency to stream zones.

Medium threat ranking results when (1) ecosystem function and process are (or are expected to be) moderately altered or (2) impacts to the population are moderate. Medium threats occur when the consequences of this threat are largely irreversible but could be ameliorated.

Low threat ranking results when (1) ecosystem function and process remain largely intact or (2) are slightly altered, and easily reversible.

Resources Utilized:
For areas where timber harvest is conducted, road densities were calculated using CalFire timber harvest GIS data[13]. Topologically Integrated Geographic Encoding and Referencing (TIGER) data generated by the U.S. Census Bureau provided additional data (2000)[14].

Threat: Severe Weather

This threat includes short-term extreme variations such as severe droughts and major floods, and long-term climatic changes outside the range of natural variation that may be linked to global warming and other large scale climatic events. These natural events exacerbate already degraded conditions.

Impacts to Salmonids:
Droughts can have a variety of negative impacts on salmon and other fish populations at several points of their life cycles. Adult salmon can experience difficulties reaching upstream spawning

[13] http://www.fire.ca.gov/resource_mgt/resource_mgt_forestpractice_gis.php

[14] http://www.census.gov/geo/www/tiger/

grounds during certain low flow conditions. Low flows can also increase pre-spawn mortality rates in returning adult salmon when high adult escapement coincides with elevated water temperatures, low dissolved oxygen levels, and increased disease transmission between fish (CDFG 2003). Drying streams can severely reduce juvenile rearing habitat which in turn reduces carrying capacity. Some salmon species spawn in channel margins, side channels and smaller tributaries, and spawning for those species would have to occur in mainstem waters if off channel and tributary habitat is unavailable because of low flows. Where this occurs, salmon redds within the mainstem river channel may be more susceptible to bed scour during the fall and winter (Washington Dept. Fish and Wildlife)[15]. In other cases, instream flow can drop after the salmon spawn, dewatering the redds and desiccating the eggs.

High flows associated with major storms and floods can result in complete loss of eggs and alevins as they are scoured from the gravel or buried in sediment (Sandercock 1991; NMFS 1998b). Juveniles and smolts can be stranded on the floodplain, washed downstream to poor habitat such as isolated side channels and off-channel pools, or washed out to sea prematurely. Peak flows can induce adults to move into isolated channels and pools and prevent their migration because of excessive water velocities (CDFG 2004) .

Climate change may profoundly affect salmonid habitat on a regional scale by altering streamside canopy structure, increasing forest fire frequency and intensity, elevating instream water temperatures; and altering rainfall patterns that in turn affect water availability. These impacts are likely to negatively impact salmonid population numbers, distribution, and reproduction.

Application to the ESU:
Droughts are a natural phenomenon in the Mediterranean climate of the CCC coho salmon ESU. Nonetheless, droughts can result in depressed salmons runs three years later, when those salmonids would be returning as adults. The drought of 1976/1977 is believed to have significantly impacted coho populations south of San Francisco Bay (Hope 1993; Smith 2011). Flooding also has beneficial effects, including: cleaning and scouring of gravels; transporting sediment to the flood plain; recruiting, moving and rearranging LWD; recharging flood plain aquifers (Spence *et al.* 1996); allowing salmonids greater access to a wider range of food sources (Pert 1993); and maintaining the active channel.

Streams can be drastically modified by erosion and sedimentation in large flood flows almost to the extent of causing uniformity in the stream bed (Spence *et al*., 1996). After major floods, streams can take years to recover pre-flood equilibrium conditions. Flooding is generally not as devastating to salmon in morphologically complex streams, because protection is afforded to the fish by the natural in-stream structures such as LWD and boulders, stream channel features such as pools, riffles, and side channels and an established riparian area (Spence *et al*., 1996).

Salmonids in the CCC ESU are at the southern extent of the species range, and may be more vulnerable to changes in water availability and instream temperatures. Climate change is discussed in more detail in Appendix A: Marine and Climate. Significant alteration in the instream and near-stream environments due to climate change may result in further range

[15] http://wdfw.wa.gov/drought/index.htm

contraction for salmonids and a reduction in overall habitat availability in the more resilient watersheds.

Threat Context:
In the ESU there is increased pressure for limited water resources in many of the focus watersheds. This problem is most severe in the southern part of the ESU where rainfall is generally less than in the northern part of the ESU. Compounding this problem is a larger human population in the southern watersheds with a higher number of instream water diversions.

Streams can be drastically modified by erosion and sedimentation in large flood flows almost to the extent of causing uniformity in the stream bed (Spence *et al*., 1996). After major floods, streams can take years to recover pre-flood equilibrium conditions. Flooding is generally not as devastating to salmon in streams with complex habitat features, because protection is afforded to the fish by the natural in-stream structures such as LWD and boulders, stream channel features such as pools, riffles, and side channels and an established riparian area (Spence *et al*., 1996).

NMFS has reviewed extensive data and modeling sources, and assumes the future effects of climate change and the expected sea level rise in California could include: lost estuarine habitat; reduced groundwater recharge and base-flow discharge; and associated rises in stream temperature and demand for water supplies. Smaller (remnant) salmonid populations in such areas are likely at most risk from climate change.

Threats Evaluated and Ranked:
Threats related to droughts were evaluated for their potential to effect:

1. Insufficient flows to facilitate egg incubation, adult escapement, juvenile rearing, smolt emigration, and juvenile immigration;
2. Poor water quality leading to increased instream temperatures, low dissolved oxygen, decreased food availability, increased concentrations of pollutants, *etc*.;
3. Earlier than normal water diversion for anthropogenic purposes; and
4. Insufficient flows to breach sandbars at river mouths.

Threats related to flooding were evaluated for their potential to:

1. Increase the frequency, duration, and magnitude of flooding beyond natural conditions;
2. Require flood control or management actions;
3. Cause loss of riparian and instream habitat attributes;
4. Increase frequency of channel scour beyond natural conditions; and
5. Increase turbidity beyond natural conditions.

Threats related to climate change were evaluated for their potential effects to managing limited water storage to provide cool water refugia, additional demands on existing water supplies, and changes in vegetation patterns.

Threats were evaluated for their potential to:

1. Elevate instream water temperatures and alter historical hydrologic patterns; and
2. Alter the composition of native plant communities, which may adversely alter riparian process and function.

High or very high threat rankings result when ecosystem function and process are (or are expected to be) severely altered. High or very high threat rankings may occur in heavily urbanized watersheds subjected to extensive diversion, historical and ongoing instream modification conducted for flood control purposes, and where circumstances preclude future opportunities to protect critical refugia habitats.

Medium threat ranking results when ecosystem function and process are (or are expected to be) moderately altered but could be reversed or ameliorated.

Low threat ranking results when ecosystem function and process are (or are expected to be) largely intact, slightly altered, and easily reversible. Low threat ranking may occur in watersheds with little urban interface, few diversions, intact floodplains, and where instream habitat forming features (such as LWD) are present and are not routinely removed.

Resources Used:
Droughts were evaluated in the context of available information regarding ongoing water diversions coupled with the effects of drought. A variety of resources were used to evaluate this potential impact, including individual watershed assessments, briefings with NMFS, CDFG, and others familiar with individual watersheds and existing diversions, *etc*.

For the threat of flooding, staff knowledgeable on specific watersheds and ongoing practices, *etc*., ranked this threat. In addition, NMFS reviewed models related to climate change where they predicted increased storms or flooding.

NMFS has considered future habitat condition scenarios for salmonids based on projected climate change impacts as described in Appendix A: Marine and Climate. We used existing information on the current distribution of extant populations and areas targeted for recovery, and evaluated current stresses into the future.

Threat: Water Diversion and Impoundment

This threat includes appropriative and riparian surface water diversions and groundwater pumping resulting in changes to water flow patterns outside the natural range of variation. This threat includes use, construction, and maintenance of seasonal dams for water diversions, as well as the operations of larger dams affecting the natural hydrograph and watershed processes such as sediment transport.

Impacts to Salmonids:
According to CDFG (2004) losses of coho salmon result from a wide range of conditions related to unscreened water diversions and substandard fish screens. Primary concerns and considerations for fish at diversions that are unscreened or equipped with poorly functioning screens include; delay of downstream migration and a reduction in the overall survival of downstream migrants, entrainment of juvenile coho salmon into the diversion, impingement of juvenile coho salmon on

the screen surfaces because of high approach velocities or low sweeping velocities, predator holding areas created by localized hydraulic effects of the fish screen and related facilities, entrapment of juvenile coho salmon in eddies or other hydraulic anomalies where predation can occur, elevated predation levels due to concentrating juveniles at diversion structures, and disruption of normal fish schooling behavior caused by diversion operations, fish screen facilities, or channel modifications. Dam operations also affect salmonids by altering the natural hydrograph, typically by reducing winter flows that provide cues to migrate, and altering summer flows to levels that may reduce the survival of rearing juveniles.

Application to the ESU:
Water is often handled in the regulatory or legal arena due to its relative scarcity in California's Mediterranean climate. Summer baseflow is a critical attribute that is degraded in many streams across the ESU. A substantial amount of coho salmon habitat has been lost or degraded as a result of water diversions and groundwater extraction (KRBFTF 1991; CDFG 1997). The nature of diversions varies from major water developments which can alter the entire hydrologic regime in a river, to small domestic diversions which may only have a localized impact during the summer low flow period. In some streams the cumulative effect of multiple small legal diversions may be severe. Illegal diversions are also believed to be a problem in some streams within the range of coho salmon (CDFG 2004).

Threat Context:
Water is the most important of all habitat attributes necessary to maintain a viable fishery and, based on the last 150 years of water development in California, one of the most difficult threats to address effectively. Few restoration projects address water because; in large part it is a very divisive issue. Diversions are subject to regulation by the State Water Resources Control Board through the appropriative water rights process, and by CDFG under Fish and Game Code § 1600 *et seq*. (which requires an agreement with the Department for any substantial flow diversion), Fish and Game Code § 2080 *et seq*. (California Endangered Species Act take authorization), and Fish and Game Code § 5937 (which requires sufficient water below a dam to maintain fish in good condition). NMFS has authority under ESA to regulate the take of coho salmon at diversions.

In some watersheds, the demand for water has already exceeded the available supply and some water rights have been allocated though court adjudication. These adjudications usually did not consider coho salmon habitat needs at a level that could be considered protective under the California Endangered Species Act or the Federal ESA. The use of wells adjacent to streams is also a significant and growing issue in some parts of the coho salmon range. Extraction of flow from such wells may directly affect the adjacent stream, but is often not subject to the same level of regulatory control as diversion of surface flow. Site specific groundwater studies are required to determine a direct connection between surface flow and groundwater, and these are often very costly and take a significant amount of time to complete.

Threats Evaluated and Ranked:
Threats were evaluated for their potential to:

1. Increase water diversion and withdrawal, both legal and illegal;
2. Increase chronic and acute sediment inputs from surface erosion and drainage;

3. Impair passage or create blockages;
4. Alter drainage channels and hydrology;
5. Alter riparian zone diversity, function, and composition;
6. Alter channel and streambank stability;
7. Alter or eliminate floodplain and/or estuarine habitats due to reduced freshwater inflow;
8. Introduce water-borne pollutants, such as sediment and chemicals, into the aquatic environment, and adversely alter nutrient levels;
9. Facilitate increased development and associated consequences;
10. Cause changes in water flow, fish habitat, and temperature;
11. Reduce gravel recruitment to downstream areas;
12. Cause dewatering and/or flow reductions;
13. Cause secondary effects to salmonids (*e.g*., increasing disease such as bacterial kidney disease); and
14. Delay sandbar breaching (*e.g*., Scott Creek).

High or very high threat rankings result when (1) ecosystem function and process are (or are expected to be) severely altered or (2) impacts to the population are severe. High or very high threats occur when amelioration of the consequences of this threat are largely irreversible.

Medium threat ranking results when (1) ecosystem function and process are (or are expected to be) moderately altered or (2) impacts to the population are moderate. Medium threats occur when the consequences of this threat are largely irreversible but could be ameliorated.

Low threat ranking results when (1) ecosystem function and process remain largely intact or (2) are slightly altered, and easily reversible.

Resources Utilized:
Fisheries biologists from CDFG and Regional Water Quality Control Boards were invited to participate in a structured decision-making process to provide individual opinions regarding flow conditions for specific habitat attributes, and also considered diversion and impoundments for each watershed. Workshop participants were asked to individually rate the hydrologic setting, the degree of exposure to flow impairments, and the intensity of those impacts for each CCC coho salmon population. GIS analysis of known diversion points, and the CDFG Passage Assessment Database (PAD)[16] were reviewed. NMFS GIS watershed characterizations, NMFS staff knowledge of watersheds and ongoing practices, *etc*., were also examined.

[16] http://nrm.dfg.ca.gov/PAD/Default.aspx

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APPENDIX C

DESCRIPTION OF ATTRIBUTES IN DATA TABLES PRODUCED IN THE STREAM SUMMARY APPLICATION

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Description of Attributes in Tables produced in the Stream Summary Application

The following report provides descriptions of attributes for the Stream Summary Application output database that was created for the California Department of Fish and Game - Hopland Office. The application was developed in 2008 by UC:ANR:Hopland Research Extension and Center GIS Lab under the Fisheries Restoration Grant Program (FRGP) grant number PO430411. The stream summary application was modified to provide additional information needed by the National Marine Fisheries Service (NMFS) to inform federal recovery planning underway in the North Central California Coast Recovery Domain: a geographic area encompassing the federally listed Distinct Population Segments (DPS) of Northern California steelhead and Central California Coast steelhead and the Evolutionarily Significant Units (ESU) of California Coastal Chinook and the Central California Coast coho salmon. This work was made possible under Sonoma County Water Agency (SCWA) Contract TW 08/09-125.

The Stream Summary Application was developed to provide additional information to regional biologists when assessing salmonid habitat based on stream habitat surveys. The Application produces 4 tables standard (stream summary, habitat criteria, ranked manual criteria, and reachsum_x), that contain all of the metrics in the Stream Habitat Program report (text, tables, and graphs) and some additional calculations from various Department of Fish and Game planning documents. For the SCWA contract we produced three additional tables (noaa_table, Units, and Populations), these additional tables were requested by NMFS planning team.

STANDARD TABLES:

The “stream summary” table reports the metrics in the text, tables, and graphs found in Stream Habitat Reports. Data is reported at specific habitat levels (1 - 4, California Salmonid Stream Habitat Restoration Manual III-30, and an additional habitat level of 0, this summarizes the data either at the stream or reach level without taking into account a habitat type.). Additionally data is reported for all metrics for all habitat types (Habitat Type Level field). The “stream summary” table provides the metrics at both the stream and the reach level (StreamOrReach field). In the “stream summary” table we also provide the sample sizes and sums of values for all of the metrics provided.

The “habitat criteria” table contains additional metrics and habitat criteria that can be used to evaluate stream condition. The criteria have been gleaned from various Department planning documents (see end of document for a detailed list of the metrics and source documents). The “habitat criteria” table provides the metrics at both the stream and the reach level (StreamOrReach field).

The “ranked manual criteria” table contains information about 6 habitat criteria as described in the California Salmonid Stream Habitat Restoration Manual. The table provides a boolean score, depending on whether they do (value 1) or do not meet (value 0) the criteria. The seventh value in the table is the numeric sum of criteria scores by each reach or stream. The table provides the metrics at both the stream and the reach level (StreamOrReach field).

The "reachsum_x" table is loosely based on the data reported in Stream Habitat Program table number 8. The "reachsum_x" table provides the metrics at the reach level. This table has been replaced by the "stream summary" table produced by the Stream Summary Application. "Reachsum_x," is provided as a reference to help older projects transition to the new "stream summary" table.

SCWA TABLES:

The "noaa_table" table contains additional metrics and habitat criteria that can be used to evaluate stream condition for salmonids species. These criteria have been developed by NMFS planning team through literature reviews and consultation with experts in the field of salmonid ecology. The "noaa_table" table provides the metrics at both the stream and the reach level (StreamOrReach field).

The "Units" table contains information that can be used to relate the stream and the reach level data to common aggregating layers, such as, county boundaries, USGS hydrologic unit codes (HUCs), ecoregional boundaries, and CALWATER boundaries.

The "Populations" table contains information that can be used to relate the stream and the reach level data to the NMFS salmonid populations planning dataset.

The data produced in this application can be joined to spatial data representing the streams or reaches surveyed by the California Department of Fish and Game. The spatial data available includes:

- Reach lines – Line shapefile that represents the surveyed reaches.
- Reach Sheds – Polygon shapefile that represents the surveyed reaches as watersheds.

How to link tables to GIS:

- Join the tables to the GIS data through two different fields. For the reach level data join based on the common field code and for the stream level join based on the Table field code to spatial data field code1.

Contact Information –

- For questions about data structure and database design, etc.
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Acknowledgments: UC:ANR:Hopland Research Extension and Center (HREC) would like to acknowledge the California Department of Fish and Game for the original Stream Habitat program which provided the initial data structure used for the stream summary application. Modifications were made to accommodate changes and additional information as requested by the California Department of Fish and Game personnel in Region 3 and the National Marine Fisheries Service.

Application Table: Stream Summary – All metrics in report (text, tables, and graphs).

The "stream summary" table contains all of the metrics in the Stream Habitat Program report (text, tables, and graphs). The "stream summary" table provides the metrics at both the stream and the reach level (StreamOrReach field). The Stream Habitat Program reports the metrics in the text, tables, and graphs at specific habitat levels (1 - 4, California Salmonid Stream Habitat Restoration Manual III-30, in the "stream summary" table we provided an additional habitat level of 0, this summarizes the data either at the stream or reach level without taking into account a habitat type.), in the "stream summary" table we provide the metrics at all habitat levels (Habitat Type Level field). In the "stream summary" table we also provide the sample sizes and sums of values for all of the metrics provided.

Example Record

What are we looking at – Definition or explanation

Reported in: *Where in the stream habitat program outputs do these values appear*

Inclusions: *What is included in the calculations*

Used in Calculations: *Where is this information used in calculations*

Attribute	Description
Field Name	Description of field name (if necessary) and calculation

General Survey Information

This section contains basic information about the stream habitat survey such as the Site ID, site name, stream name, year of record, the duration of the sample, etc.

Reported in: All Tables

Inclusions:

Used in Calculations:

Attribute	Description
SurveyId	Survey identification number
Pname	Stream name
Pnmcd	Stream number
Year	Year of survey
StreamOrReach	Code used to delineate whether the measurements are at the stream or reach level
Code	Stream code or ReachID depending on StreamOrReach Value
Habitat Type Level	Habitat level 1 - 4 (figure 3-8, habitat manual)
MinOfL4_Number	Value used to sort data based on habitat type

Dates – The dates of the habitat surveys

Reported in: All Tables

Inclusions:

Used in Calculations:

Attribute	Description
Minimum Date	The minimum date of the survey in the reach or stream
Maximum Date	The maximum date of the survey in the reach or stream

Channel Type - Rosgen channel type classification. The channel type of the reach or stream based on the Stream Channel Type Work Sheet (Part III)

Reported in: Table 8

Inclusions:

Used in Calculations:

Attribute	Description
Channel Type	Rosgen channel type classification. The channel type of the reach or stream based on the stream channel type work Sheet (part III)

Base Flow (cfs) - The base flow is the flow that the stream reduces to during the dry season or a dry spell. This flow is supported by ground water and subsurface seepage into the channel.
Reported in: Table 8
Inclusions:
Used in Calculations:

Attribute	Description
Base Flow (cfs)	The mean base flow in cubic feet per second, measured at the beginning of the survey. If flows change significantly during the survey they are again measured at the end of the survey at the same location. The average of the two measurements is recorded.

Temperature Data – Temperature of the water and air taken during the surveys. Temperatures are taken at the beginning of each page record and recorded to the nearest degree Fahrenheit. Temperatures are taken in the shade and within one foot of the water surface.
Reported in: Table 8
Inclusions:
Used in Calculations: Temperature values > 0

Attribute	Description
Minimum Water Temperature ˚F	For those water temperatures greater than zero, the minimum water temperature during survey
Maximum Water Temperature˚F	For those water temperatures greater than zero, the maximum water temperature during survey
Average Water Temperature˚F	For those water temperatures greater than zero, the average water temperature during survey
Minimum Air Temperature˚F	For those air temperatures greater than zero, the minimum air temperature during survey
Maximum Air Temperature˚F	For those air temperatures greater than zero, the maximum air temperature during survey
Average Air Temperature˚F	For those air temperatures greater than zero, the average air temperature during survey

Bankfull Width (W_{bkf}) – The width of the stream at bankfull discharge (Q_{bkf}) is measured by stretching a level tape from one bank to the other, perpendicular to the stream and at the Q_{bkf} line of demarcation on each bank. Q_{bkf} is determined by changes in substrate composition, bank slope, and perennial vegetation caused by frequent scouring flows. Bankfull discharge is the dominant channel forming flow with a recurrence interval within the 1 to 2 year range.

Reported in: Table 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
Minimum Bankfull Width (ft)	The minimum Bankfull width in reach or stream
Maximum Bankfull Width (ft)	The maximum Bankfull width in reach or stream
Mean Bankfull Width	The mean Bankfull width in reach or stream

(ft)	
StDev Of Bankfull Width (ft)	The standard deviation of Bankfull width in reach or stream

Large Woody Debris – Wood debris is defined as a piece of wood having a minimum diameter of twelve inches and a minimum length of six feet. Root wads must meet the minimum diameter criteria at the base of the trunk but need not be at least six feet long.
Reported in: Table 8 and 10; Graph 7
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
Sum of LWD	For those units with Large Woody Debris (LWD), the sum of the number of LWD in the stream or reach
Occurrence of LWD (%)	For those units with Large Woody Debris (LWD), the sum of the percent cover of LWD in the stream or reach divided by the number of habitat units with percent canopy values in reach or stream multiplied by 100
LWD per 100 ft	For those units with Large Woody Debris (LWD), the sum of the number of LWD in the stream or reach divided by the number of sum length of reach or stream multiplied by 100

Stream Order - The Strahler Stream Order is a simple hydrology algorithm used to define stream size based on a hierarchy of tributaries.
Reported in:
Inclusions:
Used in Calculations: Primary pool and mean residual depth by nth stream order calculations.

Attribute	Description
Stream Order Minimum	The minimum stream order of the stream or reach. Stream order is calculated based on the Shreve ordering system.
Stream Order Maximum	The maximum stream order of the stream or reach. Stream order is calculated based on the Shreve ordering system.
Stream Order Majority	The majority stream order of the stream or reach. Stream order is calculated based on the Shreve ordering system.

Habitat Units Counts and Information – Habitat units are delineated in the field and represent different habitat types as defined in chapter III of the California Salmonid Stream Habitat Restoration Manual (Part III, Page 27).
Reported in: Table 1, 2, 3, 4, 5 and 6; Graph 1, 3
Inclusions:
Used in Calculations:

Attribute	Description
Units Fully Measured	Number of habitat unit fully measured (width measurements taken)
Total Units Fully Measured	Total number of habitat unit fully measured (width measurements taken)
Habitat Units	Number of habitat units by type
Total Habitat Units	Total number of habitat units surveyed
Habitat Type At Level	Habitat Level Name (Figure 3-8, Habitat Manual)

Habitat Occurrence (%) – Percent of the habitat type within the reach of stream surveyed, based on the frequency of occurrence
Reported in: Table 1, 2, 3, 4, 5, and 6; Graph 1, 3
Inclusions:
Used in Calculations:

Attribute	Description
Habitat Occurrence (%)	Percent of the habitat type within the reach of stream surveyed based on the frequency of occurrence. The number of each habitat unit type divided by the total number of habitat units surveyed multiplied by 100.
Total N Of Pool Units Table 3	Total Number of Pool Habitat Units at Level III
Total N Of Pool Units Table 4	Total Number of Pool Habitat Units at Level IV
Pool Occurrence (%) Table 3	Percent of the pool habitat types within the reach of stream surveyed based on the frequency of occurrence. The number of each habitat unit type divided by the total number of pool units at Level III surveyed multiplied by 100.
Pool Occurrence (%) Table 4	Percent of the pool habitat types within the reach of stream surveyed based on the frequency of occurrence. The number of each habitat unit type divided by the total number of pool units at Level IV surveyed multiplied by 100.

Mean Length – Length for the surveys is defined as the thalweg length of the habitat unit, measured in feet. Side channel units are included in calculating the mean length.
Reported in: Table 1, 2, 3 and 8; Graph 2
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Area, Mean Volume, Mean Residual Pool Volume,*** All Area, Pool depth, and volume calculations.

Attribute	Description
Sum Length (ft)	Sum of lengths for each habitat type
Mean Length (ft)	Mean length was obtained by taking the sum of lengths for each habitat type divided by the total number of habitat units
Dry Length (ft)	Sum of lengths classified as dry (7.0)
Total Length	Total length of all units
Total Length (%)	Sum of lengths for each habitat type divided by the total length of all habitat units including side channels.

Mean Width – Mean Width is defined as the mean of two or more wetted channel widths. Width measurements are recorded in feet.
Reported in: Table 1, 2, 3 and 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Area, Mean Volume, Mean Residual Pool Volume,*** All Area, Pool depth, and volume calculations.

Attribute	Description
Sum Mean Width (ft)	For the units that were fully surveyed, the summation of Mean Widths
N Of Mean Width	For the units that were fully surveyed, the number of Mean Widths
Mean Width (ft)	Sum Mean Width values divided by the number of units fully surveyed

Mean Depth - Mean Depth for the surveys is defined as the mean of several random depth measurements across the unit with a stadia rod in feet. Mean depths for pools are the mean residual depth that is the mean depth value from the survey minus the pool tail crest value.
Reported in: Table 1,2, and 3; Graph 5
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: All volume calculations

Attribute	Description
N Of Mean Depth (ft)	For the units that were fully surveyed and not null, the number of Mean Depth Values
Sum Mean Depth (ft)	For the units that were fully surveyed, for all types other than pools (see residual depth) the sum of mean depth values
N Of Residual Depth (ft)	For the units that were fully surveyed and not null, the number of Mean Depth Values. For the units that were fully surveyed and not null, the number of mean depth values minus pool tail crest depth value
Sum Residual Depth (ft)	For the units that were fully surveyed and not null, the sum of mean depth values minus pool tail crest depth value
Mean Depth (ft)	For pools the mean depth is the sum of residual depth (pool depths minus pool tail crest) divided by the number of units fully measured, for other types it is the sum of mean depth values divided by the total number of units that were fully measured.

Mean Maximum Depth - Enter the measured maximum depth for each habitat unit, in feet. Mean maximum depth for the surveys is defined as the mean maximum depth measurements in the unit in feet. Mean maximum depths for pools are the mean maximum residual depths (mean maximum depth value from the survey minus the pool tail crest value).
Reported in: Table 1,4 and 8; Graph 5
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
N Of Maximum Depth	For the units that were fully surveyed and not null, the number of Maximum Depth Values
Sum Maximum Depth (ft)	For units that were fully measured, the sum of maximum depth of all units
N Of Residual Maximum Depth (ft)	For the units that were fully surveyed and not null, the number of Residual Max Depth Values
Sum Residual Maximum Depth (ft)	For the units that were fully surveyed and not null, the sum of maximum depth values minus pool tail crest depth value
Mean Maximum Residual Depth (ft)	For the units that were fully surveyed and not null, the number of Residual Max Depth Values divided by the total number of residual max depth values
Mean Maximum Depth (ft)	For pools the mean maximum depth is the sum of residual maximum depth values divided by the total number of units fully measured, for other types it is the sum of maximum depth values divided by the total number of units fully measured

Maximum Depth - Enter the measured maximum depth for each habitat unit, in feet. Maximum depth for the surveys is defined as the maximum depth measurements in the unit in feet.

Maximum depths for pools is the maximum residual depths that is the maximum depth value from the survey minus the pool tail crest value.
Reported in: Table 2
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
Maximum Depth for Non-Pools	For non pool units, maximum depth of any unit
Maximum Depth (ft)	For the units that were residual max depth > 0, the maximum depth value

Depth Pool tail Crest - Depth pool tail crest for the surveys is defined as the maximum thalweg depth of pool tail crest, in feet. This measurement is only taken in pool habitat units.
Reported in: Not Reported
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Depth, Mean Residual Pool Volume,*** All Pool depth and volume calculations

Attribute	Description
N Of Residual Maximum Depth (ft)	For the units that were fully surveyed and not null, the number of Residual Max Depth Values
Sum Residual Maximum Depth (ft)	For the units that were fully surveyed and not null, the sum of maximum depth values - pool tail crest depth values

Maximum Residual Pool Depths by Strata – The number and the percent of pools with maximum residual depths less than or equal to 5 strata (less than 1 foot, between 1 foot and 2 feet, between 2 feet and 3 feet, between 3 feet and 4 feet, greater than 4 feet).
Reported in: Table 4 and 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
N Of Pools <1 Foot Maximum Residual Depth	For those units classified as pool, total number of pools with maximum residual depth < 1 foot
<1 Foot Percent Occurrence	The number of pools < 1 foot divided by the total number of pools with a residual maximum depth > 0 feet
N Of Pools 1<2 Feet Maximum Residual Depth	For those units classified as pool, total number of pools with maximum residual depth >= 1 Foot and < 2 Feet
1<2 Feet Percent Occurrence	The number of pools >= 1 foot and < 2 feet divided by the total number of pools with a residual maximum depth > 0 feet
N Of Pools 2<3 Feet Maximum Residual Depth	For those units classified as pool, total number of pools with maximum residual depth >= 2 Feet and < 3 Feet
2<3 Feet Percent Occurrence	The number of pools >= 2 feet and < 3 feet divided by the total number of pools with a residual maximum depth > 0 feet
N Of Pools 3<4 Feet Maximum Residual Depth	For those units classified as pool, total number of pools with maximum residual depth >= 2 Feet and < 3 Feet
3<4 Feet Percent Occurrence	The number of pools >= 3 feet and < 4 feet divided by the total number of pools with a residual maximum depth > 0 feet
N Of Pools >=4 Feet Maximum Residual Depth	For those units classified as pool, total number of pools with maximum residual depth >= 4 feet
>=4 Feet Percent Occurrence	The number of pools >= 4 feet divided by the total number of pools with a residual maximum depth > 0 feet

Mean Area - Mean Area is calculated for all habitat types and reported in square feet. Area calculations are based on the wetted width of the habitat units, that is the mean width multiplied by the product of 1 minus the percent exposed substrate. The wetted width is then multiplied by the length.
Reported in: Table 1, 2, and 3
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Volume, Mean Residual Pool Volume,*** All volume calculations

Attribute	Description
N Of Area (sqft)	For the units that were fully surveyed and had a mean depth > 0, the number of mean width values
Sum Of Area (sqft)	For the units that were fully surveyed and had a mean depth > 0, the sum of unit areas multiplied by the wetted width (mean width times (1 - percent exposed substrate)) times length
Mean Area (sqft)	For the units that were fully surveyed and had a mean depth > 0, the sum of unit areas multiplied by the wetted width (mean width times (1 - percent exposed substrate) times length times divided by the number of area values
Estimated Total Area (cuft)	The mean area of surveyed units multiplied by the total number of habitat units
Total Area (sqft)	Summed the estimated total area for the reach or streams

Mean Volume - Mean Volume is calculated for all habitat types and reported in cubic feet. Volume calculations are based on the wetted width of the habitat units, that is the mean width multiplied by the product of 1 minus the percent exposed substrate. The wetted with is than multiplied by the length and then multiplied by mean depth. Mean depths for pools are the mean residual depth that is the mean depth value from the survey minus the pool tail crest value.
Reported in: Table 1,2, and 3
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
N Of Volume (cuft)	For the units that were fully surveyed and had a mean depth > 0, the number of mean width values
Sum Of Volume (cuft)	For the units that were fully surveyed and had a mean depth > 0, the sum of unit volumes (multiplied the wet width (mean width * (1 - percent exposed substrate)) times length time the mean depth)
Mean Volume (cuft)	For the units that were fully surveyed and had a mean depth > 0, the sum of unit volumes (multiplied the wet width (mean width * (1 - percent exposed substrate)) times length time the mean depth) divided by the number of volume values
Estimated Total Volume (cuft)	The mean volume of surveyed units multiplied by the total number of habitat units
Total Volume (cuft)	Summed the estimated total area for the reach or streams
Sum Of Residual Pool Volume (cuft)	For pools the units that were fully surveyed and had a residual mean depth > 0, the sum of unit volumes (multiplied the wetted width (mean width * (1 - percent exposed substrate)) times length times the residual mean depth)
Mean Residual Pool Volume (cuft)	For pools the units that were fully surveyed and had a residual mean depth > 0, the sum of unit volumes (multiplied the wetted width (mean width * (1 - percent exposed substrate)) times length times the residual mean depth) divided by the number of volume values

Estimated Total Residual Volume (cuft)	The mean residual volume of surveyed units multiplied by the total number of habitat units
Total Residual Volume (cuft)	Summed the estimated total residual volume for the reach or streams

Riffle/Flatwater Mean Width (ft) - Riffle/Flatwater Mean Width for the surveys is defined as the mean of two or more wetted channel widths measurements in feet within the habitat unit.

Reported in: Table 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Depth,*** volume calculations

Attribute	Description
N Of Riffle/Flatwater Mean Width	For the units that were fully surveyed and classified as riffles/flat water, the number of mean width values
Sum Riffle/Flatwater Mean Width (ft)	For the units that were fully surveyed and classified as riffles/flat water, the sum of mean width values
Riffle/Flatwater Mean Width (ft)	For the units that were fully surveyed and classified as riffles/flat water, the sum of mean width values and divided by the number of mean width values

Pool Tail Embeddedness - Percent cobble embeddedness is determined at pool tail-outs where spawning is likely to occur. Sample at least five small cobbles (2.5" to 5.0") in diameter and estimate the amount of the stone buried in the sediment.

This is done by removing the cobble from the streambed and observing the line between the "shiny" buried portion and the duller exposed portion. Estimate the percent of the lower shiny portion using the corresponding number for the 25% ranges. Average the samples for a mean cobble embeddedness rating. Additionally, a value of 5 is assigned to tail-outs deemed unsuited for spawning due to inappropriate substrate particle size, having a bedrock tail-out, or other considerations:

Embeddedness Value	Amount of stone buried in sediment
1	0 to 25%
2	26 to 50%
3	51 to 75%
4	76 to 100%
5	unsuitable for spawning

Reported in: Table 8 and 9; Graph 6
Inclusions: Unit Mean Width > 0 feet, with embeddedness > 0
Used in Calculations:

Attribute	Description
N Of Embeddedness Values	For those units classified as pool, total number of embeddedness values >0
Sum Of Embeddedness Value 1	For those units classified as pool, summed the number of units with an Embeddedness value of 1
% Embeddedness Value 1	For those units classified as pool, the number of units with an Embeddedness value of 1 divided by the total number of Embeddedness Values > 0
Sum Of Embeddedness Value 2	For those units classified as pool, summed the number of units with an Embeddedness value of 2

% Embeddedness Value 2	For those units classified as pool, the number of units with an Embeddedness value of 2 divided by the total number of Embeddedness Values > 0
Sum Of Embeddedness Value 3	For those units classified as pool, summed the number of units with an Embeddedness value of 3
% Embeddedness Value 3	For those units classified as pool, the number of units with an Embeddedness value of 3 divided by the total number of Embeddedness Values > 0
Sum Of Embeddedness Value 4	For those units classified as pool, summed the number of units with an Embeddedness value of 4
% Embeddedness Value 4	For those units classified as pool, the number of units with an Embeddedness value of 4 divided by the total number of Embeddedness Values > 0
Sum Of Embeddedness Value 5	For those units classified as pool, summed the number of units with an Embeddedness value of 5
% Embeddedness Value 5	For those units classified as pool, the number of units with an Embeddedness value of >= 5 divided by the total number of Embeddedness Values > 0
Mean Embeddedness	For those units classified as pool, the sum of Embeddedness value of > 0 divided by the total number of Embeddedness Values > 0
Mean Embeddedness Integer	The integer value of the Mean Embeddedness Value

Pool tail Substrate – Pool substrate for the surveys is entered based on the code (A through G) for the dominant substrate composition of tail-out for all pools.
Reported in: Table 8; Graph 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: None

Attribute	Description
N Of Pool tail Silt/Clay Substrate	Number of units with a Pool tail Substrate of Silt/Clay (value A)
N Of Pool tail Sand Substrate	Number of units with a Pool tail Substrate of Sand (value B)
N Of Pool tail Gravel Substrate	Number of units with a Pool tail Substrate of Gravel (value C)
N Of Pool tail Small Cobble Substrate	Number of units with a Pool tail Substrate of Small Cobble (value D)
N Of Pool tail Large Cobble Substrate	Number of units with a Pool tail Substrate of Large Cobble (value E)
N Of Pool tail Boulder Substrate	Number of units with a Pool tail Substrate of Boulder (value F)
N Of Pool tail Bedrock Substrate	Number of units with a Pool tail Substrate of Bedrock (value G)
N Of Total Pool tail Substrate Values	The total count of all Pool tail Substrate Values
% Silt/Clay Pool tail Substrate	Number of units with a Pool tail Substrate of Silt/Clay (value A) divided by the total count of all Pool tail Substrate Values
% Sand Pool tail substrate	Number of units with a Pool tail Substrate of Sand (value B) divided by the total count of all Pool tail Substrate Values
% Gravel Pool tail Substrate	Number of units with a Pool tail Substrate of Gravel (value C) divided by the total count of all Pool tail Substrate Values
% Small Cobble Pool tail Substrate	Number of units with a Pool tail Substrate of Small Cobble (value D) divided by the total count of all Pool tail Substrate Values

% Large Cobble Pool tail Substrate	Number of units with a Pool tail Substrate of Large Cobble (value E) divided by the total count of all Pool tail Substrate Values
% Boulder Pool tail Substrate	Number of units with a Pool tail Substrate of Boulder (value F) divided by the total count of all Pool tail Substrate Values
% Bedrock Pool tail Substrate	Number of units with a Pool tail Substrate of Bedrock (value G) divided by the total count of all Pool tail Substrate Values

Shelter Value – Shelter value for the surveys is entered based on the number code (0 to 3) that corresponds to the dominant instream shelter type that exists in the unit (Part III- Instream Shelter Complexity).
Reported in:
Inclusions: shelter value >= 0 and cover >=0
Used in Calculations: Shelter Rating

Attribute	Description
N Of Shelter Values	For the units that had a shelter value >= 0, the number of shelter values
Sum Shelter Value	For the units that had a shelter value >= 0, the sum of shelter values
Mean Shelter Value	For the units that had a shelter value >= 0, the sum of shelter values divided by the number of shelter values

Percent Shelter Cover – Percent shelter cover for the surveys is the percentage of the stream area that is influenced by instream shelter cover.
Reported in: Table 2 and Table 8
Inclusions: Unit Cover >= 0
Used in Calculations: Shelter Rating

Attribute	Description
N Of Shelter Cover	Number of shelter cover values that were >= 0
Sum Of Shelter Cover	For those units classified with a shelter cover >= 0, take the sum of all shelter cover values
Mean Shelter Cover %	For those units classified with a shelter cover > 0, take the sum of all cover values and divide by the number of shelter cover values that were > 0

Shelter Rating – The product of shelter value multiplied by the percent shelter cover of the unit.
Reported in: Table 1, 2, 3, and 8
Inclusions: shelter value >= 0 and shelter cover >=0
Used in Calculations:

Attribute	Description
N Of Shelter Rating	For the units that had a shelter value >= 0, the number of shelter values
Sum Shelter Rating	For the units that had a shelter value >= 0, the sum of (shelter values times cover)
Mean Shelter Rating	For the units that had a shelter value >= 0, the sum of (shelter values times cover) divided by the number of shelter ratings

Instream Shelter – Instream shelter for the surveys is entered based on the percentage of the unit occupied by the instream shelter types. The totals per unit will equal 100 percent. Note: bubble curtain includes white water.
Reported in: Table 5 and 8; Graph 7 and 10

Inclusions: Unit Mean Width > 0 feet
Used in Calculations: LWD for Table 8

Attribute	Description
N Of Percent Cover	For those units with a shelter value > 0, summed the number of units with shelter values
Mean % Undercut Banks Cover	For those units with a mean width value > 0, summed the values for undercut bank cover and divided by the total number of percent cover values
Mean % SmallWood Cover	For those units with a mean width value > 0, summed the values for small wood cover and divided by the total number of percent cover values
Mean % LargeWood Cover	For those units with a mean width value > 0, summed the values for large wood cover and divided by the total number of percent cover values
Mean % RootMass Cover	For those units with a mean width value > 0, summed the values for root mass cover and divided by the total number of percent cover values
Mean % TerrestrialVeg Cover	For those units with a mean width value > 0, summed the values for terrestrial vegetation cover and divided by the total number of percent cover values
Mean % AquaticVeg Cover	For those units with a mean width value > 0, summed the values for aquatic vegetation cover and divided by the total number of percent cover values
Mean % WhiteWater Cover	For those units with a mean width value > 0, summed the values for whitewater cover and divided by the total number of percent cover values
Mean % Boulder Cover	For those units with a mean width value > 0, summed the values for boulder cover and divided by the total number of percent cover values
Mean % Bedrock Ledges Cover	For those units with a mean width value > 0, summed the values for bedrock cover and divided by the total number of percent cover values
% No Shelter Cover	100 minus the sum of all cover types

Substrates Composition – Substrate composition for the surveys tracks the dominant substrate (1) and co-dominant substrate (2). Note: changes in the dominant and co-dominant substrate may indicate that the channel type has changed.
Reported in: Table 6; Graph 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
N Of Dominant Substrate Values	Total number of dominant substrate values of units with substrate values > 0
Sum Of Silt/Clay Dominant Values	For those units with a mean width value > 0, summed the values of silt/clay
% Total Silt/Clay Dominant	For those units with a mean width value > 0, summed the values of silt/clay and divided by the total number of units with substrate values > 0
Sum Of Sand Dominant Values	For those units with a mean width value > 0, summed the values of sand
% Total Sand Dominant	For those units with a mean width value > 0, summed the values of sand and divided by the total number of units with substrate values >

	0
Sum Of Gravel Dominant Values	For those units with a mean width value > 0, summed the values of gravel
% Total Gravel Dominant	For those units with a mean width value > 0, summed the values of gravel and divided by the total number of units with substrate values > 0
Sum Of Small Cobble Dominant Values	For those units with a mean width value > 0, summed the values of small cobble
% Total Small Cobble Dominant	For those units with a mean width value > 0, summed the values of small cobble and divided by the total number of units with substrate values > 0
Sum Of Large Cobble Dominant Values	For those units with a mean width value > 0, summed the values of large cobble
% Total Large Cobble Dominant	For those units with a mean width value > 0, summed the values of large cobble and divided by the total number of units with substrate values > 0
Sum Of Boulder Dominant Values	For those units with a mean width value > 0, summed the values of boulder
% Total Boulder Dominant	For those units with a mean width value > 0, summed the values of boulder and divided by the total number of units with substrate values > 0
Sum Of Bedrock Dominant Values	For those units with a mean width value > 0, summed the values of Bedrock
% Total Bedrock Dominant	For those units with a mean width value > 0, summed the values of bedrock and divided by the total number of units with substrate values > 0

Percent Total Canopy – Percent total canopy for the surveys is the percentage of the stream area that is influenced by the tree canopy. The canopy is measured using a spherical densiometer at the center of each habitat unit.
Reported in: Table 8; Graph 9
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
N Of Canopy Cover	Number of canopy cover values that were >= 0
Sum Of Canopy Cover	For those units classified with a canopy cover >= 0, take the sum of all canopy cover values
Mean % Canopy	For those units classified with a canopy cover > 0, take the sum of all canopy cover values and divide by the sum of canopy cover values that were > 0

Percent Hardwood and Coniferous Trees - Percent hardwood and coniferous trees for the surveys estimates the percent of the total canopy consisting of Broadleaf and coniferous trees. Note: there are semantic differences in some of the terms for this category. Broadleaf, Hardwood and Deciduous are synonymous and Evergreen is synonymous with Coniferous.
Reported in: Table 7, 8; Graph 9
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
N Of Canopy > 0	Number of canopy cover values that were > 0
Sum Of Deciduous Cover	For those units classified with a canopy cover > 0, take the sum of all deciduous cover values

Sum Of Coniferous Cover	For those units classified with a canopy cover > 0, take the sum of all coniferous or evergreen cover values
Mean Percent Hardwood	For those units classified with a canopy cover > 0, take the sum of all deciduous cover values and divide by the number of canopy cover values that were > 0
Mean Percent Conifer	For those units classified with a canopy cover > 0, take the sum of all coniferous cover values and divide by the number of canopy cover values that were > 1
Sum Of Open Cover	Number of canopy cover values that were = 0
Mean Percent Open Units	For those units with a canopy cover > 0, take the sum of all open cover values and divide by the number of canopy cover values that were > 0
Percent Mean Open Canopy Graph 9	For those units with a % mean canopy >0, take 100 - % mean cover
Percent Mean Coniferous Canopy Graph 9	For those units with a % coniferous > 0, take % mean cover multiplied by the % coniferous divided by 100
Percent Mean Deciduous Canopy Graph 9	For those units with a % deciduous > 0, take % mean cover multiplied by the % deciduous divided by 100

Bank Composition - Bank Composition for the surveys enter the number (1 through 4) for the dominant bank composition type as observed at the bankfull discharge level corresponding to the list located on the lower left hand side of the form. Enter one number only.
Reported in: Table 8 and 9; Graph 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
Number of Bedrock Units Right Bank	Count the number of units with a right bank composition of Bedrock (value 1)
Number of Bedrock Units Left Bank	Count the number of units with a Left bank composition of Bedrock (value 1)
Number of Boulder Units Right Bank	Count the number of units with a right bank composition of Boulder (value 2)
Number of Boulder Units Left Bank	Count the number of units with a Left bank composition of Boulder (value 2)
Number of Cobble/Gravel Units Right Bank	Count the number of units with a right bank composition of Cobble/Gravel (value 3)
Number of Cobble/Gravel Units Left Bank	Count the number of units with a Left bank composition of Cobble/Gravel (value 3)
Number of Sand/Silt/Clay Units Right Bank	Count the number of units with a right bank composition of Sand/Silt/Clay (value 4)
Number of Sand/Silt/Clay Units Left Bank	Count the number of units with a Left bank composition of Sand/Silt/Clay (value 4)
Total Mean (%) Bedrock	For those units with a composition value, summed the right and left banks unit counts for bedrock (value 1) and divided this value by the total number of composition values
Total Mean (%)	For those units with a composition value, summed the right and left

Boulder	banks unit counts for Boulder (value 2) and divided this value by the total number of composition values
Total Mean (%) Cobble/Gravel	For those units with a composition value, summed the right and left banks unit counts for Cobble/Gravel (value 3) and divided this value by the total number of composition values
Total Mean (%) Sand/Silt/Clay	For those units with a composition value, summed the right and left banks unit counts for Sand/Silt/Clay (value 4) and divided this value by the total number of composition values

Bank Dominant Vegetation - Bank Composition for the surveys enter the number (5 through 9) for the dominant vegetation type, from bankfull to 20 feet upslope, corresponding to the list located on the lower left hand side of the form. Enter one number only.
Reported in: Table 8 and 9; Graph 11
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
Number of Grass Units Right Bank	Number of units with a right bank Dominant Vegetation of Grass (value 5)
Number of Grass Units Left Bank	Number of units with a Left bank Dominant Vegetation of Grass (value 5)
Number of Brush Units Right Bank	Number of units with a right bank Dominant Vegetation of Brush (value 6)
Number of Brush Units Left Bank	Number of units with a Left bank Dominant Vegetation of Brush (value 6)
Number of Hardwood Tree Units Right Bank	Number of units with a right bank Dominant Vegetation of Hardwood (value 7)
Number of Hardwood Tree Units Left Bank	Number of units with a Left bank Dominant Vegetation of Hardwood (value 7)
Number of Coniferous Tree Units Right Bank	Number of units with a right bank Dominant Vegetation of Coniferous Trees (value 8)
Number of Coniferous Tree Units Left Bank	Number of units with a Left bank Dominant Vegetation of Coniferous Trees (value 8)
Number of No Vegetation Units Right Bank	Number of units with a right bank Dominant Vegetation of No Vegetation (value 9)
Number of No Vegetation Units Left Bank	Number of units with a Left bank Dominant Vegetation of No Vegetation (value 9)
Total Mean (%) Grass	For those units with a Dominant Vegetation value, summed the right and left banks unit counts for Grass (value 5) and divided this value by the total number of Dominant Vegetation values
Total Mean (%) Brush	For those units with a Dominant Vegetation value, summed the right and left banks unit counts for Brush (value 6) and divided this value by the total number of Dominant Vegetation values
Total Mean (%) Hardwood Trees	For those units with a Dominant Vegetation value, summed the right and left banks unit counts for Hardwood (value 7) and divided this value by the total number of Dominant Vegetation values
Total Mean (%) Coniferous Trees	For those units with a Dominant Vegetation value, summed the right and left banks unit counts for Coniferous Trees (value 8) and divided this value by the total number of Dominant Vegetation values
Total Mean (%) No Vegetation	For those units with a Dominant Vegetation value, summed the right and left banks unit counts for No Vegetation (value 9) and divided this value by the total number of Dominant Vegetation values

Percent Veg Cover	The sum of right and left bank values divided by the total number of left and right bank values

Percent Bank Vegetated – Estimate the total percentage of the bank covered with vegetation from the bankfull discharge elevation to 20 feet upslope.
Reported in: Table 7 and Table 8; Graph 9
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
N Of Right Bank Cover	Number of right bank cover values that were >= 0
N Of Left Bank Cover	Number of left bank cover values that were >= 0
Sum Of Right Bank Cover	For those units with a right bank cover value > 0, take the sum of all right bank cover values
Sum Of Left Bank Cover	For those units with a left bank cover value > 0, take the sum of all left bank cover values
Mean Right Bank % Cover	For those units with a right bank cover value > 0, take the sum of all right bank cover values and divide by the total number of both left and right bank cover values > 0
Mean Left Bank % Cover	For those units with a left bank cover value > 0, take the sum of all left bank cover values and divide by the total number of both left and right bank cover values > 0

Application Table: Habitat Criteria – Select stream habitat criteria that can be used to evaluate stream condition.

The "habitat criteria" table contains additional metrics and habitat criteria that can be used to evaluate stream condition. The criteria have been gleaned from numerous plans and sources. For a list of sources contact Derek Acomb (note contact information page 2). The "habitat criteria" table provides the metrics at both the stream and the reach level (StreamOrReach field).

Example Record
What are we looking at – Definition or explanation
Reported in: *Where in the stream habitat program outputs do these values appear*
Inclusions: *What is included in the calculations*
Used in Calculations: *Where is this information used in calculations*

Attribute	Description
Field Name	Description of field name (if necessary) and calculation

General Information
This section contains basic information about the stream habitat survey such as the Site ID, site name, stream name, year of record, the duration of the sample, etc.

Attribute	Description
SurveyId	Survey Identification Number
Pname	Stream Name
Pnmcd	Stream Number
StrOrRch	Code used to delineate whether the measurements are at the stream or reach level
Code	Stream code or ReachID depending on StreamOrReach Value
Year	Year of Survey

Channel Type - Rosgen channel type classification. The channel type of the reach or stream based on the Stream Channel Type Work Sheet (Part III)
Reported in: Table 8
Inclusions:
Used in Calculations:

Attribute	Description
Chnl_Type	Rosgen channel type classification. The channel type of the reach or stream based on the Stream Channel Type Work Sheet (Part III)

Stream Order - The Strahler Stream Order is a simple hydrology algorithm used to define stream size based on a hierarchy of tributaries.
Reported in:
Inclusions:
Used in Calculations: Primary pool and mean residual depth by nth stream order calculations.

Attribute	Description
StrOrMin	The minimum stream order of the stream or reach. Stream order is calculated based on the Shreve ordering system.
StrOrMax	The maximum stream order of the stream or reach. Stream order is calculated based on the Shreve ordering system.
StrOrMaj	The majority stream order of the stream or reach. Stream order is calculated based on the Shreve ordering system.

Temperature Data - Temperature of the water and air taken during the surveys. Temperatures are taken at the beginning of each page record and recorded to the nearest degree Fahrenheit. Temperatures are taken in the shade and within one foot of the water surface.
Reported in: Table 8
Inclusions:
Used in Calculations: Temperature values > 0

Attribute	Description
WtempMin	For those water temperatures greater than zero, the minimum water temperature during survey
WtempMax	For those water temperatures greater than zero, the maximum water temperature during survey
WtempAve	For those water temperatures greater than zero, the average water temperature during survey
AtempMin	For those air temperatures greater than zero, the minimum air temperature during survey
AtempMax	For those air temperatures greater than zero, the maximum air temperature during survey
AtempAve	For those air temperatures greater than zero, the average air temperature during survey

Pool Tail Embeddedness - Percent cobble embeddedness is determined at pool tail-outs where spawning is likely to occur. Sample at least five small cobbles (2.5" to 5.0") in diameter and estimate the amount of the stone buried in the sediment.

This is done by removing the cobble from the streambed and observing the line between the "shiny" buried portion and the duller exposed portion. Estimate the percent of the lower shiny portion using the corresponding number for the 25% ranges. Average the samples for a mean cobble embeddedness rating. Additionally, a value of 5 is assigned to tail-outs deemed unsuited for spawning due to inappropriate substrate particle size, having a bedrock tail-out, or other considerations:

Reported in: Table 8 and 9; Graph 6
Inclusions: Unit Mean Width > 0 feet, with embeddedness > 0
Used in Calculations:

Attribute	Description
MeanEmb	Mean Embeddedness Integer, For those units classified as pool, the sum of Embeddedness value of > 0 divided by the total number of Embeddedness Values > 0, converted to an integer value
DomEmb	Dominant Embeddedness Value(s), the most common embeddedness value, there may be more then one dominant value showing co-dominance.
EmbRange	Embeddedness Range of Value(s)
PerEmb12_pn	Percent Pools Embeddedness 1 and 2, the number of value 1 and 2 embeddedness values in pools, divided by the total number of embeddedness values in pools.
PerEmb12_sn	Percent Pools Embeddedness 1 and 2, the number of value 1 and 2 embeddedness values in pools, divided by the total number of habitat units in the stream.
PerEmb12_pl	Percent Pools Embeddedness 1 and 2 by length, the total length of value 1 and 2 embeddedness values in pools, divided by the total length of pools.

PerEmb12_sl	Percent Pools Embeddedness 1 and 2 by length by Stream, the total length of value 1 and 2 embeddedness values in pools, divided by the total length of the surveyed stream.
PerEmb34_pn	Percent Pools Embeddedness 3 and 4, the number of value 3 and 4 embeddedness values in pools, divided by the total number of embeddedness values in pools.
PerEmb34_sn	Percent Pools Embeddedness 3 and 4, the number of value 3 and 4 embeddedness values in pools, divided by the total number of habitat units in the stream.

Mean Residual Depth by Stream Order – Residual depth is the mean depth of the pools minus the pool tail crest depth.
Reported in:
Inclusions: Mean width > 0 feet
Used in Calculations:

Attribute	Description
MnResDpth1	Mean Residual depth of first order streams pools for the units that were fully surveyed and not null, the sum of mean depth values - pool tail crest depth value
MnResDpth2	Mean Residual depth of second order streams pools for the units that were fully surveyed and not null, the sum of mean depth values - pool tail crest depth value
MnResDpth3	Mean Residual depth of third order streams pools for the units that were fully surveyed and not null, the sum of mean depth values - pool tail crest depth value
MnResDpth4	Mean Residual depth of fourth order streams pools for the units that were fully surveyed and not null, the sum of mean depth values - pool tail crest depth value

Riffles - Shallow stretch of a river or stream, where the current is above the average stream velocity and where the water forms small rippled waves as a result. It often consists of a rocky bed of gravels or cobbles. This portion of a stream is often an important habitat for small aquatic invertebrates and juvenile fishes.
Reported in:
Inclusions:
Used in Calculations:

Attribute	Description
PerDomRif_n	Dominant Riffle Substrate Percent, the percent of most common Riffle Substrate value.
DomRifSub	Dominant Riffle Substrate Value(s), the most common Riffle Substrate value, there may be more than one dominant value showing co-dominance.
PerRif_l	Riffle Length Percent, Sum of lengths for riffle habitat types divided by the total length of all habitat units
RifRange_l	Riffle Substrate Range of Value(s)

Low-Gradient Riffle (LGR) – Shallow reaches with flowing, turbulent water with some partially exposed substrate. Gradient < 4%, substrate is usually cobble dominated.
Reported in:
Inclusions:

Used in Calculations:

Attribute	Description
PerDomLGR	Dominant LGR Substrate Percent, the percent of most common LGR Substrate value.
DomLGRVal	Dominant LGR Substrate Value(s), the most common LGR Substrate value, there may be more than one dominant value showing co-dominance.
LGRRngVal	LGR Substrate Range of Value(s)

Mean Shelter Value - Shelter value for the surveys is entered based on the number code (0 to 3) that corresponds to the dominant instream shelter type that exists in the unit (Part III- Instream Shelter Complexity).
Reported in:
Inclusions: shelter value >= 0 and Shelter Cover >=0
Used in Calculations: Shelter Rating

Attribute	Description
MnShVal_s	Mean Shelter Value Stream, for the units that had a shelter value >= 0, the sum of shelter values divided by the number of shelter values.
MnShVal_p	Mean Shelter Value Pools, for the units that had a shelter value >= 0, the sum of shelter values divided by the number of shelter values in pools.

Mean Percent Shelter Cover - Percent shelter cover for the surveys is the percentage of the stream area that is influenced by instream shelter cover.
Reported in: Table 2 and Table 8
Inclusions: Unit Shelter Cover >= 0
Used in Calculations: Shelter Rating

Attribute	Description
PerMnCov_s	Mean percent shelter cover, for those units classified with a cover > 0, take the sum of all cover values and divide by the number of cover values that were > 0
PerMnCov_p	Mean percent shelter cover, for those pool units classified with a cover > 0, take the sum of all cover values and divide by the number of pool cover values that were > 0

Mean Shelter Rating – The product of Shelter Value multiplied by the Percent unit covered.
Reported in: Table 1, 2, 3, and 8
Inclusions: shelter value >= 0 and Shelter Cover >=0
Used in Calculations:

Attribute	Description
MnShRat_s	Mean Shelter Rating Stream, for the units that had a shelter ratings >= 0, the sum of shelter ratings divided by the number of shelter ratings.
MnShRat_p	Mean Shelter Rating Pools, for the units that had a shelter ratings >= 0, the sum of shelter ratings divided by the number of shelter ratings in pools.

Percent Total Canopy – Percent total canopy for the surveys is the percentage of the stream area that is influenced by the tree canopy. The canopy is measured using a spherical densiometer at the center of each habitat unit.
Reported in: Table 8; Graph 9
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
PerMnCan_s	Percent total canopy, for those units classified with a canopy > 0, take the sum of all canopy values and divide by the number of canopy values that were > 0
PerMnCan_p	Percent total canopy of pools, for those pool units classified with a canopy > 0, take the sum of all canopy values and divide by the number of pool canopy values that were > 0

Mean Maximum Depth by Stream Order - Enter the measured maximum depth for each habitat unit, in feet. Mean maximum depth for the surveys is defined as the mean of the maximum depth measurements. Mean maximum depths for pools are the mean maximum residual depths (mean maximum depth value minus the pool tail crest value).
Reported in: Table 1,4 and 8; Graph 5
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
AveMxDpth12	Mean Maximum Depth of 1 and 2 order streams, for the units that were fully surveyed and not null, the number of residual max depth values divided by the total number of residual max depth values
AveMxDpth34	Mean Maximum Depth of 3 and 4 order streams, for the units that were fully surveyed and not null, the number of residual max depth values divided by the total number of residual max depth values

Percent Maximum Pool Depths by Strata – The percent of pools with maximum residual depths in two strata (greater than or equal to 2 feet and greater than or equal to 3 feet).
Reported in: Table 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
PerPoolMxDgt1	Pool Max Depth >= 2 feet Percent Pool Freq
PerPoolMxDgt2	Pool Max Depth >= 3 feet Percent Pool Freq

Residual Pool Depths by Strata – The number and the percent of pools with maximum residual depths in two strata (greater than or equal to 2 feet and greater than or equal to 3).
Reported in: Table 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
PerPoolResDgt1	Residual Pool Depth >= 2 feet Percent Pool Freq
PerPoolResDgt2	Residual Pool Depth >= 3 feet Percent Pool Freq

Percent Conifer Canopy – For the surveys estimates the percent of the total canopy consisting of coniferous trees.
Reported in: Table 7; Graph 9

Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
PerMnCon_s	Mean Percent Conifer, for those units classified with a canopy cover > 0, take the sum of all coniferous cover values and divide by the number of canopy cover values that were > 1

Bank Substrate – (Bank Composition) Bank substrate for the surveys enter the number (1 through 4) for the dominant bank composition type observed at the bankfull discharge elevation corresponding to the list located on the lower left hand side of the form. Enter one number only.
Reported in: Table 8 and 9; Graph 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
DomBSubType	Dominant Bank Substrate Value(s), the most common Bank Substrate value, there may be more than one dominant value showing co-dominance.
BSubRngVal	Bank Substrate Range of Value(s)

Bank Substrate Not Meeting Canopy - (Bank Composition) Bank substrate for the surveys enter the number (1 through 4) for the dominant bank composition type corresponding to the list located on the lower left hand side of the form. Enter one number only.
Reported in: Table 8 and 9; Graph 10
Inclusions: Unit Mean Width > 0 feet and Mean canopy < 80%
Used in Calculations:

Attribute	Description
DomBSubVal_nc	Dominant Bank Substrate Value(s) not meeting canopy, the most common Bank Substrate value, there may be more then one dominant value showing co-dominance.
BSubRange_nc	Bank Substrate Range of Value(s) not meeting canopy

Percent Bank Cover - Estimate the total percentage of the bank covered with vegetation from the bankfull discharge elevation to 20 feet upslope.
Reported in: Table 7 and Table 8; Graph 9
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
PerMnBCov_s	The sum of right and left bank values divided by the total number of left and right bank values

Substrates Composition – Substrate composition for the surveys tracks the dominant substrate (1) and co-dominant substrate (2). Note: changes in the dominant and co-dominant substrate may indicate that the channel type has changed.
Reported in: Table 6; Graph 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
PerDomSub	Substrate Dominant Percent
DomSubVal	Substrate Dominant Value(s)

SubRange	Substrate Range

Pool tail Substrate - Pool substrate for the surveys is entered based on the code (A through G) for the dominant substrate composition of tail-out for all pools.
Reported in: Table 8; Graph 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: None

Attribute	Description
PerDomPTSub	Dominant Pool tail Substrate Percent
DomPTSubVal	Dominant Pool tail Substrate Value(s)
PTSubRngVal	Pool tail Substrate Range of Value(s)

Percent Pools – The percent pools based on area, frequency, and length.
Reported in: Table 1, 2, 3, 4, and 8; Graph 1, 2, 3, and 4
Inclusions:
Used in Calculations:

Attribute	Description
PerPoolArea	Percent pools by area, the sum of pool areas in square feet divided by the total area in square feet.
PerPoolFreq	Percent pools by frequency, the number of pool habitat units divided by the total number of habitat units.
PerPoolLen	Percent pools by length, the sum of pool lengths in feet divided by the total length in feet.

Percent Primary Pools - Primary pools are defined differently based on the stream order. First through 2nd order streams primary pools have a maximum depth >=2 feet and 3rd through 4th (nth) order streams primary pools have a maximum depth >=3 feet.
Reported in:
Inclusions:
Used in Calculations:

Attribute	Description
PerPrimP_p	Percent primary pools by total pools, the sum of pools that are classified as primary pools divided by the number of pool units.
PerPrimP_s	Percent primary pools, the sum of pools that are classified as primary pools divided by the number of habitat units.

Mean Depth - Mean Depth for the surveys is defined as the mean of several random depth measurements taken with a stadia rod across the unit recorded in feet. Mean depths for pools are the mean residual depth, that is the mean depth value minus the pool tail crest value.
Reported in: Table 1, 2, and 3; Graph 5
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: All volume calculations

Attribute	Description
AveMnDepth	For pools the mean depth is the sum of residual depth (pool depths - pool tail crest) divided by the number of units fully measured, for other types it is the sum of mean depth values divided by the total number of units that were fully measured.

Application Table: Ranked Manual Criteria - Evaluation of selected California Department of Fish and Game restoration manual criteria based on selected "Habitat Criteria" table fields.

The "ranked manual criteria" table contains information about 6 criteria that some biologist feel are important for salmonids in the region. The table provides that boolean score, depending on whether they do (value 1) or do not meet (value 1) the criteria. The seventh value in the table is the numeric sum of criteria Scores by each reach or stream. The table provides the metrics at both the stream and the reach level (StreamOrReach field).

Example Record
Criteria
Criteria from: *Where does the criteria come from.*

Attribute	Description
Field Name	Description of field name (if necessary) and ranking criteria

General Survey Information

This section contains basic information about the stream habitat survey such as the Site ID, site name, stream name, year of record, the duration of the sample, etc.

Attribute	Description
SurveyId	Survey Identification Number
Pname	Stream Name
Pnmcd	Stream Number
StrOrRch	Code used to delineate whether the measurements are at the stream or reach level
Code	Stream code or ReachID depending on StreamOrReach Value
Year	Year of Survey

Percent Primary Pools (Length)

Criteria from: California Salmonid Stream Habitat Restoration Manual VI-6, V-15

Attribute	Description
PerPrimP_s	Percent Primary Pools, if the percent primary pools of the stream was >= 45% a value of one was assigned, if the percent of primary pools was < 45% a value of zero was assigned.

Mean Embeddedness

Criteria from: California Salmonid Stream Habitat Restoration Manual VI-8

Attribute	Description
MeanEmb	Mean Embeddedness, if the Mean Embeddedness of the stream was <= 1 a value of one was assigned, if the Mean Embeddedness was > 1 a value of zero was assigned.

Mean Canopy Cover of the Stream

Criteria from: California Salmonid Stream Habitat Restoration Manual VI-7and V-22

Attribute	Description
PerMnCan_s	Mean Canopy Cover of the Stream, if the Mean Canopy Cover of the Stream was >= 80% a value of one was assigned, if the Mean Canopy Cover of the Stream was < 80% a value of zero was assigned.

Mean Shelter Rating of Pools

Criteria from: California Salmonid Stream Habitat Restoration Manual VI-7and V-15

Attribute	Description
MnShRat_p	Mean Shelter Rating of Pools, if the Mean Shelter Rating of Pools in the stream was >= 80% a value of one was assigned, if the Mean Shelter Rating of Pools in the stream was < 80% a value of zero was assigned.

Coho Salmon Temperature

Criteria from: California Salmonid Stream Habitat Restoration Manual V-21

Attribute	Description
CohoTemp	Assigned a value of 1 if temperature between 48-60˚ F, a value of zero was assigned if the temperature was not within this range.

Steelhead Salmon Temperature

Criteria from: California Salmonid Stream Habitat Restoration Manual V-22 and V-23

Attribute	Description
SHTemp	Assigned a value of 1 if temperature between 40-65˚ F, a value of zero was assigned if the temperature was not within this range

Stream Rating – Based on the six criteria mentioned above

Attribute	Description
Criteria_cnt	Total of the six values in the criteria table, the higher the final count the more suitable the stream may be for salmonids.

Application Table: Reachsum_x – Based on report table 8

The "reachsum_x" table contains all of the metrics in the Stream Habitat Program table number 8. The "reachsum_x" table provides the metrics at the reach level. This table is being replaced by the other tables produced by the Stream Summary Application. The table will directly join to the GIS data mentioned in the introduction on Page 1.

Example Record
What are we looking at – Definition or explanation
Reported in: *Where in the stream habitat program outputs do these values appear*
Inclusions: *What is included in the calculations*
Used in Calculations: *Where is this information used in calculations*

Attribute	Description
Field Name	Description of field name (if necessary) and calculation

General Survey Information
This section contains basic information about the stream habitat survey such as the Site ID, site name, stream name, year of record, the duration of the sample, etc.

Attribute	Description
StreamName	Stream name as recorded in the reachsum database.
LLID	Latitude-Longitude identifier of stream
Reach	Reach number (standardized to two digits, i.e. 01, 02, etc.).
ReachLLId	Alternative unique reach identifier, based on Llid
St_unit	Starting (minimum), main channel or primary side channel, habitat unit number.
End_unit	Ending (maximum), main channel or primary side channel, habitat unit number.

Channel Type - Rosgen channel type classification. The channel type of the reach or stream based on the Stream Channel Type Work Sheet (Part III)
Reported in: Table 8
Inclusions:
Used in Calculations:

Attribute	Description
Chan_typ	Rosgen channel type classification.

Length of Survey - Thalweg length of the habitat unit, in feet.
Reported in: Table 1,2,3, and 8; Graph 2
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Area, Mean Volume, Mean Residual Pool Volume,*** All Area, Pool depth, and volume calculations.

Attribute	Description
Chan_len	Total length of all main channel habitat units.
Side_len	Total length of all side channel habitat units.

Riffle/Flatwater Mean Width (ft) - Riffle/Flatwater Mean Width for the surveys is defined as the mean of two or more wetted channel widths measurements in feet within the habitat unit.
Reported in: Table 8
Inclusions: Unit Mean Width > 0 feet

Used in Calculations: ***Mean Depth*** and volume calculations

Attribute	Description
Rf_fl_wdth	Average of the surveyed mean width for main channel riffle and flatwater habitat units (habitat types 1.x, 2.x and 3.x). Average not weighted by habitat unit length.

Mean Pool Depth - Mean pool depth for the surveys is defined as the mean of several random depth measurements using a stadia rod and recorded in feet. Mean depths for pools are the mean residual depth, that is the mean depth value from the survey minus the pool tail crest value.
Reported in: Table 8
Inclusions: shelter value >= 0 and cover >=0
Used in Calculations: Shelter Rating

Attribute	Description
Pool_dpth	Average of the surveyed mean depth for main channel pool habitat units (habitat types 4.x, 5.x and 6.x). Average not weighted by pool area.

Base Flow (cfs) - The base flow is the flow that the stream reduces to during the dry season or a dry spell. This flow is supported by ground water and subsurface seepage into the channel.
Reported in: Table 8
Inclusions:
Used in Calculations:

Attribute	Description
Flow	The mean base flow in cubic feet per second, measured at the beginning of the survey. If flows change significantly during the survey they are again measured at the end of the survey at the same location. The average of the two measurements is recorded.

Temperature Data - Temperature of the water and air taken during the surveys. Temperatures are taken at the beginning of each page record and recorded to the nearest degree Fahrenheit. Temperatures are taken in the shade and within one foot of the water surface.
Reported in: Table 8
Inclusions:
Used in Calculations: Temperature values > 0

Attribute	Description
Lwater	Minimum surveyed water temperature ˚F
Uwater	Maximum surveyed water temperature ˚F
Lair	Minimum surveyed air temperature ˚F
Uair	Maximum surveyed air temperature ˚F

Bank Dominant Vegetation - Bank Vegetation for the surveys enter the number (5 through 9) for the dominant vegetation type, from bankfull to 20 feet upslope, corresponding to the list located on the lower left hand side of the form. Enter one number only. The dominant bank vegetation of the reach is highlighted.

Reported in: Table 8 and 9; Graph 11
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description

Dom_bk_veg	Vegetation class (Grass, Brush, Deciduous Trees, Coniferous Trees or No Vegetation) most frequently identified as dominant vegetation type in habitat units surveyed for dominant vegetation.

Percent Vegetative Cover – Average percent vegetative cover for habitat units surveyed for vegetative cover.
Reported in: Table 8
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
Veg_cov	Average percent vegetative cover for habitat units surveyed for vegetative cover. Average not weighted.

Dominant Bank Composition – Bank Composition for the surveys enter the number (1 through 4) for the dominant bank composition type corresponding to the list located on the lower left hand side of the form. Enter one number only. The dominant bank composition reach is highlighted.
Reported in: Table 8 and 9; Graph 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations:

Attribute	Description
Dom_bk_sub	Bank substrate class (Bedrock, Boulder, Cobble/Gravel or Silt/Clay/Sand) most frequently identified as dominant bank substrate in habitat units surveyed for bank composition.

Pool Tail Embeddedness - Percent cobble embeddedness is determined at pool tail-outs where spawning is likely to occur. Sample at least five small cobbles (2.5" to 5.0“) in diameter and estimate the amount of the stone buried in the sediment.

This is done by removing the cobble from the streambed and observing the line between the "shiny“ buried portion and the duller exposed portion. Estimate the percent ofthe lower shiny portion using the corresponding number for the 25% ranges. Average the samples for a mean cobble embeddedness rating. Additionally, a value of 5 is assigned to tail-outs deemed unsuited for spawning due to inappropriate substrate particle size, having a bedrock tail-out, or other considerations:

Reported in: Table 8 and 9; Graph 6
Inclusions: Unit Mean Width > 0 feet, with embeddedness > 0
Used in Calculations:

Attribute	Description
Emb_one	Percentage of main channel pool tail-outs, surveyed for embeddedness and containing suitable spawning substrate (not classified with pool tail embeddedness = 5), with an embeddedness classification of 1 (0% to 25% embeddedness).
Emb_two	Percentage of main channel pool tailouts, surveyed for embeddedness and containing suitable spawning substrate (not classified with pool tail embeddedness = 5), with an embeddedness classification of 2 (25% to 50% embeddedness).
Emb_three	Percentage of main channel pool tailouts, surveyed for embeddedness and containing suitable spawning substrate (not classified with pool tail embeddedness = 5), with an embeddedness classification of 3 (50% to 75% embeddedness).

Emb_four	Percentage of main channel pool tailouts, surveyed for embeddedness and containing suitable spawning substrate (not classified with pool tail embeddedness = 5), with an embeddedness classification of 4 (75% to 100% embeddedness).

Percent Hardwood and Coniferous Trees - Percent hardwood and coniferous trees for the surveys estimates the percent of the total canopy consisting of Broadleaf and coniferous trees. Note: there are semantic differences in some of the terms for this category. Broadleaf, Hardwood and Deciduous are synonymous and Evergreen is synonymous with Coniferous.
Reported in: Table 7, 8; Graph 9
Inclusions: Unit Canopy >= 0
Used in Calculations:

Attribute	Description
Canopy	Average canopy density for habitat units surveyed for canopy cover. Average not weighted.
Conif	Average percent evergreen canopy for habitat units surveyed for canopy cover. Average not weighted.
Decid	Average percent deciduous canopy for habitat units surveyed for canopy cover. Average not weighted.

Mean Length - Length for the surveys is defined as the thalweg length of the habitat unit, in feet.
Reported in: Table 1, 2, 3 and 8; Graph 2
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Area, Mean Volume, Mean Residual Pool Volume,*** All Area, Pool depth, and volume calculations

Attribute	Description
Pct_pls_ln	Percent of main channel, by length, composed of pools (habitat types 4.x, 5.x and 6.x). Includes dry (habitat type 7.0) and recorded but not non-surveyed (habitat type 9.x) habitat units.
Dry	Total length of main channel habitat units surveyed as Dry (habitat type = 7.0).
Wet	Total length of main channel habitat units not surveyed as Dry (habitat type = 7.0). Units recorded, but not surveyed (habitat types 9.0 and 9.1), are not included in this total.

Residual Pool Depths by Strata – The number and the percent of pools with residual depths in two strata (greater than or equal to 2 feet, greater than or equal to 3 feet).
Reported in: Table 8
Inclusions: shelter value >= 0 and cover >=0
Used in Calculations: Shelter Rating

Attribute	Description
Pools_2ft	Percent of main channel pools (habitat types 4.x, 5.x and 6.x) greater than, or equal to, two feet deep.
Pools_3ft	Percent of main channel pools (habitat types 4.x, 5.x and 6.x) greater than, or equal to, three feet deep.

Shelter Rating of Pools – The product of shelter value multiplied by the percent shelter cover of the pool unit.
Reported in: Table 1, 2, 3, and 8
Inclusions: shelter value >= 0 and cover >=0

Used in Calculations:

Attribute	Description
Pol_sh_rtn	Average shelter rating (ShelterValue x Cover) for main channel pools surveyed for in-stream shelter.

Dominant Instream shelter – Instream shelter for the surveys is entered based on the percentage of the unit occupied by the instream shelter types. The totals per unit will equal 100 percent. Note: bubble curtain includes white water. The dominant instream shelter of the reach is highlighted.
Reported in: Table 5 and 8; Graph 7 and 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: LWD for Table 8

Attribute	Description
Dom_shel	Shelter type (Undercut Banks, Small Woody Debris, Large Woody Debris, Root Masses, Terrestrial Vegetation, Aquatic Vegetation, White Water, Boulders and Bedrock Ledges) representing highest total percent composition of instream shelter in all habitat units surveyed.

Riffle/Flatwater Mean Width (ft) - Riffle/Flatwater Mean Width for the surveys is defined as the mean of two or more wetted channel widths measured within the habitat unit and recorded in feet.
Reported in: Table 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Depth,*** volume calculations

Attribute	Description
Rf_fl_mean	Weighted average of the surveyed mean width for main channel riffle and flatwater habitat units (habitat types 1.x, 2.x and 3.x). Average weighted by habitat unit length.

Mean Pool Area - Mean pool area is calculated for all Pool habitat types and reported in square feet. Area calculations are based on the wetted width of the habitat units, that is the mean width multiplied by the product of 1 minus the percent exposed substrate. The wetted with is than multiplied by the length.
Reported in: Table 1,2,3 and 8
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: ***Mean Volume, Mean Residual Pool Volume,*** All volume calculations

Attribute	Description
Pool_area	Proportion of main channel surface area composed of pools (habitat types 4.x, 5.x and 6.x). Pool surface area calculated as the sum of length x average width for each main channel pool. Remaining (non-pool) surface area calculated as non-pool wet length x adjusted mean riffle/flatwater width.

Instream shelter - Instream shelter for the surveys is entered based on the percentage of the unit occupied by the instream shelter types. The totals per unit will equal 100 percent. Note: bubble curtain includes white water.
Reported in: Table 5 and 8; Graph 7 and 10
Inclusions: Unit Mean Width > 0 feet
Used in Calculations: LWD for Table 8

Attribute	Description
Cov_under	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by undercut banks.
Cov_swood	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by small woody debris.
Cov_lwood	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by large woody debris.
Cov_root	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by root mass.
Cov_tveg	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by overhanging terrestrial vegetation.
Cov_aveg	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by aquatic vegetation.
Cov_water	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by white water or bubble curtain.
Cov_bould	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by boulders.
Cov_bed	The proportion of main channel pool (habitat types 4.x, 5.x and 6.x) area which is provided shelter by bedrock edges.

Large Woody Debris – Large Wood is defined as a piece of wood having a minimum diameter of twelve inches and a minimum length of six feet. Root wads must meet the minimum diameter criteria at the base of the trunk but need not be at least six feet long.
<u>Reported in:</u> Table 8
<u>Inclusions:</u> shelter value >= 0 and cover >=0
<u>Used in Calculations:</u> Shelter Rating

Attribute	Description
Lod	Percentage of habitat units containing shelter from large woody debris or root mass (LargeWood > 0 or RootMass > 0).
Lwd_pools	Number of main channel pools enhanced by large woody debris (habitat types 5.2, 5.3, 6.3 and 6.4).
Prob_lwdp	Number of main channel pools that are probably enhanced by large woody debris (habitat types 5.2, 5.3, 6.3, 6.4 and 6.5).
Pot_lwdp	Number of main channel pools that are potentially enhanced by large woody debris (habitat types 5.2, 5.3, 5.6, 6.3, 6.4 and 6.5).
Part_lwdp	The proportion of main channel pools enhanced by large woody debris (habitat types 5.2, 5.3, 6.3 and 6.4).

Application Table: NOAA_Table - The “noaa_table” table contains additional metrics and habitat criteria that can be used to evaluate stream condition for salmonids species. These criteria have been developed by NMFS planning team through literature reviews and consultation with experts in the field of salmonid ecology. The “noaa_table” table provides the metrics at both the stream and the reach level (StreamOrReach field).

Example Record
Criteria
Criteria from: *Where does the criteria come from.*

Attribute	Description
Field Name	Description of field name (if necessary) and ranking criteria

General Survey Information

This section contains basic information about the stream habitat survey such as the Site ID, site name, stream name, year of record, the duration of the sample, etc.

Attribute	Description
SurveyId	Survey Identification Number
Pname	Stream Name
StrOrRch	Code used to delineate whether the measurements are at the stream or reach level
Code	Stream code or ReachID depending on StreamOrReach Value

Spawning Substrate (Area) – The amount of spawning substrate is defined as riffle habitat directly below a primary pool that is potentially used by spawning salmonids. Primary pools are defined differently based on the stream order. First through 2nd order streams primary pools have a maximum depth >=2 feet and 3rd through 4th (nth) order streams primary pools have a maximum depth >=3 feet. The spawning substrate values are further divided by the embeddedness value of the primary pool, which is an estimate of the amount of sediment in the spawning habitat.

Attribute	Description
SpawningSub_lt5	The area of spawning substrate in square meters, where the primary pools have an embeddedness value < 5. The value is the product of the sum of the area of riffle habitat multiplied by the count of primary pools with riffles below.
spavearea_lt5	For those primary pools with embeddeness values < 5 and a riffle unit below, the area of the riffle (the mean width ^2).
spembcnt_lt5	The count of primary pools with embeddeness values < 5 and a riffle unit below.
spvalueft_lt5	The area of spawning substrate in square feet, where the primary pools have an embeddedness value < 5. The value is the product of the sum of the area of riffle habitat multiplied by the count of primary pools with riffles below.
spavearea_lt4	For those primary pools with embeddeness values < 4 and a riffle unit below, the area of the riffle (the mean width ^2).
spembcnt_lt4	The count of primary pools with embeddeness values < 4 and a riffle unit below.
SpawningSub_lt4	The area of spawning substrate in square meters, where the primary pools have an embeddedness value < 4. The value is the product of the sum of the area of riffle habitat multiplied by the count of primary pools with riffles below.
spvalueft_lt4	The area of spawning substrate in square feet, where the primary pools have an embeddedness value < 4. The value is the product of

	the sum of the area of riffle habitat multiplied by the count of primary pools with riffles below.
spavearea_lt3	For those primary pools with embeddeness values < 3 and a riffle unit below, the area of the riffle (the mean width ^2).
spembcnt_lt3	The count of primary pools with embeddeness values < 3 and a riffle unit below.
SpawningSub_lt3	The area of spawning substrate in square meters, where the primary pools have an embeddedness value < 3. The value is the product of the sum of the area of riffle habitat multiplied by the count of primary pools with riffles below.
spvalueft_lt3	The area of spawning substrate in square feet, where the primary pools have an embeddedness value < 3. The value is the product of the sum of the area of riffle habitat multiplied by the count of primary pools with riffles below.

Pool to Riffle Ratio

Attribute	Description
PR Ratio Length	The sum of pool lengths divided by the sum of riffle lengths.
PR Ratio Freq	The number of pool units divided by the number of riffle units.
Pool_L	For those pool units (habitat type >= 4 and < 7), the sum of the length of pool units
RiffleL	For those riffle units (habitat type >= 1 and < 4), the sum of the length of riffle units
RiffleF	For those riffle units (habitat type >= 1 and < 4), the sum of the number of riffle units
Pool_F	For those pool units (habitat type >= 4 and < 7), the sum of the number of pool units

Percent Total Canopy – Percent total canopy for the surveys is the percentage of the stream area that is influenced by the tree canopy. The canopy is measured using a spherical densiometer at the center of each habitat unit.

Attribute	Description
N Of Canopy Cover	Number of canopy cover values that were >= 0
Sum Of Canopy Cover	For those units classified with a canopy cover >= 0, take the sum of all canopy cover values
Mean % Canopy	For those units classified with a canopy cover > 0, take the sum of all canopy cover values and divide by the sum of canopy cover values that were > 0

Large Woody Debris – Wood debris is defined as a piece of wood having a minimum diameter of twelve inches and a minimum length of six feet. Root wads must meet the minimum diameter criteria at the base of the trunk but need not be at least six feet long.

Attribute	Description
Sum of LWD	For those units with Large Woody Debris (LWD), the sum of the number of LWD in the stream or reach
Occurrence of LWD (%)	For those units with Large Woody Debris (LWD), the sum of the percent cover of LWD in the stream or reach divided by the number of habitat units with percent canopy values in reach or stream multiplied by 100
LWD per 100 ft	For those units with Large Woody Debris (LWD), the sum of the

	number of LWD in the stream or reach divided by the number of sum length of reach or stream multiplied by 100

Instream Shelter – Instream shelter for the surveys is entered based on the percentage of the unit occupied by the instream shelter types. The totals per unit will equal 100 percent. Note: bubble curtain includes white water.

Attribute	Description
N Of Percent Cover	For those units with a shelter value > 0, summed the number of units with shelter values
Mean % Undercut Banks Cover	For those units with a mean width value > 0, summed the values for undercut bank cover and divided by the total number of percent cover values
Mean % SmallWood Cover	For those units with a mean width value > 0, summed the values for small wood cover and divided by the total number of percent cover values
Mean % LargeWood Cover	For those units with a mean width value > 0, summed the values for large wood cover and divided by the total number of percent cover values
Mean % RootMass Cover	For those units with a mean width value > 0, summed the values for root mass cover and divided by the total number of percent cover values
Mean % TerrestrialVeg Cover	For those units with a mean width value > 0, summed the values for terrestrial vegetation cover and divided by the total number of percent cover values
Mean % AquaticVeg Cover	For those units with a mean width value > 0, summed the values for aquatic vegetation cover and divided by the total number of percent cover values
Mean % WhiteWater Cover	For those units with a mean width value > 0, summed the values for whitewater cover and divided by the total number of percent cover values
Mean % Boulder Cover	For those units with a mean width value > 0, summed the values for boulder cover and divided by the total number of percent cover values
Mean % Bedrock Ledges Cover	For those units with a mean width value > 0, summed the values for bedrock cover and divided by the total number of percent cover values

Shelter Rating – The product of shelter value multiplied by the percent shelter cover of the unit.

Attribute	Description
N Of Shelter Rating	For the units that had a shelter value >= 0, the number of shelter values
Sum Shelter Rating	For the units that had a shelter value >= 0, the sum of (shelter values times cover)
Mean Shelter Rating	For the units that had a shelter value >= 0, the sum of (shelter values times cover) divided by the number of shelter ratings

Mean Depth - Mean Depth for the surveys is defined as the mean of several random depth measurements across the unit with a stadia rod in feet. Mean depths for pools are the mean residual depth, that is the mean depth value from the survey minus the pool tail crest value.

Attribute	Description

N Of Mean Depth (ft)	For the units that were fully surveyed and not null, the number of Mean Depth Values
Sum Mean Depth (ft)	For the units that were fully surveyed, for all types other than pools (see residual depth) the sum of mean depth values
Sum Residual Depth (ft)	For the units that were fully surveyed and not null, the sum of mean depth values minus pool tail crest depth value
Mean Depth (ft)	For pools the mean depth is the sum of residual depth (pool depths minus pool tail crest) divided by the number of units fully measured, for other types it is the sum of mean depth values divided by the total number of units that were fully measured.

Mean Maximum Depth - Enter the measured maximum depth for each habitat unit, in feet. Mean maximum depth for the surveys is defined as the mean maximum depth measurements in the unit in feet. Mean maximum depths for pools are the mean maximum residual depths (mean maximum depth value from the survey minus the pool tail crest value).

Attribute	Description
N Of Maximum Depth	For the units that were fully surveyed and not null, the number of Maximum Depth Values
Sum Maximum Depth (ft)	For units that were fully measured, the sum of maximum depth of all units
N Of Residual Maximum Depth (ft)	For the units that were fully surveyed and not null, the number of Residual Max Depth Values
Sum Residual Maximum Depth (ft)	For the units that were fully surveyed and not null, the sum of maximum depth values minus pool tail crest depth value
Mean Maximum Depth (ft)	For pools the mean maximum depth is the sum of residual maximum depth values divided by the total number of units fully measured, for other types it is the sum of maximum depth values divided by the total number of units fully measured

Maximum Depth - Enter the measured maximum depth for each habitat unit, in feet. Maximum depth for the surveys is defined as the maximum depth measurements in the unit in feet. Maximum depths for pools is the maximum residual depths, that is the maximum depth value from the survey minus the pool tail crest value.

Attribute	Description
Maximum Depth (ft)	For non pool units, maximum depth of any unit
Residual Maximum Depth (ft)	For the units that were residual max depth > 0, the maximum depth value

Channel Type - Rosgen channel type classification. The channel type of the reach or stream based on the Stream Channel Type Work Sheet (Part III)

Attribute	Description
Channel Type	Rosgen channel type classification. The channel type of the reach or stream based on the stream channel type work Sheet (part III)

Percent Primary Pools - Primary pools are defined differently based on the stream order. First through 2nd order streams primary pools have a maximum depth >=2 feet and 3rd through 4th (nth) order streams primary pools have a maximum depth >=3 feet.

Attribute	Description
Percent Primary Pools by Pools by Stream	Sum of primary pool habitat lengths divided by the total length of all units.

Percent Primary Pools by Pools	Sum of primary pool habitat lengths divided by the total length of all pool units.
Primary Pool Length	Total length of all primary pool units.
Total Length	Total length of all habitat units.
Total Length Pools	Total length of all pool units.

Percent Off Channel Habitat – Off Channel Habitat Types (3.1, 3.5, >= 5 and <7)

Attribute	Description
LengthOfOffChannel	Sum of lengths for off channel habitat types
TotalLength	Total length of all units
OffChannelRatio	Sum of off channel habitat lengths divided by the total length.

Application Table: Units – The “Units” table contains information that can be used to relate the stream and the reach level data to common aggregating layers, such as, county boundaries, USGS hydrologic unit codes (HUCs), ecoregional boundaries, and CALWATER boundaries.

Example Record
Unit Descriptions
Source: *Where does the data come from.*

Attribute	Description
Field Name	Description of field name (if necessary) and ranking criteria

Bailey's Ecoregions and Subregions of the United States, Puerto Rico Attributes
Source: USDA Forest Service

Attribute	Description
OBJECTID	Internal feature number.
ECOREGP075	A five-character code that corresponds to the narrative description in the attribute Section. Ecocode and Section represent the lowest mapping level in the hierarchy of ecoregions and subregions. The first character is an indication of whether the section is mountainous. The next three digits are a code identifying the province, and the last character is a letter identifying the section within the province.
ECOCODE	A major ecoregion distinguished from other domains by climate, precipitation and temperature. This is the highest level in the hierarchy of ecoregions.
DOMAIN_	A subdivision of a domain. A division represents a climate within a domain and is differentiated from other divisions based on precipitation levels and patterns as well as temperature. This is the second level in the hierarchy of ecoregions.
DIVISION	A subdivision of a division. A province represents variations in vegetation or other natural land covers within a division. Mountainous areas that exhibit different ecological zones based on elevation (elevational zonation) are distinguished according to the character of the zonation by listing the elevational zones from lower to upper. This is the third level in the hierarchy of ecoregions.
PROVINCE	A subdivision of a province. A section represents different landform groupings within a province. This is the lowest level in the hierarchy of ecoregions and subregions. Narrative descriptions of sections correspond to unique Ecocode values, above.
SECTION_	A code used to identify mountainous ecoregions with variations due to elevation.
MCODE	A numeric code identifying the Province.
PCODE	A code identifying the section within the Province. This is the last character of Ecocode. This field is designed for cartographic production.
SCODE	The first three characters of the Section value.
KEY_	The last four digits of Ecocode. This is a cartographic production field for labeling Sections.
FDIGIT	The first four digits of Ecocode. This code identifies mountainous and non-mountainous Provinces.
MTEXT	String field

California County Boundaries Attributes
Source: California Department of Forestry and Fire Protection

Attribute	Description
CNTY24K97_	Internal feature number.
CNTY24K971	User-defined feature number.
NAME	County name
NAME_CAP	County name in capitals
NUM	County number (1 - 58)

California Interagency Watersheds Attributes

Source: California Interagency Watershed Map of 1999 (Calwater 2.2.1)

Attribute	Description
CALW221_	Internal feature number.
CALW221_ID	User-defined feature number.
CALWNUM	Unique identifier (type=character) of watershed polygon; concatenates HR+RB+HU+"."+HA+HSA+SPWS+PWS
SWRCBNUM21	Unique identifier (type=character) of watershed polygon as published by SWRCB on HBPA Map Series (revised 1986); concatenates RB+HU+"."+HA+HSA
HRC	Hydrologic Region Code
HBPA	Hydrologic Basin Planning Area
RBU	Concatenates HR+RB+HU into single integer
RBUA	Concatenates HR+RB+HU+HA
RBUAS	Concatenates HR+RB+HU+HA+HAS
RBUASP	Concatenates HR+RB+HU+HA+HSA+SPWS
RBUASPW	Concatenates HR+RB+HU+HA+HSA+SPWS+PWS
HR	Hydrologic Region (as a number)
RB	Region Water Quality Control Board number
HU	Hydrologic Unit
HA	Hydrologic Area
HSA	Hydrologic Sub-Area
SPWS	Super-Planning Watershed
PWS	Planning Watershed
HRNAME	Hydrologic Region Name
RBNAME	Regional Water Quality Control Board Name
HBPANAME	Hydrologic Basin Planning Area Name
HUNAME	Hydrologic Unit Name
HANAME	Hydrologic Area Name
HSANAME	Hydrologic Sub-Area Name
CDFSPWNAME	CDF Super-Planning Watershed Name
CDFPWSNAME	CDF Planning Watershed Name
ACRES	Acreage of watershed polygon
HUC_8	SubBasin (USGS Hydrologic Unit Code, HUC)
HUC_8_NAME	SubBasin Name
HUC_8_ALT2	If populated, is an additional SubBasin that overlaps a State-designated watershed
HUC_8_ALT3	If populated, is a 3rd SubBasin that overlaps a State-designated watershed
DWRNUM20	DWR Alternate watershed identifier
DWRHUNAME	DWR Alternate Hydrologic Unit Name
DWRHANAME	DWR Alternate Hydrologic Area Name
DWRHSANAME	DWR Alternate Hydrologic Sub-Area Name
CDFNUM22	CDF Unique identifier (character) of watershed polygon; concatenates HR+RB+HU+"."+HA+HSA+SPWS+PWS

OUT	Binary
NOTES	String field

Join Fields

Source: Hopland Research and Extension Center

Attribute	Description
Code	Join the code field of the output tables to this field to query the data based on surveyed reaches
Code1	Join the code field of the output tables to this field to query the data based on surveyed stream

Application Table: Populations - The “Populations” table contains information that can be used to relate the stream and the reach level data to the NMFS salmonid populations planning dataset.

Salmonid Populations Planning Dataset

Source: National Marine Fisheries Service (NMFS)

Attribute	Description
OBJECTID	Internal feature number.
POPULATION	Salmonid Population Name
STRATUM	Population Stratum
ESU_DPS	The name of the ecological significant unit (ESU) or distinct population segment (DPS, for steelhead)
POP_ID	Internal coding that combines the species with the population name (ST = steelhead, CO = coho, CH = Chinook, SS = Steelhead (summer), CW = Chinook (winter (Sacramento River winter-run only))
WS_ID	What watershed the population falls into (often a population is a watershed but occasionally the population is a subset of the watershed)
IS_WS	Indicates whether the population and watershed boundaries are coincident (1 = population and watershed are one and the same, 0 = population and watershed boundaries are different (pop is probably a small subset of the watershed)
PLAN_NAME	What Recovery Plan is addressing that population (CCV multi = Central Valley Multispecies Plan, NCCC Multi = NCCC domain multispecies plan, NCCC coho = NCCC domain coho plan, SONCC coho = SONCC domain coho plan, SCCC steelhead = South-central CA Coast steelhead plan. SC steelhead = Southern CA steelhead recovery plan.

Join Fields

Source: Hopland Research and Extension Center

Attribute	Description
Code	Join the code field of the output tables to this field to query the data based on surveyed reaches
Code1	Join the code field of the output tables to this field to query the data based on surveyed stream

Detailed list of the metrics and source documents

Parameter	Level	description	Does-Not Meet Criteria	Meets Criteria	source	document	page	object	species	range	manual page
Pool	1	% primary pools by length compared to all others	<40%	>=40%	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	VI-6, V-15	2a	all, coho	all, coastal	VI-6, V-15
		Primary pool Definition: 1st through 2nd order streams, max depth >=2'	<2'	>=2'	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-15		all	all	V-15
		Primary pool Definition: 3rd through 4th (nth) order streams, max depth >=3'	<3'	>=3'	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-15		all	all	V-15
Pool	1	% pool area compared to all others	<40%	>=50%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Pool	1	% pool frequency number compared to all others	<40%	>=50%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Pool	1	% stream length consisting of primary pools	<40%	>=40%	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	all	Mendocino Coast Hydrologic Unit	VI-6, V-15
Pool		% pool length [stream] of primary pools	undefined	undefined	undefined	undefined					
Pool	1	% pool length compared to all others	<43%	43-50%	Doug Albin	personal communication			coho	Mendocino Coast Hydrologic Unit	
Pool	1	% pool length compared to all others	<40%	>=40%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19		all	North Coast	
Pool	1	% pool depth frequency, number pools >= 2' max depth for order 1 and 2 compared to all other pools	<40%	>=40%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	
Pool	1	% pool depth frequency, number pools >= 2' residual depth for order 1 and 2 compared to all other pools	<40%	>=40%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	
Pool	1	% pool depth frequency, number pools >= 3' max depth for order 3 and 4 compared to all other pools	<40%	>=40%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	
Pool	1	% pool depth frequency, number pools >= 3' residual depth for order 3 and 4 compared to all other pools	<40%	>=40%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	
Pool	1	residual pool depth for first order stream	<1.0	>1.5	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Pool	1	residual pool depth for second order stream	<1.5	>2.0	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	

Pool	1	residual pool depth for third order stream	<2.5	>3.0	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Pool	1	residual pool depth for fourth order stream	<2.6	>3.1	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	86	Table 17	all	Russian River	
Pool	1	mean pool depth (all pools)	<1.25'	>=1.25'	Doug Albin	personal communication			coho	Mendocino Coast Hydrologic Unit	
Pool	1	average maximum pool depth 1st and 2nd order stream	<2'	>=2'	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	all	Mendocino Coast Hydrologic Unit	V-15
Pool	1	average maximum pool depth 3rd and 4th order stream	<3'	>=3'	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	all	Mendocino Coast Hydrologic Unit	V-15
Pool	0	Minimum Stream Order	undefined	undefined	undefined	undefined					
Pool	0	Maximum Stream Order	undefined	undefined	undefined	undefined					
Pool	0	Majority Stream Order	undefined	undefined	undefined	undefined					
Embededness	0	average embededness rating	>1	<=1	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	VI-8	7a	all	all	VI-8
Embededness	0	dominant embededness rating	undefined	undefined	undefined	undefined					
Embededness	1	pool embededness value (not value 5?)	>50%	<25%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Embededness	1	%pools [pools] (number) <50% embeded (1 and 2)	<50%	>=50%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	
Embededness	1	%pools [stream] (number) <50% embeded (1 and 2)	undefined	undefined	undefined	undefined					
Embededness	1	%pools [pools] (length) <50% embeded (1 and 2)	<50%	>=50%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	
Embededness	1	%pools [stream] (length) <50% embeded (1 and 2)	undefined	undefined	undefined	undefined					
Embededness	1	% pools [Pools] (number) having fines (3-4)	>25%	<=25%	Doug Albin	personal communication			coho	Mendocino Coast Hydrologic Unit	VI-8
Embededness	1	% pools [Stream] (number) having fines (3-4)	undefined	undefined	undefined	undefined					
Embededness	0	cobble embededness	2,3,4	1	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	all	Mendocino Coast Hydrologic Unit	VI-8
Riffle	4	LGR dominant substrate	A,B,E,F,G	C,D	California Salmonid Stream Habitat Restoration Manual	Salmon, in the Mendocino Coast Hydrologic Unit	VI-9	8b	all	all	VI-9

Riffle	1	riffle substrates, list %, chose dominant	sand/silt	gravel/small cobble	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85		all	Russian River	VI-9
Riffle	2	% riffle length compared to all others	<10%, >30%	15-30%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85		all	Russian River	
Canopy	0	canopy density	<80%	>=80%	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	VI-7, V-22	4b	all, coho	all	VI-7, V-22
Canopy	0	canopy	<70%	>80%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Canopy	1	pool canopy	<60%	>80%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Canopy	0	% coniferous	<30%	>=50%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Canopy	0	canopy	<93%	>=93%	Doug Albin	personal communication			coho	Mendocino Coast Hydrologic Unit	
Canopy	0	%canopy	<80%	>=80%	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	VI-7, V-22
Canopy	0	% canopy	<80%	>=80%	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	all	Mendocino Coast Hydrologic Unit	VI-7, V-22
Shelter	1	mean pool shelter rating	<80	>=80	California Salmonid Stream Habitat Restoration Manual	Salmon, in the Mendocino Coast Hydrologic Unit	VI-7, V-15	3a	all	all	VI-7, V-15
Shelter	0	stream shelter rating	<80	>100	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Shelter	0	stream complexity value (Shelter Value)	<=1	2-3	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Shelter	0	stream %coverage	<40%	>=40%	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	all	Russian River	
Shelter	1	pool shelter rating	<80	>=80	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	19	Table 8	all	North Coast	VI-7, V-15
Shelter	1	pool complexity value (Shelter Value)	undefined	undefined	undefined	undefined					
Shelter	1	pool % coverage	undefined	undefined	undefined	undefined					
Shelter	1	mean shelter rating all pools	<80	>=80	Doug Albin	personal communication			coho	Mendocino Coast Hydrologic Unit	VI-7, V-15
Shelter	0	shelter rating	<80	>=80	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	all	Mendocino Coast Hydrologic Unit	VI-7, V-15

Bank	0	dominant banks substrate	undefined	undefined	undefined	Salmon, in the Mendocino Coast Hydrologic Unit					
	0	*dominant banks substrate [where canopy does not meet criteria] (*criteria for planting projects)	1,2	3,4	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	VI-8	4c	all	all	VI-8
Bank	0	mean % of stream banks vegetation (both banks)	<65%	>=65%	Doug Albin	personal communication			coho	Mendocino Coast Hydrologic Unit	
Substrate	0	chinook dominant substrate, 1-3"	A,B,E,F,G	C,D	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-21		chinook	all	V-21
Substrate	0	chinook substrate range, 0.5-10"	A,B,F,G	C,D,E	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-21		chinook	all	V-21
Substrate	0	steelhead dominant substrate, 2-3"	C,D	C,D	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-22		steelhead	all	V-22
Substrate	0	steelhead substrate range, 0.5-6"	C,D	C,D	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-22		steelhead	all	V-22
Substrate	1	dominant pool tail substrate	undefined	undefined	undefined	undefined					
Temperature	0	chinook temperature	>65	40-65	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	chinook	Russian River	
Temperature	0	coho temperature	>65	48-60	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	coho	Russian River	
Temperature	0	steelhead temperature	>70	40-65	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft	85	Table 16	steelhead	Russian River	
Temperature	0	coho temperature		48-60	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-21		coho	all	V-21
Temperature	0	steelhead temperature	>65	40-65	California Salmonid Stream Habitat Restoration Manual	California Salmonid Stream Habitat Restoration Manual	V-22,23		steelhead	all	V-22,23
Temperature	0	MWAT	>65	50-60	NCWAP	Gualala River Watershed Assessment Report, Appendix 5	4-6		all	North Coast	
Temperature	0	coho temperature		48-60	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	coho	Mendocino Coast Hydrologic Unit	V-21
Temperature	0	steelhead temperature	>65	<65	Doug Albin	Assessment of Environmental Effects on Salmonids, with Emphasis on Habitat Restoration for Coho Salmon, in the Mendocino Coast Hydrologic Unit	61	Table 7	steelhead	Mendocino Coast Hydrologic Unit	V-22,23
Survey Year	0	Survey Year	undefined	undefined							

Channel Type	0	Channel Type, suitable for fish	D, F1,2,6	B,C,E,G,F3-5	Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft			all	Russian River	
Habitat Diversity		Manual Pages V-3, V-19, V-20 and associated other pages			Bob Coey	Russian River Basin Fisheries Restoration Plan, 2002 Draft			all	Russian River	

APPENDIX D

COST DEVELOPMENT PROTOCOL

North Central California Coast Recovery Domain
CCC Coho ESU Recovery Plan

Cost Assumption Tables

Prepared by:

NOAA's National Marine Fisheries Service, Southwest Region

Protected Resources Division, NCCC Recovery Domain

Santa Rosa, California

COSTS ASSUMPTION TABLES

In order to develop recovery costs, a standardized method was developed to assign costs to recovery actions. The assumptions are based on DFG's "Cost and Socioeconomic Impacts of Implementing the California Coho Recovery Strategy" (2004) and NMFS "Habitat Restoration Cost References for Salmon Recovery Planning" (2008), assessed addition information such as aggregate costs, wage rates, and socioeconomic impacts and created assumption tables for specific categories of actions and action types. The following assumption tables were used to assign costs to specific action steps for the population specific implementation tables.

Table 1. Recovery Implementation Cost

Action	Cost	Unit
Stream Complexity	25,000	Mile
	101,120	ELJ
Riparian Vegetative Cover	20,057	Acre
Vegetative Ground Cover	1,422	Acre
	39,574[1]	Acre
Floodplain Connectivity	36,046	Mile
Estuarine Ecology	272,120	Acre

[1] Source: CDFG 2004 (p. 1-16)
[2] Source: NMFS 2008, p. 43-44

[1] Cost for treating non-native species in freshwater and riparian environments.

Table 2. Fish Passage Improvement ($/Project)				
Stream Crossing	**Land Use**			
	Forest	**Agriculture**	**Suburban**	**Urban**
Tributary: Total Barrier	63,636	159,090	318,181	556,818
Tributary: Partial/Temporal Barrier	31,818	79,545	159,090	278,409
Stream : Total Barrier	159,090	381,818	556,818	795,454
Stream: Partial/Temporal Barrier	79,545	190,909	278,409	397,727

[1]Source: CDFG 2004, p. 1-16

Table 3. Dam Removal	
Size of Dam	**$; $/ft**
one cost estimate for <15ft dam	568,181
>15 ft high -cost/ft	17,045
one estimate - unknown height; complete barrier	1,022,727
one estimate - unknown height; partial/temporal or unknown barrier	511,363

[1] Source: CDFG 2004, p.11

Table 4. New Fish Ladder[1]	
Waterway Size	**Cost ($)**
Large	1,022,727
Small	568,181

1

Source: NMFS 2008, p. 9

Table 5. Culvert Replacement ($/Culvert)[1]				
Size of Waterway	**Road Type**			
	Forest Road	**Minor 2 Lane**	**Major 2 Lane**	**Hwy 4+ Lane**
Small (0-10')	31,976	87,209	174,419	319,767
Medium (10-20')	87,209	220,930	319,767	436,047
Large (20-30')	133,721	267,442	406,977	813,953

[1]Source: NMFS 2008, p. 10

Table 6. Replacing a Culvert w/ a New Type of Structure[1]	
New Type of Crossing	**Avg. Cost ($)**
Bridge <40ft	51,546
Bridge >40ft	103,093
Bottomless/Open Bottom Arch	193,961
Natural Bottom Pipe Arch	215,776
Box Culvert	248,352

[1]Source: NMFS 2008, p. 10

Table 7. Floodplain and Tributary Reconnection ($/acre)[1]			
Materials	**Extent of Earth Moving**		
	Minimal	**Moderate**	**Substantial**
Minimal	8,721	17,442	40,698
Moderate	17,442	29,070	58,140
Substantial	40,698	58,140	81,395

[1]

Source: NMFS 2008, p.26

Table 8. Riparian Planting ($/acre)[1]			
Materials/Site Accessibility	**Level of Site Preparation***		
	Flat/Light Clearing	**Avg. Slope/Avg. Clearing**	**Steep/Heavy Clearing**
Low Cost	17,442	40,698	93,023
Medium Cost	26,163	63,954	110,465
High Cost	46,512	78,488	1,366,279

[1] Source: NMFS 2008, p. 32

Table 9. Upslope Riparian Thinning[1]	
Type	**$/acre***
Mechanical	876
Hand 15-30% slope 40-60% cover	928
Hand 30-50% slope 60-90% cover	1,237
Chemical	155
Average	799

[1]Source: NMFS 2008, p. 64

Table 10. Road Inventories[1]	
Location	**$/mi**
Humboldt County	829
Eel River	538
Mattole River	635
Russian River	936
Salmon Creek	1068
Gualala River	837
Avg. all Inventories	807

[1]Source: NMFS 2008, p. 61

Table 11. Erosion Assessments[1]	
Location	**$/acre***
Humboldt County	9.5
Del Norte County	11.9
Average all assessments in CA**	10.7

[1]Source: NMFS 2008, pg. 61

Table 12. Removal of Invasive Plant Species[1]		
Species	**$/acre***	**Source**
Arundo	29,762	Neil 2002
Himalayan Blackberry	990	Bennet 2007 (avg)
Purple Loosestrife and Water Chestnut	361	USFWS 2001
Pepperweed and Giant Reed	1,000	Northern California Conservation Center 2010
Average (excluding outlier of Arundo)	784	

Establishing a Multiplier

The recovery costs established by DFG in 2004 are for CCC coho salmon ESU and portions of the SONC coho ESU, which include Del Norte to Santa Cruz counties. Recovery costs were not standardized across the CCC coho salmon ESU due to the variability between each of the three regions, such as extent of urbanization, labor wages, access, and material costs. To attempt to encapsulate the anticipated increased cost of implementing recovery actions, we applied a multiplier of 0.20 to the standard costs for the San Francisco Region, and a multiplier of 0.14 in the Central Coast Region to reflect the variability in wages between the regions. It is uncertain if this will apply in all circumstances, watersheds, or recovery actions.

Table 13. Multiplier of Recovery Cost to Regions: North Central Coast Office

Region	Multiplier
North Coast	none
San Francisco Bay	0.20
Central Coast	0.14

APPENDIX E

BIOLOGICAL VIABILITY REPORT
SPENCE ET AL. 2008

NOAA Technical Memorandum NMFS

APRIL 2008

A FRAMEWORK FOR ASSESSING THE VIABILITY OF THREATENED AND ENDANGERED SALMON AND STEELHEAD IN THE NORTH-CENTRAL CALIFORNIA COAST RECOVERY DOMAIN

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U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southwest Fisheries Science Center

NOAA Technical Memorandum NMFS

NOAA Technical Memorandum NMFS
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A FRAMEWORK FOR ASSESSING THE VIABILITY OF THREATENED AND ENDANGERED SALMON AND STEELHEAD IN THE NORTH-CENTRAL CALIFORNIA COAST RECOVERY DOMAIN

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NOAA-TM-NMFS-SWFSC-423

U.S. DEPARTMENT OF COMMERCE
Carlos M. Gutierrez, Secretary

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Vice Admiral Conrad C. Lautenbacher, Jr., Under Secretary for Oceans and Atmosphere

National Marine Fisheries Service
James W. Balsiger, Acting Assistant Administrator for Fisheries

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Acronyms and Abbreviations

CC-Chinook salmon	California Coastal Chinook salmon Evolutionarily Significant Unit
CCC-coho salmon	Central California Coast coho salmon Evolutionarily Significant Unit
CCC-steelhead	Central California Coast steelhead Distinct Population Segment
DPS	distinct population segment
DP	dependent population
DS	diversity stratum
ESA	U.S. Endangered Species Act
ESU	evolutionarily significant unit
FIP	functionally independent population
NC-steelhead	Northern California steelhead Distinct Population Segment
NCCC	North-Central California Coast
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
PIP	potentially independent population
PVA	population viability analysis
TRT	Technical Recovery Team

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The Technical Recovery Team would like to thank Dr. Steve Lindley, Dr. Pete Adams, and Dr. David Boughton, for many thoughtful discussions about the viability framework. Dr. Lindley and Dr. Adams also provided helpful reviews of the final draft of the manuscript. Matthew Jones and Tom Pearson provided GIS support for this project. Heidi Fish reviewed the document to ensure accuracy of tables, figures, and references cited in the report. We also thank the California Department of Fish and Game, who reviewed an earlier draft of this document.

Executive Summary

The Technical Recovery Team (TRT) for the North-Central California Coast Recovery Domain has been charged with developing biological viability criteria for each listed Evolutionarily Significant Unit (ESU) of salmon and Distinct Population Segment (DPS) of steelhead within the recovery domain. The viability criteria proposed in this report represent the TRT's recommendations as to the minimum population and ESU/DPS characteristics indicative of an ESU/DPS having a high probability of long-term (> 100 years) persistence. Our approach employs criteria representing three levels of biological organization: populations, diversity strata, and the ESU or DPS as a whole. Populations include both independent and dependent populations defined in Bjorkstedt et al. (2005), as modified in Appendix A of this report. Diversity strata are groups of geographically proximate populations that reflect the diversity of selective environments, phenotypes, and genetic variation across an ESU or DPS (Bjorkstedt et al. 2005). A viable ESU or DPS comprises sets of viable (and sometimes nonviable) populations that, by virtue of their size and spatial arrangement, result in a high probability of persistence over the long term.

We provide background critical to understanding the context for viability criteria development in Chapter 1 of this report. Chapters 2 and 3 define viability criteria at the population and ESU/DPS levels, respectively. In Chapter 4, we apply the criteria to assess current viability, though with limited success due to the lack of appropriate, population-level time series of abundance. We emphasize that the focus of this document is looking forward to evaluating recovery, not assessment of current conditions.

Population Viability Criteria

Our approach to population viability extends the "viable salmonid population" concept of McElhany et al. (2000), who proposed that four parameters are critical to evaluating population status: abundance, population growth rate, spatial structure, and diversity. Our approach classifies populations into various extinction risk categories based on a set of quantitative and qualitative criteria related to these parameters. Both the approach and the specific criteria have their roots in the IUCN (1994) red list criteria (derived in part from Mace and Lande 1991) and subsequent modifications made by Allendorf et al. (1997) to address populations of Pacific salmon. We have extended the Allendorf criteria, adding criteria related to spawner density and to the potential effects of hatchery activities on wild populations.

In this document, we consider population viability from two distinct but equally important perspectives. The first perspective relates to the goal of defining the minimum viable population (MVP) size for which a population can be expected to persist with some specified probability over a specified period of time.

The minimum viable population size identifies the approximate lower bounds for a population, above which risks associated with demographic stochasticity, environmental stochasticity, severe inbreeding, and long-term genetic losses are negligible. The second perspective views viability in terms of how a population is currently functioning in relation to its historical function. This latter perspective recognizes the critical role that large, productive populations historically played in ESU viability, both as highly persistent parts of an ESU and as sources of strays that influenced the dynamics and extinction probabilities of neighboring populations. Central to this view is the idea that historical patterns of abundance, productivity, spatial structure, and diversity form the reference conditions about which we have high confidence that ESUs and their constituent independent populations had a high probability of persisting over long periods of time. As populations depart from these historical conditions, their probability of persistence declines and their functional role with respect to ESU viability may be diminished. The criteria we propose in this document encompass both of these perspectives, addressing immediate demographic and genetic risks, as well longer-term risks associated with loss of spatial structure and diversity, both of which contribute to population resilience and the ability of populations to fulfill their functional roles within the ESU.

Evaluation of extinction risk is done either based on rigorous, model-based population viability analysis (PVA) or, in the absence of sufficient data to construct a credible PVA model, using five surrogate criteria related to effective population size per generation, population declines, effects of recent catastrophes on abundance, spawner density, and hatchery influence (Table 1). Population viability analyses produce direct estimates of extinction probability over a specified time frame. The effective population size criteria address the loss of genetic diversity that can occur in small populations. Effective population size can be estimated directly from demographic or genetic data, or absent such data, by assuming a specific ratio of effective population size to total population size. The population decline criteria address increased demographic risks associated with rapid or prolonged declines in abundance to small population sizes. The catastrophe criteria seek to capture effects of large environmental perturbations that produce rapid declines in abundance. Such events are distinct from environmental stochasticity that arises from a series of small or moderate perturbations that affect population growth rate. The density criteria are intended to capture several distinct processes not explicitly addressed in the Allendorf et al. (1997) criteria. The high-risk thresholds identify densities at which populations are at heightened risk of a reduction in per capita growth rate (i.e., depensation). Populations exceeding the low-risk density thresholds are expected to inhabit a substantial portion of their historical range, which serves as a proxy indicator that resultant spatial structure and diversity will reasonably represent the

Table 1. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids. Overall risk is determined by the highest risk score for any category. See Table 2 for definitions of N_g, N_e, *and* N_a. Modified from Allendorf et al. (1997) and Lindley et al. (2007).

Population Characteristic	Extinction Risk: High	Extinction Risk: Moderate	Extinction Risk: Low
Extinction risk from population viability analysis (PVA)	≥ 20% within 20 yrs	≥ 5% within 100 yrs but < 20% within 20 yrs	< 5% within 100 yrs
	- or any ONE of the following -	- or any ONE of the following -	- or ALL of the following -
Effective population size per generation *-or-* Total population size per generation	$N_e \leq 50$ *-or-* $N_g \leq 250$	$50 < N_e < 500$ *-or-* $250 < N_g < 2500$	$N_e \geq 500$ *-or-* $N_g \geq 2500$
Population decline	Precipitous decline[a]	Chronic decline or depression[b]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[c]	Not apparent
Spawner density	$N_a/IPkm$[d] ≤ 1	$1 < N_a/IPkm <$ MRD[e]	$N_a/IPkm \geq$ MRD[e]
Hatchery influence[f]	Evidence of adverse genetic, demographic, or ecological effects of hatcheries on wild population		No evidence of adverse genetic, demographic, or ecological effects of hatchery fish on wild population

[a] Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size $N_a \leq 500$ spawners (historically small but stable populations not included) *or* $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.
[b] Annual run size N_a has declined to ≤ 500 spawners, but is now stable *or* run size $N_a > 500$ but continued downward trend is evident.
[c] Annual run size decline in one generation < 90% but biologically significant (e.g., loss of year class).
[d] *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).
[e] MRD = minimum required spawner density and is dependent on species and the amount of potential habitat available. Figure 5 summarizes the relationship between spawner density and risk for each species.
[f] Risk from hatchery interactions depends on multiple factors related to the level of hatchery influence, the origin of hatchery fish, and the specific hatchery practices employed.

historical condition. The hatchery criteria are narrative criteria that address potential genetic, demographic, and ecological risks that occur when hatchery fish interact with wild fish.

ESU-Level Criteria

ESU-level criteria specify the number and distribution of viable and, in some cases, nonviable populations that would constitute a viable ESU or DPS. The three primary goals of the ESU/DPS level criteria are 1) to ensure sufficient genetic and phenotypic diversity within the ESU or DPS to maintain its evolutionary potential in the face of changing environmental conditions; 2) to maintain sufficient connectivity among populations within the ESU or DPS to maintain long-term demographic and evolutionary processes; and 3) to buffer the ESU or DPS against catastrophic loss of populations by ensuring redundancy (i.e., multiple viable populations). Four criteria are developed to address these concerns.

Representation Criteria

1. a. All identified diversity strata that include historical functionally or potentially independent populations within an ESU or DPS should be represented by viable populations for the ESU or DPS to be considered viable.

-AND-

b. Within each diversity stratum, all extant phenotypic diversity (i.e., major life-history types) should be represented by viable populations.

Representation of all diversity strata achieves the primary goal of maintaining a substantial degree of the ESU's or DPS's historical diversity, as well as ensuring that the ESU or DPS persists throughout a significant portion of its historical range. The second element of the representation criteria specifically addresses the persistence of major life-history types (i.e., summer-run steelhead) as an important component of ESU viability.

Redundancy and Connectivity Criteria

2. a. At least fifty percent of historically independent populations (functionally or potentially independent) in each diversity stratum must be demonstrated to be at low risk of extinction according to the population viability criteria developed in this report. For strata with three or fewer independent populations, at least two populations must be viable.

-AND-

b. Within each diversity stratum, the total aggregate abundance of populations selected to satisfy this criterion must meet or exceed 50% of the aggregate viable population abundance (i.e., meeting density-based criteria for low risk) for all functionally independent and potentially independent populations.

The first element of this criterion provides a buffer against the loss of diversity due to catastrophic loss of populations within a stratum. The second element recognizes the differing roles that various populations historically played in ESU or DPS viability depending on their size and location. The criterion emphasizes the importance in having some large, resilient populations serve as the foundation of a persistent ESU or DPS.

3. Remaining populations, including historical dependent populations and any historical functionally or potentially independent populations that are not expected to attain a viable status, must exhibit occupancy patterns consistent with those expected under sufficient immigration subsidy arising from the 'core' independent populations selected to satisfy the preceding criterion.

This criterion acknowledges that, while certain populations may no longer fulfill their historical role in ESU viability, the remaining portions of these populations can contribute substantially to connectivity among populations within the ESU, as well as represent important parts of the ESU's evolutionary legacy.

4. The distribution of extant populations, regardless of historical status, must maintain connectivity within the diversity stratum, as well as connectivity to neighboring diversity strata.

This criterion stresses the importance of ensuring connectivity within and among diversity strata to maintain long-term evolutionary and demographic processes that result from natural dispersal.

Assessment of Current Viability

Attempts to assess current viability of salmon and steelhead populations and ESUs/DPSs in the North-Central California Coast Recovery Domain using our approach were hampered by the lack of data, especially long-term time series of population abundance, for the vast majority of populations within the domain. Few populations within the domain are monitored, and most ongoing monitoring programs are either not designed to obtain population-level abundance estimates or are relatively new programs that have not produced the 12+ years of data required to apply the criteria as outlined. As a result, strict application of the criteria results in almost all populations being classified as "data deficient." However, in many cases, ancillary data strongly suggest certain populations would currently fail to meet one or more of the identified low-risk or moderate-risk thresholds. In these instances, we assign a population-level risk designation, identifying the specific criteria that we believe the population is unlikely to satisfy and the data that justify the particular risk rating. Populations addressed below are outlined by Bjorkstedt et al. as modified in Appendix A of this report.

Central California Coast Coho Salmon

The Central California Coast (CCC) coho salmon ESU historically comprised twelve independent populations, as well as a number of dependent populations, representing five diversity strata. There are no population data of sufficient quality to rigorously assess the current viability of any of the twelve independent coho salmon populations within the CCC ESU using the proposed criteria. However, recent ancillary data on occupancy of historical streams within the ESU indicates that at least half of the independent populations within the ESU are extinct or nearly so, including the San Lorenzo River, Pescadero Creek, Walker Creek, Russian River, Gualala River, and Garcia River populations. Furthermore, all dependent populations within the San Francisco Bay diversity stratum have been extirpated. Populations continue to persist in Lagunitas Creek, Navarro River, Albion River, Big River, Noyo River, and Ten Mile River, as well as a few smaller watersheds; however, the available data are inadequate for assigning risk according to the viability criteria, and these populations were thus classified as data deficient. The lack of demonstrably viable populations (or the lack of data from which to assess viability) in any of the diversity strata, the lack of redundancy of viable populations in any of the strata, and the substantial gaps in the current distribution of coho salmon, particularly in the southern two-thirds of the CCC ESU, clearly indicate that the ESU fails to satisfy diversity stratum and ESU-level criteria and is at high risk of extinction.

California Coastal Chinook Salmon

The California Coastal Chinook salmon ESU historically consisted of fifteen independent populations of fall-run Chinook, as many as six spring-run populations, and an unknown number of dependent population representing four diversity strata. Current population abundance data are insufficient to rigorously evaluate the viability of any of the fifteen putative independent populations of fall-run Chinook salmon in the ESU using the proposed criteria. Ancillary data indicate that fall-run populations continue to persist in watersheds in the northern part of the ESU, including Redwood Creek, Little River, Mad River, Humboldt Bay tributaries, the upper and lower Eel River, Bear River, and the Mattole River. However, all of these populations are classified as data deficient, with the exception of the Mattole River, where we concluded that the population was at least at moderate risk of extinction based on low adult abundances and apparent population declines in recent years. Over the last 10–15 years, fall Chinook salmon have been reported sporadically in the Ten Mile River, Noyo River, and Navarro River, but there is no evidence that these watersheds support persistent runs. Additionally, we found no evidence of recent occurrence of Chinook salmon in the Big River, Garcia River, or Gualala River. Consequently, all six of these populations are believed to be either at high risk of extinction or extinct. The Russian River population appears to be the only extant population of Chinook salmon south of the Mattole River within this ESU. Recent (since 2002) adult counts made at Mirabel Dam have ranged from 1,300 to 6,100. Lacking longer time series of data, we categorized this population as data deficient; however, should counts continue to fall in this range, the Russian River population would likely meet all but the density criterion for low risk. All six putative spring-run independent populations of Chinook salmon within the ESU are believed extinct.

The lack of reliable information on abundance for any fall Chinook populations in the northern half of the ESU precludes us from ascertaining whether either the North Coastal or North Mountain Interior diversity strata are represented by one or more viable populations. Populations appear extinct in the North-Central stratum, and only the Russian River population persists in the Central Coastal stratum. Consequently, there is a 200 km stretch of coastline between the Mattole and Russian Rivers where Chinook salmon no longer appear present. Additionally, spring Chinook salmon within the ESU are thought to be extinct, indicating loss of diversity within the ESU. The lack of demonstrably viable populations in any of the diversity strata, the apparent loss of populations from all watersheds between the Mattole and Russian rivers, and the loss of important life-history diversity (i.e. spring-run populations) all indicate that this ESU fails to meet our representation, redundancy, and connectivity criteria.

Northern California Steelhead

Historically, the Northern California steelhead DPS consisted of at least 42 independent populations of winter-run steelhead, perhaps as many as ten summer-run populations, and an unknown number of dependent populations representing five diversity strata. Currently available data are insufficient to rigorously evaluate the current viability of any of the 42 independent populations of winter steelhead in the NC-steelhead DPS using our viability criteria, and ancillary data that allow classification of populations is available for only a few populations. Populations persist in many watersheds from Redwood Creek (Humboldt Co.) to the Gualala River (Sonoma Co.), but few time series of adult abundance span more than a few years, and those that do represent only a portion of the population and thus do not allow inference about the population at large. Based on spawner estimates made since 2000 and 2001, we classified four populations as at moderate risk: Pudding Creek, Noyo River, Caspar Creek, and Hare Creek. Three additional populations, Soda Creek, Bucknell Creek, and the Upper Mainstem Eel River, were classified as at moderate or high risk based on counts at Van Arsdale Station, which potentially samples fish from all three populations. Low adult returns and a substantial hatchery influence justified these rankings. All remaining winter-run steelhead populations were classified as data deficient.

Abundance data for summer-run populations are somewhat more available, but population-level estimates of abundance spanning a period of four generations or more are available for only one population: the Middle Fork Eel River. This population falls short of low-risk thresholds for effective population size, and the long-term downward trend, if it continues, would bring the annual run size below 500 spawners within two generations. Consequently, we categorized this population as at moderate risk of extinction. Limited data from Redwood Creek and Mattole River suggest that these populations likely number fewer than 30 fish, and we thus concluded both are at high risk of extinction. The Mad River population appears somewhat larger (geometric mean of 250 spawners from 1994-2002) but has declined in recent years. Thus, we concluded it was at moderate risk. Little is known about potential summer-run steelhead populations in the Van Duzen River, South Fork Eel River, Larabee Creek, North Fork Eel River, Upper Middle Mainstem Eel River, or Upper Mainstem Eel River. All were categorized as data deficient, though the lack of even anecdotal reports in recent years suggests that many of these populations are either extirpated or extremely depressed.

Although steelhead persist in many of their historical watersheds in the NC-Steelhead DPS, the almost complete lack of data with which to assess the status of virtually all of the 42 independent populations of winter steelhead within the NC-Steelhead DPS precludes evaluation of ESU viability using the criteria developed in this paper. For summer steelhead, the limited available data provide no evidence of viable summer steelhead populations within the ESU. Consequently, it is highly likely that, at a minimum, the representation and redundancy criteria are not being met for summer-run steelhead. It is unclear if any diversity strata are represented by multiple viable populations or if connectivity goals are being met.

Central California Coast Steelhead

The Central California Coast steelhead DPS historically comprised 37 independent winter-run populations representing five diversity strata. The lack of data on spawner abundance for steelhead populations in the DPS precludes a rigorous assessment of current viability for any of these populations, and in only a few cases do ancillary data provide sufficient information to allow reasonable inference about population risk at the present time. Overall, we classified 30 populations as data deficient. Six populations, all in tributaries to San Francisco Bay (Walnut Creek, San Pablo Creek, San Leandro Creek, San Lorenzo Creek, Alameda Creek, and San Mateo Creek), were classified as at high risk of extinction. In all six cases, dams preclude access to substantial proportion of historical habitat, and what habitat remains downstream is poor quality and insufficient to support viable populations. We categorized one population, Scott Creek (Santa Cruz Co.), as at moderate risk based on recent (2004-2007) estimated adult returns numbering between 230 and 400, with about 34% of these fish being of hatchery origin.

Because of the extreme data limitations, we are unable to assess the viability of CCC-Steelhead DPS using our criteria. All populations within North Coastal, Interior, and Santa Cruz Mountains strata were categorized as data deficient, as were many of the populations in the Coastal and Interior San Francisco Bay strata. The presence of dams that block access to substantial amounts of historical habitat (particularly in the east and southeast portions of San Francisco Bay), coupled with ancillary data, suggest that it is highly unlikely that the Interior San Francisco Bay strata has any viable populations, or that redundancy criteria would be met. The data are insufficient to evaluate representation and connectivity criteria elsewhere in the DPS.

1 Introduction

1.1 Background

Since 1989, the National Marine Fisheries Service (NMFS) has listed twenty-seven Evolutionarily Significant Units (ESUs) or Distinct Population Segments (DPSs)[1] of coho salmon, Chinook salmon, sockeye salmon, chum salmon, and steelhead in the states of Idaho, Washington, Oregon, and California as threatened or endangered under the federal Endangered Species Act (ESA). Among the provisions of the ESA, as amended in 1988, are requirements that NMFS develop recovery plans for listed species and that these recovery plans contain "*objective, measurable criteria which, when met, would result in a determination… that the species* [or ESU] *be removed from the list*." (ESA Sec 4(f)(1)(B)(ii)). The ESA, however, provides no detailed guidance on how to define these recovery criteria.

In 2000, NMFS organized recovery planning for listed salmonid ESUs[2] into geographically coherent units termed "recovery domains." Subsequently, Technical Recovery Teams (TRTs) consisting of scientists from NOAA Fisheries; other federal, tribal, state, and local agencies; academic institutions; and private consulting firms were convened for each recovery domain to provide technical guidance in the recovery planning process. Among their responsibilities, the TRTs have been charged with developing biological viability criteria for each listed ESU within their respective domains. The North-Central California Coast (NCCC) Recovery Domain, which is the focus of this report, encompasses four ESA-listed ESUs and DPSs of anadromous salmon and steelhead: California Coastal Chinook salmon (CC-Chinook salmon ESU), listed as threatened in 1999; Central California Coast coho salmon (CCC-Coho salmon ESU), listed as threatened in 1996 and revised to endangered in 2005; Northern California steelhead (NC-Steelhead DPS), listed as threatened in 1997; and Central California Coastal steelhead (CCC-Steelhead DPS), also listed as threatened in 1997. These ESUs cover a geographic area extending from the Redwood Creek watershed (Humboldt County) in the north, to tributaries of northern Monterey Bay in

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[1] The ESA allows listing not only of species, but also "distinct population segments" of species. Policies developed by NMFS have defined distinct population segments as populations or groups of populations that are reproductively isolated from other conspecific population units and that are an important component in the evolutionary legacy of the species. NMFS has termed these distinct population segments "Evolutionarily Significant Units" or ESUs (Waples 1991). More recently, NMFS revisited the distinct population segment question as it pertains to populations of *O. mykiss*, which may have both resident and anadromous forms living sympatrically. Although at the time of the original listings of Central California Coast and Northern California steelhead, both resident and anadromous forms were considered part of these ESUs, only the anadromous forms were listed (62 FR 43937, at 43591). A court ruling (Alsea Valley Alliance v. Evans, 161 F. Supp. 2d 1154 (D. Or. 2001)) concluded that listing a subset of a delineated group, such as the anadromous form of an ESU, was not allowed under ESA. Thus, existing federal policy regarding DPSs (61 FR 4722) was applied to delineate resident and anadromous forms of *O. mykiss* as separate DPSs. Subsequently, the CCC and NC steelhead DPSs were listed as threatened under ESA (71 FR 834).

[2] Throughout this document, we frequently use the term ESU to encompass both ESUs and DPSs when speaking in general terms about listed salmonid units in order to avoid awkward or cumbersome language. When referring to a specific ESU or DPS, we use the appropriate term.

the south, inclusive of the San Francisco Bay estuary east to the confluence of the Sacramento and San Joaquin rivers (Figure 1)[3].

The first step in the development of viability criteria was to define the historical population structure for each ESU within the domain (Bjorkstedt et al. 2005). The biological organization of salmonid species is hierarchical, from species and ESUs down to local breeding groups or subpopulations, reflecting differing degrees of reproductive isolation. For example, by virtue of their close proximity and shared migratory pathways, subpopulations within the same watershed are likely to exchange individuals through the process of straying on a regular basis (i.e., annually), whereas for populations or larger groups (i.e., diversity strata[4]) such interactions may occur much less frequently. The level of exchange of individuals among spawning aggregations can have significant bearing on the population dynamics and extinction risk of such groups, which in turn may influence the persistence of higher-level groups, on up to ESUs. For recovery planning purposes, it is particularly important to identify the minimum population units that would be expected to persist in isolation of other such populations, as recovery strategies focused solely on smaller units would have a high likelihood of failure. Additionally, over the spatial scale typical of an ESU, reproductive isolation of populations and exposure of these reproductively isolated populations to unique environmental conditions are likely to result in local adaptations and genetic diversity. This diversity, coupled with spatial structure at levels above the population, is important to the long-term persistence of the ESU. Development of appropriate viability criteria and recovery goals requires some understanding of and accounting for this hierarchical structure, and it was therefore necessary to explore probable historical relationships among various spawning groups of salmonids within each ESU. The NCCC TRT (Bjorkstedt et al. 2005) has provided the foundation for viability criteria at these spatial scales by defining both population units and diversity strata (i.e., groups of populations that likely exhibit genotypic and phenotypic similarity due to exposure to similar environmental conditions or common evolutionary history) important to consider in the development of ESU viability criteria. Further consideration by the TRT has led to some modifications to the structures proposed in Bjorkstedt et al. (2005); revised summaries for each ESU and DPS are presented in Appendix A of the present report.

[3] A fifth listed ESU, the Southern Oregon-Northern California Coast coho salmon ESU, extends into the geographic region of the NCCC Recovery Domain; however, viability criteria for this ESU are being developed by the Southern Oregon-Northern California Coast workgroup of the Oregon-Northern California Coast Technical Recovery Team.

[4] Diversity strata are generally defined by Bjorkstedt et al. (2005) as groups of populations that inhabit regions of relative environmental similarity and therefore presumed to experience similar selective regimes.

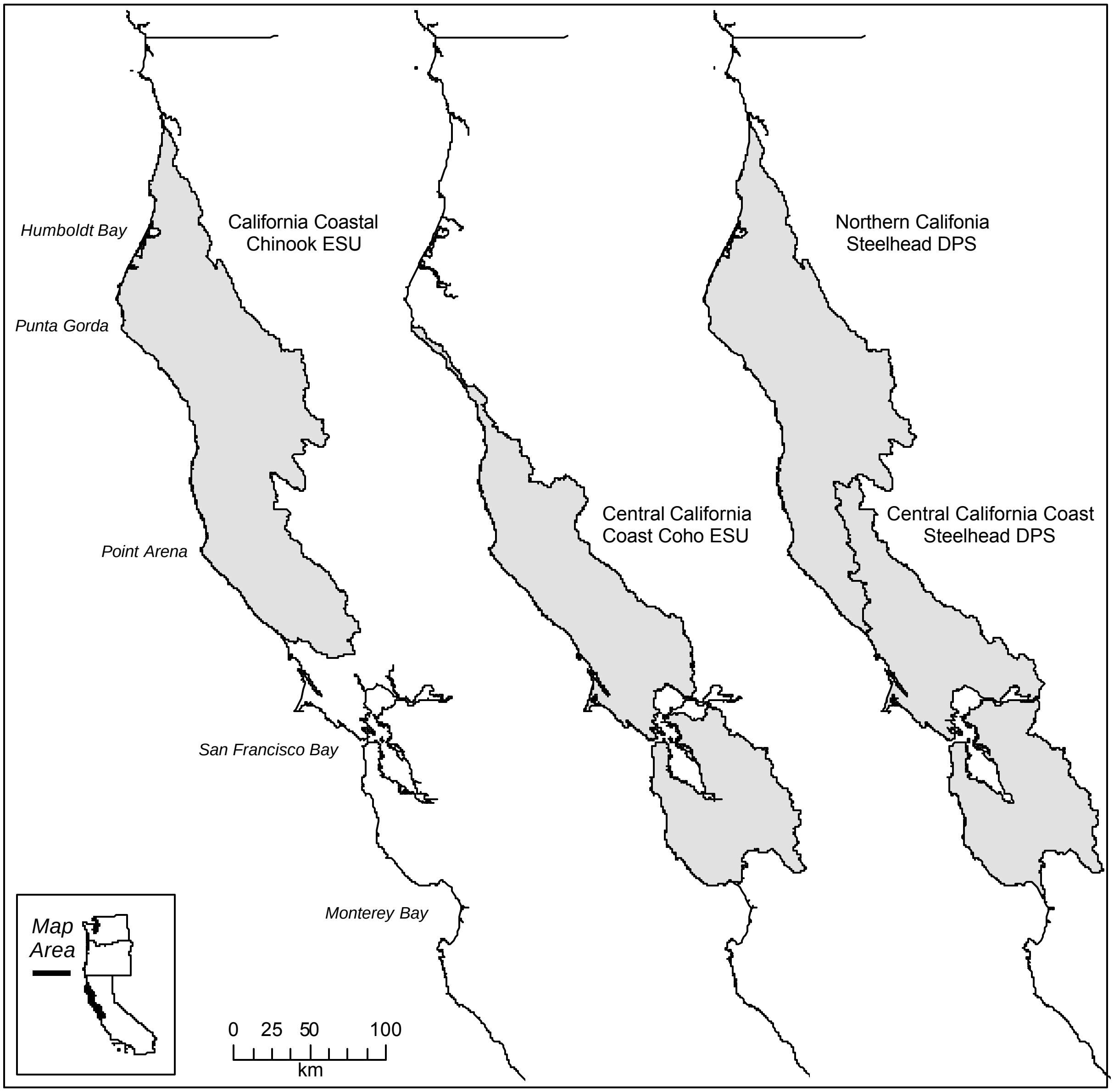

Figure 1. Approximate historical geographic boundaries of ESA-listed salmon and steelhead ESUs and DPSs in the North-Central California Coast Recovery Domain.

The TRT's second report, *Framework for Assessing Viability*, comprises the next step in development of viability criteria for ESUs and DPSs within the NCCC Recovery Domain. Specifically, we develop an approach for assessing viability using criteria representing three levels of biological organization and processes that are important to persistence and sustainability: populations, diversity strata, and the ESU as a whole. Ideally, population-level criteria would be tailored to each population, taking into account specific biological characteristics of populations and differences in the inherent productive capacities of the habitats that may underlie these biological differences. In most cases, however, such population-specific information is not currently available and likely will not be available in the foreseeable future. In the absence of extensive quantitative population data, the Recovery Science Review Panel[5] (RSRP 2002) and Shaffer et al. (2002) have recommended using general, objective population-based criteria such as those used by the IUCN (IUCN 2001). In response to both data limitations and recommendations by the RSRP, we have adopted (with modifications) the conceptual approach of Allendorf et al. (1997), who proposed a series of general criteria for assessing extinction risk and prioritizing the conservation of populations of Pacific salmonids. The Allendorf et al. approach includes criteria related to population size (effective and total) and recent trends in abundance (catastrophic and longer term), to which we have added criteria related to population density and hatchery effects. Other TRTs within California have likewise adopted the Allendorf et al. (1997) framework, with various modifications (Lindley et al. 2007; Boughton et al., 2007; Williams et al., in prep.).

Our criteria for diversity strata emphasize the need for within-strata redundancy in viable populations so as to minimize the risks of losing a significant component of the overall genetic diversity of an ESU due to a single catastrophic disturbance. At the ESU level, criteria are intended to ensure that the range of genetic diversity of the ESU is adequately represented and to foster connectivity among the constituent populations and diversity strata. For diversity strata and ESU-level criteria, we draw heavily from the work of the Puget Sound (PSTRT), Willamette and Lower Columbia (WLCTRT), Interior Columbia (ICTRT), Oregon/Northern California Coast (ONCCTRT) technical recovery teams, all of which have published or are producing criteria incorporating similar, though not identical, elements (PSTRT 2002; WLCTRT 2003; ICTRT 2005; Boughton et al. 2007; Wainwright et al., in press.; Williams et al., in prep.).

The primary intent of our framework for assessing population and ESU viability is to guide future determinations of when populations and ESUs are no longer at risk of extinction. To implement the framework, it is necessary to have fairly lengthy time-series of adult abundance (at least 10-12 years to evaluate populations using the general criteria, and even longer time series to conduct credible population viability analyses) at appropriate spatial scales (i.e., population-level estimates for most historically independent populations that have been identified within each ESU). The practical reality in California is that few such datasets exist. Although there are a number of ongoing salmonid monitoring activities, few are designed to generate estimates of abundance at the population level; thus, there is an urgent need to initiate monitoring programs that will generate data of sufficient quality to rigorously assess progress toward population and ESU recovery. Development of a comprehensive coastal monitoring plan for

[5] The Recovery Science Review Panel was convened by NMFS to provide guidance on technical aspects of recovery planning.

salmonids has been underway for several years by the California Department of Fish and Game, with input from NMFS; however, datasets that will allow assessment of status using the criteria described herein are likely more than a decade away. Consequently, the present values of the criteria put forth in this document are to inform the development of such a monitoring plan and to provide preliminary targets for recovery planners.

1.2 Relationship Between Biological Viability Criteria and Delisting Criteria

Before elaborating on our approach to developing biological viability criteria, it is important to distinguish *biological viability criteria* proposed herein from the *recovery criteria* that will ultimately be put forth in a recovery plan. Although the ESA provides no detailed guidance for defining recovery criteria, subsequent NMFS publications including *Recovery Planning Guidance for Technical Recovery Teams* (NMFS 2000), and *Interim Endangered and Threatened Species Recovery Planning Guidance* (NMFS 2006) have elaborated on the nature of recovery criteria and underlying goals and objectives. NMFS (2006) clearly affirms that the primary purpose of the Federal Endangered Species Act is to *"...provide a means by whereby the ecosystems upon which endangered species and threatened species depend may be conserved"* (16 U.S.C. 1531 et sec., section 2(a)), noting that "*in keeping with the ESA's directive, this guidance focuses not only on the listed species themselves, but also on restoring their habitats as functioning ecosystems.*" To this end, NMFS (2006) directs that recovery criteria must address not only the biological status of populations and ESUs, but also the specific threats and risk factors that contributed to the listing of the ESU. These threats and risks can include (a) current or threatened destruction, modification or curtailment of the ESU's habitat or range; (b) overutilization for commercial, recreational, scientific or educational purposes; (c) disease or predation; (d) the inadequacy of existing regulatory mechanisms; (e) other natural or manmade factors affecting the ESU's continued existence (16 USC 1533). Thus, formal recovery or delisting criteria for Pacific salmonids will at a minimum likely include at least two distinct elements: (1) criteria related to the number, sizes, trends, structure, recruitment rates, and distribution of populations, as well as the minimum time frames for sustaining specified biological conditions; and (2) criteria to measure whether threats to the ESU have been ameliorated (NMFS 2006)[6]. The latter criteria have been referred to as "administrative delisting criteria" (NMFS 2000), and may require that management actions be taken to address specific threats before a change in listing status would be considered (NMFS 2006). Recovery plans may also set

[6] The need to address each listing factor when developing delisting criteria has been affirmed in Court, which concluded that "since the same five statutory factors must be considered in delisting as in listing…in designing objective, measurable criteria, the FWS must address each of the five delisting factors and measure whether threats to the [species] have been ameliorated." (Fund for Animals v. Babbitt, 903 F. Supp. 96 (D.D.C 1995), Appendix B).

recovery goals higher than those needed to achieve delisting of the species under ESA in order to allow for other uses (e.g., commercial, recreational, or tribal harvest) or to provide ecological benefits (e.g., maintenance of ecosystem productivity). These additional goals have been termed "broad-sense" recovery goals (NMFS 2000). Where such recovery goals are established, NMFS (2006) indicates that they should be clearly distinguished from ESA-specific recovery goals.

The *biological viability criteria* proposed in this document represent the NCCC TRT's recommendations as to the minimum population and ESU characteristics indicative of an ESU having a high probability of long-term (> 100 years) persistence. Population viability criteria define sets of conditions or rules that, if satisfied, we believe would suggest that the population is at low risk of extinction. ESU viability criteria define sets of conditions or rules related to the number and configuration of viable populations across a landscape that would be indicative of low extinction risk for the ESU as a whole. The ESU criteria do not explicitly specify which populations must be viable for the ESU to be viable (though in some cases, certain populations will likely be critical for achieving viability, given their current status or functional role), but rather they establish a framework within which there may be several ways by which ESU viability can be achieved.

The biological viability criteria can be viewed as indicators of biological status and thus are likely to be directly related to the biological delisting criteria that will be defined in a recovery plan. However, the criteria are independent of specific sources of mortality (natural or human-caused) or specific threats to populations and ESUs that led to their listing under ESA; thus, the criteria should not be construed as sufficient, by themselves, for determining the ESA status of ESUs. These threats, and associated administrative delisting criteria, are to be addressed through a formal "threats assessment" process in the second phase of recovery planning. Likewise, development of "broad-sense" recovery goals is to occur during the next phase of recovery planning. These latter processes will provide the basis for determining which populations have the highest likelihood of being recovered to viable levels (based on current status, practicality and cost of restoring habitat or otherwise ameliorating threats) or to levels that will achieve broad-sense recovery goals. Thus, formal biological delisting criteria contained in a recovery plan are likely to have greater specificity about which populations may need to be viable before the ESU is considered so.

NMFS (2006) recovery planning guidance document highlights a number of objectives that are relevant to the TRT's task of developing biological viability criteria. Recovery and long-term sustainability of endangered or threatened species depends on the following:

- Ensuring adequate reproduction for replacement of losses due to natural mortality factors (including disease and stochastic events)
- Maintaining sufficient genetic diversity to avoid inbreeding depression and to allow adaptation
- Providing sufficient habitat (type, amount, and quality) for long-term population maintenance
- Elimination or control of threats (which may include having adequate regulatory mechanisms in place).

The NMFS interim guidance document further states that, in order to meet these general objectives, recovery criteria should at a minimum address three major issues related to long-term persistence of populations and ESUs: representation, resiliency, and redundancy (NMFS 2006). Representation involves conserving the breadth of the biological diversity of the ESU to conserve its adaptive capabilities. Resiliency involves ensuring that populations are sufficiently large and/or productive to withstand both natural and human-caused stochastic stressor events. Redundancy involves ensuring a sufficient number of populations to provide a margin of safety for the ESU to withstand catastrophic events (NMFS 2006). Each of these issues may be relevant at more than one spatial scale. For example, genetic representation may be important both within populations (i.e., maintaining genetic diversity at the population level, which can allow for the expression of phenotypic diversity and hence buffer against environmental variation) and among populations across an ESU (i.e., preserving genetic adaptations to local or regional environmental conditions to maintain evolutionary potential in the face of large-scale environmental change). The NCCC TRT has attempted to develop viability criteria that encompass these primary principles and objectives.

It is not practical for the TRT, which must necessarily focus on ESU-scale analysis, to address various threats and risk factors that contributed to the ESA listing of ESUs within the NCCC Recovery Domain or to develop criteria related to those threats and risks at the resolution and detail required for effective recovery. Nevertheless, it is important to understand the primary factors that have contributed to salmonid declines within these areas so that the proposed viability criteria can be viewed in an appropriate context. Each listed ESU within the domain has undergone one or more status reviews prior to listing, in which a number of general factors for decline were identified. Federal Register notices containing the final listing determinations likewise have identified factors contributing to the declines of listed species[7]. All of these reviews have identified habitat loss and degradation associated with land-use practices as a primary cause of population declines within the listed salmon and steelhead ESUs (Weitkamp et al. 1995;

[7] For the most part, published status reviews and Federal Register Notices have provided only general lists of factors that affect multiple populations within an ESU or DPS; they typically do not provide details on population-specific risk factors.

Busby et al. 1996; Myers et al. 1998; NMFS 1999; Good et al. 2005). Almost all watersheds within the domain have experienced extensive logging and associated road building, which have wide-reaching effects on hydrology, sediment delivery, riparian functions (e.g., large wood recruitment, fine organic inputs, bank stabilization, stream temperature regulation), and channel morphology. Activities such as splash damming and "stream cleaning," though no longer practiced, have had substantial effects on channel morphology that continue to affect the ability of streams and rivers to support salmonids. Impacts of agricultural practices on aquatic habitats, though spatially perhaps not as widespread as those associated with forest practices, are often more severe since they typically involve repeated disturbance to the landscape, often occur in historical floodplains or otherwise in close proximity to streams, commonly involve diversion of water in addition to the land disturbance, and frequently involve intensive use of chemical fertilizers and pesticides that degrade water quality. Urbanization has severely impacted streams, particularly in the San Francisco Bay area, portions of the Russian River basin, and the Monterey Bay area, often involving stream channelization, modification of hydrologic regime, and degradation of water quality, among other adverse effects. Hard rock (mineral) and aggregate (gravel) mining practices have also substantially altered salmonid habitats in certain portions of the domain. For example, gravel extraction in the Russian River has substantially altered channel morphology both in the mainstem and in tributaries entering the mainstem (Kondolf 1997). Loss and degradation of estuarine and lagoon habitats—which are important juvenile rearing and feeding habitats (Smith 1990; Bond 2006; Hayes et al. in review), as well as being critical areas of acclimation while smolts make the transition from fresh to salt water—have likely also contributed to declines of salmon and steelhead in the region. Published status reviews have also noted that severe floods, such as the 1964 flood, have exacerbated many impacts associated with land use (Busby et al. 1996; Myers et al. 1998).

In certain watersheds and regions (e.g., Mad River, Eel River, Russian River, and many San Francisco Bay tributaries), dams have blocked access to historical spawning and rearing habitats (Busby et al. 1996), although compared with other regions, such as California's Central Valley and the Columbia Basin, the fraction of historical habitat lost behinds dams is relatively small in most of the NCCC Recovery Domain. In addition to preventing access to historical spawning and rearing habitats, dams disrupt natural hydrologic patterns, sediment transport dynamics, channel morphology, substrate composition, temperature regimes, and dissolved gas concentrations in reaches downstream, potentially affecting the suitability of these reaches to salmonids. Water withdrawals for agricultural, industrial, and domestic use have resulted in reduced stream flows, increased water temperatures, and otherwise diminished water quality. Water diversions are widespread throughout the domain but are a particularly acute problem in portions of the domain with intense agriculture or urbanization, such as portions of the

Russian River, upper Navarro River, tributaries of San Francisco and Monterey bays, and the lower reaches of many coastal watersheds.

Excessive commercial and sport harvest of salmonids is also believed to have contributed to the declines of populations within the region, though little information on harvest rates is provided in published status reviews for ESUs or DPSs within the NCCC Recovery Domain. In addition to affecting the number of adults that return to their natal streams to spawn, harvest can also affect the age- and size-structure of returning adults through selective harvest of older individuals, which are vulnerable to fishing for a longer period or to size-selective fishing gear (Ricker 1981). This selectivity usually results in a reduction in the proportion of larger, older individuals in a population, particularly for Chinook salmon, which are vulnerable to ocean fisheries for several years. Selection on size- and age-at-maturity can result not only in immediate demographic consequences (e.g., reductions in spawner abundance, decreased average fecundity of spawners, and increased variability in abundance; Anderson et al. 2008), but may potentially result in genetic selection for early maturation (Hankin et al. 1993). Such changes in population attributes may have longer-term demographic consequences. Though directed commercial and sport harvest of listed salmonids in the NCCC Recovery Domain has decreased since populations were first listed in the mid-1990s, incidental take of listed ESUs continues to occur in fisheries targeting non-listed ESUs, including Central Valley and Klamath River fall Chinook salmon. Although no direct estimates of harvest rates are currently available for listed ESUs or DPSs in the NCCC Recovery Domain, it seems unlikely that harvest rate of CC-Chinook salmon stocks is less than that for Klamath River Chinook, and it is possible that some of these populations (e.g., Eel River Chinook salmon) are harvested at very high rates in the Central California fishery.

Status reviews have identified hatchery practices, including out-of-basin transfers of stocks, as important risk factors in all four listed ESUs (Weitkamp 1995; Busby et al. 1996; Myers et al. 1998; Good et al. 2005). While the status reviews emphasize potential genetic risks associated with hatcheries, there are demographic and ecological risks as well (see Section 2.2 of this report for further discussion). Additionally, the introduction or invasion of nonnative fishes may also pose a significant threat to salmonids within the domain. Busby et al. (1996) identified the introduction of nonnative species (e.g. Sacramento pikeminnow) as a significant threat to NC steelhead populations in the Eel River, and it is likely a threat to Chinook and coho salmon populations in this basin as well (CDFG 2002). Numerous other nonnative species, including various cyprinids, centrarchids, ictalurids, and clupeids, have been introduced into coastal watersheds within the domain and may influence listed populations through predation or competition. The Redwood Creek, Mad River, Eel River, Russian River, and Tomales Bay systems may be the most likely systems affected by such introductions, as nonnative fishes currently make up 30% or more of the total fish species present in these watersheds (Moyle 2002). Many tributaries of San Francisco Bay likewise have a high percentage of nonnative fishes (Leidy 2007).

All of the factors listed above have likely contributed to declines in the abundance and distribution of listed salmon and steelhead within the NCCC Recovery Domain and will need to be addressed in the development of recovery plans. Although attainment of the biological criteria proposed herein would suggest that some of the conditions that led to listing have been ameliorated, natural variation in environmental conditions in both the freshwater and marine environments can produce substantial changes in abundance of salmon and steelhead, even without fundamental improvement in habitat quality (Lawson 1993). Consequently, complementary analyses of both biological status and existing or future threats will need to form the basis of future status assessments.

1.3 Population Delineations and Biological Viability Criteria

Scientists from NMFS' Northwest Fisheries Science Center and Southwest Fisheries Science Center developed a series of guidelines for setting viability objectives in a document titled "*Viable Salmonid Populations and the Recovery of Evolutionarily Significant Units*" (McElhany et al. 2000). The viable salmonid population (VSP) concept developed in McElhany et al. (2000) forms the foundation upon which the draft viability criteria proposed here rests. McElhany et al. (2000) defined a viable salmonid population as "*an independent population of any Pacific salmonid (genus* Oncorhynchus*) that has a negligible risk of extinction due to threats from demographic variation (random or directional), local environmental variation, and genetic diversity changes (random or directional) over a 100-year time frame*." They defined an independent population to be "*any collection of one or more breeding units whose population dynamics or extinction risk over a 100-year time period is not substantially altered by exchanges of individuals with other populations*." Their conceptualization thus distinguishes between independent populations, as defined above, and dependent populations, whose dynamics and extinction risk *are* substantially affected by neighboring populations.

For our purposes, we found it useful to further distinguish among independent populations based on both their viability in isolation and their degree of self-recruitment (i.e., the proportion of spawners of natal origin), which assists in identifying the functional role different populations historically played in ESU persistence (Bjorkstedt et al. 2005). We defined *functionally independent* populations as "those with a high likelihood of persisting over 100-year time scales and [that] conform to the definition of independent 'viable salmonid populations' offered by McElhany et al. (2000, p. 3)". We defined *potentially independent populations* as those that "have a high likelihood of persisting in isolation over 100-year time scales, but are too strongly influenced by immigration from other populations to exhibit independent dynamics." Thus, whereas the McElhany et al. definition of independence explicitly requires sufficient isolation for demographic independence, the NCCC TRT definition of independence encompasses populations that could conceivably persist in isolation in the absence of adjacent populations that at one time may have substantially influenced their extinction risk (Bjorkstedt et al. 2005). We also define dependent populations as those that have a substantial likelihood of going extinct within a 100-year time period in isolation, but that receive sufficient immigration to alter their dynamics and reduce their extinction risk (Bjorkstedt et al. 2005).

These distinctions are important to consider in developing a recovery strategy for two reasons. First, certain historical functionally independent populations likely had disproportionate influence on ESU persistence. By definition, functionally independent populations are net sources of strays that influence the dynamics of neighboring populations. Loss or reduction of such populations thus may have greater impact on ESU persistence, since associated potentially independent and dependent populations are also negatively affected. Second, recovery planners will need to consider the functional role a population is playing or might play in the future, relative to its historical role. For example, dams that block access to a significant proportion of a population's habitat might preclude that population from behaving as a functionally independent population. While such a population may continue to persist, it should not be viewed as providing the same contribution to ESU viability as the historical population. Conversely, there may be certain circumstances where functionally or potentially independent populations have been lost or severely depleted, but neighboring dependent populations continue to persist. In these instances, dependent populations, while not expected to persist indefinitely in isolation, may provide the only reasonable opportunity for recovering nearby populations classified as functionally or potentially independent under historical conditions. Dependent populations may also provide reservoirs of genetic diversity that has been lost from depleted independent populations or provide connectivity among independent populations that is important for long-term ESU viability. And finally, it may be possible for a collection of spatially proximate dependent populations to function as a metapopulation that is viable without input from independent populations. Thus, when prioritizing recovery efforts among watersheds, recovery planners will need to evaluate the full context of the historical and current population structure.

1.4 Report Organization

In the remaining chapters of this report, we present both the general framework for assessing population and ESU viability, and application of the framework to the four listed ESUs within the NCCC Recovery Domain. Chapter 2 describes an approach for categorizing populations according to extinction risk that extends the framework proposed by Allendorf et al. (1997). Extinction risk is evaluated based on six metrics intended to address issues of abundance, productivity, spatial structure, and diversity identified in McElhany et al. (2000). We briefly summarize the rationale for inclusion of each viability criterion and then discuss some assumptions and caveats associated with each. The TRT augmented the Allendorf et al. (1997) criteria by adding criteria related to spawner densities and hatchery influences. In these two instances, we provide somewhat more detailed rationale for the criteria (see Appendices B and C). These modifications to the Allendorf et al. (1997) approach have been done in coordination with other TRTs in NMFS' Southwest Region; thus, there is substantial overlap in approaches used (see Lindley et al. 2007; Boughton et al. 2007; Williams et al. in prep.).

Chapter 3 puts forth viability criteria at the levels of diversity strata and entire ESUs. Diversity strata were identified in the *Population Structure Report* (Bjorkstedt et al. 2005), and have subsequently been revised by the TRT (see Appendix A). These strata represent regional population groupings that have evolved under similar environmental conditions, as well as life-history diversity expressed within a particular watershed (e.g., spring- and fall-run Chinook salmon). Criteria at the level of diversity strata and ESUs are directed toward increasing the likelihood that genetic and phenotypic diversity is represented across the ESU, that there is redundancy in viable populations within diversity strata to reduce the risk that an entire diversity stratum is affected by a single catastrophic event, and that there is sufficient connectivity among populations to maintain long-term demographic and genetic processes.

In Chapter 4, we apply the methods described in the preceding two chapters to the four ESUs within the NCCC Recovery Domain. As noted earlier, the NCCC Recovery Domain suffers from an almost complete lack of appropriate data that can inform the risk analysis. This paucity of data precludes us from drawing firm conclusions about population or ESU status based on our framework; however, the exercise is instructive both in identifying important information gaps that need to be filled and in establishing preliminary numeric targets that can assist planners in developing recovery strategies.

2 Population Viability Criteria

2.1 Key Characteristics of Viable Populations

McElhany et al. (2000) propose a conceptual framework for both defining a viable salmonid population (VSP) and the critical parameters that should be evaluated when assessing viability of both populations and ESUs. The issue of defining populations for the NCCC Recovery Domain has been treated at length in Bjorkstedt et al. (2005). Here, we turn our attention to defining appropriate parameters to be measured when assessing viability and the development of specific metrics and criteria that would enable classification of populations according to their extinction risk.

McElhany et al. (2000) propose that four general population parameters are key to evaluating population status: abundance, population growth rate, population spatial structure, and diversity. Abundance—the number of individuals within the population at a given life stage—is of obvious importance. Other factors being equal, small populations are at greater risk of extinction than larger populations due to the fact that several deterministic and stochastic processes operate differently in small versus large populations. As discussed by McElhany et al. (2000), to be viable, a population needs to be large enough 1) to have a high probability of surviving environmental variation of the patterns and magnitude observed in the past and expected in the future; 2) to allow compensatory processes to provide resilience to natural environmental and anthropogenic disturbances; 3) to maintain its genetic diversity over the long term (i.e., avoiding inbreeding depression, fixation of deleterious alleles, genetic drift, and loss of long-term adaptive potential); and 4) to provide important ecological functions (e.g., provision of marine-derived nutrients to maintain productivity, physical modification of habitats such as spawning gravels) throughout its life cycle.

Population growth rate refers to the actual or expected ratio of abundances in successive generations, and provides information about how well the population is performing in its environment over its entire life cycle. Populations that consistently fail to replace themselves over extended periods are at greater risk of extinction than those that are consistently at or above replacement. Additionally, populations with higher intrinsic productivity (i.e., recruits per spawner when spawner densities are low, compensation is not reducing per capita productivity, and depensatory effects are absent) recover more rapidly following a decline in abundance than do those with lower intrinsic productivity. Thus, a population with lower abundance but higher intrinsic productivity may be less prone to extinction than one with greater mean abundance but lower productivity. Additionally, when comparing populations with equal mean abundance and intrinsic productivities, populations that exhibit more variability in abundance and growth rate are likewise more vulnerable to extinction than less-variable populations.

Spatial structure refers to the distribution of members in the population at a given life stage among the potentially available habitats and the processes that give rise to that structure (McElhany et al. 2000). Populations may organize themselves in a variety of ways across a watershed or landscape, depending on the spatial arrangement and quality of habitats and the dispersal characteristics of individuals within the population. Under natural conditions, the distribution of favorable habitats may shift over time in response to environmental disturbances. Consequently, local breeding groups with differing relative productivities may populate the landscape. Populations that exhibit such structure may be less vulnerable to disturbances such as fires, floods, landslides, and toxic spills that typically occur at relatively small scales (reach to subwatershed) than populations with more restricted distributions. Portions of the landscape unaffected by the disturbance may assume increased importance as disturbed areas recover and may provide sources of colonizers as habitat conditions improve, imparting greater resilience to the population. Through each of these mechanisms, spatial diversity can reduce variation in population growth rate, lowering a population's extinction risk. Maintenance of this spatial structure requires that high quality habitat patches, and suitable corridors connecting these patches to one another and the marine environment, be consistently present.

Diversity is the variety of life histories, sizes, ages, fecundity, run timing, and other traits expressed by individuals within a population, and the genetic variation that in part underlies these differences. In many respects, diversity is tied closely to spatial structure. Diversity results from the interaction of genetic and environmental factors, and it imparts several attributes to populations that influence persistence by spreading of risk through both space and time. First, genetic diversity potentially allows a population to use a wider range of habitats than it could with lower diversity; thus, loss of this diversity may diminish the productive capacity and spatial extent of a population. Additionally, distribution of populations across a heterogeneous watershed may lead to phenotypic variation in characteristics such as length of freshwater residence, resulting in more complicated age structures. Such diversity can buffer populations against poor environmental conditions in either the freshwater or marine environment, effectively spreading risk across both time and space and thereby increasing population resilience in the face of environmental stochasticity. And finally, the underlying genetic diversity of a population determines its ability to adapt to long-term changes in environmental conditions.

Although it is clear that each of the parameters described by McElhany et al. (2000) is important to assessing viability, selecting specific metrics to relate these parameters to viability is less straightforward, and defining criteria for each of these metrics proves even more challenging. For abundance and productivity parameters, relationships between various metrics and extinction risk are more fully developed in the scientific literature. For spatial structure and diversity, the theoretical basis underlying the importance of these parameters is clear, but there is substantially more uncertainty regarding quantitative relationships between these attributes and population viability. Nevertheless, the TRT felt strongly that our approach needed to address each of these issues, since failing to do so would leave a substantial gap between our approach and both the conceptual framework proposed in McElhany et al. (2000) and interim NMFS guidance on viability criteria (NMFS 2006). We also note that although the VSP framework proposed by McElhany et al. (2000) has intuitive appeal, we found it difficult to develop individual metrics that correspond to the VSP parameters in one-to-one fashion. Thus, several of the metrics we propose directly or indirectly address multiple VSP parameters.

In the VSP framework, the concept of population viability can be viewed from two distinct but equally important perspectives. The first perspective relates to the goal of defining the minimum viable population size (MVP) for which a population can be expected with some specified probability to persist over a specified period of time (Soulé 1987; Nunney and Campbell 1993). In one sense, the minimum viable population size can be thought of as identifying the approximate lower bounds for a population at which risks associated with demographic stochasticity, environmental stochasticity, severe inbreeding, and long-term genetic losses are negligible (Soulé 1987). This conceptualization of viability asks where a population is likely going in the future, but not necessarily where it has been in the past. For example, with respect to genetic diversity, criteria related to a fixed MVP size are intended to guard against further erosion of genetic diversity but do not necessarily consider diversity that may have already been lost.

A second way to consider viability is in terms of how a population is currently functioning in relation to its historical function. From this perspective, historical patterns of abundance, productivity, spatial structure, and diversity form the reference conditions about which (at least for independent populations) we have high confidence that the population had a high probability of persisting over long periods of time. This broader (and longer term) view asks how a population functioned in its historical context (e.g., what roles did spatial structure and diversity play in population persistence?), and what functional role the population played in relation to other populations within an ESU (e.g., was the population likely a key source of migrants that contributed to the persistence of other independent or dependent populations?).

As populations depart from these historical conditions, their probability of persistence likely declines and their functional role with respect to ESU viability may be diminished.

The criteria we propose in this document encompass both of these perspectives, addressing both immediate demographic and genetic risks, as well longer-term risks associated with loss of spatial structure and diversity that are important both for population resilience (and hence persistence) and the ability of populations to fulfill their roles within the ESU and thus to contribute to ESU viability. Given the technical difficulties associated with developing accurate population viability analyses that focus on a strict definition of viability (e.g., MVP), the second perspective is especially useful in that it embodies a precautionary approach through which increasing departure from historical characteristics logically requires a greater degree of proof that a population is indeed viable. Likewise, this second perspective links directly to viability criteria for higher levels of biological organization.

2.2 Population-Level Criteria

The approach we use seeks to classify populations into various extinction risk categories based on a set of quantitative criteria. Both the approach and the specific criteria employed have their roots in the IUCN (1994) red list criteria (derived in part from Mace and Lande 1991) and subsequent modifications made by Allendorf et al. (1997) to specifically deal with populations of Pacific salmon. The Allendorf et al. (1997) framework defines four levels of extinction risk according to the probability of extinction over a specified time frame:

Very high: 50% probability of extinction within 5 years
High: 20% probability of extinction within 20 years
Moderate: 5% probability of extinction within 100 years
Special concern: Historically present, believed to still exist, but no current data

Evaluation of extinction risk is then done either based on population viability analysis (PVA) or, in the absence of sufficient data to construct a credible PVA model, using four surrogate criteria related to population size and trend in abundance. These surrogate criteria address effective population size per generation (or, in the absence of data on effective population size, total population size), population declines, and the effects of recent catastrophes on abundance (see Table 1 in Allendorf et al. 1997).

For our purposes, we make several modifications to the Allendorf et al. (1997) approach—in both the risk categories and the metrics used to evaluate risk—to deal with our specific needs in recovery planning (Table 1). First, we add a "low risk" category, which is implicit in Allendorf et al. (1997), defining criteria we believe are indicative of a high likelihood (>95%) of persistence over a 100-year time frame. Second, we collapse the "very high risk" and "high risk" categories of Allendorf et al. (1997) into a single "high risk" category. Whereas discriminating between "high risk" and "very high risk" was critical to Allendorf et al.'s emphasis on prioritizing stocks for conservation, the distinction is less important for our purposes, since either categorization would clearly indicate populations that should not be considered viable over short-to-moderate time frames.

The practical effects of collapsing these two categories are relatively minor, though they lead to a configuration and implementation of the viability criteria table that differs somewhat from that of Allendorf et al. (1997). Foremost, we adopt a rule that the assignment of risk to the population is based on the highest risk category for any individual risk metric. For example, a population rated at "high risk" based on effective population size, but moderate or low risk for the other metrics would receive the "high risk" rating. Allendorf et al. (1997) employ a similar strategy but have an additional rule whereby populations that rank at a certain risk level for more than one metric get elevated to the next highest risk level when categorizing the population (e.g., a population rated at moderate risk for two metrics is considered at high risk overall). For this reason, the criteria listed in our "high risk" and "moderate risk" categories superficially align themselves with the "very high risk" and "high risk" categories, respectively, in Allendorf et al. (1997). In actual application, a population that satisfies a single criterion (as opposed to two or more) receives the same ranking using either the Allendorf et al. (1997) or the NCCC TRT approach. We viewed our configuration of the risk matrix to be somewhat simpler to apply and understand, but we note that populations that rank at a given level for multiple metrics should be considered more vulnerable to extinction than populations that rank at that level for a single metric. Finally, we define as "data deficient" populations that are believed to still persist but where data for evaluating risk are partially or entirely lacking. This category equates to the "special concern" category of Allendorf et al. (1997).

Two extensions we made to the Allendorf et al. (1997) approach were the addition of criteria related to spawner density and to the potential effects of hatchery activities on wild populations. The density criteria are intended to address aspects of spatial structure and diversity that are important to population viability (McElhany et al. 2000) but not explicitly addressed by the Allendorf et al. metrics. We believe there is a compelling theoretical basis for including these criteria, though we acknowledge that, as with

Table 1. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids. Overall risk is determined by the highest risk score for any category. See Table 2 for definitions of N_g, N_e, *and* N_a. Modified from Allendorf et al. (1997) and Lindley et al. (2007).

Population Characteristic	**Extinction Risk**		
	High	**Moderate**	**Low**
Extinction risk from population viability analysis (PVA)	$\geq$ 20% within 20 yrs	$\geq$ 5% within 100 yrs but < 20% within 20 yrs	< 5% within 100 yrs
	- or any ONE of the following -	- or any ONE of the following -	- or ALL of the following -
Effective population size per generation *-or-* Total population size per generation	$N_e \leq 50$ *-or-* $N_g \leq 250$	$50 < N_e < 500$ *-or-* $250 < N_g < 2500$	$N_e \geq 500$ *-or-* $N_g \geq 2500$
Population decline	Precipitous decline[a]	Chronic decline or depression[b]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[c]	Not apparent
Spawner density	$N_a/IPkm$[d] ≤ 1	$1 < N_a/IPkm <$ MRD[e]	$N_a/IPkm \geq$ MRD[e]
Hatchery influence[f]	Evidence of adverse genetic, demographic, or ecological effects of hatcheries on wild population		No evidence of adverse genetic, demographic, or ecological effects of hatchery fish on wild population

[a] Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size $N_a \leq 500$ spawners (historically small but stable populations not included) <u>*or*</u> $N_a > 500$ but declining at a rate of $\geq$10% per year over the last two-to-four generations.
[b] Annual run size N_a has declined to $\leq$ 500 spawners, but is now stable *or* run size $N_a > 500$ but continued downward trend is evident.
[c] Annual run size decline in one generation < 90% but biologically significant (e.g., loss of year class).
[d] *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).
[e] MRD = minimum required spawner density and is dependent on species and the amount of potential habitat available. Figure 5 summarizes the relationship between spawner density and risk for each species.
[f] Risk from hatchery interactions depend on multiple factors related to the level of hatchery influence, the origin of hatchery fish, and the specific hatchery practices employed.

other metrics, there is considerable uncertainty surrounding the relationship between the specific metrics and extinction risk. The hatchery criteria consider potential genetic, demographic, and ecological risks associated with the interaction between hatchery and wild fish. Here, the NCCC TRT concluded that simple numerical criteria relating hatchery influence to risk were inappropriate given the substantial variation in how individual hatcheries are operated and the fact that impacts associated with hatcheries are often highly context-dependent. Instead, we propose general narrative criteria related to hatcheries under the assumption that each case will require independent analysis of risks. Allendorf et al. (1997) address the issue of hatchery influence in a separate analysis that evaluates the biological consequences of extinction for populations that have been free from such introductions, but they do not attempt to develop criteria linking hatchery influence to risk.

Several points of clarification regarding terminology used in this report are required before beginning our discussion of the population viability criteria. First, we use the term “risk category” to describe the possible status (i.e., extinct, high risk, moderate risk, low risk, or data deficient) of a population in relation to either a particular population characteristic or the full suite of characteristics. We use the term “risk metric” to mean those attributes of a population that are measured in order to evaluate risk, and the term “risk criteria” to indicate the specific values of a metric that are used to place a population into a particular risk category for that metric. We also note that in describing population size, our criteria use three different terms: N_a, which is number of annual spawners; N_g, the number of spawners per generation; and N_e, the effective population size per generation (Table 2). The inclusion of population size metrics expressed as functions of both annual run size and the numbers of spawners per generation creates some potential for confusion; however, it is necessary both to provide a generalized table that can be used across all three species (each with a unique mean generation time) within our domain and to reflect the different time scales over which the specific processes addressed by these criteria occur (e.g., demographic processes that operate at an annual time scale versus genetic processes where generational time scales are more relevant). Table 2 summarizes these different terms for population abundance.

Table 2. Description of variables used to describe population size in the population viability criteria. All expressions of population size refer to naturally spawning adults, inclusive of jacks but exclusive of hatchery fish.

Population Variable	Description
N_a	Total abundance of adult spawners in a year. Related forms that appear in this report include $N_{a(t)}$ = the number of adult spawners in year t; and $\overline{N}_{a(geom)}$ = the geometric mean of adult spawner abundance over a specified period (see equation 3, pg. 27).
N_e	Effective population size per generation.
N_g	Total number of spawners for the generation. Related forms that appear in this report include $N_{g(t)}$ = the running sum of adult abundance at time t for a period equal to one generation (rounded to nearest whole year; see equation 2, pg. 24); and $\overline{N}_{g(harm)}$ = the harmonic mean of the running sums of abundance, $N_{g(t)}$, calculated over a specified period (see equation 1, pg. 24).

In the sections that follow, we provide a discussion of each criterion listed in the modified Allendorf et al. (1997) table, including the rationale for inclusion of the criteria, the specific criteria associated with low-, moderate-, and high-risk populations, and guidance on metrics and estimators used in application of the criteria. We also discuss additional considerations that need to be made in evaluating viability using this generalized framework.

Extinction Risk Based on Population Viability Analysis (PVA)

Rationale: The first set of criteria in Table 1 follow directly from Allendorf et al. (1997) and deal with direct estimates of extinction risk over a specified time frame based on population viability models. If PVAs are available and considered reasonable, then such analyses may be sufficient for assessing risk. In fact, Allendorf et al. (1997) intended the remaining criteria in the table to be used as surrogates if models for estimating extinction probability were not available or if parameters required in such models could not be estimated with acceptable accuracy. A number of models for population viability analysis have been proposed (e.g., Samson et al. 1985; Simberloff 1988; Ferson et al. 1988, 1989; Ginzburg et al. 1990; Dennis et al. 1991; Lee and Hyman 1992; Lacy 1993; Lindley 2003). We note, however, that there is considerable discussion in the literature about the value and limitations of PVA models, particularly as it relates to predicting extinction risk in small populations (see review by Beissinger and Westphal 1998; Mann and Plummer 1999; Coulson et al. 2001; Reed et al. 2002). Some specific concerns are discussed under *Metrics and Estimation* below. We also note that if data sufficient to construct a credible PVA model are available, then it is likely that the population can be assessed in relation to most or all of the alternative metrics within Table 1 as well. We therefore recommend using both approaches and comparing the outcomes, as these comparisons may illuminate potential limitations of either approach.

Criteria: Consistent with Allendorf et al. (1997), we define high-risk populations as those with greater than a 20% probability of extinction within 20 years; moderate-risk populations as those with at least a 5% probability of extinction within 100 years but less than 20% probability of extinction within 20 years; and low-risk populations as those with less than a 5% extinction probability within 100 years (Table 1).

Metrics and Estimation: Population viability models produce estimates of extinction probability over a specified time frame and are thus directly comparable to the criteria. The Oregon Coast TRT (OCTRT; Wainwright et al., in press) recommends applying a variety of models and averaging the results of those models, due to the fact that outcomes may differ substantially depending on underlying assumptions of the model and the suite of factors considered. Data needs for PVAs vary with the specific model or models used. In general, however, most PVAs estimate extinction risk based on at least four factors: current population abundance, intrinsic population growth rate, habitat capacity, and variability in growth rate arising from variation in fecundity, growth, or survival (Lande and Orzack 1988, Lande 1993; Wainwright et al., in press). Thus, at a minimum, data for estimating these population attributes are required.

Although PVAs allow incorporation of population-specific information that can help refine assessment of viability, the use of PVAs must be done cautiously, as there are many limitations to these models. The OCTRT (Wainwright et al., in press) identifies several issues to consider when using PVAs to evaluate the status of Pacific salmon. First, PVAs for salmonids are typically based on stock-recruitment models, of which there are several commonly used forms (e.g., Ricker, Beverton-Holt, and hockey-stick). PVA outcomes may differ depending on the underlying stock-recruitment model, and there is no general consensus among scientists about which of these models are most appropriate for salmonids. Second, PVAs are subject to statistical error and bias in parameter estimates that may arise from high measurement error in spawner abundance estimates or high environmental variation. Coulson et al. (2001) note that for PVAs to be meaningful, data must be of sufficiently high quality that estimates of the shape, mean, temporal variance, and autocorrelation (which could be caused by density-dependent processes) of the distribution of vital rates or population growth rate are accurate. Third, most models incorporate only a small subset of factors that may influence extinction risk. More complicated PVA models require more data, though it is not always clear that increasing complexity of models leads to superior performance, particularly when dispersal plays a role in population dynamics (Hill et al. 2002). Fourth, because PVA models represent projection into the future, the results depend critically on assumptions about future conditions, which cannot possibly be known (Coulson et al. 2001). Models that assume that the future will be similar to the recent past (i.e., the period during which data used to parameterize PVA models are collected) may be inaccurate or misleading if, as climate models suggest, the future climate is likely to differ substantially from that of the present. And fifth, obtaining reliable absolute predictions of extinction probability is difficult, as is verifying model predictions. These limits have caused some authors to suggest that PVAs should not be used to determine minimum viable population size or the specific probability of reaching extinction (Reed et al. 2002). Nevertheless, despite these limitations and concerns, PVAs represent an important tool for incorporating population-specific differences in vital rates, habitat quantity and quality, and other factors influencing persistence into assessments of extinction risk.

Effective Population Size/Total Population Size Criteria

Rationale: The first two surrogate extinction risk criteria—the effective population size criterion and the total population size criterion—are intended to address risks associated with inbreeding and the loss of genetic diversity within a population. Genetic variability is the source of adaptive potential of a population; thus, losses of genetic variability decrease the ability of a population to respond to changing environmental conditions (Allendorf et al. 1997). Furthermore, as populations decrease in size, demographic stochasticity becomes more important (Lande 1998), and inbreeding depression and genetic drift may reduce the average fitness of the population (Meffe and Carroll 1997), resulting in a greater extinction risk over short time scales. These deleterious genetic effects are a function of N_e, the effective population size (i.e., the size of an idealized population, where every individual has an equal probability of contributing genes to the next generation, having the same rate of genetic change as the population under study; Wright 1931), rather than the total number of spawners per generation, N_g. For most organisms, effective population sizes are substantially smaller than total population size because of variance in family size, unequal sex ratios, and temporal variation in population size (Lande 1995; Hartl and Clark 1997; Meffe and Carroll 1997).

The total population size criteria serve as alternative criteria when reliable direct estimates of effective population size are not available, which is likely to be the case for most populations. The criteria are based on an assumption that the ratio of effective spawners to total spawners (N_e/N_g) in most salmonid populations is on the order of 0.2 (Allendorf et al. 1997); thus, they are directly related to the proposed effective population size criteria.

Criteria:

Effective population size per generation (N_e) — We adopt three criteria related to effective population size to reflect these genetic risks. Populations are rated at high risk of extinction when $N_e \leq 50$. Below N_e of 50, populations are believed to be at high risk from genetic effects, such as inbreeding depression, genetic drift, and fixation of deleterious alleles (Franklin 1980; Soulé 1980; Nelson and Soulé 1987). Populations are considered at moderate risk of extinction when $50 < N_e < 500$, and populations are at low risk of extinction when $N_e \geq 500$ (Table 1).

Selection of $N_e = 500$ as a threshold between low and moderate risk has been the subject of considerable discussion in the literature. Allendorf et al. (1997) proposed that long-term adaptive potential begins to be compromised due to random genetic drift at $N_e < 500$, though they note that if populations are reproductively isolated from other populations then the N_e required to prevent loss of genetic variation might be as much as an order of magnitude greater (i.e., N_e = 5,000; Nelson and Soulé 1987). Lande (1995) has argued that the models used to derive the $N_e > 500$ rule assume that all mutations are mildly deleterious, whereas subsequent work suggests that most mutations with large effects are strongly detrimental, with perhaps only 10% being mildly deleterious. Thus, Lande (1995) proposed that N_e of 5,000, rather than 500, may be necessary to maintain normal levels of adaptive genetic variance in quantitative characters under a balance between mutation and genetic drift. On the other hand, the models of Franklin (1980) and Soulé (1980) also assume that populations are closed to immigration (Lindley et al. 2007). Low levels of immigration—as few as one or two individuals per generation—can be sufficient to prevent the loss of genetic diversity through drift (Lacy 1987). For most salmon and steelhead populations within the NCCC recovery domain, such rates of migration among populations are reasonable, or at least were so under historical conditions. Because violations of the assumptions discussed act in opposition to one another, we accept the $N_e = 500$ recommendation of Allendorf et al. (1997) as a reasonable criterion for defining the threshold between populations at low and moderate risk.

Total population size per generation (N_g) **—** The total population size criteria assume that the N_e/N_g ratio for salmonids is approximately 0.2; thus, the criteria are directly proportional (five-fold higher) than those for effective population size based on the rationale given above. Populations are considered at high risk of extinction at $N_g \leq 250$, moderate risk of extinction where $250 < N_g < 2500$, and low risk of extinction where $N_g \geq 2500$. We re-emphasize that the total population size criteria are directed at genetic concerns and that reliance on N_g as a metric incurs greater uncertainty as a consequence of uncertainty in the N_e/N_g ratio.

Metrics and Estimation:

Effective population size per generation (N_e) — The specific metric to be evaluated will depend on which approach to N_e estimation is used (see below). For genetic methods, the precision of the N_e estimate is dependent on numerous factors, including sample sizes, number of alleles surveyed, and number of generations between samples (Waples 1989); thus, it is difficult to generalize about an appropriate formulation or temporal scale of sampling.

Although direct estimates of N_e based on genetic or demographic methods are theoretically the most accurate for evaluating genetic risks to populations, N_e is extremely difficult to estimate in natural populations (Waples 1989, 2002; Heath et al. 2002). Estimation of N_e from demographic data requires detailed information on the mean and variance among individuals of relative reproductive success (Nunney and Elam 1994; Waples 2002). Such information is difficult to obtain even in cultured populations and impossible to gather in wild populations without complete, genetically determined pedigrees. To overcome these difficulties, several authors have developed methods for indirectly estimating N_e using molecular genetic data. One such approach, the temporal method, involves estimating changes in allelic frequencies through time, with the change expected to be proportional to N_e (Waples 1989, 1990; Williamson and Slatkin 1999). Such methods require collection of genetic data from two points in time that are separated by at least a full generation (preferably longer), may produce estimates that are either biased or have large variance, can be computationally complex, and are typically based on a set of assumptions (e.g., populations are isolated and genetic markers are selectively neutral) that may not be true (Williamson and Slatkin 1999). Thus, while estimates of N_e derived from genetic data can be valuable, care must be taken in their interpretation.

Total population size per generation (N_g) — We recommend that N_g be approximated as the harmonic mean of the running sum of adult spawner abundance over the mean generation time for the species and population (Li 1997). Mathematically, this can be expressed as follows:

(1) $$\overline{N}_{g(harm)} = \frac{1}{\frac{1}{n}\sum_{t=1}^{n}\frac{1}{N_{g(t)}}}$$

where $N_{g(t)}$ is the running sum of adult abundance at time t for a period equal to the mean generation time k of the population (rounded to the nearest whole year)

(2) $$N_{g(t)} = \sum_{i=t-k}^{t} N_{a(i)}$$

and n is the number of years for which the running sum can be calculated. The estimate should be based on counts of naturally spawning fish (exclusive of hatchery-origin fish, but inclusive of jacks[8]) over a period representing at least four generations. Use of the harmonic mean, which gives greater weight to low values of N_g, reflects concern over the potential long-term consequences of a genetic bottleneck on population persistence; populations that have experienced a recent bottleneck may require extended periods of relatively high abundance to be considered no longer at risk (see discussion on page 25).

[8] Allendorf et al. (1997) note that spawner survey data frequently exclude jacks in counts of adult fish. However, jacks may contribute genetically to subsequent generations and thus need to be accounted for. For example, Van Doornik et al. (2002) estimated that the effective proportion of two-year-old males was 35% in two wild coho populations. Some adjustment for the relative reproductive success of jacks versus older adults may be warranted.

Satisfying the low-risk criterion also requires demonstration that N_g remains above critical thresholds during periods of low marine survival due to unfavorable ocean conditions.

As noted above, the total population size criteria are based on an assumption that the N_e/N_g for Pacific salmonids is generally about 0.2. This ratio is based on the recommendation of Allendorf et al. (1997), who cite personal communication with R. Waples (NMFS, Northwest Fisheries Science Center). Subsequent work with Chinook salmon (Waples 2004), steelhead (Heath et al. 2002), and coho salmon (Wainwright et al., in press) has suggested that for many populations, the N_e/N_g ratio likely falls within a range of approximately 0.05 to 0.30, though Ardren and Kapucinski (2003) reported a substantially higher ratio (0.5–0.7) for a steelhead population in Washington. Based on these studies, we conclude that the value of 0.2 suggested by Allendorf et al. (1997) remains a reasonably precautionary default value for relating total population size per generation to effective population size in the absence of other information, but it should be adjusted as information on the N_e/N_g ratios for specific populations becomes available.

In applying the total population size criteria, we note that conditions that may lead to violations in the 0.2 N_e/N_g assumption should be evaluated. Factors that likely contribute to an N_e/N ratio of less than 0.2 include highly skewed sex ratios, sex-biased differences in dispersal, and substantial among-family variation in survival rates (Gall 1987). The ratio of census size and effective population size may also be affected (both increasing and decreasing it) by the spatial structure of a population (Whitlock and Barton 1997), as well as by the degree of isolation of the population and hence the level of exchange of individuals among populations. And finally, total population size may be a poor predictor of long-term mean effective population size in populations that have undergone a recent population bottleneck. Where severe population bottlenecks have occurred, recovery in total population size may occur rapidly, whereas recovery of genetically effective population size may take a much longer time. The rate of recovery from genetic bottlenecks depends on the natural mutation rate and, perhaps more importantly for many salmonid populations, infusion of new variation from immigrants into the population. However, there is little information with which to speculate about how long it may take these processes to replace genetic variation in salmon and steelhead populations. Nevertheless, we advise that when there are clear indications that populations have recently declined below the proposed viability thresholds, additional genetic evidence should be gathered to demonstrate that populations are no longer at appreciable risk. We discuss this issue further in the section title *Critical Considerations for Implementation* on page 51.

Population Decline Criteria

Rationale: The population decline criteria address increased demographic risks associated with rapid or prolonged declines in abundance to small population size. Populations that experience unchecked declines may reach levels at which the probability of extinction from random demographic or environmental events increases substantially (Soulé and Simberloff 1986), and if declines continue unabated, deterministic extinction results. As defined by Allendorf et al. (1997), the criteria have two components: a downward trend in population size (an indication that the population is not replacing itself) and a minimum annual adult run size. Each of these components is evaluated in the context of the other.

Criteria: We adopt criteria consistent with Allendorf et al. (1997), with minor modifications. A population is considered at high risk if it meets any of the following three conditions: (1) the population has undergone a recent decline in abundance (within the last two generations) to an annual run size, N_a, of fewer than 500 fish; (2) the population currently has an average annual run size of $N_a > 500$ but is declining at a rate of $\geq$10% per year over the last two–four generations[9], or (3) the population currently has an annual average run size of $N_a > 500$ but has been declining at a rate that, if it continued, would cause N_a to fall below 500 within two generations. In this high-risk category, the progeny/parent ratio is less than one, indicating that populations are failing to replace themselves. Populations that have declined to annual run sizes at or below 500 spawners but that are currently stable (i.e., progeny/parent ratio is ≥ 1) *or* populations that are above 500 spawners but continue on a downward trajectory (i.e., progeny/parent ratio is < 1) are considered at moderate risk of extinction. By extension, populations at low risk of extinction are those with annual run sizes of greater than 500 and mean progeny/parent ratios of ≥ 1 (Table 1). Although Allendorf et al. (1997) do not specifically discuss their rationale for choosing 500 fish as the threshold between risk categories, we adopt their criteria to foster consistency between the two approaches.

We note that the abundance threshold suggested by Allendorf et al. (1997) as indicative of high risk ($N_a <$ 500 spawners per year) is adopted as appropriate in the absence of information on intrinsic growth rate (i.e., growth rate at low population density, when populations are released from intraspecific competition). Population models that predict extinction probability can be highly sensitive to assumptions about intrinsic growth rate and environmental stochasticity, which causes year-to-year

[9] We note that it might be reasonable to argue that populations at high abundance (e.g., $N_a > 10{,}000$ individuals) might experience declines on the order of 10% or more per year for two generations without appreciably increasing the risk of extinction. However, currently within the NCCC Recovery Domain, there is little evidence to suggest that any salmon or steelhead populations approach such abundances. Should such circumstances arise in the future, it would be appropriate to re-evaluate this element of the population decline criteria, particularly if information on potential sources of variation in population size is available.

variation in population growth rate (see e.g. Lande 1993; Foley 1994; Boughton et al. 2007). A population with $N_a < 500$ might have a relatively low probability of extinction if the intrinsic growth rate were high and variation in growth rate low, but a high probability of extinction if the reverse conditions were true. Consequently, relaxing this criterion would require demonstration that a population of fewer than 500 spawners would not be at heightened risk of extinction[10].

Metrics and Estimation: The population decline criteria require estimation of two parameters: mean annual population abundance, $\overline{N}_a$, and population trend, *T*. We recommend using the geometric mean of spawner abundance for the most recent 3–4 generations as an estimator for $\overline{N}_a$:

(3) $$\overline{N}_{a(geom)} = \left(\prod_{i=1}^{n} N_{a(i)} \right)^{1/n}$$

where $N_{a(i)}$ is the total number of adult spawners in year *i*, and *n* is the total number of years of available data. The geometric mean is slightly more conservative than the arithmetic mean, in that low values have greater influence on the mean. Mean spawner abundance should be based on counts of naturally spawning fish, exclusive of hatchery-origin fish. Our recommendation to use this estimator is consistent with analyses developed for previously published status reviews (e.g., Good et al. 2005).

Population trend, *T*, is estimated as the slope of the number of natural spawners (log-transformed) regressed against time. To accommodate for zero values, 1 is added to the number of natural spawners before log-transforming the value. The regression is calculated as follows:

(4) $$ln(N_a + 1) = \beta_0 + \beta_1 X + \epsilon$$

where N_a is the annual spawner abundance, β_0 is the intercept, β_1 is the slope of the equation, and ϵ is the random error term (Good et al. 2005). Estimation of trend requires a time series of adult abundance for at least two generations and up to four generations[11]. It may be possible to estimate population trends using indices of abundance, so long as the indices truly reflect overall population trends. However, as estimates

[10] Results from Lindley (2003) suggest that a minimum of 30 years of data is likely needed to obtain unbiased estimates of variance in population growth rate within reasonable confidence limits. Such lengthy time series may be needed to accurately estimate variance when there are longer-term trends in abundance and productivity.

[11] The population decline criteria are intended to capture recent, relatively rapid declines in abundance. Over longer periods of time, populations declining at less than 10% per year may still be at high risk of extinction. In the NCCC Recovery Domain, there are few existing time series of population abundance spanning longer than 10 years. In these cases, long-term trends should be evaluated independently of the proposed population decline thresholds.

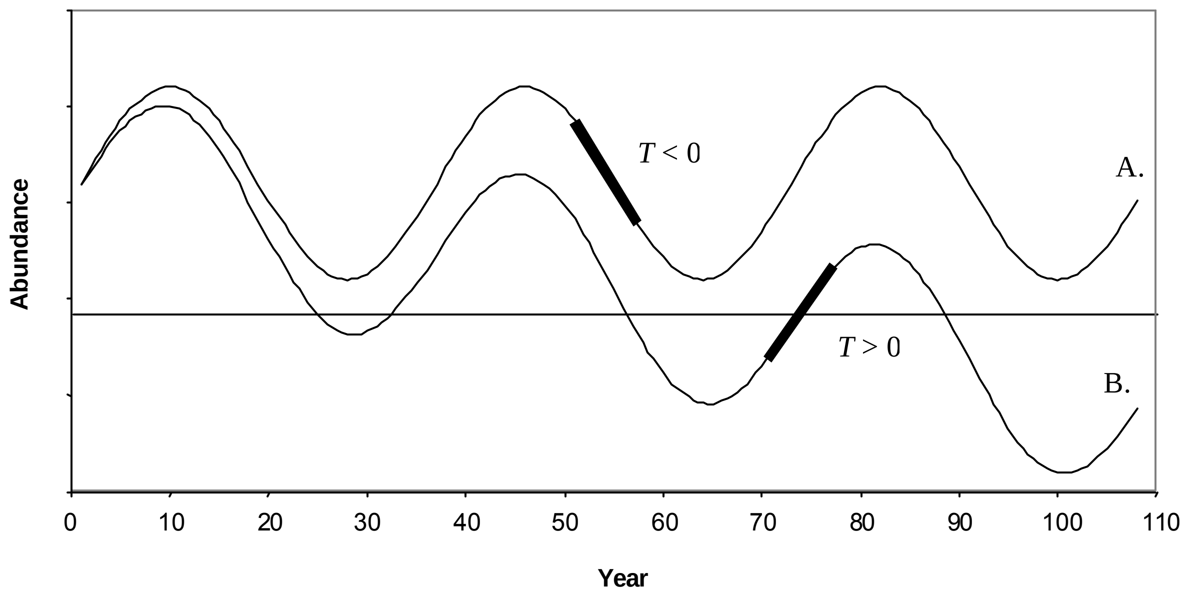

Figure 2. Hypothetical fluctuations in the abundance for a healthy population showing no long-term trend in abundance (A) versus a population undergoing a long-term decline (B). Thick lines depict periods where short-term population growth rates are in opposition to the long-term patterns. Figure based on a conceptual model by Lawson (1993).

of total abundance are needed to evaluate other criteria in Table 1, use of total population estimates will generally be preferable to indices.

Interpretation of population trends is confounded by the fact that salmonid populations may undergo natural fluctuations at time scales ranging from annual to decadal or longer, leading to highly variable estimates of trend. As most estimates of T for populations of salmonids within the NCCC Recovery Domain are likely to be based on relatively short time series of abundance, interpretation of T needs to be made in the context of marine and freshwater survival during the period of record and other population metrics of viability. For instance, healthy populations at little risk of extinction almost certainly experience periods of negative population growth without being at heightened risk of extinction (Figure 2, Line A). Conversely, populations experiencing a long-term downward trend in abundance may exhibit a short-term positive trend response to periods of favorable ocean conditions (Figure 2, line B). These scenarios underscore the need to both understand the causes of population fluctuations and to evaluate population trend and abundance simultaneously, as short-term population trend by itself can be misleading as a metric of viability. Our requirement that low-risk populations be stable or increasing also considers the fact that the criteria proposed herein are being developed for ESUs that have already been listed under ESA. In the vast majority of cases, most populations within these ESUs are considered depressed, often severely so. In this context, it would seem unreasonable to conclude that a population has recovered if it continues to decline in abundance. In future scenarios, demonstration that populations can remain above viability thresholds for other population metrics (e.g., population size, effective population size, and population density) during periods of both favorable and unfavorable conditions and that the population responds positively and rapidly to improvement in marine conditions might justify relaxation of the population trend requirement. In contrast, for populations that otherwise satisfy viability criteria, short-term declines that lack an obvious mechanism (e.g., change in ocean conditions) would be cause for renewed concern.

Catastrophe, Rate and Effect Criteria

Rationale: Catastrophes are large environmental perturbations that produce rapid and dramatic declines in population abundance (Shaffer 1987; Lande 1993). Such events are distinct from environmental stochasticity that arises from the continuous series of small or moderate perturbations that affect population growth rate (e.g., interannual variation in climate, ocean conditions, food resources, populations of competitors, etc.). Some population modelers have suggested that catastrophes may be more important than either environmental or demographic stochasticity in determining average persistence times of populations (Shaffer 1987; Pimm and Gilpin 1989; Soulé and Kohm 1989), though Lande (1993) argues that the relative risks of environmental stochasticity and catastrophes cannot be generalized, being dependent on the mean and variance of population growth rate and the magnitude and frequency of catastrophes. Regardless, there is agreement that populations are at increased risk of extinction following a major reduction in abundance.

Criteria: Within the Allendorf et al. (1997) framework, the goal of the catastrophe criteria is to capture situations where a population has experienced a sudden shift from a no-risk or low-risk status to a higher risk level. Allendorf et al. (1997) defined the very high-risk criterion for catastrophic declines as a 90% decline in population abundance within one generation, and the high-risk criterion as "any lesser but significant reduction in abundance due to a single event or disturbance." These criteria depart to some degree from the IUCN criteria (Mace and Lande 1991), which proposed average population reductions over 2–4 generations of 50%, 20%, and 10% to correspond to critical, endangered, and vulnerable status, respectively. Allendorf et al. (1997) offer limited discussion of the reasoning behind these differences, noting only that Pacific salmonid stocks often exhibit substantial natural variation in abundance. We surmise that Allendorf et al. felt that declines of the magnitude specified in the IUCN criteria may be well within the range of natural variation for salmonid populations and thus adopted more stringent criteria. Further, we note that the rates of decline listed in the IUCN criteria for catastrophic risk are generally subsumed by the Allendorf et al. (1997) population decline criteria, which are adopted in this report.

We adopt the criteria of Allendorf et al. (1997) as they stand, considering populations that have experienced a 90% decline in abundance within one generation to be at "high risk" of extinction and those experiencing a lesser but significant decline to be at "moderate risk" (Table 1). Although Allendorf et al. (1997) do not explicitly define what constitutes a "lesser but significant decline" in abundance, we consider events such as the failure of a year class due to a catastrophic disturbance to be an example of such an event.

Metric and Estimation: We define the estimator of catastrophic decline, *C*, as the maximum proportional change in abundance from one generation to the next. Formally, this can be expressed as follows:

$$\hat{C} = \text{maximum}\left(1 - \frac{N_{g(t)}}{N_{g(t-2h)}}\right) \quad (4)$$

where $N_{g(t)}$ is the running generational sum of adult spawners in year *t*, and $N_{g(t-2h)}$ is the running generational sum at time *t-2h,* where *h* is mean generation time (rounded to the nearest whole year)[12]. By this formulation, estimation of $\hat{C}$ requires a time series of adult spawner abundance of at least 3 generations (but see exception below), and should be based on naturally spawning fish, exclusive of hatchery origin fish. As with the population decline criteria, it may be possible to evaluate catastrophic declines using an index of abundance (rather than a total population estimate), provided that the index faithfully reflects the characteristics of an entire population.

Although it may seem more intuitive to use the running sum in the most recent generation, $N_{(t-h)}$, in the denominator of equation (3), the value of $\hat{C}$ is highly influenced by the pattern of abundance during the transition from a period of high abundance to a period of low abundance since it is based on a running sum of abundance. For example, consider the two time series of abundance depicted in Figure 3. Line A illustrates a situation where population hovering around an average of about 50,000 spawners in years 1 through 13, drops in a single year to an average of about 5,000 spawners from year 14 to 30. Line B illustrates the same scenario, but where the decline occurs over a generation (3 years), rather than in a

[12] For example, for a coho salmon population with a mean generation time of three years, C at *t* = 9 would be 1 minus the sum of adult abundance for years 7, 8, and 9 divided by the sum of abundance for years 1, 2, and 3.

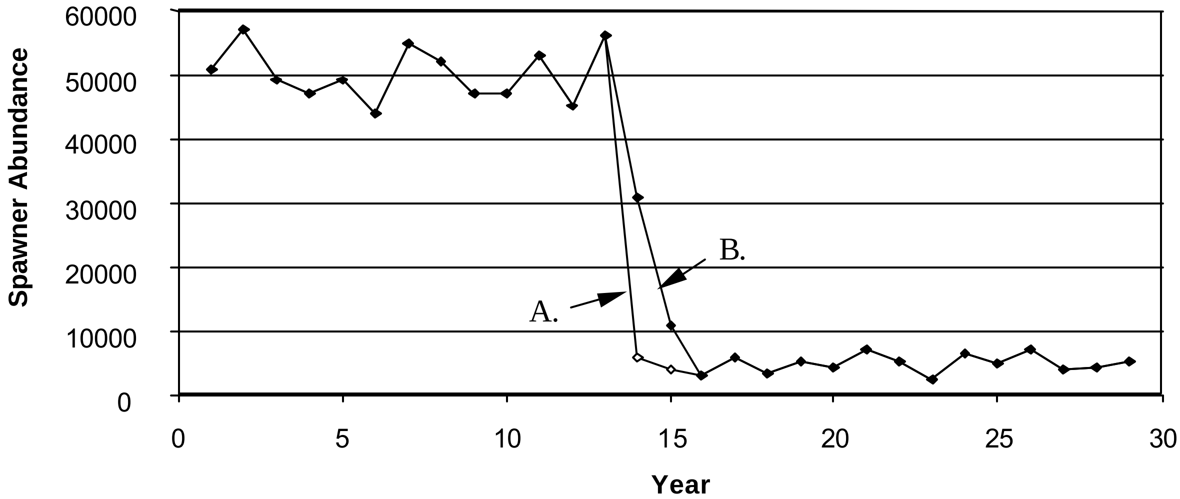

Figure 3. Hypothetical example where an order of magnitude decline in abundance occurs over a single year (A) versus three years (B). See text for elaboration.

single year. Were $N_{(t-h)}$ used in the denominator, value of $\hat{C}$ would exceed the threshold (90%) only for the scenario shown in line A, where the decline occurs over a single year. In scenario B, the intermediate population abundances in years 14 and 15 effectively moderate the value of $\hat{C}$, such that the 90% criterion is never exceeded, despite the order of magnitude drop in abundance that occurred within 3 years. Use of $N_{(t-2h)}$ in the denominator assures that both scenarios are captured by the criteria.

We note that there may be instances where a population either exhibits a clear and precipitous decline in abundance or suffers a major loss or alteration of habitat (e.g., landslide causing a passage blockage, chemical spill affecting an entire year class, or some other catastrophic event). Clearly, in such cases, an immediate elevated risk designation could be warranted, even in the absence of a longer time series of data.

For longer time series where a population experienced a catastrophic decline in abundance at some time during the past, consideration needs to be given to the response of the population following the catastrophic decline. For example, in Figure 4, we depict three distinct trajectories in population abundance following a catastrophe, including an increasing trend in abundance (Line A), a relatively stable abundance (Line B), and a decreasing trend in abundance (Line C). Because the catastrophic decline criteria are intended to capture heightened demographic risks associated with a rapid decline in

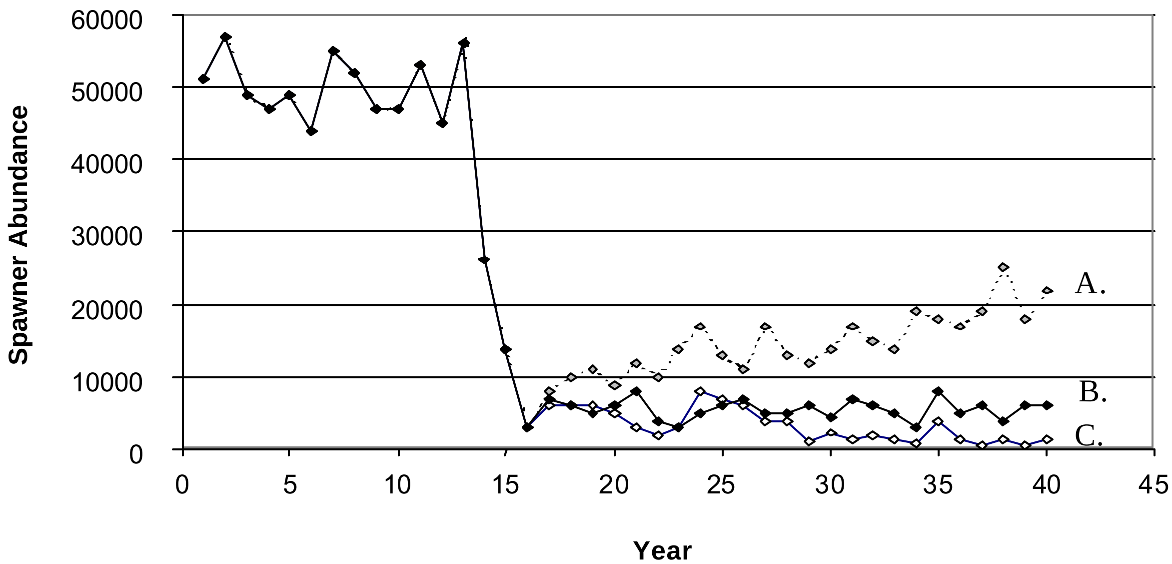

Figure 4. Hypothetical example catastrophic decline in abundance, showing three possible trajectories: A) apparent trend toward recovery from the decline, B) relatively stable abundance following the decline, and C) continued downward trend in abundance.

abundance, scenarios A and B are suggestive that, while the population did experience a rapid declines exceeding the low-risk threshold, the population has since exhibited signs of stabilizing or increasing. In such instances, the castastrophic decline criteria needs to be evaluated in the context of information on patterns of marine survival or more-or-less permanent, naturally caused changes in system capacity (for example, blockage of habitat due to a natural landslide or other disturbance where the blockage is expected to persist for hundred or thousands of years).

Allendorf et al. (1997) provide no details about what might be considered a "lesser but significant decline in abundance." We conclude that the most likely occurrence that would qualify as a moderate risk of extinction would be the loss or severe reduction in an individual year class due to a catastrophic disturbance (e.g., fire, landslide, severe flood or drought, chemical spill, or some other similar catastrophe). Because the risk associated with such an event is likely to vary substantially depending on specific circumstances such as the size of the population in other year classes and the degree of life-history variation (which influences how rapidly a population might recover from such a loss), we do not propose numeric thresholds for moderate risk and instead suggest that such risk will need to be evaluated on a case-by-case basis.

Spawner Density Criteria

Rationale: The spawner density element of the viability criteria is intended primarily to fill a perceived gap in the Allendorf et al. (1997) framework with respect to population attributes identified as important to persistence in the VSP framework: spatial structure and diversity. These characteristics of populations influence viability by spreading risk through time and space and by contributing to the resiliency of populations to natural and human-caused disturbances. Historically, populations making up an ESU undoubtedly differed in average abundance as a function of differences in both the total habitat available for spawning and rearing and the relative capacities of those habitats. Additionally, the distribution of individuals across large and potentially diverse watersheds likely further enhanced the probability of populations persisting over the long term. For example, populations where spawning occurs in multiple, relatively discrete areas are less vulnerable to localized (reach or subwatershed) disturbances such as fires or landslides and have greater potential to recovery from such disturbances, since unaffected portions of the population can both sustain the population following the disturbance and provide colonizers to repopulate the affected habitats. Further, populations distributed over a large watershed have the potential to experience a broader range of environmental conditions, leading to greater phenotypic and genotypic diversity. Life-history variation (e.g., variation in the age and size of individuals at smoltification and maturity) potentially buffers populations from natural fluctuations in both freshwater and marine conditions, spreading risk through both space and time (den Boer 1968; Hankin and Healey 1986; Hankin et al. 1993; Mobrand et al. 1997; Hill et al. 2003). Greater genetic diversity increases the ability of a population to adapt to changes in environmental conditions over the long term. As a population departs from its historical pattern of distribution and abundance, through loss or degradation of habitat, the probability of the population persisting decreases as well, though numerous factors will determine how far a population can depart from historical conditions and still remain viable.

At the opposite end of the spectrum, populations that have been reduced due to severe and widespread degradation of habitat may be subject to directional demographic processes that result in heightened extinction risk. Specifically, at very low densities, populations may experience a reduction in per capita growth rate with declining abundance, a phenomenon referred to as *depensation*. Most population growth models typically assume that per-capita growth rate increases as population density decreases, a result of reduced intraspecific competition. However, if populations are reduced to extremely low densities, a variety of mechanisms can lead to reduced per-capita growth rate, including reduced probability of fertilization (e.g., failure of spawners to find mates), inability to saturate predator populations, impaired group dynamics, or loss of environmental conditioning (Allee 1931; Liermann and Hilborn 2001;

Montgomery et al. 1996). Depensation can result in a postitive feedback that, if unchecked, accelerates a decline toward extinction.

High densities of spawning salmonids serve the additional role of providing marine-derived nutrients from salmon carcasses, which help maintain the productivity of aquatic ecosystems. A growing body of literature has documented the substantial contribution that salmon carcasses play in the nutrient budgets of streams in the Pacific Northwest (Bilby et al. 1996, 1998, and 2001; Cederholm et al. 1999; Gresh et al. 2000; Gende et al. 2002; Naiman et al. 2002; Schindler et al. 2003). Carcasses constitute important sources of nitrogen and phosphorous, which fuel primary production in stream ecosystems, and provide a direct source of food to juvenile salmon (Bilby et al. 1998). Reductions in abundance and spatial distribution of salmonid populations may thus fundamentally reduce the capacity of the streams to support salmonids, creating a feedback loop that could negatively affect long-term population persistence or slow recovery. For example, Scheuerell et al. (2005) suggest that the reductions in the abundance of spring/summer Chinook salmon in the Snake River basin may have resulted in a shift to a less productive state, as evidenced by compensatory mortality in Chinook juveniles even though populations were far below their historical abundance (Achord et al. 2003), as well as failure of smolt recruits per spawner to rebound in years of higher adult abundance. Recognition of this important role has led to a growing call for the link between salmon-derived nutrients and system productivity to be considered when setting salmon recovery goals (Gende et al. 2002; Peery et al. 2003; Scheuerell et al. 2005). And though additional research will be needed before escapement goals for ensuring maintenance of ecosystem (and salmon) productivity based on nutrient subsidies can be established (Bilby et al. 1998; Gende et al. 2002), requiring minimum spawner densities increases the likelihood that such benefits will be maintained or at least not further eroded.

As fixed values, other metrics in the viability table (the effective population size criteria and population size element of the population decline criteria) do not account for these historical among-population differences in total habitat available for spawning and rearing, the relative productive capacity of those habitats, the potential role of spatial structure and diversity in population persistence, the role of nutrient subsidies in maintaining ecosystem productivity, or the possibility of depensation if individuals are sparsely distributed across the landscape. It seems particularly problematic, for example, to conclude that a population is viable at an N_e of about 500 (or N_g of 2,500) when historically that population was much, much larger. An effective population size of 500 fish per generation in a small watershed might seem reasonable, but a population with the same number of fish spread at low densities throughout a much larger watershed could be at moderate or high risk of extinction. Even if the 500 fish per generation were consistently concentrated in a core habitat within a watershed, reducing the risk of depensation, the risk of extinction from a single catastrophe (e.g., flood, landslide, fire) would be higher. Equally important, in either scenario the smaller population's functional contribution to ESU viability would be substantially diminished, even if the population remained viable.

We propose using criteria related to spawner density to address these issues of spatial structure and depensation risk. In developing these criteria, we operate from the following set of assumptions:

- **For independent populations, the historical distribution and abundance of adult spawners represents reference conditions for which extinction risk was likely low and the population made its greatest contribution to ESU viability.** Under these conditions, populations likely tended toward their carrying capacity, and the resilience imparted by spatial structure, diversity, and ecosystem productivity (i.e., contribution of marine-derived nutrients) made it unlikely that the population would go extinct in the absence of a large-scale catastrophe.

- **The farther a population departs from its historical condition, the greater its extinction risk and the higher the uncertainty associated with its viability[13].** Although some departure from historical conditions due to diminished habitat quality or reduced spatial distribution (with incumbent effects on diversity) may have minimal influence on population persistence, the more restricted and/or fragmented the distribution of the population becomes, the higher its extinction risk.

- **How far a population can deviate from its historical condition and remain viable depends, in part, on how large the population was and how it was distributed historically.** Thresholds defined for the minimum amount of intrinsic habitat potential (*IPkm*[14]) required for viability in isolation are based on an assumption that, under historical conditions, populations were at or near a carrying capacity. For historically small populations (i.e., those near the IP threshold for independence), reductions in abundance or distribution would likely move these populations below levels required for viability. For populations in larger watersheds, a comparable percentage reduction in habitat is less likely to result in a substantial increase in extinction risk.

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[13] Theoretically, human modifications that increased the amount of available habitat, such as construction of fish passage structures around natural barriers, could constitute an exception to this generalization.

[14] *IPkm* is an estimate of the accessible stream kilometers, weighted by their intrinsic potential, as estimated by the model of Burnett et al. (2003) and modified by Agrawal et al. (2005). See Bjorkstedt et al. (2005) for details.

- **At extremely low densities, populations may be at heightened risk of extinction due to depensation.** Although demographic and environmental variability can make it very difficult to detect depensation in fish populations, the consequences of depensation are sufficiently severe to warrant consideration of depensatory processes when populations are at very low densities.

The first three assumptions relate directly to the establishment of low-risk thresholds, where the key question is "how far can a population depart from historical conditions and still remain viable?" This is a difficult question to answer, given that the quantitative basis for relating spatial structure, diversity, and ecosystem productivity is presently limited. The last assumption deals directly with establishment of a high-risk threshold, where the key question is "at what densities is depensation likely to occur in salmonid populations?" This too is a challenging question, as detecting depensatory processes in natural populations has proven difficult, though not impossible. Despite these acknowledged uncertainties, the NCCC TRT believes that reasonable criteria can be developed from these general principles.

Criteria: The spawner density criteria define two thresholds. The first, which distinguishes between populations at high versus moderate risk, is based on potential depensation effects. The second defines the threshold between moderate and low risk based on spatial structure, diversity, and productivity concerns. Populations potentially at high risk of depensation are defined as those with average spawner densities of fewer than 1 adult spawner per *IPkm*. For the low-risk threshold, we propose density criteria that vary as a function of both species and population-specific estimates of potential habitat capacity (Figure 5).

For the smallest watersheds capable of supporting viable populations (as estimated based on *IPkm*), low-risk populations are defined as those exceeding 40 spawners per *IPkm*, a value assumed to approximate a natural carrying capacity for salmonids systems (see discussion below). For larger watersheds, required densities decrease to a minimum of 20 spawners/*IPkm* (Figure 5) based on the assumption that larger populations can depart farther from historical conditions before extinction risk is substantially increased.

Defining the density at which depensation is likely to occur is difficult due to high variability and few observations at low abundances in most spawner-recruit datasets (Liermann and Hilborn 1997, 2001). Nevertheless, several authors have attempted to define thresholds at which depensation appears to occur in salmonids. Based on spawner-recruit data for coho populations, Barrowman (2000; cited in Chilcote et al. 2005 and Wainwright et al., in press), suggested that depensation may become a factor at spawner

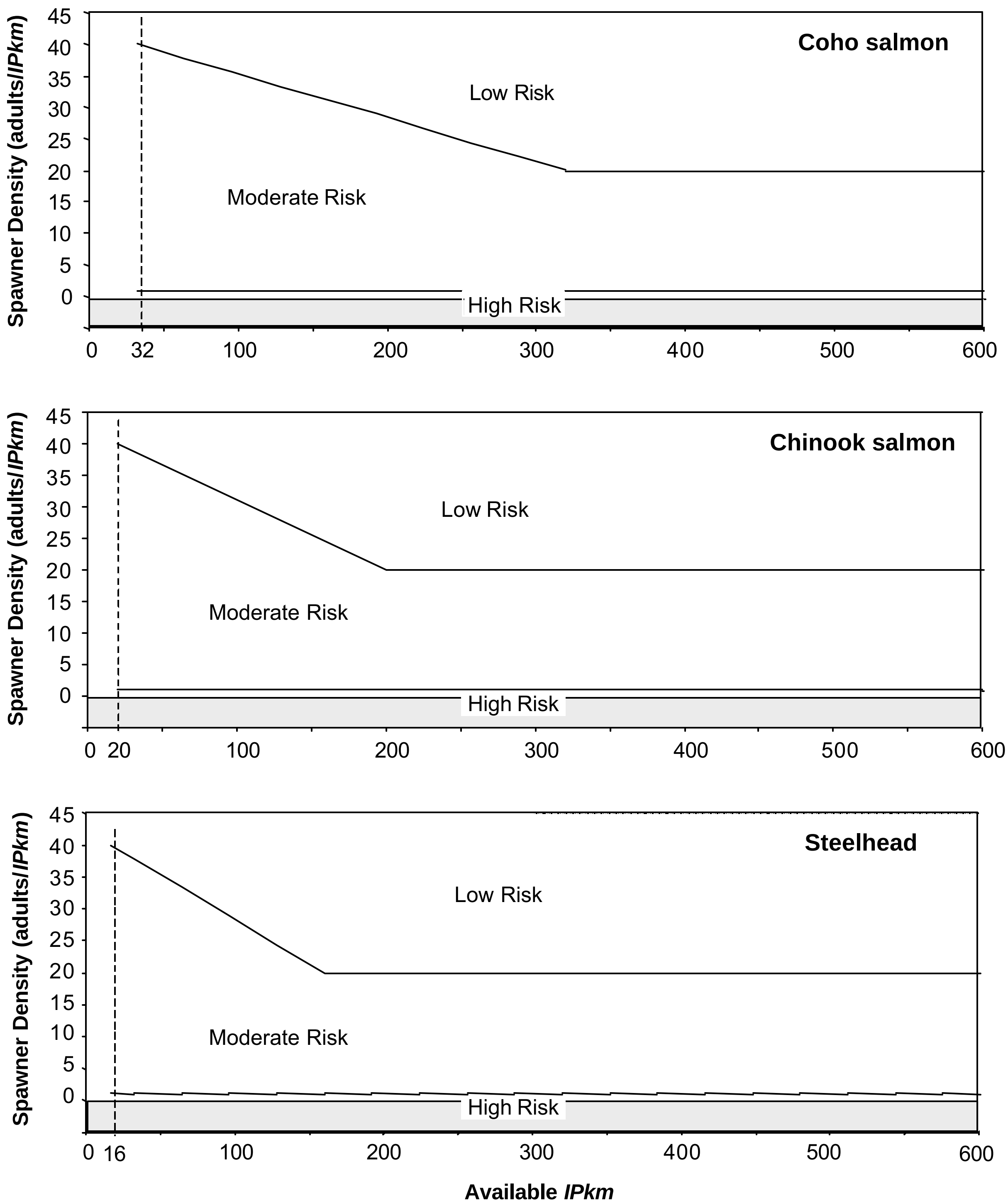

Figure 5. Relationship between risk and spawner density as a function of total intrinsic habitat potential for coho salmon, Chinook salmon, and steelhead. Values above upper lines indicate populations at low risk; values below this line are at moderate risk. Values below 1 spawner/*IPkm* are at high risk for all species. Dashed vertical lines indicate minimum *IPkm* for independent populations.

densities of 1 female per km. Likewise, Barrowman et al. (2003) found little evidence of depensation in coho salmon unless densities were less than 1 female/km. Assuming a 50:50 sex ratio, these values equate to 2 adults per km. Based on analysis of coho populations that went extinct in the lower Columbia River during the 1990s, Chilcote (1999) suggested that populations were unlikely to recover if their densities fell below about 2.4 adults/km. Similarly, Sharr et al. (2000) suggested that coho populations at densities of fewer than 2.4 adults per km should be considered “critical” based on potential risks of depensation. Based on these data, the OCTRT (Wainwright et al., in press) concluded that depensation risks were very likely at spawner densities of 0.61 spawners per km (1 spawner per mile). For our purposes, we chose to use $IPkm$ in the denominator in order to account for potential differences in habitat quality among watersheds[15]. Since the ratio of $IPkm$ to total km is about 0.6 for coho salmon, the OCTRT rule of 0.6 fish per km equates to approximately 1 fish per $IPkm$, the criterion we propose. In adopting this criterion, we recognize that the empirical evidence supporting depensation in salmonid populations remains somewhat limited. However, we heed the recommendation of Liermann and Hilborn (2001) who noted that the paucity of evidence “should not be interpreted as evidence that depensatory dynamics are rare or unimportant.” In practical application of our population viability criteria, the depensation criterion is likely to play a significant role in population risk classification only for the largest populations within the domain, as other criteria (e.g., effective population size, and population decline criteria) are likely to be more conservative in watersheds where potential habitat is estimated to be less than 500 $IPkm$.

The low-risk density criteria were defined based on the following rationale. First, recall that for each species, Bjorkstedt et al. (2005) defined a minimum threshold of potential habitat (expressed as $IPkm$) that was required for the population to be considered viable-in-isolation (32 $IPkm$ for coho salmon, 20 $IPkm$ for Chinook salmon, and 16 $IPkm$ for steelhead), with the among-species differences in $IPkm$ thresholds reflecting differences in life-history variation. These thresholds assume that populations historically operated at something close to the natural carrying capacity of the system. By extension, for populations in the smallest watersheds (in $IPkm$ terms) capable of supporting a viable population to remain viable, they must function at something close to this historical carrying capacity, as any reduction in abundance would drop them below thresholds for viability. Consequently, the average spawner density at natural carrying capacity serves as a reasonable basis for establishing the threshold for low-risk in the smallest watersheds.

[15] The decision to use *IPkm* was based on an assumption that *IPkm* provides a reasonable measure of the relative productive potential of a watershed. For watersheds that have comparable *IPkm* but somewhat different total km, the average density, expressed as fish/km might be expected to be lower in the less productive watershed, potentially leading to greater depensation risk. However, we assume that in most cases, fish distribute themselves somewhat according to habitat quality; thus, we consider these two scenarios as having comparable risk.

The difficulty lies in estimating this value. For coho salmon, we relied on the work of Bradford et al. (2000), who examined stock-recruit relationships for 14 historical data sets of coho salmon in the Pacific Northwest. Fitting a hockey stick model to these data, they found that, on average, the plateau in the stock-recruit relationship, which identifies number of spawners at which full smolt recruitment occurs (an estimate of carrying capacity), occurred on average at 19 females per kilometer. Assuming a sex ratio that is slightly biased in favor of males, we round this number to approximately 40 adult spawners per kilometer. For Chinook salmon and steelhead, we lack the same kind of empirical basis for setting the spawner density for watersheds with the minimum IP required for viability, and so we default to the 40 spawners/km value recommended for coho salmon.

For coho salmon, we find some support for our recommended spawner density in population viability models developed for coho salmon on the Oregon Coast. Recall that the NCCC TRT estimated that at least 32 *IPkm* was required for a population of coho salmon to be considered viable-in-isolation (Bjorkstedt et al. 2005). This threshold value was based on the simulation analyses of Nickelson and Lawson (1998), who used a life-cycle model to predict extinction risk for a population of coho salmon as a function of the amount of "high quality" habitat available (Bjorkstedt et al. 2005). The Nickelson-Lawson model produces quantitative extinction probabilities. These probabilities are sensitive to many of the model parameters; thus, determining an absolute extinction probability for any population is difficult. Nevertheless, the model consistently shows that extinction probabilities begin to rise rapidly when the available high-quality habitat falls below 24 kilometers. The NCCC TRT set the viability-in-isolation threshold based on an assumption that watersheds with at least 32 *IPkm* would have sufficient high-quality habitat to support a viable population (Bjorkstedt et al. 2005). These estimates assume that this quantity of habitat would be expected to produce sufficient numbers of smolts to yield 1,500 spawners during a period of 1% marine survival (Wainwright et al., in press). For the smallest population (i.e., in a watershed with 32 *IPkm*), 1,500 spawners would result in a density of about 47 spawners per *IPkm*, a value in reasonable agreement with the 40 spawners/*IPkm* chosen for our criteria.

For Chinook salmon the default value of 40 spawners/km value is consistent with the rationale of Bjorkstedt et al. (2005). Based on reported values for average Chinook salmon redd densities, they argued that a redd density of 20 per km (and thus a spawner density of 40 fish/km assuming a 50:50 sex ratio) over 20 *IPkm* would be required for a population to be viable. We also note that although the density required for viability in the smallest watersheds is the same for coho salmon, Chinook salmon, and steelhead, the absolute abundance requirements would differ, since the *IPkm* threshold for viability differs (i.e., the smallest watershed for viable coho salmon, Chinook salmon, and steelhead populations would require annual run sizes of 1,280, 800, and 640 spawners, respectively). This result is consistent with the hypothesis that the greater life-history diversity exhibited by steelhead and Chinook salmon enables them to persist at somewhat lower absolute abundances than coho salmon, which have a more rigid life history.

With the spawner density criteria of 40 fish/*IPkm* for the smallest populations serving as an anchoring point, the next step was to generate a function representing our general conclusion that the larger the population historically was, the more it can depart from historical conditions and still remain viable. Here, we assume that a population with ten-fold more habitat potential than the smallest population requires an average spawner density half that of the smallest population and that the required density declines linearly between these two reference point (Figure 5). For watersheds with greater than ten-fold the habitat potential of the minimum watershed, we assume that spawner density must be at least 20 fish/*IPkm* for the population to be at low risk.

We acknowledge that selection of the latter reference point is based largely on expert opinion and that there is room for debate about both the shape of the density function and the floor density that is used for large watersheds. However, we believe that application of the density criteria yields results that are qualitatively consistent with general hypotheses relating watershed size and density to spatial structure, diversity, and other factors that influence population persistence. First, a result of application of the density criteria is that it establishes a watershed-specific abundance target that is scaled to the amount of potential habitat. This overcomes the unsatisfying outcome of “fixed” abundance criteria, where a remnant of a historically very large population might still be considered “viable” in the sense of having a low extinction risk over some time frame, even though the population clearly plays a much-diminished role in ESU viability. A second desirable outcome is that the density criteria substantially increase the likelihood that elements of spatial structure and diversity that contribute to viability will be maintained, without rigidly asserting what that spatial structure must look like. For example, in a large watershed, the density criteria could be attained in a variety of ways, ranging from having roughly half the available habitat occupied at something near carrying capacity, with little use of remaining habitats, to having fish distributed at moderate densities throughout the watershed. Each of these scenarios offers some potential advantages and disadvantages from a population persistence standpoint. For example, populations anchored in a subset of watersheds that are functioning at or near carrying capacity may provide for greater resilience during periods of low ocean productivity (Nickelson and Lawson 1998) but be at somewhat more risk of localized disturbances than populations distributed more broadly but at lower average densities. Because these tradeoffs do not seem to be quantifiable given our current state of knowledge, the density criteria seem preferable to more stringent requirements related to spatial structure.

Metrics and Estimation: For the high risk of depensation threshold, we propose estimating average spawner density (expressed as spawners/*IPkm*) in the *h* consecutive years of lowest abundance within the last four generations, where *h* is mean generation time for the species. Mathematically, we express this as follows:

(5) $$\hat{D}_{dep} = \left(\min \left[\frac{N_{g(t)}}{h} \right] \right) \Big/ IPkm$$

where $N_{g(t)}$ is running generational sum of spawner abundance at time *t*, and *IPkm* is the estimate of potential habitat capacity for the watershed in which the population resides (see Chapter 4 for *IPkm* estimates for each independent population). The decision to evaluate average spawner density in the *h* consecutive years of lowest abundance (as opposed a single year or over all years) balances several considerations. Foremost, we seek an indicator that is sensitive to the possibility that a population is at risk of depensatory mortality, without being overly sensitive to natural fluctuations in abundance. For example, a population that experiences a single year of low abundance may be at minimal risk of slipping into an accelerating pattern of depensation, especially for species with overlapping generations, which may be able to rebound more rapidly after a poor year. On the other hand, a metric that uses average abundance over a longer period could be insensitive to depensation risks if a few relatively good years elevate the average to levels above the depensation threshold and thereby mask these risks. Selecting the lowest *h* consecutive years looks for recurring evidence of population numbers sufficiently low that there is heightened potenential for depensatory dynamics that could rapidly deteriorate into a feedback situation. We note also that the proposed metric assumes that fish are distributed relatively uniformly across the available spawning habitats. Were spawner densities consistently higher in certain locations within a watershed, it would suggest that risks associated with depensation due to the difficulty of spawners finding mates might be low and that the criterion could therefore be relaxed, though other possible depensation mechanism (e.g., lack of predator saturation) must also be considered.

For the low-risk density threshold, we propose as a metric the arithmetic mean of adult spawner density, expressed as adult spawners per *IPkm*, for all years over the last four generations:

$$(6) \qquad \hat{D}_{ssd} = \frac{1}{4h} \sum_{t=1}^{4h} \frac{N_a}{IPkm}$$

where N_a and *IPkm* are as defined above, and h is the mean generation time for the population (rounded to the nearest whole year). The estimated density is then evaluated against thresholds that are a function of both species and populations-specific estimates of potential habitat capacity or *IPkm*, as outlined in Figure 5.

Density estimates are likely to be derived in two different ways. First, where weirs or other fish passage structures exist, average density can be estimated by dividing either total fish count (if all upstream migrating fish are captured) or a total population estimate (if only a portion of adults are captured, but where the proportion can be accurately estimated)—both of which estimate annual run size, N_a—by the number of stream *IPkm* accessible in the watershed. Second, where randomized spawner surveys allow for population estimation, again the total population estimate, N_a, can be divided by total accessible *IPkm* in the basin to yield an average density over the entire watershed.

Of the criteria proposed in this document, the density criteria perhaps generated the most discussion among TRT members about both the selection of the specific criteria and the most appropriate way to apply them. Among the specific issues debated were (1) the relationship between density and viability in populations where a significant amount of historical habitat is now inaccessible behind dams or severely degraded (which becomes a question of selecting an appropriate habitat-based denominator when estimating density); (2) whether the proposed criteria were sufficiently precautionary or overly so; (3) whether it was more appropriate to express density criteria in terms of fish per *IPkm* or fish per total accessible kilometers; and (4) whether adjustments to the criteria should be made to account for potential bias in estimates of IP. We discuss the first of these issues in the paragraphs that follows, since resolution of this issue is integral to subsequent discussion of ESU-level viability criteria that comes in Chapter 3. The remaining topics we treat in Appendix B.

An important issue in estimating density is how to handle situations where substantial historical habitat now lies behind impassible dams or other human-caused barriers to fish migration. This raises the question as to whether, in estimating density using the two methods above, it is more appropriate to use historical versus currently available *IPkm* in the denominator. In some instances, where significant historical habitat has been lost, use of historical *IPkm* would, in all likelihood, preclude such populations from ever attaining viable status in relation to historical standards. This seems problematic, in that there may be sufficient habitat downstream of impassible barriers (i.e., more than the minimum threshold for the population to be considered viable in isolation) to support a viable population. (Put another way, it seems illogical to conclude that a population below human-created barriers that still has access to substantial habitat cannot be viable, if a population in a watershed with comparable habitat but no such barriers *can* be considered viable.) On the other hand, excluding areas upstream of barriers from consideration violates one of our fundamental assumptions: that the spatial structure and diversity resulting from the distribution of individuals broadly and over diverse habitats contributes significantly to population persistence. We therefore recommend that populations be evaluated based on both historical (pre-barrier) and current (post-barrier) conditions. Populations that fail to satisfy density criteria based on historical habitat availability but that do satisfy the density criteria as applied to current conditions could potentially be considered viable in the sense of having a relatively high probability of persistence. But these “partial populations” represent something other than the historically defined population. Such populations could be at greater risk than if criteria for the historical habitat were met (due to loss of diversity or spatial structure), and their contribution to ESU persistence might be substantially diminished, requiring reassessment of their role in ESU viability.

A related issue is how to deal with situations where fish still have access to portions of a watershed, but where habitat alterations are both severe and permanent (e.g., intensive urbanization), effectively precluding use by salmonids. In principle, arguments similar to those discussed above could be used to make the case that density should only be estimated in those habitats that still are capable of supporting salmonids. However, whereas in the case of dams, habitat losses are relatively easy to quantify, habitat degradation is a matter of degree, and thus defining boundaries around areas that are no longer suitable becomes problematic. We conclude that, assuming such areas could be clearly defined[16], one could evaluate density criteria using only “accessible and suitable” habitats; however, again such “partial populations” represent something other than the historical population, having substantially departed from their historical spatial structure and diversity. In no case should a population be considered viable, by any standard, when the remaining habitat that is deemed suitable does not meet the minimum viability thresholds set for each species (i.e., 32 *IPkm* for coho salmon, 20 *IPkm* for Chinook salmon, and 16 *IPkm* for steelhead). How “partial populations” may relate to viability at the levels of diversity strata and ESUs is discussed further in Chapter 3.

[16] Defining such areas may be complicated if fish from relatively good habitats periodically “leak” into poor habitats.

Hatchery Criteria

Rationale: The hatchery criteria are intended to address potential impacts of hatchery operations on the viability of wild populations of salmon and steelhead. Hatchery operations can affect wild populations through a variety of ecological, demographic, and genetic mechanisms, thereby influencing their probability of persistence.

The potential ecological effects of hatchery operations and hatchery fish on wild fish are many and varied. When released into the wild, hatchery fish may compete for food, space, or mates with wild fish in both the freshwater (Nickelson et al. 1986) and marine (Levin et al. 2001; Ruggerone et al. 2003; Ruggerone and Nielsen 2004) environments. Hatchery fish can alter predator-prey dynamics by preying directly on wild salmonids (Sholes and Hallock 1979) or by attracting or supporting increased numbers of avian, mammalian, or piscine predators, resulting in increased predation rates on wild fish (Collis et al. 2001; Ryan et al. 2003; Major et al. 2005). Conditions within hatcheries can increase the vulnerability of fish to infection by pathogens, cause pathogen amplification, and increase opportunities for disease transmission (Moffitt et al. 2004). These diseases can then be transferred to wild populations (Kurath et al. 2004). Marine or estuarine netpen rearing of such hatchery fish can also result in transfer of pathogens and parasites to nearby wild fish (Naylor et al. 2005; Krkosek et al. 2006). Stocking of large numbers of hatchery smolts in streams containing wild fish can also alter the behavior of wild fish, resulting in premature emigration of wild fish (Hillman and Mullan 1989). Additionally, hatchery facilities themselves may pose risks to wild populations by diverting water from natural streams in order to supply hatcheries, releasing polluted effluent (e.g., fish wastes, antibiotics) waters from hatcheries back into streams and rivers, and creating barriers to migration through installation of weirs or other fish collection structures (White et al. 1995; Pearsons and Hopley 1999; Reisenbichler 2004).

Hatchery programs also potentially pose direct demographic risks to wild populations. Production of large numbers of hatchery fish can result in increased human harvest of wild fish in mixed-stock fisheries, resulting in reduced spawning escapement (McIntyre and Reisenbichler 1986; Hilborn 1992; NRC 1996; Reisenbichler 2004). Additionally, hatchery programs that draw broodstock from wild populations, so-called broodstock mining, also pose direct demographic risks to the wild population if the survival and subsequent reproductive success of hatchery-origin fish that spawn in the wild does not at least replace production lost due to the removal of natural-origin fish for broodstock (ISAB 2003). Broodstock mining may also compromise the ability of a wild population to maintain its genetic character if too few adults are allowed to spawn naturally, increasing the risk for adverse effects associated with small population size (effects that may be exacerbated if broodstock suffer a catastrophic loss in the hatchery). In very small populations, removal of wild fish for hatchery broodstock may result in depensation, through Allee effects and other mechanisms, in the remaining wild population if too few individuals are left to spawn.

Genetic risks of hatcheries arise when wild fish interbreed with genetically dissimilar hatchery fish, which can result in changes in genetic composition of wild populations, as well as genetic structure across larger spatial scales. Under natural conditions, accurate homing to natal streams tends to result in the formation of distinct breeding groups or populations that, over time, become locally adapted to the environmental conditions they experience during their life cycle. This local adaptation and the diversity it creates over larger spatial scales are important for the long-term persistence of populations and ESUs (NRC 1996; Hendry 2001; McElhany et al. 2000; Reisenbichler et al. 2003). Within populations, interbreeding of wild fish with hatchery-origin fish can alter the genetic characteristics of the wild population, reducing the (average) individual fitness and hence overall population productivity (ISAB 2003). When hatchery fish stray into other watersheds and interbreed with wild fish, patterns of genetic variation can likewise be altered.

Genetic differences between hatchery and wild populations can arise in several non-mutually exclusive ways. First, they may result when nonnative (i.e., out-of-basin or out-of-ESU) broodstock are used in the hatchery. Second, genetic differences can arise when hatchery broodstock are subject to various artificial selection processes, sometimes referred to as domestication selection, that result either through hatchery practices or from exposure to unnatural hatchery environments. Artificial selection processes may be intentional, such as when hatchery managers select for certain desirable traits (e.g., size of broodstock or progeny, timing of return, etc.) or inadvertent, such as when selected broodstock randomly differ in some trait from wild populations or when the hatchery environment favors (and therefore selects for) traits that improve survival in the hatchery but that may lead to reduced fitness in the wild. And third, genetic modification may occur through hybridization of distinct subspecies, races, runs or phenotypes that co-occur in the same stream or basin. For example, hybridization of spring- and fall-run Chinook in the Feather and Trinity rivers appears to have occurred in response to broodstock collection during periods of overlap in run timing (Blankenship et al., in prep; Kinziger et al., in review). Regardless of the specific mechanism, the result is hatchery populations that differ in their genetic composition from wild populations.

Another genetic risk of hatcheries is the "Ryman-Laikre effect", whereby the admixture of hatchery fish into a natural population causes a reduction in the effective population size of the combined population (Ryman and Laikre 1991). This occurs because a group of hatchery fish generally have a smaller number of parents than a similar-sized group of natural fish, due to higher juvenile survival within the hatchery. When these hatchery fish reach reproductive age and interbreed with wild fish, the average number of genetic lineages in their offspring will be lower than if they were all wild fish. The magnitude of the reduction in effective size is proportional to the percentage of spawners that are hatchery fish and the difference in the average number of parents for the hatchery and wild fish.

Of particular concern within hatchery broodstock is inbreeding depression, which is when interbreeding between closely related individuals causes a decrease in average fitness of offspring, usually resulting from increased frequency of homozygotes for deleterious recessive alleles, fixation of deleterious alleles within a population, or loss of overdominance. Outbreeding depression is a reduction in fitness of hybrid progeny when genetically dissimilar fish interbreed. It can result when wild fish interbreed with nonnative (e.g., out-of-basin or out-of-ESU) fish or when wild fish interbreed with hatchery fish that have undergone domestication selection. Processes that contribute to outbreeding depression include the introduction of alleles from the hatchery stock that are maladaptive in the local environment or the breakdown in co-adapted gene complexes (Fleming and Petersson 2001; ISAB 2003). Evolutionary models suggest that genetic exchange between hatchery fish and wild fish has the potential to erode the fitness of wild populations, with effects depending on the strength of selection and the magnitude of the hatchery contribution to total production (Ford 2002; Goodman 2004, 2005). Such changes may occur even if a large proportion of the hatchery broodstock consists of natural-origin fish (Ford 2002). Collectively, these processes can result in a variety of population-level and ESU-level changes in genetic diversity, including decreased within-population diversity resulting from insufficient numbers of broodstock and inappropriate mating protocols; loss or dilution of distinct, locally adapted populations; and increased homogenization of populations within an ESU (through increased straying). Such changes may affect the long-term persistence of both populations and the ESUs comprising those populations.

Although the ecological, demographic, and genetic effects of hatcheries on wild populations are well documented (see NRC 1996 for a review), quantitatively relating these effects to the probability of extinction of populations is difficult. Many of the ecological impacts of hatcheries are highly context-dependent. For example, competitive interactions between hatchery and wild fish are likely to vary with the carrying capacities of different ecosystems, the size of the wild population at the time of introduction, the number of hatchery fish released, the average size of stocked fish relative to wild fish, whether fish are planted in a few locations or distributed broadly across a watershed, or any number of other confounding factors. Likewise, genetic impacts on wild populations will depend on many factors including the origin of broodstock, how the hatchery is operated (e.g., mating protocols, rearing practices), and the number and effectiveness of hatchery fish that spawn in the wild, among other things. Further complicating matters in the NCCC Recovery Domain is the fact that hatchery programs at many facilities have changed substantially in the past decade or so, from predominately large-scale production-oriented programs to smaller-scale supplementation or captive broodstock programs. For example, out-of-basin coho salmon were planted for a number of years in the Russian River basin; however, the program was terminated in the mid 1990s, and there is now a captive broodstock program in operation intended to conserve what appears to be a remnant native population. Consequently, assessing potential hatchery risks involves evaluating not only current practices, but potential lingering genetic effects resulting from historical operations as well.

Criteria: Because of the numerous and complex ways in which artificial propagation activities may affect wild populations of salmonids, and because of the unique histories of ongoing and recently terminated hatchery programs within the recovery domain, the NCCC TRT concluded that simple numeric criteria for assessing hatchery risk would be difficult to justify. Acknowledging both the potentially significant risks that hatcheries pose to wild populations and the uncertainty in quantitatively relating these risks to extinction risk, the NCCC TRT adopts the following narrative criteria for hatcheries: populations are considered at low risk if there is demonstrably no or negligible evidence for ecological, demographic, or genetic effects resulting from current or past hatchery operations; populations are at elevated risk (moderate-high) if there is evidence of significant ecological, demographic, or genetic effects or high uncertainty surrounding these potential effects (Table 1).

The NCCC TRT notes that other Technical Recovery Teams have developed quantitative criteria specifically addressing *genetic* risks of hatcheries. For example, the OCTRT (Wainwright et al., in press) and Southern Oregon-Northern California Coast TRT (Williams et al., in prep.) propose assessing genetic risk based on the fraction of natural spawners that are of hatchery origin. The Interior Columbia (ICTRT 2005) and Central Valley TRT (Lindley et al. 2007) propose a somewhat more complicated approach in which risk is assessed based on the fraction of natural spawners of hatchery origin in relation to the degree of genetic divergence between hatchery and wild stocks, the management practices used at the hatchery, and the duration of interaction between hatchery and wild populations.

We considered using such approaches but concluded, for the reasons noted above, that few hatchery programs (current or recent) could be effectively evaluated by those criteria, and that case-by-case assessment of hatchery impacts is more appropriate for the NCCC Recovery Domain. Nevertheless, from these documents and others, we have drawn a number of important principles that can assist in guiding such assessments of risk. These principles are discussed in *Metrics and Estimation* below. Our decision not to adopt numeric criteria, as done by other TRTs, should not be construed as contradictory, but instead reflects substantial differences in the number and types of hatchery programs found in the different recovery domains. Within other recovery domains, existing programs are predominately large-scale production hatcheries that have been operated for many decades. In contrast, only two large-capacity production hatchery programs (Mad River and Warm Springs/Coyote Valley steelhead) are currently operating within the NCCC domain, the remainder being conservation hatcheries (e.g., captive broodstock programs) or small-scale cooperative supplementation hatcheries (Table 3).

Metrics and Estimation: Because analysis of risks associated with hatcheries should be done on a case-by-case basis, we do not propose specific metrics for assessing risk. To a substantial degree, the types of risks and hence the associated risk indicators depend on the type of hatchery program being considered. The Hatchery Scientific Review Group (HSRG 2004; Mobrand et al. 2005) suggests that, for the purposes of assessing risk, it is useful to distinguish between two types of hatchery programs based on management goals and protocols for propagating the hatchery broodstock. *Integrated* hatchery programs seek to minimize genetic divergence between the hatchery broodstock and a naturally spawning wild populaton by systematically incorporating wild fish into the hatchery broodstock. *Segregated* hatchery programs, in contrast, strive to maintain hatchery broodstock that are distinct from their wild counterparts by using predominately or exclusively hatchery-origin adults returning to the hatchery in subsequent broodstock. These general categories can be further subdivided based on the specific purposes of the hatchery (e.g., harvest augmentation, supplementation, restoration, rescue, etc.). The specific genetic, demographic, and ecological risks associated with various hatchery program types will differ, as can the approaches for minimizing such risks and the data needed for risk evaluation. We provide general guidance on issues that should be considered when evaluating risks associated with hatcheries, the types of information that are needed to evaluate these risks, and some basic principles that can inform risk assessment in Appendix C of this report. Without a thorough evaluation of hatchery risks, populations affected by hatcheries should generally be considered at risk because of the high uncertainty surrounding these potential effects.

Summary of Population Metrics and Estimators

Most of the metrics for evaluating populations against the proposed population viability criteria require time series of adult spawner abundance spanning three to four generations (but see preceeding discussion for possible use of abundance indices for estimation of population trends and catastrophic declines). Table 4 presents a summary of the metrics proposed in this paper and the data needs for estimating each.

Table 3. Current salmon and steelhead hatchery programs operating within the NCCC Recovery Domain, their purpose, mode of operation, and status.

Species, facility, and agency	River basin	Program type	Years of operation	Description and status
Chinook salmon				
Hollow Tree Creek (Eel River Restoration Project)	South Fork Eel River	Supplementation	1983 to present	Supplementation program that uses local broodstock to boost populations in Hollow Tree Creek, tributary to the South Fork Eel River. Development of hatchery genetic management plan ongoing.
Coho salmon				
Don Clausen Warm Springs (CDFG)	Russian River	Rescue/captive broodstock and restoration	1979 to present; captive broodstock since 2001	Historically a production program that used out-of-basin and out-of -ESU (primarily Noyo River) fish for broodstock. Captive broodstock program was initiated in 2001; juveniles are collected from tributaries (Green Valley Creek) are reared to the adult stage at the hatchery and then spawned. Juveniles are subsequently released into Russian River tributaries to re-establish depleted or extirpated subpopulations.
Big Creek (Monterey Bay Salmon and Trout Project)	Scott Creek	Rescue/captive broodstock, restoration, and supplementation	1982 to present; captive broodstock since 2001	Historically a supplementation program. Currently, a combined supplementation/captive broodstock/ restoration program. Broodstock are collected from Scott Creek; broodstock collection is prioritized so that only wild fish are taken in strong year classes, returning hatchery fish are used if wild fish are unavailable, and captive broodstock are used as last resort. Progeny are released into Scott Creek for supplementation, as well as in other watersheds to re-establish depleted or extirpated populations.
Steelhead				
Mad River winter steelhead (Friends of Mad River/CDFG)	Mad River	Production	1971 to present	Historically operated as a production program to support fisheries that was established with out-of-basin (Eel River) broodstock. Currently operating as a cooperative hatchery with a goal of releasing 150,000 yearlings annually. Development of hatchery genetic management plan ongoing.
Warm Springs/ Coyote Valley winter steelhead (CDFG)	Russian River	Production	1982 to present	Large-scale production program with goal of releasing 300,000 yearlings annually from Warm Springs and 200,000 yearlings from Coyote Valley. Some history of out-of-basin transfers (Eel and Mad River fish) pre-dating hatchery construction and continuing to the early 1990s (Busby et al. 1996). Development of a hatchery genetic management plan ongoing.
Big Creek winter steelhead (Monterey Bay Salmon and Trout Project)	Scott Creek/ San Lorenzo River	Supplementation	1982 to present	Supplementation program that uses local broodstock to boost populations in Scott Creek and the San Lorenzo River. Historically involved outbasin planting, but in recent years Scott Creek and San Lorenzo River fish have been planted only in their stream of origin.

Table 4. Estimation methods and data requirements for population viability metrics. Note that all references to population abundance refer to naturally produced adults (i.e., exclusive of hatchery returns).

Population Characteristic	Metric	Estimator	Data Needs
Effective population size per generation	$\overline{N}_e$	Variable: several direct and indirect methods for estimating N_e (see text).	Variable
-or-			
Total population size per generation	$\overline{N}_{g(harm)}$	Harmonic mean of spawner abundance per generation: $$\overline{N}_{g(harm)} = \frac{1}{\frac{1}{n} * \sum_{t=1}^{n} \frac{1}{N_{g(t)}}}$$ where n is the number of years, where $N_{g(t)}$ is the running sum of adult abundance over period equal to the population's mean generation time (rounded to the nearest whole year) at time t*	Time series of adult spawner abundance, N_a, for a minimum of 4 generations; demonstration that N_g remains above threshold during periods of low marine survival
Population decline Critical run size	$\overline{N}_{a(geom)}$	Geometric mean annual adult run size: $$\overline{N}_{a(geom)} = \left(\prod_{i=1}^{n} N_{a(i)} \right)^{1/n}$$	Time series of adult spawner abundance, N_a, for a minimum of 4 generations; demonstration that N_a remains above threshold during periods of low marine survival
Population trend	T	Slope of natural log of the g-year running sum of abundance v. time: $\hat{T}$ = slope ln(N_a+1) v. time where N_a is as defined above	Time series of adult spawner abundance, N_a, for 2-4 generations; demonstration that increasing trend is not result of short-term increases in marine survival
Catastrophic decline	C	Maximum 1-generation decline (proportion) in abundance: $$\hat{C} = \text{maximum} \left(1 - \frac{N_{g(t)}}{N_{g(t-2h)}} \right)$$ where $N_{g(t)}$ is as defined above, and h is the mean generation time (rounded to the nearest whole year)	Time series of adult spawner abundance, N_a; minimum of 3 generations to estimate short-term catastrophic risk; for longer time series, need analysis of trends following catastrophic decline and information on marine survival
Population density		Mean spawner density expressed as spawners per IP kilometer (see text).	Time series of adult spawner abundance, N_a, or mean spawner density from randomized survey locations; 4 generations
Depensation	$\overline{D}_{dep}$	Arithmetic mean of spawner density for lowest h consecutive years within the last 4 generations where h is mean generation time. $$\hat{D}_{dep} = \left(\min \left[\frac{N_{g(t)}}{h} \right] \right) \Big/ IPkm$$	

Table 4. (continued)

Population density Spatial structure and diversity	$\overline{D}_{ssd}$	Arithmetic mean of spawner density for past 4 generations $\hat{D}_{ssd} = \frac{1}{4h}\sum_{t=1}^{4h}\frac{N_a}{IPkm}$ where $IPkm$ is the sum of available stream kilometers of habitat multiplied by their IP value, and h is mean generation time.	Time series of either adult spawner abundance, N_a, or mean spawner density from randomized survey locations; minimum of 4 generations. $IPkm$ estimates for each population.
Hatchery influence	No specific metrics of estimators proposed. See text for guidance on potentially appropriate analyses.		

* In the absence of population-specific information, mean generation time is assumed to be 3 yrs for coho salmon, and 4 yrs for steelhead and Chinook salmon, which constitute the most common ages at spawning for these species within the domain. For more southerly winter steelhead populations, 3 yr-olds may constitute the majority of adult spawners (Busby et al. 1996).

Critical Considerations for Implementation

The TRT cautions that the generalized criteria proposed here are subject to substantial uncertainty arising from many different sources. For example, there is debate in the scientific literature regarding the appropriateness of the effective population size criteria of N_e > 500 for low risk, with some authors suggesting values as much as an order of magnitude higher. Likewise, various authors have suggested depensation thresholds ranging anywhere from 1 to 5 spawners/km. Perhaps even greater uncertainty surrounds the low-risk density criteria established for the purpose of maintaining spatial structure and diversity. In this case, although we believe the density criterion serves as a useful proxy for addressing spatial structure and diversity, quantitatively relating these parameters to extinction risk remains a challenge. Adding to this uncertainty is the fact that populations may fundamentally differ in their productive potential; hence, populations of comparable size may have different extinction risks. It is entirely conceivable that some of the criteria may ultimately turn out to be overly conservative in some cases and not precautionary enough in others.

Because of these uncertainties, we strongly caution against treating the recommended thresholds as “absolutes” or “knife-edge” decision points. More accurately, the criteria represent a set of viability indicators, which, if all low-risk thresholds were met, would suggest that a population has a relatively high likelihood of persisting into the future. Obviously, we are most certain about the status of populations that are far above or below the low- and high-risk thresholds, respectively. Likewise, we have greater certainty about the status of populations that lie close to identified thresholds for one metric, than we do for populations that are marginal for multiple metrics. Ultimately, however, decreasing uncertainty about the viability of populations will require a better understanding of the dynamics of individual populations, which can only come about with increased attention to research and monitoring within the recovery domain. In the interim, we believe that, collectively, the criteria provide a reasonably precautionary approach to assessing viability.

We also note that there will likely be situations where implementation of the criteria is confounded by special circumstances. The general framework we have adopted assumes that the historical (pre-EuroAmerican settlement) population abundance, distribution, and diversity represent reference conditions under which populations had a high probability of persisting over long periods of time. With respect to diversity, we foresee situations where assessing genetic risk will require considerations outside the scope of the proposed viability criteria. One such case is where a population has undergone a severe population bottleneck but has since recovered to levels that, from a demographic standpoint, suggest low risk. Low genetic diversity resulting from the bottleneck would indicate that the population remains at elevated risk of extinction. However, managers will need to assess at what point the risk no longer appears significant. An example of such a case is the northern elephant seal, which was hunted to near extinction in the 19th century, but has since rebounded to population sizes of about 175,000 individuals (Weber et al. 2000). The population displays extremely low genetic variation, but apparently with minimal consequences for fitness. It remains unclear whether such a population may be prone to disease outbreaks or substantial changes in environmental conditions. Similar questions will need to be addressed in cases where populations that have been extirpated or reduced to low levels and subsequently restored through hatchery activities. Clearly, such cases will need a more rigorous assessment process than that proposed in our relatively simple approach.

While we acknowledge that there are uncertainties around the proposed population viability criteria, we do not believe these uncertainties should seriously impede recovery planning. The proposed population viability criteria represent our best judgment given the available scientific information, and we fully acknowledge that these should be considered preliminary and subject to change if credible scientific evidence suggests that the criteria are inappropriate, either as general criteria or on a case-by-case basis as population-specific information becomes available. The simple reality is that the vast majority of independent populations of all listed species within the NCCC Recovery Domain are far from reaching the proposed targets, and resolving whether the ultimate recovery target should be 2,000 or 3,000 fish does little to advance recovery planning. Regardless of the specific targets, the critical actions needed for recovery will, in the majority of cases, be the same irrespective of the viability target. Should we ever get to the point where (a) we have sufficient data to estimated population abundances with reasonable precision, and (b) we begin to approach the proposed viability targets, the questions about the uncertainties can and undoubtedly will be reassessed.

3 ESU Viability Criteria[17]

3.1 Characteristics of Viable ESUs

At the ESU level, viability criteria focus primarily on maintaining the ESU as an integrated, functioning biological unit by seeking to buffer the ESU against catastrophic loss of populations by ensuring redundancy, provide sufficient connectivity among populations to maintain long-term demographic and evolutionary processes, and ensure sufficient genetic and phenotypic diversity to maintain the ESU's evolutionary potential in the face of changing environmental conditions. Because we are most certain that an ESU would have persisted more or less indefinitely under conditions that existed prior to the impacts stemming from European-American settlement of the West Coast, the historical population structure of an ESU provides a template against which proposed ESU viability criteria can be evaluated. Although ESU viability almost certainly declines with increasing departure from historical ESU structure, the precise nature of this relation is unknown. To accommodate this uncertainty in a precautionary manner, we therefore suggest that the degree of proof required to demonstrate that a proposed ESU configuration is consistent with ESU viability should increase with increasing departure from historical ESU structure. Bjorkstedt et al. (2005) identified historical population structure that explicitly recognizes variation in the functional roles that populations filled within the historical ESU (i.e., functionally independent, potentially independent, and dependent populations) and, in anticipation of the present report, proposed a general structure for ESU viability criteria that accommodates this variation. We expand upon their proposal below.

The arrangement and status of populations within an ESU must balance between populations sharing common catastrophic risks and maintaining sufficient connectivity via dispersal among populations. Thus, viable populations need to be distributed across the landscape, yet not to be so distant from one another that dispersal is ineffective in maintaining connectivity across an ESU. Moreover, in order to maintain or restore connectivity patterns similar to those that historically underlay ESU structure, some populations must be sufficiently large to produce dispersers (strays) in sufficient numbers (1) to support adequate exchange among populations and subsidies to dependent populations; (2) to increase overall abundance in the ESU; and (3) to provide additional capacity to buffer the ESU against catastrophic disturbance. Based on their historical roles in the ESU, functionally independent populations (FIPs) and potentially independent populations (PIPs) are essential to ensuring connectivity. However, dependent populations (DPs) and the smaller watersheds they occupy also contribute substantially to ESU connectivity and therefore provide an essential contribution to ESU viability. Likewise, dependent

[17] Again, we remind the reader that we use the term ESU to mean both salmon ESUs and steelhead DPSs.

populations may provide important temporary refugia and potential sources of colonizers or broodstock for restoration of nearby FIPs and PIPs that have been extirpated (e.g., Scott and Waddell creeks are extant dependent populations in the Santa Cruz Mountains diversity stratum of the Central California Coast Coho Salmon ESU).

ESU structure should maintain representative diversity within the ESU and thus maintain the evolutionary potential of the ESU. To satisfy this requirement, we propose that a viable ESU include representation across diversity strata, as defined in Bjorkstedt et al. (2005) and revised in this report (see Appendix A). These diversity strata are intended primarily to reflect diversity arising from variation in environmental conditions in freshwater habitats, a major component of the selective regime affecting salmon and steelhead. Because genetic and geographic distances appear to be strongly correlated for anadromous salmonids within coastal regions of California (Bjorkstedt et al. 2005; Bucklin et al. 2007; Garza et al., in review), we expect that the occurrence of viable populations in all diversity strata will result in a spatial arrangement that contributes to maintenance of genetic diversity at the ESU scale.

3.2 ESU-level Criteria

In the following sections, we propose ESU viability criteria intended to ensure representation of the diversity within an ESU across much of its historical range, to buffer an ESU against potential catastrophic risks, and to provide sufficient connectivity among populations to maintain long-term demographic and genetic processes. We specify these criteria not in terms of specific sets of populations but rather as a set of conditions to be satisfied by a configuration of populations. In some cases, attainment of these conditions will require that certain populations be included in any specific scenario of ESU viability. More often, however, there will exist several plausible scenarios of population viability that could satisfy ESU-level criteria.

As with the population-level criteria, the proposed set of ESU-level criteria represent conditions for which we believe an ESU would have a high likelihood of persisting over long time frames (hundreds of years). The criteria are based on general principles of conservation biology and are intended to serve as precautionary guidelines that incorporate uncertainty about the rates at which populations historically interacted, both within and among diversity strata, as well as across ESU boundaries. Consequently, we note that there may be specific population and diversity strata configurations that could lead to ESU viability without strictly meeting all of the proposed criteria for every diversity stratum. For example, the geography of the California coastline makes certain diversity strata more important than others for fostering within-ESU connectivity or providing representation of a significant portion of the ESUs historical range or evolutionary potential. We emphasize, however, that in evaluating such alternatives, demonstration that the primary goals of representation, redundancy, and connectivity are not compromised would be essential, and that adopting such configurations without further information on larger-scale processes necessarily entails accepting greater risk of extinction for the ESU.

Representation Criteria

1. a. All identified diversity strata that include historical FIPs or PIPs within an ESU or DPS should be represented by viable populations for the ESU or DPS to be considered viable.

-AND-

b. Within each diversity stratum, all extant phenotypic diversity (i.e., major life-history types) should be represented by viable populations.

Representation of all diversity strata achieves the primary goal of maintaining a substantial degree of the ESU's historical diversity (i.e., genetic diversity, exposure and responses, including presumed adaptation, to diverse environmental conditions). Representation of all diversity strata, by virtue of the geographical structure of diversity strata, also contributes to ensuring that the ESU persists throughout a significant portion of its historical range and that connectivity is maintained across this distribution. The second element of the representation criteria (1.b) specifically addresses the persistence of major life-history types, specifically summer steelhead, as an important component of ESU viability.

In the NCCC Recovery Domain, evaluation of ESU viability must consider an additional complexity. Coho salmon and Chinook salmon reach their southernmost (coastal) limits within the NCCC Domain. Likewise, in two species the expression of major life-history types, spring-run Chinook and summer steelhead, also reach their southernmost extent within coastal basins[18]. Species ranges and life-history distribution patterns represent ESU edges in a geographic and evolutionary sense, respectively, which raises the issue of how much an ESU can contract and remain viable.

In two cases, the TRT expressed high uncertainty regarding whether populations were ever historically persistent in areas that lie near the edge of the species range: coho salmon in watersheds tributary to the

[18] Interior populations of spring Chinook salmon occur to the south in the Sacramento River basin. Likewise, summer steelhead may also have inhabited Central Valley streams draining the west slope of the Sierra Nevada at one time (McEwan 2001).

San Francisco Bay Estuary[19] (with the possible exception of a few watersheds that enter the Bay relatively close to the Golden Gate and that drain the eastern slopes of the coastal mountains) and Chinook salmon in coastal basins from the Navarro River to the Gualala River[20] (Bjorkstedt et al. 2005). In both cases, analysis of long-term average environmental characteristics of these areas suggests that environmental conditions were substantially less favorable for these species and were possibly favorable only on an inconsistent basis. Requiring viable populations where none may have existed historically as a prerequisite for ESU viability is obviously problematic, and it is therefore possible that a viable ESU might not include full representation of populations in these 'edge' regions. Nevertheless, persistent occurrence or frequent observation of the species in these areas would be strong evidence that nearby strata were producing dispersers and that habitat quality within these source watersheds was improving, which would also bode well for other species (e.g., steelhead).

In the case of life-history types that have experienced tremendous reduction in abundance (e.g., summer steelhead in the NC-steelhead ESU) or extirpation (e.g., spring Chinook in the CC-Chinook ESU), it is also possible that such losses do not necessarily indicate substantial risk to ESU viability in demographic terms, and that a viable ESU lacking this diversity might be possible. However, these populations represent unique components of ESU diversity and the evolutionary legacy of the ESU, and it is difficult to justify ignoring this diversity in ESU viability criteria focused on diversity, particularly if recovery planning follows the precautionary approach of requiring increasingly stronger proof of viability to counter increasing departure from the template of historical ESU structure (Lesica and Allendorf 1995). It appears that, in coastal ESUs, spring-run Chinook salmon arose from fall-run Chinook salmon in the same basin (Waples et al. 2004). Loss of these populations therefore may not be irrevocable if the genetic variability that underlies their origin has not been lost in extant fall-run populations. Likewise, coastal summer steelhead appear to be derived from local winter steelhead populations, which might retain a genetic legacy that will support re-expression of summer-run populations. In both cases, however, demonstration that this potential has not been lost would require restoration of environmental conditions (i.e., coldwater refugia that allow adults to oversummer) that allow expression of these life-history types and an unknown period of time for populations to express these phenotypes. It is worth noting that Chinook salmon from a common source (Battle Creek, CA) introduced into rivers of New Zealand during the early 1900s currently exhibit a broad range of phenotypes, including differences in the period of

[19] Note that the uncertainty is not about whether coho salmon occurred in the San Francisco Bay Area, which is well documented (see Leidy et al. 2005a), but rather whether any populations were sufficiently large to function independently.

[20] In contrast to the coastal basins of moderate size, the Russian River is likely to have provided adequate access and spawning habitat for fall-run Chinook salmon on a consistent basis. Thus, the TRT concluded, with little uncertainty, that the population of fall-run Chinook salmon in the Russian River was a functionally independent population under historical conditions (Bjorkstedt, et al. 2005).

freshwater residency and timing of adult migration (Quinn and Unwin 1993; Quinn et al. 2001), suggesting that re-expression of life-history variation over periods of a few tens of generations may be possible. However, whether re-expression of clearly defined spring Chinook runs in the NCCC Recovery Domain is possible remains highly uncertain.

Efforts to set the stage for recovery of locally extirpated life-history types are independently justified by a slight extension of the 'historical template' argument to consider the role of these life- history types as sensitive indicators of habitat conditions. Because of their need for low summer water temperatures (for adult holding), spring-run Chinook salmon and summer steelhead are likely to be substantially more sensitive to factors that affect freshwater habitat quality than are fall-run and winter populations. Fall Chinook salmon and winter steelhead spend less time as adults in freshwater, do so under relatively benign seasonal conditions, and, in the case of fall-run Chinook salmon, usually (though not always) leave freshwater as juveniles before more stressful conditions develop during the summer. Restoration of habitat conditions that will presumably allow re-emergence of the more sensitive life-history types (even in the absence of such re-emergence) or recovery of those populations that remain extant is almost certain to benefit populations of fall-run Chinook or winter steelhead in the same watershed, and thus to provide additional assurances that these populations are, in fact, viable and contributing as expected to ESU viability. Such habitat restoration will increase the potential range of life-history variation (e.g., age at ocean-entry) that can complete the life cycle in such populations and thus increase the ability of such populations to persist in the face of a broader range of environmental perturbations. Thus, although the representation criteria do not require re-expression of diversity that has been lost due to extirpation, we encourage recovery planners to pursue actions that would benefit these more sensitive life-history types.

Redundancy and Connectivity Criteria

Three additional and interrelated criteria for ESU viability are proposed for guarding against catastrophic risk (redundancy) and ensuring sufficient connectivity across and ESU. For each diversity stratum:

2. a. At least fifty percent of historically independent populations (FIPs or PIPs) in each diversity stratum must be demonstrated to be at low risk of extinction according to the population viability criteria developed in this report. For strata with three or fewer independent populations, at least two populations must be viable.

-AND-

b. **Within each diversity stratum, the total aggregate abundance of independent populations selected to satisfy this criterion must meet or exceed 50% of the aggregate viable population abundance (i.e., meeting density-based criteria for low risk) for all FIPs and PIPs.**

In developing strategies to satisfy this requirement, recovery planners should seek ESU configurations that emphasize historical populations that, by virtue of their size and location, formed the foundation of the ESU. Ideally, this will mean that the first criterion is satisfied directly, thereby satisfying the second criterion as well. In some cases, however, it may prove infeasible to implement a strategy that will include restoration of the larger FIPs or PIPs in an ESU to a state relative to their historical status that will consequently lead to sufficient abundance within the stratum. An example might be if a substantial proportion of historical habitat was either no longer accessible due to a dam or so degraded as to have a very low likelihood of being restored. In such cases, recovery planners may need to identify stratum-scale recovery strategies that include (1) restoring some (presumably historically large) FIPs so that they are demonstrably viable but occupy only a remnant of the historical population's range, and so cannot be considered as being entirely representative of the historical population, and (2) restoring additional (presumably smaller) FIPs, or PIPs, to a sufficient degree for stratum abundance to satisfy the second part of this criterion.

Note that any FIP or PIP contributing to the aggregate stratum abundance must be a viable population[21], and must (1) have abundance above the minimum viable level for a small basin (e.g., N_a > 40 fish x minimum IP requirement = 1,280 for coho, 800 for Chinook, 640 for steelhead) with the distribution of fish such that the density criterion is satisfied within the remaining useable habitat[22], and (2) meet minimum thresholds for low genetic risk (N_g > 2500).

3. **Remaining populations, including historical DPs and any historical FIPs and PIPs that are not expected to attain a viable status, must exhibit occupancy patterns consistent with those expected under sufficient immigration subsidy arising from the 'core' independent populations selected to satisfy the preceding criterion.**

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[21] Dependent populations, as well as independent populations that fail to meet minimum standards for viability, by definition are not expected to persist over long time frames in the absence of subsidies from other neighboring populations. Consequently, only populations that are expected to persist and could do so in isolation are counted toward the aggregate population criterion.

[22] In the case of populations affected by impassible dams or other human-caused barriers to fish passage, the remaining useable habitat will consist of habitat downstream of the obstruction. In areas still accessible to anadromous fish, but affected by severe and irreversible habitat modification, recovery planners will need to explicitly define those portions of a watershed expected to contribute to a viable population.

Within this set of populations, we recommend that recovery planners place a high priority on populations that are remnants of historical FIPs and PIPs, and, that, at a minimum, most historically independent populations should be at no greater than moderate risk of extinction when evaluated as independent populations. Although such populations no longer fully serve their historical role within the ESU, remaining elements of these populations can contribute substantially to connectivity and, in general, are more likely than dependent populations to represent major parts of the ESUs evolutionary legacy. Additionally, planners should place high priority on maintaining dependent populations in situations where associated historical FIPs and PIPs are at high risk of extinction or have been extirpated. In these situations, dependent populations may be vital as sources of colonizers and genetic diversity to support restoration of adjacent FIPs and PIPs, and afterwards to buffer these larger populations against future disturbances. Indeed, during the recovery process, dependent populations may act (temporarily) as source populations for nearby FIPs and PIPs that have been reduced to sink status. Likewise, dependent populations can be expected to contribute to maintaining genetic diversity within a stratum and providing a source of colonizers that can reduce both genetic and demographic risks to adjacent FIPs and PIPs.

4. The distribution of extant populations, regardless of historical status, must maintain connectivity within the diversity stratum, as well as connectivity to neighboring diversity strata.

To ensure this, it might prove necessary to identify key watersheds that fill what would otherwise be substantial spatial gaps in the diversity stratum. Such watersheds might harbor populations considered to have been historically dependent on immigration from other populations. Ensuring that such populations persist requires ensuring that their source populations are also at a sufficient status to maintain connectivity. Currently, data on both the distances that Pacific salmonids within California's coastal region stray from their natal streams and the rates at which they do so is insufficient to provide concrete guidance on how close adjacent populations should be to maintain connectivity. However, a limited number of studies of straying by Chinook salmon (Hard and Heard 1999), pink salmon (Wertheimer et al. 2000), chum salmon (Tallman and Healey 1994), and Atlantic salmon (Jonsson et al. 2003) in other regions suggest that the majority of salmon that stray enter streams within a few tens of kilometers from their natal stream (or stream of release). Assuming that salmon and steelhead populations in coastal California exhibit similar tendencies, unoccupied gaps along the coastline of more than 20–30 km may be sufficient to disrupt normal patterns of dispersal and connectivity.

3.3 Example Scenarios of Application of ESU-Viability Criteria

In this section, we present a series of hypothetical scenarios to illustrate how ESU viability criteria for individual diversity strata (DS) might be applied to evaluate DS configurations proposed as the goal for recovery efforts. We propose a hypothetical diversity stratum that historically comprised three FIPs, three PIPS, and nine dependent populations (Figure 6), and then identify various scenarios of distribution and abundance to evaluate whether each would be considered viable according to the criteria proposed in this document (Table 5). The set of scenarios identified below is hardly exhaustive and serves simply to highlight a range of possible proposals and where such proposals might be expected to succeed or fail in establishing a DS that contributes to a viable ESU. Specifics regarding the cause of populations' status are left intentionally vague. Proposed reduction in habitat capacity from current measurements may arise from planned loss of habitat, or perhaps more likely, will stem from redefinition of the extent of occupied or habitable habitat to allow population viability criteria to be based on densities in occupied areas.

Current Conditions

In its current state (column labeled "Actual N_a in Table 5), the DS does not contribute to ESU viability. All historically independent populations fail to satisfy requirements for population viability, some dependent populations are no longer extant, and those dependent populations that remain are at low density. Connectivity is not necessarily eroded as a consequence of disruption to the spatial arrangement of populations in the DS. However, substantial declines in abundance are likely to underlie reductions in the number of dispersers, especially emigrants from historically independent populations, and therefore to compromise connectivity among populations. The spatial arrangement of populations continues to maintain a degree of independence among populations with respect to catastrophic disturbance and is likely to maintain a substantial portion of historical diversity associated with environmental variation.

Scenario I

In this scenario, recovery actions are directed at increasing the quality of available habitat in historically independent populations and thus boosting abundance, but there is no effort to restore access to areas that have been effectively lost to the DS, or to improve conditions in watersheds occupied by historically dependent populations. Three historically independent populations are recovered to viability (two historically FIP and one historically PIP), but these populations do not include sufficient abundance to satisfy overall DS abundance requirements. Connectivity is likely to improve, as most populations are included in the configuration, and abundance in the larger source populations is increased.

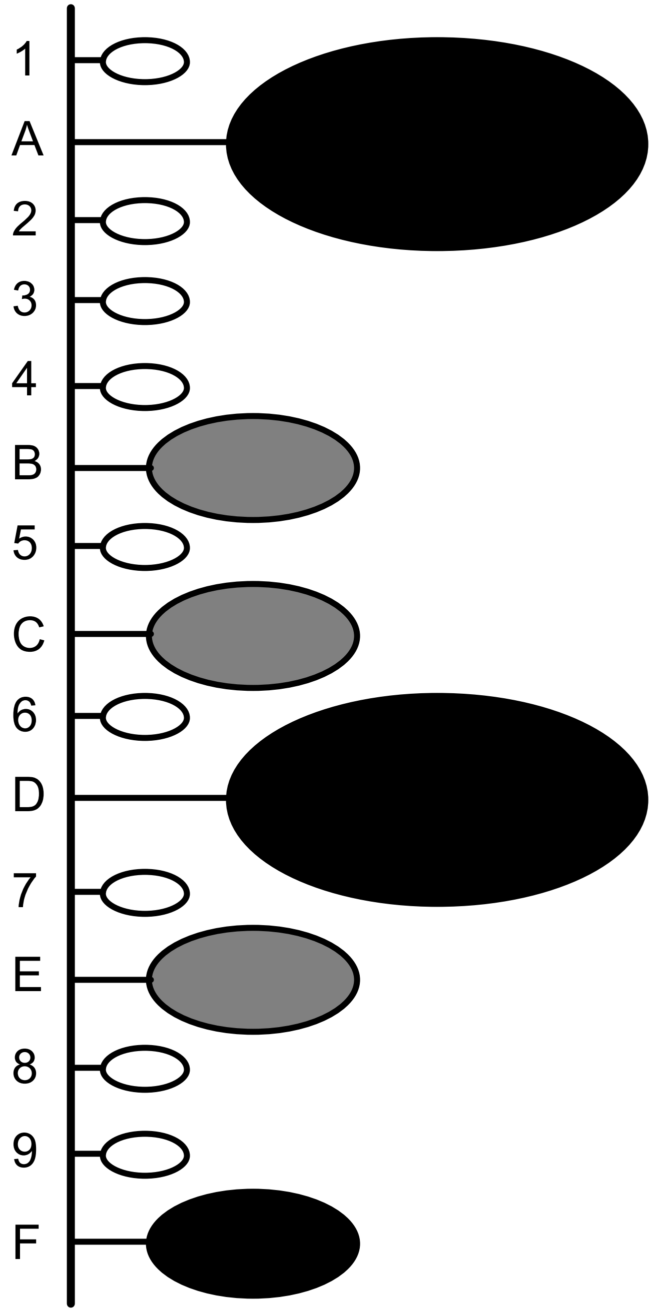

Figure 6. Historical population structure of a hypothetical diversity stratum within an ESU. Oval size is crudely proportional to historical population size. Black ovals are historical functionally independent populations. Grey ovals are historical potentially independent populations. White ovals are dependent populations. Population IDs correspond to those in Table 5.

Table 5. Historical structure, current conditions, and potential recovery planning scenarios for a hypothetical diversity stratum in a listed ESU (illustrated in Figure 6). N_a = average annual number of spawners. Under Scenarios, 'Pot' refers to target potential N_a based on accessible habitat, 'Real' refers to realized N_a. Scenarios are described in greater detail and evaluated in text. Minimum N_a, which corresponds to a minimum extent of habitat and associated density criterion, is set at 1,200.

Population		Potential N_a		Actual N_a	Scenario I		Scenario II		Scenario III		Scenario IV		Scenario V		Scenario VI		Scenario VII	
		Historic	Curr		Pot.	Real.	Pot.	Real.	Pot.	Real.	Pot.	Real.	Pot.	Real.	Pot.	Real.	Pot.	Real.
FIPs	A	8,500	2,500	500	2,500	2,500	2,500	2,500	4,000	4,000	6,000	6,000	5,000	5,000	1,000	1,000	1,500	1,500
	D	6,000	3,000	1,000	3,000	3,000	3,000	3,000	4,000	4,000	5,000	5,000	4,000	4,000	1,000	1,000	3,000	3,000
	F	2,000	2,000	200	500	500	1,200	1,200	1,100	1,100	2,000	2,000	2,000	2,000	500	500	1,500	1,500
PIPs	B	2,200	1,500	300	1,500	1,500	1,500	1,500	1,500	1,500	0	0	1,000	1,000	2,200	2,200	2,200	2,200
	C	1,800	1,000	700	1,000	1,000	1,200	1,200	1,200	1,200	0	0	500	500	1,800	1,800	1,800	1,800
	E	1,500	500	500	500	500	1,200	1,200	1,200	1,200	0	0	500	500	1,500	1,500	1,500	1,500
DPs	1	200	50	50	50	50	50	50	50	50	0	0	0	0	50	50	50	50
	2	150	100	0	100	0	100	0	100	0	0	0	0	0	100	100	100	0
	3	300	100	100	100	100	100	0	100	100	0	0	100	100	100	100	100	100
	4	100	50	50	50	50	50	0	50	50	0	0	0	0	50	50	50	50
	5	200	100	0	100	0	100	0	100	0	0	0	0	0	100	100	100	0
	6	300	50	50	50	50	50	50	50	50	0	0	0	0	50	50	50	50
	7	200	100	0	100	0	100	0	100	0	0	0	0	0	100	100	100	0
	8	400	150	0	150	0	150	0	150	0	0	0	150	150	150	150	150	0
	9	150	100	100	100	100	100	100	100	100	0	0	0	0	100	100	100	100
Total DS N_a		24,000	11,300	3,550		9,350		10,800		13,350		13,000		13,250		8,800		11,850
% Hist. N_a			47	15		39		45		56		54		55		37		49
N_a in IPs		22,000		0		7,000		10,600		11,900		13,000		11,000		5,500		11,500
% Hist. N_a in IPs				0		32		48		54		59		50		25		52
Viable FIPs & PIPs				0		3		6		5		3		3		3		6
% Hist. FIPs & PIPs				0		50		100		83		50		50		50		100

Scenario II

In this scenario, recovery actions are directed at restoring all historically independent populations to viable status but increasing access to habitat only as necessary to meet the minimum abundance requirement for viability. Watersheds that harbor dependent populations are not restored, and some (DPs 2 and 3) decline further. The three viable historically independent populations recovered in Scenario I are now joined by three additional viable populations that satisfy the minimum requirements for viability, yet this configuration still does not satisfy the overall DS abundance criterion, since its historically large populations are only partially recovered. Connectivity is likely to be locally enhanced by increased abundance in source populations, but the lack of dependent populations 2, 3, and 4 leaves a substantial spatial gap between populations A and B (Figure 6).

Scenario III

In this scenario, recovery actions are directed at restoring all but one of the historically independent populations to viable status, with additional effort to increase habitat access (and therefore abundance) in historical FIPs. Watersheds that harbor dependent populations are not restored, nor are they allowed to degrade further. This configuration satisfies redundancy, and the viable populations include a satisfactory proportion of the historical potential N_a of the DS. Connectivity is good due to the occupancy of all populations. Connectivity with the rest of the ESU to the south of this DS must be evaluated in light of the projected non-viable status of the southernmost historically independent population (population F).

Scenario IV

In this scenario, recovery actions are directed solely at restoring the historically large populations in the DS, and as a tradeoff, populations elsewhere are effectively allowed to go extinct (or to decline to negligible abundance). Although the number of viable populations and the abundance of fish in these populations satisfy the relevant criteria for the DS to contribute to ESU viability, the loss of connectivity (i.e., substantial gaps between the three viable populations; Figure 6) and diversity within the DS precludes concluding that this configuration allows the DS to contribute to ESU viability.

Scenario V

In this scenario, recovery actions are directed primarily at restoring historical FIPs, but some effort is also directed at maintaining a selected set of populations as non-viable dependent populations, including populations in watersheds historically occupied by PIPs. This configuration satisfies the criteria for number of viable populations and proportion of fish in historically independent populations. The configuration also reduces risk to the DS by distributing populations across the landscape, and presumably increasing connectivity within the ESU. Diversity may also be increased, in terms of the habitats occupied, but the degree to which diversity is preserved in the dependent populations (including the non-viable PIPs) may be limited.

Scenario VI

In this scenario, recovery actions are focused on maintaining the status quo in historical FIPs, while restoring historical PIPs to something approaching their original status. In addition, recovery focuses on maintaining occupancy of dependent populations throughout the DS. This scenario satisfies criteria for number of viable populations and connectivity, but it fails to include a sufficient abundance of fish in viable populations. Diversity might also be compromised, depending on the character of the remnants of the historical FIPs.

Scenario VII

In this scenario, viable populations are restored in all historically independent populations, although the viable populations in watersheds historically occupied by FIPs are now spatially restricted viable remnants of the historical populations. This scenario satisfies criteria for number of populations, abundance within viable populations, and connectivity. Again, diversity issues need to be considered in light of the fact that historical FIPs are now represented as viable remnant populations, and diversity associated with lost portions of their watersheds might not be represented elsewhere in the DS.

3.4 Other Considerations

The proposed criteria for DS to contribute ESU viability represent an approach that, while precautionary, is intended to correspond to what the TRT believes is a maximum acceptable level of risk for the ESU to be susceptible to future decline, disintegration, and extinction, and as such represent the minimum conditions that must be achieved in each DS for an ESU to be considered viable. Achieving these minimum conditions is not sufficient for long-term viability—these conditions must be maintained. As a consequence, recovery actions that lead to ESU configurations that exceed ESU viability criteria, even slightly, are likely to decrease the risk facing the ESU and thus the risk that future recovery crises will arise.

Although the scenarios discussed above are measured against these minimal benchmarks, comparisons among some of the scenarios illustrate how going beyond minimal viability requirements can provide additional buffering against future events. For example, the differences between Scenario IV and

Scenario V involves a trade-off between concentrating efforts (and fish) in the three largest populations (Scenario IV) and distributing fish among dependent populations while retaining a focus on historical FIPs (Scenario V). The latter scenario is likely to reduce risk by increasing the resiliency of the DS as a whole through increased connectivity and thus the potential for the other populations to buffer individual populations that experience disturbance or a temporary decline. In general, increasing the number of extant populations will contribute to viability, even when those populations would not be considered viable independently.

One caution that must also be kept in mind is that viable ESUs and their component DSs cannot be considered as static entities. Relative abundance in populations within an ESU or DS can fluctuate substantially in response to natural environmental variation, and populations that were once numerically dominant can decline and be replaced by others as the most productive populations (see e.g., Hilborn et al. 2003). A prudent recovery strategy will accommodate this potential by creating conditions that allow populations not included in configurations designed to meet the minimum ESU/DS criteria to recover as a buffer against loss or decline of populations that are the focus of intense recovery efforts. For this reason, a recovery plan that begins with Scenario II, III or V as an initial goal (and thus avoids a trade-off such as illustrated in Scenario IV) is preferable, as it allows for the development of an ESU with greater flexibility to respond to disturbance of an extant population and does not shut down options for future restoration to further increase ESU resiliency.

Finally, we note that the proposed ESU-level criteria are based on certain assumptions about historical population structure, which in turn were based on assumptions about both the minimum habitat needed to support a viable population in isolation and the level of interaction among populations. The TRT acknowledges the possibility of more complex population structures. For example, although we defined populations occupying smaller watersheds (i.e., below minimum IP thresholds) to be “dependent”, it is possible that geographically proximate dependent populations may interact to a degree sufficient to collectively form a larger unit with a likelihood of persistence comparable to a viable independent population. Should such population structures be demonstrated to exist, it is conceivable that rules regarding stratum viability could be modified accordingly (e.g., a viable group of “mutually dependent” populations might be considered comparable to a viable independent population). We draw attention to this scenario to alert recovery planners to the need to consider such possibilities when developing recovery strategies. Our concern is that although historically independent populations should almost certainly form the core of any recovery strategy, there are specific instances where it may be more prudent to focus initial restoration and recovery efforts on extant dependent populations than on independent populations that have been extirpated or that inhabit watersheds that are so degraded as to have a low probability of supporting persistent populations for the foreseeable future.

At the present time, data are not available to identify specific instances of where sets of mutually dependent populations might function as plausible recovery units. Support of such a delineation would require substantial information on all populations involved. First, there would need to be direct estimates of straying among putative constituent dependent populations to demonstrate that exchange of individuals among these populations is sufficiently high to warrant consideration of the group as a single unit. Second, a determination would have to be made about the amount of total habitat that would be needed to support an aggregate group of dependent populations. The minimum IP thresholds to support viable coho salmon, Chinook salmon, and steelhead populations are estimated to be approximately 32 $IPkm$, 20 $IPkm$, and 16 $IPkm$, respectively. However, the amount of habitat needed to support a network of dependent populations depends on a number of factors, including the rate of exchange of individuals among populations, the variability in population abundance, and the degree of correlation in the dynamics of contributing populations, which is a function of heterogeneity of habitats and temporal synchrony in environmental conditions. Consequently, the total aggregate habitat needed to support a viable unit might be substantially different (either higher or lower) than the identified $IPkm$ thresholds and would not likely simply be an additive effect. Consequently, demonstrating that a group of populations functions as an independent unit with a specific extinction risk is not a simple undertaking.

4 Assessment of Current Viability of Salmon and Steelhead Populations within the NCCC Recovery Domain

The criteria presented in the preceding two chapters are intended to provide a framework for planners both to set general biologically based targets for recovery and to guide future evaluations of the status of ESA-listed salmonids within the NCCC Recovery domain. In this chapter, we apply the population-level and ESU-level viability criteria developed in Chapters 2 and 3 to salmon and steelhead within ESUs of the North-Central California Coast Recovery Domain to assess current viability. Theoretically, application of the criteria should occur in two steps. First, because the spawner density criteria for each population depend on specific watershed attributes (i.e., historical intrinsic habitat potential, expressed as *IPkm*), specific criterion values are estimated for each population. Determination of appropriate density criteria is confounded by the fact that, in some instances, habitat that was historically accessible to anadromous salmonids now lies behind impassible dams or other barriers. In some instances, remaining habitat, even if functioning properly, may be insufficient to support a viable population (i.e., available *IPkm* is less than the thresholds for viability-in-isolation established by Bjorkstedt et al. 2005). In other cases, it may be possible for a population to be viable without access to this historical habitat, though its functional role in relation to other populations in the ESU may have been substantially altered. For this reason, we estimate density criteria and associated population abundances (estimated as density multiplied by *IPkm*) for both historical (pre-barrier) and current (post-barrier) conditions[23]. In addition to allowing evaluation of whether or not a below-barrier population could be considered viable in its current habitat, this also highlights situations where access to blocked habitat may be either a necessary step to restore a population's viability or a desirable step for enhancing the population's role in maintaining ESU-viability. Appendix B provides further discussion of the relationship between population viability and the current accessibility and condition of habitats.

The second step involves evaluating risk according to the criteria. In reality, we have virtually no instances where currently available data are of sufficient quality and duration to rigorously assess population viability according to our criteria. Most of the population viability metrics require adult time series of abundance sufficient for estimating total population size of wild populations for a period of at least three or four generations. The few available time series of adult abundance for populations within the NCCC Recovery Domain generally are either too short in duration to apply the criteria, inadequate for estimating total population abundance, influenced to an unknown degree by hatchery fish, or otherwise

[23] Our estimates of habitat lost behind barriers include only major obstructions to fish passage and do not factor in the hundreds, if not thousands, of culverts and other smaller barriers that may partially or completely prevent fish passage.

deficient. As a result, strict application of the criteria results in most, if not all, populations being classified as "data deficient." However, in some circumstances, we have ancillary data (often highly qualitative) that strongly suggest that populations would currently fail to meet one or more of the identified low-risk or moderate-risk thresholds. It seems unsatisfying to simply describe these populations as data deficient when the collective body of data strongly suggests that populations are currently at elevated risk of extinction. In these instances, we assign a population-level risk designation, identifying the specific criteria that we believe the population is unlikely to satisfy and the data we believe justifies the particular risk rating. We caution, however, that while we occasionally used this ancillary data to assign a probable moderate or high risk, in no instances did we feel that such data were sufficient to assign a low-risk designation.

4.1 Central California Coast Coho Salmon

Population Viability

Summary of density-based criteria.

Within the Central California Coast Coho Salmon ESU, Bjorkstedt et al. (2005) identified eleven functionally independent populations (FIPs) and one potentially independent population (PIP). Table 6 summarizes proposed density-based criteria for these populations and the estimated population abundances (rounded to the nearest 100 spawners) that would result if density criteria were met under both historical (pre-dam) and current (post-dam) conditions. For each population, the high-risk abundance values indicate population-specific abundances below which populations are likely at substantial risk due to depensation. The low-risk estimates based on historically accessible habitat can be viewed as preliminary abundance targets that, if consistently exceeded, we believe would lead to a high probability of persistence over a 100-year time frame and would likely result in a population fulfilling its historical role in ESU viability.

Comparison of historical versus current *IPkm* provides a rough estimate of the proportion of historical habitat that is no longer accessible to the population and the affect this has on density and abundance targets. For the CCC ESU, the largest percentage losses of potential habitat have occurred in the Lagunitas Creek (49%) and Walker Creek (27%) watersheds. Estimated losses of *IPkm* due to dams in the San Lorenzo and Russian River watersheds are 7% and 3%, respectively. The relatively minor influence of dams in the Russian River is due to the fact that most of the predicted habitat lies in the lower coastal portions of the watershed, below the influence of major dams such as Coyote and Warm Springs dams. Losses of potential habitat due to dams for the remaining populations are estimated to be less than

Table 6. Projected population abundances (N_a) of CCC-Coho Salmon independent populations corresponding to a high-risk (depensation) thresholds of 1 spawner/*IPkm* and low-risk (spatial structure/diversity=SSD) thresholds based on application of spawner density criteria (see Figure 5). Values listed under "historical" represent criteria applied to the historical landscape in the absence of dams that block access to anadromous fish. Values listed under "current" exclude areas upstream from impassible dams. The IP-bias index is a qualitve measure of possible hydrologic bias in the IP model that could potentially lead to overprediction of historical habitat for juvenile coho salmon (Bjorkstedt et al. 2005).

					High Risk		Low Risk			
					Historical	Current	Historical SSD		Current SSD	
Population	Historical *IPkm*	Current *IPkm*	*IPkm* Lost	IP-bias index	Depens. N_a	Depens. N_a	Density spawner/*IPkm*	N_a	Density spawner/*IPkm*	N_a
Ten Mile River	105.1	105.1	0%	moderate	105	105	34.9	3700	34.9	3700
Noyo River	119.3	118.0	1%	moderate	119	118	33.9	4000	34.0	4000
Big River	193.7	191.8	1%	moderate	194	192	28.8	5600	28.9	5500
Albion River	59.2	59.2	0%	high	59	59	38.1	2300	38.1	2300
Navarro River	201.0	201.0	0%	high	201	201	28.3	5700	28.3	5700
Garcia River	76.0	76.0	0%	high	76	76	36.9	2800	36.9	2800
Gualala River	252.2	251.6	0%	high	252	252	24.7	6200	24.8	6200
Russian River	779.4	757.4	3%	high	779	757	20.0	15600	20.0	15100
Walker Creek	103.7	76.2	27%	high	104	76	35.0	3600	36.9	2800
Lagunitas Creek	137.0	70.4	49%	high	137	70	32.7	4500	37.3	2600
Pescadero Creek	60.6	60.6	0%	high	61	61	38.0	2300	38.0	2300
San Lorenzo River	135.3	126.4	7%	high	135	126	32.8	4400	33.4	4200

1%. Overall, Lagunitas and Walker creeks provide the only two instances where abundance targets change appreciably due to loss of historical habitat (Table 6).

Evaluation of current population viability

There are virtually no data of sufficient quality to rigorously assess the current viability of any of the twelve independent coho salmon populations within the CCC ESU using the proposed criteria. Consequently, many populations are identified as data deficient (Table 7). However, recent information on occupancy of historical streams within the CCC ESU indicates that wild populations of coho salmon are extinct or nearly so in a number of watersheds within the CCC ESU (Good et al. 2005). In the San Lorenzo River, annual summer surveys conducted on the San Lorenzo River and many of its tributaries failed to produce evidence of successful reproduction by coho salmon from 1994 to 2004 (D.W. Alley and Associates, 2005). After reports of approximately 50 adult spawners passing the Felton Diversion Dam (mostly marked hatchery fish) during the 2004–2005 spawning season, a few juvenile coho salmon were independently observed in a single tributary (Bean Creek) by Don Alley (D. W. Alley and Associates, pers. comm.) and by NMFS biologists (Brian Spence, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). However, extensive snorkel and electrofishing surveys elsewhere in the San Lorenzo River basin produced no other evidence of successful reproduction. Based on the apparent long-term absence of coho salmon form this watershed, we classified the San Lorenzo population as extinct (Table 7).

Pescadero Creek has been surveyed only sporadically over the last 10 years. Between 1995 and 2004, small numbers of juvenile coho salmon have occasionally been observed in the mainstem of Pescadero Creek, one of its tributaries (Peters Creek), and in the Pescadero estuary (Jennifer Nelson, CDFG, pers. comm..; Brian Spence and Tom Laidig, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). All but one of these observations come from the same brood cycle (1999, 2002, 2005). Planting of hatchery smolts (from Scott Creek) into Pescadero Creek in spring of 2003 apparently resulted in successful reproduction in the 2004–2005 spawning season, as approximately 1,600 juveniles were observed in snorkel surveys conducted in pools along 21 km of the mainstem of Pescadero Creek (roughly 33% of the accessible habitat in the watershed) by NMFS biologists in summer 2005. However, surveys conducted in 2006 and 2007 over approximately 8 km of both mainstem and tributary habitats revealed no juvenile coho salmon (Brian Spence, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). We categorized the extinction risk of this population as high, assuming that current abundance is sufficiently low that it would rate at high risk for three metrics: effective population

Table 7. Current viability of CCC-Coho Salmon independent populations based on metrics outlined in Tables 1 and 4. na indicates data of sufficient quality to estimate the population metric are not available. In some cases, risk categories have been designated for populations where ancillary data strongly suggest populations are extinct or nearly so, despite the lack of quantitative estimates of any of the viability metrics. Metrics for which we believe ancillary data support the assigned risk category are denoted with asterisks. See text for justification of risk rankings.

	PVA result	Effect. pop. size per generation	Tot. pop. size per generation	Population decline		Catastrophe	Density		Hatchery	Risk Category
Population		$\overline{N}_e$	$\overline{N}_{g(harm)}$	$\overline{N}_{a(geo)}$	$\hat{T}$	$\hat{C}$	$\hat{D}_{dep}$	$\hat{D}_{ssd}$		
Ten Mile River	na	na	na	na	na	na	na	na	na	Data deficient
Noyo River	na	na	na	na	na	na	na	na	na*	Moderate/High
Big River	na	na	na	na	na	na	na	na	na	Data deficient
Albion River	na	na	na	na	na	na	na	na	na	Data deficient
Navarro River	na	na	na	na	na	na	na	na	na	Data deficient
Garcia River	na	na*	na*	na*	na	na	na*	na*	na	High
Gualala River	na	na*	na*	na*	na	na	na*	na*	na	High
Russian River	na	na*	na*	na*	na	na	na*	na*	na*	High
Walker Creek	na	na*	na*	na*	na	na	na*	na*	na*	Extinct?
Lagunitas Creek	na	na	na	na	na	na	na	na	na	Data deficient*
Pescadero Creek	na	na*	na*	na*	na	na	na*	na*	na*	High
San Lorenzo River	na	na*	na*	na*	na	na	na*	na*	na	Extinct?

[*] See text for discussion of existing data for Lagunitas Creek.

size, population decline (mean annual spawner abundance), and spawner density (i.e., depensation risk; Table 7). The planting of Scott Creek fish into Pescadero Creek potentially poses a genetic risk to any remnant population that may still exist in the watershed, though these genetic risks may be trivial compared with the existing demographic risks given the population's apparent small size. Adult abundance of one dependent population of coho salmon, Scott Creek, has also been estimated from weir counts over the last four years (Sean Hayes, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). These estimates have averaged about 163 adults (range 6 to 329), though the 2005-2006 and 2006-2007 estimates were only 49 and 6 fish, respectively, and preliminary reports from 2007-2008 indicate very few returning adults. Hatchery fish accounted for about 34% of returning fish during the past four years. This is believed to be the largest remaining population south of San Francisco Bay.

The most reliable set of population data for any independent population in the CCC ESU comes from Lagunitas Creek, where spawner surveys have been conducted on a regular basis (flows permitting) since 1995. These surveys involve multiple visits to reaches representing a substantial portion of the available spawning habitats (Ettlinger et al. 2005). Redd counts from these surveys appear to provide the most consistent measure of abundance, as estimates of live spawners are likely biased high due to double-counting of individuals on successive surveys. Over the last 12 years, an average of about 260 coho redds (range 86-496) have been observed annually in the mainstem and upper tributaries of Lagunitas Creek. Additionally, National Park Service surveys of Olema Creek (a tributary to Lagunitas Creek), where maximum live/dead fish counts are recorded, indicate that a minimum of 86 fish have, on average, spawned in Olema Creek over the last eight years. These data did not meet our minimum requirements for application of viability metrics for several reasons. First, redd counts may lead to biased (both high and low) estimates of spawner abundance for a number of reasons, such as failure of observers to detect redds do to poor viewing conditions, redd superimposition, loss of redds due to scouring, individual females constructing multiple redds, or unequal sex ratios. Consequently, they may provide only an indicator of abundance[24]. Second, there is no information about spawner abundance in unsurveyed areas; thus, obtaining a total population estimate from these data is not currently possible. And finally, the 10-year time series does not yet meet the minimum data requirement of 4 generations for estimating effective population size, population decline, or density criteria. Consequently, we categorized the population as data deficient (Table 7). However, we note that with two additional years of data collection, additional analysis of the relationship between redd counts and total spawner abundance, and analysis of the relative

[24] Note that under the most favorable conditions (i.e., clear observation conditions throughout the spawning season, densities sufficiently low that superimposition is unlikely, and absence of scouring events), redd counts may prove to be an appropriate means for estimating adult spawner abundance; however, additional data are needed to establish a relationship between redd counts and total spawner abundance.

densities in surveyed versus unsurveyed reaches, these data could provide a reasonable basis for assessing population viability. We also note that the existing data suggest that, if current patterns continue, and assuming that one redd translates to approximately two spawning adults on average, the Lagunitas Creek population might satisfy low-risk criteria for the effective population size criteria and perhaps the population decline criteria as well. On the other hand, the population would likely be considered at moderate risk based on the density criteria. Lagunitas Creek and its tributaries received plantings of hatchery fish, primarily from the Noyo River but also from some out-of-ESU stocks, on numerous occasions between 1960 and 1987 (Bjorkstedt et al. 2005). Analysis of DNA microsatellite data from coho populations in California indicate some affinity between Lagunitas Creek and Noyo River coho salmon (J. Carlos Garza, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data); however, it is unclear whether this is the consequence of past hatchery plants or natural straying. Thus, it is difficult to assess potential residual hatchery-related risk for Lagunitas Creek. To our knowledge, there have been no recent plantings of hatchery fish into the Lagunitas watershed, suggesting that ongoing risks due to hatchery operations are minimal.

Naturally occurring coho salmon have not been observed in Walker Creek in several decades, though this stream was planted with 80 adult coho salmon (Olema Creek origin) from the Russian River captive broodstock program in January of 2004, and fingerlings—confirmed through genetic analysis to be primarily progeny of the planted adults—were observed in summer of 2004 (CDFG 2004; J. Carlos Garza, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). We categorized this population as “extinct” based on the long-term absence of naturally spawning coho salmon from this basin (Table 7).

In the Russian River basin, only one tributary (Green Valley Creek) has produced coho salmon annually in recent years, with salmon observed only sporadically in a few other tributaries (Merritt Smith Consulting 2003). Concerns over the decline of coho salmon in the Russian River basin have led to the establishment of a captive broodstock program at the Warm Springs (Don Clausen) Hatchery. Based on the sparse distribution (Good et al. 2005), the low apparent abundance, recent evidence of a genetic bottleneck (Libby Gilbert-Hovarth et al., NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data, cited in Bjorkstedt et al. 2005), and the perceived need for intervention with a captive broodstock program, we categorized the Russian River population as at high risk, assuming that it would rank at high risk for at least four of five population metrics (Table 7)

Limited surveys in the Garcia and Gualala rivers have documented occasional occurrence of coho salmon in the last 15 years, but the distribution of fish has been sparse in both river systems (Good et al. 2005). Observations in the Gualala River may have resulted from planting of young-of-the-year coho salmon from the Noyo River into the North Fork Gualala River in years 1995-1997 (Harris 2001). We categorized both the Gualala River and Garcia River populations as at least at high risk of extinction, as it is highly unlikely that either is sufficiently abundant to satisfy even the moderate risk criteria for effective population size, population decline (i.e., annual abundance), and density (depensation) criteria (Table 7).

Status of populations along the Mendocino Coast is less certain, though monitoring of one independent (Noyo River) and four dependent coho populations (Pudding Creek, Caspar Creek, Hare Creek, and Little River) was initiated by the California Department of Fish and Game in 2000 and 2001 (Gallagher and Wright 2007). Occupancy data suggest that populations in the Navarro, Albion, Big, Noyo, and Ten Mile rivers continue to persist but that their distributions have been substantially reduced (Good et al. 2005). In none of these cases are there sufficient population-level data to determine viability with any certainty; thus, we classified four of these populations (Navarro, Albion, Big, and Ten Mile) populations as data deficient (Table 7), though available occupancy data suggest that it is unlikely any are achieving the low-risk density criteria threshold and therefore may be at least at moderate risk.

In the case of the Noyo River, counts of adult spawners are available from the Noyo Egg Collecting Station on the South Fork Noyo River since 1962. These counts do not represent full counts (the station was operated irregularly in most years, and only about one-third of the avaiable habitat in the basin is located upstream of the ECS). Furthermore counts through 2005 are strongly influenced by hatchery activities that occurred from the early 1960s to 2003, when the last releases of hatchery coho salmon smolts were made. Counts from the mid 1990s to 2004 averaged about 620 fish; however, counts over the last three years have been among the lowest on record, with 79 fish in 2005-2006, 59 fish in 2006-2007, and even smaller numbers expected in 2007-2008. Estimates from Gallagher and Wright (2007) made using a variety of methods suggest that total numbers of coho spawners above the ECS likely exceed weir counts by 20% to 100%, depending on which estimator is used[25]. During the last two generations of hatchery operation, when all released hatchery yearlings were marked, returning hatchery adults constituted an average of 59% and 45%, respectively. Based on these data, and the fact the roughly one-third of the habitat in the Noyo River lies in the South Fork subbasin, we suspect that, even if straying of South Fork Noyo hatchery fish into other subbasins is low, the total percentage of hatchery

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[25] A primary goal of this research is to evaluate a wide range of estimating procedures, ranging from live fish and carcass mark-recapture estimates, redd counts (raw and adjusted based on fish-per-redd estimates), and AUC estimates.

fish in the entire basin likely exceeded 15%. This conclusion assumes that density of natural spawners in areas outside of the South Fork subbasin are not substantially higher than in the South Fork. Furthermore, the long history of stocking during which practices were not consistent with current best management practices (e.g., nonnative broodstock were occasionally used, and broodstock selection and mating protocols generally did not follow modern BMPs) suggests the potential for residual genetic effects of these operations. Thus, we classified Noyo River coho salmon as being at moderate/high risk due to past hatchery influence (Table 7). Although direct plantings of coho salmon into the Ten Mile, Big, Navarro, and Albion rivers do not currently occur, the potential exists for Noyo River hatchery fish to stray into these watersheds. The degree to which they do so is not known.

For the four dependent populations on the Mendocino Coast that are currently monitored, Pudding Creek has produced the largest numbers of spawning adults, averaging about 300 to 1200 fish, depending on which estimator is used. For the remaining three populations, average numbers of returning adults is estimated to be between 130 and 500 fish for Caspar Creek, 60-140 fish for Litte River, and 70-340 fish for Hare Creek, depending on the estimator used (Gallagher and Wright 2007).

ESU Viability

Though quantitative data on the abundance of coho salmon in the CCC ESU are scarce and many populations were described as data deficient (Table 7), ancillary data (primarily presence-absence data) clearly indicate that coho salmon in this ESU fail to meet both the representation and redundancy/connectivity criteria. The available data indicate that no populations meet low-risk criteria in three of the identified diversity strata (Santa Cruz Mountains, Coastal, and Gualala Point-Navarro Point), and that coho salmon are no longer present in an any of the San Francisco Bay dependent populations (indicating that either neighboring populations are not producing migrants in sufficient number to maintain these populations or the available habitat is incapable of supporting any migrants that do enter these systems). Status of populations along the Mendocino Coast is highly uncertain (all populations were categorized as data deficient), though we believe it is unlikely that any of these populations approach viable levels.

Connectivity among populations within and among diversity strata is a significant concern. Within the Santa Cruz Mountains stratum, the two identified functionally independent populations appear extinct (San Lorenzo River) or nearly so (Pescadero Creek). Dependent coho salmon populations still persist in three watersheds near the geographic center of the stratum, but only the Scott Creek population, which is supported by ongoing hatchery activities, has regularly produced spawners in all three brood lineages in recent years, and returns in the last two spawning seasons have been extremely poor. Both the Waddell Creek and Gazos Creek populations appear to have lost two year classes (Smith 2006; B. Spence, NMFS Santa Cruz, unpublished data). Coho salmon are occasionally observed in other watersheds (e.g., San Vicente, San Gregorio, and Laguna creeks), but these fish are likely the product of strays from either Scott Creek or hatchery fish that have been planted in area streams. Consequently, there are substantial portions of the stratum that have few or no coho salmon, and the nearest extant population to the north is Redwood Creek in Marin County, a dependent population some 100 km to the north. Likewise, in the Coastal stratum, coho salmon persist in significant numbers only in Lagunitas Creek, with a few coho found in the Russian River, as well as Redwood Creek to the south. To the north, in the Navarro Point-Gualala Point stratum, coho salmon appear scarce or extinct in all watersheds with the exception of the Navarro River. As the Lagunitas Creek and Navarro River populations are separated by an expanse of almost 160 km of coastline with almost no coho salmon, interactions among these populations may be minimal. Connectivity is currently less of a concern in the Lost Coast-Navarro Point stratum, as both independent and dependent populations of coho salmon still persist from Big Salmon Creek to the Ten Mile River (Good et al. 2005). It is unclear, however, how much recent distribution patterns have been influenced by hatchery operations within the Noyo River basin. The status of dependent populations to north of the Ten Mile River is poorly known, but it is possible that the Mattole River, in the SONCC ESU, is the nearest extant population that supports coho salmon on an annual basis. Coho salmon were observed in two consecutive years in the South Fork of Usal Creek (W. Jones, CDFG retired, personal observations), but it is uncertain whether coho salmon occur in all three brood years.

In summary, the lack of demonstrably viable populations (or the lack of data from which to assess viability) in any of the strata, the lack of redundancy in viable populations in any of the strata, and the substantial gaps in the distribution of coho salmon throughout the CCC ESU strongly indicate that this ESU is currently in danger of extinction. Our conclusion is consistent with recently published status reviews prepared by the National Marine Fisheries Service (Good et al. 2005) and the California Department of Fish and Game (CDFG 2002).

4.2 California Coastal Chinook Salmon

Population Viability

Summary of density-based criteria

The NCCC TRT (Bjorkstedt et al. 2005) proposed that the CC-Chinook ESU historically comprised fifteen independent populations of fall-run Chinook salmon (10 functionally independent and five potentially independent) and six independent populations of spring-run Chinook salmon (all functionally independent[26]). However, the TRT also noted that, due to the lack of historical data on Chinook salmon abundance within the ESU, the hypothesized population structure is subject to substantial uncertainty. Contributing to this uncertainty are 1) an incomplete understanding of historical habitat connectivity and resulting spatial structure of various breeding groups, particularly in the larger watersheds such as the Eel and Russian rivers, where plausible structures range from one or two large populations to multiple smaller populations occupying different subwatersheds; and 2) the scarcity of historical evidence of Chinook salmon in watersheds in Mendocino and Sonoma counties, which leads to some uncertainty about whether these populations functioned as independent units[27]. In the absence of definitive information, population designations were based primarily on predictions from our IP model and connectivity-viability analysis (Bjorkstedt et al. 2005). Table 8 presents proposed density-based criteria for these populations and the estimated population abundances (rounded to the nearest 100 spawners) that would result if density criteria were met under both historical (pre-dam) and current (post-dam) conditions. As before, high-risk abundance values indicate thresholds below which depensation is likely under both historical and current conditions. Low-risk estimates based on historically accessible habitat provide preliminary abundance targets that, if consistently exceeded, we believe would lead to a high probability of persistence over a 100-year time frame and the population fulfilling its historical role in ESU viability.

Comparison of historical versus current *IPkm* indicates that Chinook salmon in two populations, the Upper Eel River and Russian River populations, have lost access to appreciable amounts of habitat due to impassible dams. Scott Dam in the upper Eel River results in an estimated 11% loss of potential habitat. In the Russian River, a 15% reduction in potential habitat is attributed to dams, with Warm Springs and Coyote dams accounting for most of those losses.

[26] Evidence of historical occurrence is lacking for three of the six proposed spring-run populations (Redwood Creek, Van Duzen River, and the Upper Eel River). These populations were assumed to have existed based on environmental similarities between the upper portions of these watersheds and those believed to have supported spring Chinook, as well as by the historical occurrence of summer steelhead, which share similar oversummering habitat requirements (Bjorkstedt et al. 2005).

[27] The paucity of historical evidence of Chinook salmon in rivers of Mendocino and northern Sonoma counties may in part reflect the fact that by the late 1800s, substantial alteration to streams had already taken place as a result of logging activities. These activities included not only the harvest of redwoods forests, but also the transport of logs downstream through use of splash dams and log drives (see e.g., Jackson 1991; Downie et al. 2006). These activities undoubtedly had tremendous impact on habitat suitability for Chinook salmon, which spawn primarily in mainstems and larger tributaries where log drives occurred repeatedly.

Evaluation of current population viability

Fall-run populations

Currently available data are insufficient to rigorously evaluate the current viability of any of the fifteen putative independent populations of fall-run Chinook salmon in the CC-ESU using the proposed criteria. There are no population-level abundance estimates for any populations within the ESU that meet the minimum requirements for application of viability criteria outlined in Table 4. For certain populations, ancillary data are available, but in few cases do they allow for risk categorization. These data are reviewed below.

In the Redwood Creek watershed, spawner surveys have been conducted over approximately 17 km of Prairie Creek and its tributaries since the 1998-1999 spawning season. Population estimates for the surveyed reaches have averaged 342 (range 106-531) over six years (Walt Duffy and Steve Gough, Humboldt State University, unpublished data). However, there is no information on Chinook abundance in the mainstem of Redwood Creek or its other tributaries, which have been substantially more influenced by land-use practices. Spawner surveys have been conducted annually since the early 1980s on a 2 mi reach of Canon Creek, tributary to the Mad River (PFMC 2007). Maximum live-dead counts (including jacks) have ranged from 0 to 514 (mean = 107); however, because these surveys cover only a small portion of the available habitat and are variable from year to year in frequency, they cannot be used to derive population-level estimates of abundance or trends. Data from spawner surveys in index reaches of Tomki and Sprowl creeks in the upper Eel River are also available since the late 1970s (PFMC 2007). At Tomki Creek, maximum live-dead counts have ranged from 0 to 2,187 (mean = 244), though the average over the last twelve years has declined to 144 spawners. For Sprowl Creek, maximum live-dead counts over 4.5 mi of stream have ranged from 3 to 3,666 (mean = 741) since the late 1970s; however, over the last twelve years, counts have averaged only 68 spawners. In both these case, the estimates are most appropriately viewed as “floors” of abundance, and inconsistencies among years preclude their use as a reliable indicator of trend. Chinook salmon counts are also made at the Van Arsdale Fish Station in the upper mainstem Eel River, but these are similarly inappropriate for estimating population-level abundance (Good et al. 2005). A weir on Freshwater Creek has provided a reasonable census of adult Chinook counts for the period 1994-2004 (Good et al. 2005), with abundance averaging about 54 fish from 1994 to 2003. However, because Freshwater Creek represents only one of four Chinook-bearing streams within the putative Humboldt Bay independent population, we deem the data insufficient for assessing status at the population level. For both Bear River and Little River populations, we know of no current datasets of adult abundance. For these reasons, we categorized the Redwood Creek, Mad River, Humboldt Bay, Eel River, Little River, and Bear River populations as data deficient (Table 9).

Table 8. Projected population abundances (N_a) of CC-Chinook Salmon independent populations corresponding to a high-risk (depensation) threshold of 1 spawner/*IPkm* and low-risk (spatial structure/diversity=SSD) thresholds based on application of spawner density criteria (see Figure 5). Values listed under "historical" represent criteria applied to the historical landscape in the absence of dams that block access to anadromous fish. Values listed under "current" exclude areas upstream from impassible dams.

				High Risk		Low Risk			
				Historical	Current	Historical SSD		Current SSD	
Population	Historical *IPkm*	Current *IPkm*	*IPkm* Lost	Depens. N_a	Depens. N_a	Density Spawner/*IPkm*	N_a	Density Spawner/*IPkm*	N_a
Fall-run populations									
Redwood Creek (H)	116.1	116.1	0%	116	116	29.3	3400	29.3	3400
Little River (H)	18.6	18.6	0%	19	19	40.0	700	40.0	700
Mad River	94.0	94.0	0%	94	94	31.8	3000	31.8	3000
Humboldt Bay	76.7	76.7	0%	77	77	33.7	2600	33.7	2600
Lower Eel River	514.9	514.9	0%	515	515	20.0	10300	20.0	10300
Upper Eel River	555.9	495.3	11%	556	495	20.0	11100	20.0	9900
Bear River	39.4	39.4	0%	39	39	37.8	1500	37.8	1500
Mattole River	177.5	177.5	0%	178	178	22.5	4000	22.5	4000
Ten Mile River	67.2	67.2	0%	67	67	34.8	2300	34.8	2300
Noyo River	62.2	62.2	0%	62	62	35.3	2200	35.3	2200
Big River	104.3	104.3	0%	104	104	30.6	3200	30.6	3200
Navarro River	131.5	131.5	0%	131	131	27.6	3600	27.6	3600
Garcia River	56.2	56.2	0%	56	56	36.0	2000	36.0	2000
Gualala River	175.6	175.6	0%	176	176	22.7	4000	22.7	4000
Russian River	584.2	496.4	15%	584	496	20.0	11700	20.0	9900
Spring-run populations									
(Redwood Creek (H))	116.1	116.1	0%	*	*	*	*	*	*
Mad River	94.0	94.0	0%	*	*	*	*	*	*
(Van Duzen River)	109.5	109.5	0%	*	*	*	*	*	*
North Fk Eel River	76.8	76.8	0%	*	*	*	*	*	*
Middle Fk Eel River	188.5	188.5	0%	*	*	*	*	*	*
Upper Eel River	89.1	29.3	67%	*	*	*	*	*	*

* Density criteria are not applied to spring-run Chinook salmon; availability of oversummering pools for adults are more likely to limit abundance than IP-based predictions of spawning habitat. IP values for fall Chinook are presented for spring Chinook populations soley to provide a rough index of the percentage of habitat that lies upstream of dams.

The Mattole Salmon Group has conducted spawner and redd surveys on the Mattole River and its tributaries since 1994. Local experts have used these surveys and ancillary data to develop a rough "index" estimates of spawner escapement to the Mattole River; however, sampling intensity and spatial extent of surveys have varied from year to year, which makes them unsuitable for rigorous estimates of abundance or trend (MSG 2005; Good et al. 2005). The redd counts, which provide the best indicator of escapement, have ranged from 27 to 88 during the ten years of surveys. Based on the these data, we conclude that the population is likely at elevated risk of extinction but are unable to assess whether the population is at moderate or high risk of extinction (Table 9).

The status of Chinook salmon in coastal watersheds of the Mendocino and northern Sonoma counties, from the Ten Mile River to the Gualala River, is highly uncertain. To our knowledge, recent documented occurrences are limited to observations of a few adult spawners in the Ten Mile River during the mid-1990s (Maahs 1996)[28] and collection of juvenile Chinook salmon in downstream migrant traps located on the Noyo River (Gallagher 2001). Additionally, adult Chinook salmon are occasionally observed in the Noyo River during spawner surveys or at the Noyo Egg Collecting Station, and a single adult was observed in the Navarro River in the 2006–2007 spawning season (Scott Harris, California Department of Fish and Game, Willits, pers. comm.). Bell (2003) reports that Chinook salmon in the Garcia River are extinct. We know of no recent documented occurrences of Chinook salmon in the Big River or Gualala River basins, though anecdotal reports from fisherman suggest that Chinook salmon occasionally visit these watersheds. Based on this limited information, the TRT suspects that these six independent populations of Chinook salmon from Ten Mile River to the Gualala River are at least at high risk of extinction and in some cases may be extinct (Table 9). We chose to categorize them as high-risk (rather than extinct) because of the lack of spawner surveys conducted on mainstem portions of these rivers, where spawning by Chinook is most likely to occur.

Spawner surveys were initiated in the Russian River in 2000, and video monitoring at two fish ladders located at the Mirabel Inflatable Dam has provided counts of Chinook adults since 2002. Although the time series does not meet our minimum criteria for duration (four generations) and does not represent a full count (some adults spawn lower in the basin, and the dam is typically deflated in December when flows get too high), the data do suggest the Chinook run has been substantial in recent years. Chinook counts have averaged more than 3,600 fish (range 1,383 to 6,103) over the last six years (Cook 2005,

[28] Maahs (1996) estimated the total number of adult spawners in the Ten Mile River to be fewer than 10 in the 1995-1996 spawning season.

Table 9. Current viability of CC-Chinook salmon independent populations based on metrics outlined in Tables 1 and 4. na indicates data of sufficient quality to estimate the population metric are not available. In some cases, risk categories have been designated for populations where ancillary data strongly suggest populations are extinct or nearly so, despite the lack of quantitative estimates of any of the viability metrics. Metrics for which we believe ancillary data support the assigned risk category are denoted with asterisks. See text for justification of risk rankings.

Population Name	PVA result	Effect. pop. size per generation	Tot. pop. size per generation	Population decline		Catastrophe	Density		Hatchery	Risk Category
		$\overline{N}_e$	$\overline{N}_{g(harm)}$	$\overline{N}_{a(geo)}$	$\hat{T}$	$\hat{C}$	$\hat{D}_{dep}$	$\hat{D}_{ssd}$		
<u>*Fall-run populations*</u>										
Redwood Creek (H)	na	na	na	na	na	na	na	na	na	Data deficient
Little River (H)	na	na	na	na	na	na	na	na	na	Data deficient
Mad River	na	na	na	na	na	na	na	na	na	Data deficient
Humboldt Bay	na	na	na	na	na	na	na	na	na	Data deficient
Lower Eel River	na	na	na	na	na	na	na	na	na	Data deficient
Upper Eel River	na	na	na	na	na	na	na	na	na	Data deficient
Bear River	na	na	na	na	na	na	na	na	na	Data deficient
Mattole River	na	na*	na*	na*	na	na	na	na*	na	Moderate/High
Ten Mile River	na	na*	na*	na*	na	na	na	na*	na	High
Noyo River	na	na*	na*	na*	na	na	na	na*	na	High
Big River	na	na*	na*	na*	na	na	na	na*	na	High
Navarro River	na	na*	na*	na*	na	na	na	na*	na	High
Garcia River	na	na*	na*	na*	na	na	na	na*	na	High
Gualala River	na	na*	na*	na*	na	na	na	na*	na	High
Russian River	na	na	na	na	na	na	na	na	na	Data deficient
<u>*Spring-run populations*</u>										
(Redwood Creek (H))	-	-	-	-	-	-	-	-	-	Extinct
Mad River [5]	-	-	-	-	-	-	-	-	-	Extinct
(Van Duzen River)	-	-	-	-	-	-	-	-	-	Extinct
North Fk Eel River	-	-	-	-	-	-	-	-	-	Extinct
Middle Fk Eel River	-	-	-	-	-	-	-	-	-	Extinct
Upper Eel River	-	-	-	-	-	-	-	-	-	Extinct

2006). Were such patterns to continue, the population would likely meet most low-risk viability thresholds for all criteria except perhaps the density criterion.

Spring-run populations

All six spring-run independent populations of Chinook salmon in the CC-Chinook ESU are believed extinct.

ESU Viability

The complete lack of population-level information on the distribution and abundance of Chinook salmon throughout the CC-Chinook salmon ESU precludes application of the ESU-level viability criteria (Table 9). Most available information consists of spawning surveys in index reaches, for which the limited and non-random spatial extent, coupled with variation in survey frequency, render the data inappropriate for assessing population abundance or trend. Though more rigorous sampling has been conducted on Prairie Creek (tributary to Redwood Creek) and Freshwater Creek, in both cases the estimates represent only a portion the total population. Monitoring of spawning Chinook salmon in the Russian River has improved considerably in the last 5–6 years; however, this time series is not sufficiently long to assess trends.

With data limitations in mind, we identify several areas of significant concern as they relate to viability of the CC-Chinook salmon ESU. The current distribution of extant populations includes several watersheds in Humboldt County including Redwood Creek, Little River, Mad River, Humboldt Bay, Eel River (with two populations), Bear River, and Mattole River, as well as some smaller watersheds such as Maple Creek, Jacoby Creek, and Salmon Creek. However, the lack of population data precludes us from determining whether there are viable independent populations of fall run Chinook in the North Coastal or North Mountain Interior strata. Additionally, spring Chinook salmon within the ESU are thought to be extinct, indicating loss of diversity within the ESU. Currently, there are no known extant and persistent populations between the Mattole River in Humoldt County and the Russian River in Sonoma County, a distance of approximately 200 km. Consequently, there appears to be no representation of the North-Central Coastal stratum, and connectivity between the Mattole River population and the Russian River population is likely substantially reduced from historical patterns. Because of the lack of population data, viability of the Russian River population is uncertain. However, even if the Russian River population is eventually deemed viable, the lack of other viable populations within the Central Coastal stratum places this stratum at greater risk due to catastrophic risks, such as disturbances to the mainstem Russian River where most spawning is believed to occur.

In summary, the lack of data from which to assess viability of extant populations in the northern part of the ESU, the apparent lack of extant populations, with the exception of the Russian River, in the southern half of the ESU, the loss of important life-history diversity (i.e. spring-run populations), and the substantial gaps in the distribution of Chinook salmon throughout the CC ESU strongly indicate that this ESU fails to meet low-risk criteria and is therefore at elevated risk of extinction. Our conclusion is qualitatively consistent with recently published NMFS status reviews (NMFS 1999; Good et al. 2005).

4.3 Northern California Steelhead

Population Viability

Summary of density-based criteria

Bjorkstedt et al. (2005) proposed that the NC-Steelhead ESU historically consisted of 41 independent populations of winter-run steelhead (19 functionally independent and 22 potentially independent[29]), and as many as 10 populations of summer steelhead (all functionally independent). Table 10 summarizes proposed density-based criteria for these populations and the projected population abundances (rounded to the nearest 100 spawners) that would result if density criteria were met under both historical (pre-dam) and current (post-dam) conditions. High-risk abundance values indicate thresholds below which depensation is likely, and low-risk abundance values for historical conditions represent preliminary abundance targets that, if consistently exceeded, would likely lead to a high probability of persistence over a 100-year time frame and result in a population likely fulfilling its role in ESU viability.

Comparison of historical versus currently available *IPkm* indicates that two steelhead populations, the Mad River population and the Upper Mainstem Eel River population, have lost substantial habitat due to dams. In the Mad River, an estimated 36% of potential steelhead habitat lies above Ruth Dam, though a partial barrier well downstream of Ruth Dam may limit use of the upper watershed by steelhead in some years. For the upper mainstem Eel River, the Scott Dam blocks access to more than 99% of available habitat upstream of Soda Creek. The remaining 2.7 *IPkm* of habitat is insufficient to support a viable population, though the IP model predicts that this population once may have joined the South Fork Eel, North Fork Eel, Middle Fork Eel, and Van Duzen populations as the largest populations in the watershed. Outlet Creek has dams that block access to about 7% of historical potential habitat. Habitat loss attributable to dams is 1% or less for all other populations (Table 10).

[29] The TRT has since added one more potentially independent population, Soda Creek in the upper Eel River. See Appendix A.

Table 10. Projected population abundances (N_a) of NC-Steelhead independent populations corresponding to a high-risk (depensation) threshold of 1 spawner/*IPkm* and low-risk (spatial structure/diversity=SSD) thresholds based on application of spawner density criteria (see Figure 5). Values listed under "historical" represent criteria applied to the historical landscape in the absence of dams that block access to anadromous fish. Values listed under "current" exclude areas upstream from impassible dams. The IP-bias index is a qualitve measure of possible hydrologic bias in the IP model that could potentially lead to overprediction of historical habitat for juvenile coho salmon (Bjorkstedt et al. 2005).

					High Risk		Low Risk			
					Historical	Current	Historical SSD		Current SSD	
Population	Historical *IPkm*	Current *IPkm*	*IPkm* lost	IP bias index	Depens. N_a	Depens. N_a	Density Spawner/*IPkm*	N_a	Density Spawner/*IPkm*	N_a
Redwood Creek (H)	301.1	301.1	0%	low	301	301	20.0	6000	20.0	6000
Maple Creek/Big Lagoon	94.7	94.7	0%	low	95	95	29.1	2800	29.1	2800
Little River (H)	76.2	76.2	0%	low	76	76	31.6	2400	31.6	2400
Mad River	553.2	351.8	36%	low	553	352	20.0	11200	20.0	7000
Humboldt Bay	283.0	283.0	0%	low	283	283	20.0	5700	20.0	5700
Eel River - Full										
Price Creek	20.6	20.6	0%	low	21	21	39.4	800	39.4	800
Van Duzen River	363.8	363.8	0%	low	364	364	20.0	7300	20.0	7300
Larabee Creek	101.0	101.0	0%	low	101	101	28.2	2800	28.2	2800
South Fork Eel River	1182.1	1182.1	0%	low	1182	1182	20.0	23600	20.0	23600
Dobbyn Creek	52.5	52.5	0%	low	52	52	34.9	1800	34.9	1800
Jewett Creek	18.2	18.2	0%	low	18	18	39.7	700	39.7	700
Pipe Creek	18.2	18.2	0%	low	18	18	39.7	700	39.7	700
Kekawaka Creek	35.3	35.3	0%	low	35	35	37.3	1300	37.3	1300
Chamise Creek	38.0	38.0	0%	low	38	38	37.0	1400	37.0	1400
North Fork Eel River	372.8	372.8	0%	low	373	373	20.0	7500	20.0	7500
Bell Springs Creek	18.5	18.5	0%	moderate	19	19	39.6	700	39.6	700
Woodman Creek	39.4	39.4	0%	moderate	39	39	36.7	1400	36.7	1400
Outlet Creek	313.8	292.9	7%	moderate	314	293	20.0	6300	20.0	5900
Tomki Creek	131.7	131.7	0%	moderate	132	132	23.9	3200	23.9	3200
Middle Fork Eel River	584.3	581.4	0%	low	584	581	20.0	11700	20.0	11600
Bucknell Creek	21.1	21.1	0%	moderate	21	21	39.3	800	39.3	800
Soda Creek	17.6	17.6	0%	moderate	18	18	39.8	700	39.8	700
Upper Mainstem Eel River	387.3	2.7	99%	moderate	387	3	20.0	7700	-	-
Bear River	114.8	114.8	0%	low	116	116	26.1	3000	26.1	3000
Mattole River	613.9	613.9	0%	low	614	614	20.0	12300	20.0	12300
Usal Creek	19.0	19.0	0%	low	19	19	39.6	700	39.6	700
Cottaneva Creek	26.1	26.1	0%	low	26	26	38.6	1000	38.6	1000

Table 10. (continued)

					High Risk		Low Risk			
					Historical	Current	Historical SSD		Current SSD	
Population	**Historical *IPkm***	**Current *IPkm***	**IP-lost**	**IP bias index**	**Depens. N_a**	**Depens. N_a**	**Density Spawner/*IPkm***	**N_a**	**Density Spawner/*IPkm***	**N_a**
Wages Creek	19.9	19.9	0%	low	20	20	39.5	800	39.5	800
Ten Mile River	204.7	204.7	0%	moderate	205	205	20.0	4100	20.0	4100
Pudding Creek	32.0	32.0	0%	moderate	32	32	37.8	1200	37.8	1200
Noyo River	199.1	196.7	1%	moderate	199	197	20.0	4000	20.0	3900
Hare Creek	18.1	18.1	0%	moderate	18	18	39.7	700	39.7	700
Caspar Creek	16.0	16.0	0%	moderate	16	16	40.0	600	40.0	600
Russian Gulch (Me)	19.2	19.2	0%	moderate	19	19	39.6	800	39.6	800
Big River	316.6	312.9	1%	high	317	313	20.0	6300	20.0	6300
Albion River	77.1	77.1	0%	high	77	77	31.5	2400	31.5	2400
Big Salmon Creek	24.8	24.8	0%	high	25	25	38.8	1000	38.8	1000
Navarro River	458.2	457.9	0%	high	458	458	20.0	9200	20.0	9200
Elk Creek	24.3	24.3	0%	high	24	24	38.9	900	38.9	900
Brush Creek	28.3	28.3	0%	high	28	28	38.3	1100	38.3	1100
Garcia River	169.0	169.0	0%	high	169	169	20.0	3400	20.0	3400
Gualala River	478.0	476.3	0%	high	478	476	20.0	9600	20.0	9500

Evaluation of current viability

Winter-run populations

Currently available data are insufficient to rigorously evaluate the current viability of any of the 42 independent populations of winter steelhead in the NC-steelhead DPS using our viability criteria. Perhaps the best available time series of adult spawner abundance comes from Freshwater Creek, one of several streams that collectively make up the Humboldt Bay independent population. The Humboldt Fish Action Council has operated a weir on Freshwater Creek since the 1994–1995 season, and annual adult steelhead counts during this period have averaged about 73 adults (Seth Ricker, CDFG, Arcata, unpublished data). Within the last four years, mark-recapture studies have been conducted to derive escapements estimates for Freshwater Creek, and these have suggested that the weir has sampled from 38 to 74 percent of the upstream migrants. However, because the time series of escapement estimates of insufficient length to meet our criteria, and because the data represent only a portion of the Humboldt Bay population, which also includes Jacoby Creek, Elk River, and Salmon Creek (among others) we categorize the Humboldt population as data deficient (Table 11).

The Mattole Salmon Group conducts spawner surveys on the Mattole River; however, these surveys target Chinook and coho salmon, collecting only incidental data on winter steelhead (MSG 2005). On the Mendocino Coast, CDFG began monitoring steelhead in four independent populations (Pudding Creek, Noyo River, Hare Creek and Caspar Creek), as well as one dependent population (Little River) in 2000 and 2001. Estimated ranges of abundance for these streams over a three-to-six year period are as follows: Noyo River 186-364, Pudding Creek 76-265, Hare Creek 52-99, Caspar Creek 26-145, and Little River 16-34, (Gallagher and Wright 2007)[30]. Although the time series of abundances are not sufficiently long to meet our criteria, in all cases, the recent abundance ranges fall well below low-risk targets for spawner density (Table 10), suggesting that if the current patterns hold for two to three more generations, all of these populations would be considered at least at moderate risk. Thus, we classified these populations as such.

Steelhead spawner surveys on the Gualala River were initiated in 2001 (DeHaven 2005). These surveys are conducted on approximately 29 km of habitat in the Wheatfield Fork of the Gualala River and thus do not allow for estimation of total population abundance in the Gualala River basin. Consequently, we categorize these populations as data deficient as well (Table 11).

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[30] Estimates based on live fish capture-recapture estimates (where available) or fish per redd estimates, per the recommendation of Sean Gallagher, CDFG, pers. comm.

Table 11. Current viability of NC-steelhead populations based on metrics outlined in Tables 1 and 4. na indicates data of sufficient quality to estimate the population metric are not available. In some cases, risk categories have been designated for populations where ancillary data strongly suggest populations are extinct or nearly so, despite the lack of quantitative estimates of any of the viability metrics. Metrics for which we believe ancillary data support the assigned risk category are denoted with asterisks. See text for justification of risk rankings.

Population	PVA result	Effect. pop. size per generation $\bar{N}_e$	Tot. pop. size per generation $\bar{N}_{g(harm)}$	Population decline $\bar{N}_{a(geo)}$	Population decline $\hat{T}$	Catastrophe $\hat{C}$	Density $\hat{D}_{dep}$	Density $\hat{D}_{ssd}$	Hatchery	Risk category
Winter-run populations										
Redwood Creek (H)	na	na	na	na	na	na	na	na	na	Data deficient
Maple Creek/Big Lagoon	na	na	na	na	na	na	na	na	na	Data deficient
Little River (H)	na	na	na	na	na	na	na	na	na	Data deficient
Mad River	na	na	na	na	na	na	na	na	na	Data deficient
Humboldt Bay	na	na	na	na	na	na	na	na	na	Data deficient
Eel River - Full										
Price Creek	na	na	na	na	na	na	na	na	na	Data deficient
Larabee Creek	na	na	na	na	na	na	na	na	na	Data deficient
Van Duzen River	na	na	na	na	na	na	na	na	na	Data deficient
South Fork Eel River	na	na	na	na	na	na	na	na	na	Data deficient
Dobbyn Creek	na	na	na	na	na	na	na	na	na	Data deficient
Jewett Creek	na	na	na	na	na	na	na	na	na	Data deficient
Pipe Creek	na	na	na	na	na	na	na	na	na	Data deficient
Kekawaka Creek	na	na	na	na	na	na	na	na	na	Data deficient
Chamise Creek	na	na	na	na	na	na	na	na	na	Data deficient
North Fork Eel River	na	na	na	na	na	na	na	na	na	Data deficient
Bell Springs Creek	na	na	na	na	na	na	na	na	na	Data deficient
Woodman Creek	na	na	na	na	na	na	na	na	na	Data deficient
Outlet Creek	na	na	na	na	na	na	na	na	na	Data deficient
Tomki Creek	na	na	na	na	na	na	na	na	na	Data deficient
Middle Fork Eel River	na	na	na	na	na	na	na	na	na	Data deficient
Bucknell Creek	na	na	na*	na	na	na	na*	na	na*	Moderate/High
Soda Creek	na	na	na*	na	na	na	na*	na	na*	Moderate/High
Upper Mainstem Eel River	na	na	na*	na	na	na	na*	na	na*	High
Bear River	na	na	na	na	na	na	na	na	na	Data deficient
Mattole River	na	na	na	na	na	na	na	na	na	Data deficient

Table 11. (continued)

Population	PVA result	Effect. pop. size per generation $\bar{N}_e$	Tot. pop. size per generation $\bar{N}_{g(harm)}$	Population decline $\bar{N}_{a(geo)}$	Population decline $\hat{T}$	Catastrophe $\hat{C}$	Density $\hat{D}_{dep}$	Density $\hat{D}_{ssd}$	Hatchery	Risk Category
Usal Creek	na	na	na	na	na	na	na	na	na	Data deficient
Cottaneva Creek	na	na	na	na	na	na	na	na	na	Data deficient
Wages Creek	na	na	na	na	na	na	na	na	na	Data deficient
Ten Mile River	na	na	na	na	na	na	na	na	na	Data deficient
Pudding Creek	na	na	na	na	na	na	na	na*	na	Moderate
Noyo River	na	na	na	na	na	na	na	na*	na	Moderate
Hare Creek	na	na	na	na	na	na	na	na*	na	Moderate
Caspar Creek	na	na	na	na	na	na	na	na*	na	Moderate
Russian Gulch (Me)	na	na	na	na	na	na	na	na	na	Data deficient
Big River	na	na	na	na	na	na	na	na	na	Data deficient
Albion River	na	na	na	na	na	na	na	na	na	Data deficient
Big Salmon Creek	na	na	na	na	na	na	na	na	na	Data deficient
Navarro River	na	na	na	na	na	na	na	na	na	Data deficient
Elk Creek	na	na	na	na	na	na	na	na	na	Data deficient
Brush Creek	na	na	na	na	na	na	na	na	na	Data deficient
Garcia River	na	na	na	na	na	na	na	na	na	Data deficient
Gualala River	na	na	na	na	na	na	na	na	na	Data deficient
Summer-run populations										
Redwood Creek (H)	na	na	18 (high)	4.6 (high)	-0.04 (high)	0.86 (mod)	-	-	low	High
Mad River	na	na*	na*	na*	na*	na	-	-	na	Moderate
Van Duzen River	na	na	na	na	na	na	-	-	na	Data deficient
Larabee Creek	na	na	na	na	na	na	-	-	na	Data deficient
South Fork Eel River	na	na	na	na	na	na	-	-	na	Data deficient
North Fork Eel River	na	na	na	na	na	na	-	-	na	Data deficient
(Up. Mid. Mainstem Eel R)	na	na	na	na	na	na	-	-	na	Data deficient
Middle Fk Eel River	na	na	2333 (mod)	569 (low)	-0.01 (mod)	0.52 (low)	-	-	low	Moderate
(Upper Mainstem Eel R)	na	na	na	na	na	na	-	-	na	Data deficient
Mattole River	na	na*	na*	na*	na*	na	-	-	na	High

The only other time series of abundance for winter-run steelhead populations within this ESU is the count of hatchery and wild steelhead at Van Arsdale Fish Station on the upper Eel River. The counts of wild fish represent a composite of several delineated populations: Bucknell Creek, Soda Creek, the Upper Mainstem Eel River (the mainstem and tributaries upstream of Soda Creek), and various dependent populations between Van Arsdale station and Bucknell Creek. As such, the data cannot be used to evaluate any of these populations directly. However, annual counts of wild fish have averaged just under 200 fish over the last 11 years (Grass 2007). Thus, even if all fish were concentrated in Bucknell Creek, Soda Creek, or the Upper Mainstem Eel River, which does not appear to be the case (Scott Harris, CDFG, Willits, CA , pers. comm.), the abundances still would not be sufficient to meet low risk criteria (or moderate risk, in the case of the upper mainsteam Eel River) for effective population size or spawner density. Additionally, in eight of the last 11 years, there has been a substantial hatchery influence, with hatchery fish outnumbering wild fish by almost 10 to 1 during those years (Grass 1997-2007). For these reasons, we conclude that the Bucknell Creek, Soda Creek, and Upper Mainstem Eel River populations of winter steelhead are at least at moderate risk and probably at high risk of extinction.

Summer-run populations

Data on the abundance of summer-run steelhead are more readily available due to the fact that adults congregate in “resting pools” during the summer and can be observed when water is relatively clear. Currently, there are four ongoing efforts to estimate populations of summer steelhead in rivers within the NC-steelhead DPS: Redwood Creek, Mad River, Middle Fork Eel River, and Mattole River.

Summer dive surveys covering almost the entire mainstem of Redwood Creek have been conducted annually since 1981. There is some question about the reliability of some of the early counts, and it is unclear about how much summer steelhead may use tributaries to Redwood Creek for holding. However, recent abundance estimates in the mainstem clearly indicate a population that is at very high risk of extinction. Mean adult abundance has averaged only 6 fish over the past four generations, and although the recent trend over the last four generations has been just slightly negative ($T = -0.021$), the overall trend for the entire period of record has continued downward ($T = -0.046$)(Dave Anderson, Redwood National and State Parks, Crescent City, unpublished data). Effective population size is estimated to be at just 3.6 fish. Consequently, we conclude this population is at high risk of extinction (Table 11).

Diver counts of summer steelhead have been conducted on portions of the Mad River since 1982. From 1982 to 2002, the Forest Service conducted surveys on the reach from Ruth Dam to Deer Creek; however, that effort was terminated due to budget constraints. Since 1994, Green Diamond Resource Company (formerly Simpson Timber Company) and the California Department of Fish and Game have surveyed the reaches from Deer Creek to Mad River Hatchery, and from the hatchery to Cadle Hole, respectively. Although the data do not meet the minimum requirements to formally assess viability using our criteria, they do provide some indications of population status. For the period from 1994 to 2002, the period where all three reaches were surveyed, geometric mean abundance was about 250 fish and the population has declined throughout the period. Hatchery fish constituted about 2% for the two generations covered during this period (Matt House, Green Diamond Resource Company, Korbel, unpublished data; Andrew Bundschuh, US. Forest Service, Six Rivers National Forest, Eureka, unpublished data). Based on these data, we conclude that the population is at least at moderate risk of extinction (Table 11).

The Middle Fork Eel River constitutes perhaps the only population within the entire recovery domain where the existing time series of adult abundance estimates meet requirements outlined in Table 4. Summer surveys of adults in summer resting pools have provided a reasonable census of the adult population size dating back to the 1960s. Counts have ranged from 198 to 1601 during that period (Jones 1980, 1992; Jones et al. 1980; and Scott Harris, California Department of Fish and Game, Willits, unpublished data). Calculation of extinction risk metrics, shown in Table 11, indicates that the population currently ranks at low risk of extinction according to the population decline criteria (but only marginally so) and for the catastrophe criteria. For the last four generations, the geometric mean abundance has been over the 500 fish threshold, but only by a small amount, and the trend suggests a slight decline in abundance (T = -0.010). However, over the entire period of record, the downward trend is more pronounced (T = -0.025). Continued decline at this rate would have it approaching an N_a of less than 500 within two generations. The population ranks at moderate risk according to the effective population size criteria. Hatcheries do not appear to play a significant role in the current viability of this population (summer steelhead are not released into the Middle Fork Eel, and we assume that straying of summer steelhead from the Mad River is negligible). Based on the moderate risk rankings for population decline and effective population size, we conclude that the population is at moderate risk of extinction (Table 11).

Finally, the Mattole Salmon Group has conducted summer diver surveys in the mainstem Mattole and two tributaries annually since 1996 (MSG 2005). Although the data set does not meet our minimum standards for evaluation using our criteria, it does suggest that the Mattole River population is at high risk of extinction, with an average adult count of just 16 individuals (range 9-30) during the period (Table 11).

Little is known about the status of the remaining six putative summer steelhead populations in the DPS (Van Duzen River, South Fork Eel River, Larabee Creek, North Fork Eel River, Upper Middle Mainstem

Eel River, and Upper Mainstem Eel. We categorize all of these populations as data deficient (Table 11), though we note that the lack of even anecdotal reports in recent years suggests that many if not all of these populations are either extirpated or extremely depressed.

ESU Viability

The complete lack of data with which to assess the status of any of the 42 independent populations of winter steelhead within the NC-Steelhead DPS (all deemed data deficient) precludes evaluation of ESU viability using the quantitative criteria developed in this paper. For summer steelhead, the limited available data provide no evidence of viable summer steelhead populations within the ESU. Consequently, it is highly likely that representation and redundancy/connectivity criteria are not being met and that the DPS is at elevated risk of extinction. Good et al. (2005) reaffirmed the conclusion of Busby et al. (1996) that the ESU was likely to become endangered in the foreseeable future, the lack of population information being cited as a contributing risk factor. Our conclusion is consistent with their assessments.

4.4 Central California Coast Steelhead

Population Viability

Summary of density-based criteria

Bjorkstedt et al. (2005) proposed that the CCC-Steelhead ESU historically contained 11 functionally independent populations and 26 potentially independent populations. Table 12 presents proposed density-based criteria for these populations and the estimated population abundances (rounded to the nearest 100 spawners) that would result if density criteria were met under both historical (pre-dam) and current (post-dam) conditions. High-risk abundance values indicate thresholds below which depensation is likely, and low-risk estimates represent preliminary abundance targets that, if consistently exceeded, would likely lead to a high probability of persistence over a 100-year time frame and result in a population likely fulfilling its historical role with respect to ESU viability.

More so than any other ESU within the NCCC Recovery Domain, impassible dams have had a substantial effect on the available habitat of steelhead population in the CCC ESU. These effects are most pronounced for San Francisco Bay populations, Russian River populations, and coastal Marin County populations. Within San Francisco Bay, populations experiencing substantial reductions in accessible habitat include Novato Creek (22%), Napa River (17%), Walnut Creek (96%), San Pablo Creek (72%),

Table 12. Projected population abundances (N_a) of CCC-Steelhead independent populations corresponding to a high-risk (depensation) threshold of 1 spawner/*IPkm* and low-risk (spatial structure/diversity=SSD) thresholds based on application of spawner density criteria (see Figure 5). Values listed under "historical" represent criteria applied to the historical landscape in the absence of dams that block access to anadromous fish. Values listed under "current" exclude areas upstream from impassible dams. The IP-bias index is a qualitve measure of possible hydrologic bias in the IP model that could potentially lead to overprediction of historical habitat for juvenile coho salmon (Bjorkstedt et al. 2005).

					High Risk		Low Risk			
					Historical	Current	Historical SSD		Current SSD	
Population	**Historical *IPkm***	**Current *IPkm***	***IPkm* lost**	**IP bias index**	**Depens. N_a**	**Depens. N_a**	**Density Spawner/*IPkm***	**N_a**	**Density Spawner/*IPkm***	**Div/SS N_a**
Russian River	2348.8									
Austin Creek	111.9	111.9	0%	high	112	112	26.7	3000	26.7	3000
Green Valley Creek	61.7	61.3	1%	high	62	61	33.7	2100	33.7	2100
Mark West Creek	366.5	340.8	7%	high	367	341	20.0	7300	20.0	6800
Dry Creek	384.9	167.7	56%	high	385	168	20.0	7700	20.0	3400
Maacama Creek	106.9	105.2	2%	high	107	105	27.4	2900	27.6	2900
Upper Russian River	892.3	703.5	21%	high	892	704	20.0	17800	20.0	14100
Salmon Creek (S)	63.5	63.5	0%	high	63	63	33.4	2100	33.4	2100
Americano Creek	64.2	64.2	0%	high	64	64	33.3	2100	33.3	2100
Stemple Creek	73.1	73.1	0%	high	73	73	32.1	2300	32.1	2300
Tomales Bay										
Walker Creek	134.1	98.9	26%	high	134	99	23.6	3200	28.5	2800
Lagunitas Creek	170.7	87.2	49%	high	171	87	20.0	3400	30.1	2600
Northwest SF Bay										
Corte Madera Creek	41.3	41.3	0%	high	41	41	36.5	1500	36.5	1500
Miller Creek	44.4	44.4	0%	high	44	44	36.1	1600	36.1	1600
Novato Creek	78.6	61.5	22%	severe	79	62	31.3	2500	33.7	2100
North SF Bay										
Petaluma River	225.4	223.0	1%	severe	225	223	20.0	4500	20.0	4500
Sonoma Creek	268.7	268.7	0%	high	269	269	20.0	5400	20.0	5400
Napa River	593.9	491.0	17%	severe	594	491	20.0	11900	20.0	9800
Suisun Bay										
Green Val./Suisun Creek	164.0	162.2	1%	severe	164	162	20.0	3300	20.0	3200
Walnut Creek	202.2	7.5	96%	severe	202	8	20.0	4000	-	-
East SF Bay										
San Pablo Creek	67.9	18.8	72%	severe	68	19	32.8	2200	39.6	700
San Leandro Creek	80.5	16.0	80%	severe	81	16	31.0	2500	40.0	600
San Lorenzo Creek	79.8	41.5	48%	severe	80	42	31.1	2500	36.5	1500

Table 12. (continued)

Population	**Historical *IPkm***	**Current *IPkm***	***IPkm* Lost**	**IP bias index**	**High Risk Historical Depens. Na**	**Current Depens. N_a**	**Low Risk Historical SSD Density Spawner/*IPkm***	**N_a**	**Current SSD Density Spawner/*IPkm***	**Div/SS N_a**
Southeast SF Bay										
Alameda Creek	816.6	39.5	95%	severe	817	39	20.0	16300	36.7	1500
Coyote Creek	498.3	252.7	49%	severe	498	253	20.0	10000	20.0	5100
Southwest SF Bay										
Guadalupe River	157.3	124.5	21%	severe	157	125	20.4	3200	24.9	3100
Stevens Creek	39.6	18.4	54%	severe	40	18	36.7	1500	39.7	700
San Francisquito Creek	59.2	39.8	33%	severe	59	40	34.0	2000	36.7	1500
San Mateo Creek	57.6	9.9	83%	severe	58	10	34.2	2000	-	400
Pilarcitos Creek	41.9	30.6	27%	high	42	31	36.4	1500	38.0	1200
San Gregorio Creek	77.6	77.6	0%	high	78	78	31.4	2400	31.4	2400
Pescadero Creek	93.8	93.8	0%	high	94	94	29.2	2700	29.2	2700
Waddell Creek	16.5	16.5	0%	high	16	16	40.0	600	40.0	600
Scott Creek	23.5	23.5	0%	high	24	24	39.0	900	39.0	900
Laguna Creek	17.4	17.4	0%	high	17	17	39.8	700	39.8	700
San Lorenzo River	225.6	215.3	5%	high	225	215	20.0	4500	20.0	4300
Soquel Creek	66.4	66.4	0%	high	66	66	33.0	2200	33.0	2200
Aptos Creek	41.0	41.0	0%	high	41	41	36.5	1500	36.5	1500

San Leandro Creek (80%), San Lorenzo Creek (48%), Alameda Creek (95%), Coyote Creek (49%), Guadalupe River (21%), Stevens Creek (54%), San Francisquito Creek (33%), and San Mateo Creek (83%). In the Russian River basin, populations that have experienced significant reductions in habitat include the Upper Russian River (21%), Dry Creek (56%), and Mark West Creek (7%). In Lagunitas Creek, an estimated 49% of steelhead habitat lies upstream of Kent and Nicasio dams. In the Walker Creek drainage, 26% of the predicted habitat lies upstream of dams (Table 12).

Evaluation of current viability

The lack of data on spawner abundance for steelhead populations in the CCC-Steelhead ESU precludes a rigorous assessment of current viability for any of the 37 independent populations, and in only a few cases do ancillary data provide sufficient information to allow reasonable inference about population risk at the present time.

Spawner surveys have been conducted annually on Lagunitas Creek since 1994–1995 (Ettlinger et al. 2005). However, the primary purpose is to enumerate coho salmon, and surveys typically end before the steelhead spawning season is complete. Steelhead counts are made at the Noyo Egg Collecting station on the South Fork Noyo River; however, steelhead have little trouble passing over the weir, so the number passing through the counting facility is considered an unreliable indicator of total abundance (Scott Harris, CDFG, Willits, pers. comm.). Partial counts of steelhead are made at the Felton Diversion Dam on the San Lorenzo River; however, operation is inconsistent and no population estimates are made. Population estimates for Scott Creek based on weir counts and mark-recapture data have indicated that steelhead adults have numbered between 230 and 440 over the last four years, though about 34% of returning adults were hatchery fish (Sean Hayes, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). Should the current patterns of abundance and hatchery influence continue, the population would likely be classified as at moderate risk based on both density and hatchery criteria. To our knowledge, these efforts represent the only sources of information on adult abundance within the ESU, and there are few ancillary data from which to speculate about current status. Thus we classify the majority of coastal populations as data deficient (Table 13).

Likewise, within the San Francisco Bay region, there are no population-level estimates of adult abundance for any tributaries entering the Bay. However, Leidy et al. (2005b) recently completed a comprehensive review of available survey information on streams entering San Francisco Bay. For many streams, recent observations of *O. mykiss* indicate that they still persist in these watersheds. However, as noted above, several populations have been affected by dams that block access to the majority of their

Table 13. Current viability of CCC-steelhead populations based on metrics outlined in Tables 1 and 4. na indicates data of sufficient quality to estimate the population metric are not available. In some cases, risk categories have been designated for populations where ancillary data strongly suggest populations are extinct or nearly so, despite the lack of quantitative estimates of any of the viability metrics. Metrics for which we believe ancillary data support the assigned risk category are denoted with asterisks. See text for justification of risk rankings.

Population	PVA Result	Effect. pop. size per generation $\overline{N}_e$	Tot. pop. size per generation $\overline{N}_{g(harm)}$	Population decline $\overline{N}_{a(geo)}$	$\hat{T}$	Catastrophe $\hat{C}$	Spawner density $\hat{D}_{dep}$	$\hat{D}_{ssd}$	Hatchery	Risk Category
Russian River										
Austin Creek	na	na	na	na	na	na	na	na	na	Data deficient
Green Valley Creek	na	na	na	na	na	na	na	na	na	Data deficient
Mark West Creek	na	na	na	na	na	na	na	na	na	Data deficient
Dry Creek	na	na	na	na	na	na	na	na	na	Data deficient
Maacama Creek	na	na	na	na	na	na	na	na	na	Data deficient
Upper Russian River [H]	na	na	na	na	na	na	na	na	na	Data deficient
Salmon Creek (S)	na	na	na	na	na	na	na	na	na	Data deficient
Americano Creek	na	na	na	na	na	na	na	na	na	Data deficient
Stemple Creek	na	na	na	na	na	na	na	na	na	Data deficient
Tomales Bay										
Walker Creek	na	na	na	na	na	na	na	na	na	Data deficient
Lagunitas Creek	na	na	na	na	na	na	na	na	na	Data deficient
Northwest SF Bay										
Corte Madera Creek	na	na	na	na	na	na	na	na	na	Data deficient
Miller Creek	na	na	na	na	na	na	na	na	na	Data deficient
Novato Creek	na	na	na	na	na	na	na	na	na	Data deficient
North SF Bay										
Petaluma River	na	na	na	na	na	na	na	na	na	Data deficient
Sonoma Creek	na	na	na	na	na	na	na	na	na	Data deficient
Napa River	na	na	na	na	na	na	na	na	na	Data deficient
Suisun Bay										
Green Val./Suisun Creek	na	na	na	na	na	na	na	na	na	Data deficient
Walnut Creek	na	na*	na*	na*	na	na	na*	na*	na	High
East SF Bay										
San Pablo Creek	na	na*	na*	na*	na	na	na*	na*	na	High
San Leandro Creek	na	na*	na*	na*	na	na	na*	na*	na	High
San Lorenzo Creek	na	na*	na*	na*	na	na	na*	na*	na	High

Table 13. (continued)

Population	**PVA Result**	**Effect. pop. size per generation** $\overline{N}_e$	**Tot. pop. size per generation** $\overline{N}_{g(harm)}$	**Population decline** $\overline{N}_{a(geo)}$	$\hat{T}$	**Catastrophe** $\hat{C}$	**Spawner density** $\hat{D}_{dep}$	$\hat{D}_{ssd}$	**Hatchery**	**Risk Category**
Southeast SF Bay										
Alameda Creek	na	na*	na*	na*	na	na	na*	na*	na	High
Coyote Creek	na	na	na	na	na	na	na	na	na	Data deficient
Southwest SF Bay										
Guadalupe River	na	na	na	na	na	na	na	na	na	Data deficient
Stevens Creek	na	na	na	na	na	na	na	na	na	Data deficient
San Francisquito Creek	na	na	na	na	na	na	na	na	na	Data deficient
San Mateo Creek	na	na*	na*	na*	na	na	na*	na*	na	High
Pilarcitos Creek	na	na	na	na	na	na	na	na	na	Data deficient
San Gregorio Creek	na	na	na	na	na	na	na	na	na	Data deficient
Pescadero Creek	na	na	na	na	na	na	na	na	na	Data deficient
Waddell Creek	na	na	na	na	na	na	na	na	na	Data deficient
Scott Creek	na	na	na	na	na	na	na	na*	na*	Moderate?
Laguna Creek	na	na	na	na	na	na	na	na	na	Data deficient
San Lorenzo River	na	na	na	na	na	na	na	na	na	Data deficient
Soquel Creek	na	na	na	na	na	na	na	na	na	Data deficient
Aptos Creek	na	na	na	na	na	na	na	na	na	Data deficient

historical habitat, and areas below these dams are often severely impacted by urban development. In many cases, it is unclear whether the anadromous life history continues to be expressed downstream of these barriers, though resident *O. mykiss* remain present upstream (and sometimes downstream) of the dams. Based on information provided in Leidy et al. (2005b), we conclude that in six watersheds—Walnut Creek, San Pablo Creek, San Leandro Creek, San Lorenzo Creek, Alameda Creek, and San Mateo Creek—it is highly likely that, if steelhead still persist in these watersheds, they are at high risk of extinction. Steelhead appear to persist in most other functionally and potentially independent populations in the San Francisco Bay area, including Arroyo Corte Madera de Presidio, Novato Creek, Sonoma Creek, Napa River, Green Valley Creek, Coyote Creek, Guadalupe River, San Francisquito Creek, and possibly Corte Madera Creek, Miller Creek, and Petaluma River (Leidy et al. 2005b); however, data are limited to observations of occurrence. All of these populations are classified as data deficient, though some are likely at high risk or possibly even extinct (Table 13).

ESU Viability

Because of the extreme data limitations, we are unable to assess the status of the CCC-Steelhead DPS using the quantitative criteria outlined in this paper. All populations within North Coastal, Interior, and Santa Cruz Mountains strata were categorized as data deficient, as were many of the populations in the Coastal and Interior San Francisco Bay strata (Table 13). The presence of dams that block access to substantial amounts of historical habitat (particularly in the east and southeast portions of San Francisco Bay), coupled with ancillary data (see Leidy et al. 2005b) that suggest that it is highly unlikely that the Interior San Francisco Bay strata has any viable populations, or that redundancy criteria would be met. Elsewhere in the ESU, the lack of demonstrably viable populations remains a significant concern. Good et al. (2005) reaffirmed the conclusion of Busby et al. (1996) that the ESU was likely to become endangered in the foreseeable future, citing the lack of population information as a contributing risk factor. Our conclusion is consistent with their assessments.

4.5 Conclusions

In this report, we have developed a framework for assessing the viability of listed salmonid ESUs and DPSs within the NCCC Recovery Domain. Our framework follows the approach of Allendorf et al. (1997), proposing a set of general criteria by which the extinction risk of populations can be assessed. It then extends the Allendorf et al. (1997) approach, adding criteria that address population processes not explicitly addressed in the Allendorf et al. criteria, as well as criteria that consider processes occurring at higher levels of biological organization (i.e., diversity strata and ESU/DPS). The decision to use general criteria reflects, in part, the paucity of data that might allow development of models tailored specifically to individual populations. The use of general criteria or "rules of thumb" to assess extinction risk when data for developing credible population viability models are lacking has been advocated by Shaffer et al. (2002) and RSRP (2002).

We then attempted, albeit with limited success because of data limitations, to apply these criteria to four ESA-listed ESUs and DPSs within the NCCC Recovery Domain: Central California Coast Coho Salmon, California Coastal Chinook, Northern California Steelhead, and Central California Coast Steelhead. The vast majority of populations were categorized as data deficient, underscoring the critical need for development and implementation of a comprehensive monitoring plan for salmonid populations in the NCCC Recovery Domain. At a minimum, application of the proposed criteria requires estimates of population abundance for functionally and potentially independent populations within the domain that are identified in recovery plans as essential for ESU or DPS recovery, as well as information on the spatial distribution of individuals within these populations. Likewise, monitoring of trends in abundance or distribution are likely to be needed for key dependent populations that may serve as important populations for maintaining connectivity within and among strata. Historically, most monitoring programs in California targeting adult salmon and steelhead have been limited to index reaches and, as such, have not produced estimates at the population level. Without population-level estimates of abundance, assessment of risk using the proposed criteria (or any other criteria for that matter) is difficult.

The TRT fully recognizes that monitoring at a scale that would allow application of the proposed population and ESU criteria is very ambitious and would take an unprecedented (in California) commitment of effort and resources. Nevertheless, such efforts are not without precedent elsewhere. For example, the state of Oregon has developed and implemented a rigorously designed monitoring program that produces population estimates for almost all independent populations of coho salmon in the Oregon Coast ESU. This program evolved from an existing index-reach approach and has now produced time series of adult abundance dating back to the mid-1990s. In California, the California Department of Fish and Game (CDFG) has made progress in this direction through research designed to evaluate different approaches to estimating adult abundances of coho salmon and steelhead in five watersheds on the Mendocino Coast (Gallagher and Wright 2007). Such programs, if continued, will likely produce estimates sufficient to allow evaluation of population metrics proposed in this report. One ongoing CDFG monitoring program for summer steelhead in the Middle Fork Eel River provides the longest ongoing time series of adult abundance anywhere in the NCCC Recovery Domain. Additionally, there are a number of recently initiated monitoring efforts conducted by various agencies that, with refinement, can produce population-level estimates of abundance for several salmonid populations in various watersheds (e.g., Lagunitas Creek coho salmon; Scott Creek coho salmon and steelhead; Russian River Chinook salmon), and others efforts that, if augmented with additional sampling, could produce similar estimates for other populations (e.g., Gualala River steelhead, Freshwater Creek steelhead, coho salmon and Chinook salmon; Redwood Creek, Mad River, and Mattole River summer steelhead). Clearly though, comparable efforts will need to be made for many currently unmonitored populations for our criteria to be applied across ESUs or DPSs.

In addition to time series of adult abundance, information on freshwater and marine survival rates of a representative set of populations for each species is essential for ascertaining whether observed trends in abundance indicate improvement in freshwater habitat conditions or merely reflect variation in marine survival. There have been recent efforts to establish life-cycle monitoring stations to begin answering these questions (e.g., Scott Creek, Freshwater Creek, and two Mendocino Coast streams). More sophisticated viability models that would account for population-specific differences in vital rates (and therefore potentially improve on the general criteria proposed here) will have even greater data requirements. It is thus imperative that California conducts monitoring at spatial scales relevant to recovery planning in order to accurately evaluating status and progress toward recovery. A more thorough discussion of research and monitoring needs for populations in the NCCC Recovery Domain will be forthcoming in a third report being prepared by the TRT.

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Appendix A. Revisions to NCCC Population Structure Report

Introduction

The hypothesized historical population structure for two listed Evolutionarily Significant Units (ESUs) of salmon and two listed Distinct Population Segments (DPSs) of steelhead within the NCCC Recovery domain was described in detail in Bjorkstedt et al. (2005). Following publication of this report, the Technical Recovery Team discovered several errors and inconsistencies in the document that require some modification to our assessment of historical population structure. This appendix presents corrected summaries of population structure for each of the four ESUs and DPSs within the recovery domain. These revised summaries supercede previously published tables and figures and should be used as the basis for further recovery planning efforts.

Most of the errors in the *Population Structure Report* involved inconsistencies among the text, tables, and figures for each ESU with respect to population classifications (i.e., functionally independent, potentially independent, and dependent) or placement of populations into diversity strata. Minor adjustment to *IPkm* for some populations were made after these values were recalculated for all populations. These errors have been corrected in the summary tables and figures that follow. Additionally, we found two instances where historically accessible habitat above dams was not included in our estimates of *IPkm*, and several other instances where we have discovered long-standing barriers that likely prevented access to stream reaches that were assigned positive IP values. In these cases, we have since corrected estimates of *IPkm* for these populations and re-estimated self-recruitment values for each of the populations. In most cases, these changes have had a relatively minor influence on our overall conclusions, though in a few instances populations have been downgraded from potentially independent to dependent.

In addition to correcting these errors, the TRT has also revised the diversity strata for the four ESUs and DPS within the domain. In a few cases, these revisions involve minor adjustments of diversity strata boundaries to better reflect environmental similarities and differences, as well as to foster consistency in diversity strata boundaries among species. More significantly, we have restructured diversity strata for the CC-Chinook salmon ESU with respect to the treatment of fall versus spring runs and the NC-steelhead DPS with respect to summer and winter runs. These modifications are intended to more accurately represent the evolutionary history of different life-history types within each watershed. Finally, the CCC-Steelhead DPS boundary was recently modified by NMFS (71 FR 834-862) to include tributaries to Suisun Bay and Carquinez Strait; we have added a small number of populations to reflect these changes.

Central California Coast Coho Salmon Diversity Strata

Revisions to the Central California Coast coho salmon diversity strata were minor. Upon further examination of environmental data, the TRT felt that it was more appropriate to group the Gualala River population with populations to the north, including the Navarro River and Garcia River independent populations. These three basins fall within the Coast Range ecoregion, share similar geologies, and have comparable precipitation and temperature patterns. These similarities appear stronger than those between the Gualala River basin and basins farther to the south including the Russian River and smaller basins in coastal regions of southern Sonoma and northern Marin counties. Furthermore, the TRT feels that the stretch of coastline between Gualala Point and the mouth of the Russian River, which is characterized by very small watersheds few of which contain habitat that appears suitable to coho salmon, constitutes a more meaningful geographic break (i.e., potential migration barrier) than that of Point Arena. The realignment of the Gualala River required us to change the names of diversity strata to accurately reflect natural geographic breaks that define the strata. The historical population status of coho populations within the ESU is presented in Table A.1, and the placement of populations with respect to diversity strata is shown in Figure A.1 and Plate A.1.

Table A.1. Historical population structure of coho salmon in the CCC-Coho ESU. Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005). Values in parentheses are *IPkm* totals without the 21.5ºC. temperature mask. This table supercedes Table 2.2 in Bjorkstedt et al. (2005).

Population	*IPkm*	IP Bias Index	Self-Recruitment*	Historical Population Status
Jackass Creek [b]	4.3	low	0.851	dependent
Usal Creek [17]	10.6**	low	0.911	dependent
Cottaneva Creek [18]	13.8	low	0.910	dependent
Juan Creek [20]	6.0	low	0.871	dependent
Howard Creek [c]	3.3	moderate	0.817	dependent
DeHaven Creek [21]	5.7	moderate	0.919	dependent
Wages Creek [22]	10.0	low	0.897	dependent
Abalobadiah Creek [c]	4.1	low	0.614	dependent
Ten Mile River [23]	105.1	moderate	0.990	**Functionally Independent**
Mill Creek [c]	4.7	low	0.618	dependent
Pudding Creek [24]	28.9	moderate	0.936	dependent
Noyo River [25]	119.3	moderate	0.990	**Functionally Independent**
Hare Creek [26]	12.4	moderate	0.879	dependent
Jug Handle Creek [c]	4.8	moderate	0.705	dependent
Caspar Creek [27]	12.8	moderate	0.883	dependent
Russian Gulch (Me) [28]	6.4	moderate	0.727	dependent
Big River [30]	193.7 (194.8)	high	0.992	**Functionally Independent**
Little River (M) [31]	6.5	moderate	0.667	dependent
Albion River [32]	59.2	high	0.964	**Functionally Independent**
Big Salmon Creek [33]	17.0	high	0.926	dependent
Navarro River [34]	201.0 (232.5)	high	0.988	**Functionally Independent**
Greenwood Creek [35]	5.1**	high	0.633	dependent
Elk Creek [36]	9.9**	high	0.769	dependent
Mallo Pass Creek [c]	3.6	high	0.573	dependent
Alder Creek [37]	6.0**	high	0.796	dependent
Brush Creek [38]	18.0	high	0.921	dependent
Garcia River [39]	76.0 (105.3)	high	0.979	**Functionally Independent**
Point Arena Creek [d]	3.9	high	0.586	dependent
Schooner Gulch [40]	4.8	high	0.485	dependent
Gualala River [41]	252.2 (277.9)	high	0.976	**Functionally Independent**
Russian Gulch (S) [d]	6.02	moderate	0.219	dependent
Russian River [42]	779.4 (1662.0)	high	0.997	**Functionally Independent**
Scotty Creek [d]	3.8	high	0.333	dependent
Salmon Creek (S) [43]	47.6	high	0.893	dependent
Bodega Harbor [44]	11.7	high	0.672	dependent
Americano Creek [45]	60.6	high	0.938	dependent
Stemple Creek [46]	77.4	high	0.960	dependent
Tomales Bay [47]	234.5		0.969	
Walker Creek [TB1]	103.7	high		***Potentially Independent****
Lagunitas Creek [TB2]	137.0†	high		**Functionally Independent**
Drakes Bay [48]	8.0	high	0.468	dependent
Pine Gulch [49]	7.4	high	0.636	dependent
Redwood Creek (M) [50]	8.0	high	0.623	dependent

Table A.1. (continued)

Population	*IPkm*	IP Bias Index	Self-recruitment	Historical Population Status
San Francisco Bay [51]	339.2[††] (669.3)		0.996	
Arroyo Corte Madera del Presidio[S1]	10.6	high		dependent
Corte Madera Creek [S2]	35.2	high		dependent
Miller Creek [S3]	31.0	high		dependent
Novato Creek [S4]	74.0	severe		dependent
Petaluma River [S5]	233.0	severe		dependent
Sonoma Creek [S6]	227.1	high		dependent
Napa River [S7]	491.8 (500.0)	severe		dependent
San Pablo Creek [S8]	18.4	severe		dependent
Strawberry Creek [e]	4.9	severe		dependent
San Leandro Creek [S9]	21.6	severe		dependent
San Lorenzo Creek [S10]	58.9	severe		dependent
Alameda Creek [S11]	105.5 (435.6)	severe		dependent
Coyote Creek [S12]	182.8 (339.0)	severe		dependent
Guadalupe River [S13]	153.6	severe		dependent
Stevens Creek [S14]	23.3	severe		dependent
San Francisquito Creek [S15]	46.9	severe		dependent
San Mateo Creek [S16]	42.2	severe		dependent
Pilarcitos Creek [52]	31.8	high	0.818	dependent
Tunitas Creek [53]	8.3	high	0.762	dependent
San Gregorio Creek [54]	40.1	high	0.978	dependent
Pomponio Creek [55]	8.5	high	0.892	dependent
Pescadero Creek [56]	60.6	high	0.985	**Functionally Independent**
Arroyo de los Frijoles [e]	6.7	high	0.806	dependent
Gazos Creek [57]	8.2	high	0.887	dependent
Whitehouse Creek [e]	4.2	high	0.914	dependent
Cascade Creek [e]	4.2	high	0.820	dependent
Waddell Creek [58]	9.2	high	0.884	dependent
Scott Creek [59]	15.0	high	0.892	dependent
San Vicente Creek [60]	3.1	high		dependent
Wilder Creek [62]	4.9	high	0.647	dependent
San Lorenzo River [63]	135.3[†]	high	0.995	**Functionally Independent**
Soquel Creek [64]	33.0	high	0.962	dependent
Aptos Creek [65]	27.4	high	0.928	dependent

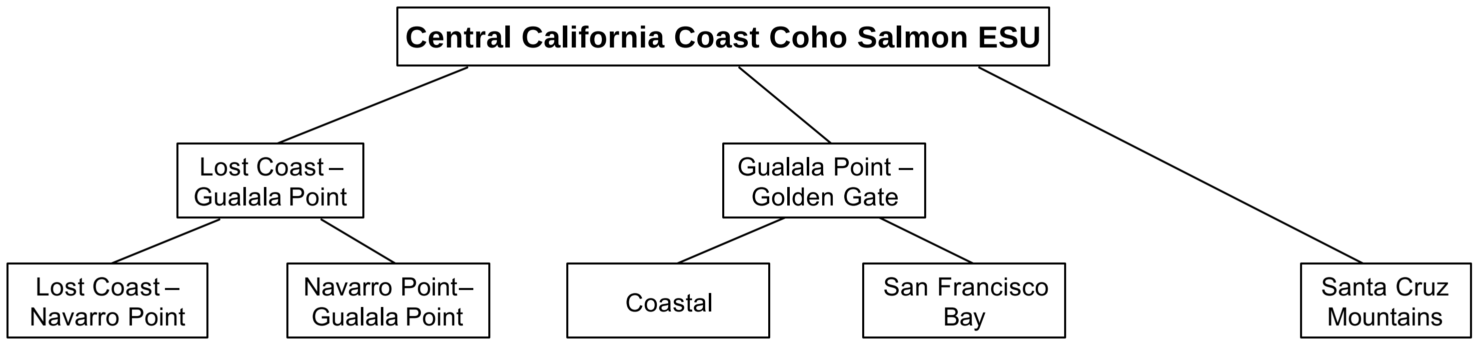
* Self-recruitment values may differ from those presented in Bjorkstedt et al. (2005) due to minor corrections in estimates of *IPkm* in several watersheds.

** The *IPkm* values for Usal Creek, Greenwood Creek, Elk Creek, and Alder Creek differ from those presented in Bjorkstedt et al. (2005) due to the subsequent identification of long-standing natural barriers on each of these streams.

*** Status of historical population in Walker Creek is especially uncertain due to environmental and ecological conditions; this population might have been dependent (mostly on the population of coho salmon in Lagunitas Creek) under historical conditions.

† The *IPkm* values for Lagunitas Creek and the San Lorenzo River differ from those presented in Bjorkstedt et al. (2005) due to corrections in IP calculations, which account for historically available habitat that currently lies behind dams.

†† IP km for San Francisco Bay is conservative, and includes only those watersheds for which there is reasonable support for historical presence of coho salmon.



Lost Coast – Navarro Point	Navarro Point– Gualala Point	Coastal	San Francisco Bay	Santa Cruz Mountains
Jackass Creek [b]	**Navarro River [34]**	Russian Gulch (S)[d]	Arroyo Corte Madera del Presidio [S1]	Pilarcitos Creek [52]
Usal Creek [17]	Greenwood Creek [35]	**Russian River [42]**	Corte Madera Creek [S2]	Tunitas Creek [53]
Cottaneva Creek [18]	Elk Creek [36]	Scotty Creek [d]	Miller Creek [S3]	San Gregorio Creek [54]
Juan Creek [20]	Mallo Pass Creek [c]	Salmon Creek (S)[43]	Novato Creek [S4]	Pomponio Creek [55]
Howard Creek [c]	Alder Creek [37]	Bodega Harbor [44]	Petaluma River [S5]	**Pescadero Creek [56]**
DeHaven Creek [21]	Brush Creek [38]	Americano Creek [45]	Sonoma Creek [S6]	Arroyo de los Frijoles [e]
Wages Creek [22]	**Garcia River [39]**	Stemple Creek [46]	Napa River [S7]	Gazos Creek [57]
Abalobadiah Creek [c]	Point Arena Creek [d]	***Walker Creek [TB1]***	San Pablo Creek [S8]	Whitehouse Creek [e]
Ten Mile River [23]	Schooner Gulch [40]	**Lagunitas Creek [TB2]**	Strawberry Creek [e]	Cascade Creek [e]
Mill Creek [c]	**Gualala River [41]**	Drakes Bay [48]	San Leandro Creek [S9]	Waddell Creek [58]
Pudding Creek [24]		Pine Gulch [49]	San Lorenzo Creek [S10]	Scott Creek [59]
Noyo River [25]		Redwood Creek (Ma)[50]	Alameda Creek [S11]	San Vicente Creek [60]
Hare Creek [26]			Coyote Creek [S12]	Wilder Creek [62]
Jug Handle Creek [c]			Guadalupe River [S13]	**San Lorenzo River [63]**
Caspar Creek [27]			Stevens Creek [S14]	Soquel Creek [64]
Russian Gulch (Me) [28]			San Francisquito Creek [S15]	Aptos Creek [65]
Big River [30]			San Mateo Creek [S16]	
Little River (Me) [31]				
Albion River [32]				
Big Salmon Creek [33]				

Figure A.1. Historical population structure of the CCC-Coho Salmon ESU, arranged by diversity strata. Functionally independent populations are listed in **bold font**. Potentially independent populations are listed in ***bold-italic*** font. Dependent populations are listed in regular font.

California Coastal Chinook Salmon Diversity Strata

Bjorkstedt et al. (2005) proposed a population structure that included major strata representing the two life-history types found in CC-Chinook salmon (i.e., fall-run and spring run), with fall-run Chinook being further subdivided into four diversity strata: North Coastal, Northern Mountain Interior, North-Central Coastal, and Central Coast. Subsequent deliberations by the TRT have led us to conclude that this proposed structure does not accurately reflect the likely evolutionary relationship between spring-run and fall-run populations. At issue is whether spring Chinook salmon populations in the ESU historically constituted a single monophyletic group, or alternatively, reflected independent parallel evolution of the spring-run life-history type from fall-run populations within each individual watershed. Because spring Chinook populations have been extirpated from the ESU, there is no way to definitively answer this question. However, analysis of genetic data from Chinook salmon in western North America indicates that, while both structures are possible, parallel evolution appears more common in coastal populations (Waples et al. 2004)[31]. The nearest extant spring Chinook populations north of the CC-Chinook ESU are found in the Klamath River basin and show stronger genetic affinity for fall-run Chinook populations in the same basin than for other spring Chinook populations to the immediate north. These data argue for independent evolution of the spring-run life history within each watershed, and we thus conclude that it is more appropriate to consider the two life-history types as substrata under the major environmentally based strata previously defined (Figure A.2). From the standpoint of implementing diversity criteria, the consequences of violating this assumption would be relatively minor. If in fact spring Chinook salmon are monophyletic, attainment of diversity strata goals would result in the monophyletic group being represented in the multiple diversity.

Finally, the TRT moved the Big Salmon Creek population from the Central Coastal stratum to the North-Central Coastal stratum. This change reflects the greater environmental similarity between Big Salmon Creek and watersheds to the immediate north (e.g., Albion River), and fosters consistency with diversity strata breaks defined for coho salmon and steelhead. The revised population structures of fall-run and spring-run Chinook salmon in the ESU are shown in Table A.2 and A.3, respectively. The arrangement of all populations with respect to diversity strata is shown in Figure A.2 and Plates A.2 and A.3.

[31] This contrasts with interior Columbia River basin spring-run populations, which form a coherent genetic group that is strongly divergent from summer- and fall-run populations in the same geographic region.

Table A.2. Historical population structure of fall-run Chinook salmon in the CC-Chinook ESU. This table supercedes Table 3.2 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005).

Population	*IPkm*	**Self-recruitment**	**Historical Population Status**
Redwood Creek (H) [1]	116.1	0.987	**Functionally Independent**
Little River (H) [3]	18.6	0.761	***Potentially Independent***
Mad River [5]	94.0	0.948	**Functionally Independent**
Humboldt Bay [6]	76.7	0.866	***Potentially Independent***
Lower Eel River*	514.9	0.993	**Functionally Independent**
Upper Eel River**	555.9		**Functionally Independent**
Bear River [10]	39.4	0.745	***Potentially Independent***
Mattole River [14]	177.5	0.968	**Functionally Independent**
Usal Creek [17]	6.1	0.530	dependent†
Cottaneva Creek [18]	5.2	0.780	dependent†
DeHaven Creek [19]	2.4	0.685	dependent†
Wages Creek [22]	5.2	0.843	dependent†
Ten Mile River [23]	67.2	0.975	**Functionally Independent**
Pudding Creek [24]	8.3	0.788	dependent†
Noyo River [25]	62.2	0.989	**Functionally Independent**
Hare Creek [26]	2.8	0.695	dependent†
Caspar Creek [27]	2.3	0.500	dependent†
Big River [30]	104.3	0.982	**Functionally Independent**
Albion River [32]	17.6	0.895	dependent†
Big Salmon Creek [33]	2.9	0.771	dependent†
Navarro River [34]	131.5	0.989	**Functionally Independent**
Greenwood Creek [35]	4.7	0.694	dependent†
Elk Creek [36]	7.8	0.747	dependent†
Alder Creek [37]	4.9***	0.647	dependent†
Brush Creek [38]	6.1	0.825	dependent†
Garcia River [39]	56.2	0.926	***Potentially Independent***
Gualala River [41]	175.6	0.923	***Potentially Independent***
Russian River [42]	584.2	0.992	**Functionally Independent**
Salmon Creek (S)[43] ††	13.8	0.639	dependent†
Americano Creek [45] ††	13.3	0.727	dependent†
Stemple Creek [46] ††	18.4	0.840	dependent†
Tomales Bay [47] ††	67.4	0.806	dependent†

* The Lower Eel River population occupied tributaries of the Eel River downstream from the confluence of the South Fork Eel River (inclusive) and is concentrated in the South Fork Eel River.
** The Upper Eel River population occupied tributaries upstream of the confluence of the South Fork Eel River (exclusive) and is concentrated in the Middle Fork Eel River.
*** The *IPkm* value for Alder Creek differs from that presented in Bjorkstedt et al. (2005) due to the subsequent identification of a long-standing natural barrier on Alder Creek.
† On the basis of environmental considerations and potential IP bias in the relation between IP km and population carrying capacity, it is unlikely that fall-run Chinook salmon consistently occupied these basins. Historical records of Chinook salmon are not available for any of these basins, save Wages Creek, from which a recent sample was collected. See Bjorkstedt et al. 2005 for further details.
†† These streams are south of the currently accepted range of the CC-Chinook ESU (Myers et al. 1998); we concur that persistent populations of Chinook salmon are not likely to have occupied these watersheds under historical conditions, although Chinook have been observed in Lagunitas Creek in recent years.

Table A.3. Historical population structure of spring-run Chinook salmon in the CC-Chinook ESU. This table supercedes Table 3.3 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005).

Population	**Historical Population Status**
(Redwood Creek (H)[1])	**(Functionally Independent)**
Mad River [5]	**Functionally Independent**
(Van Duzen River [E2])	**(Functionally Independent)**
North Fork Eel River [E5]	**Functionally Independent**
Middle Fork Eel River [E7]	**Functionally Independent**
Upper Eel River [E8]	**(Functionally Independent)**

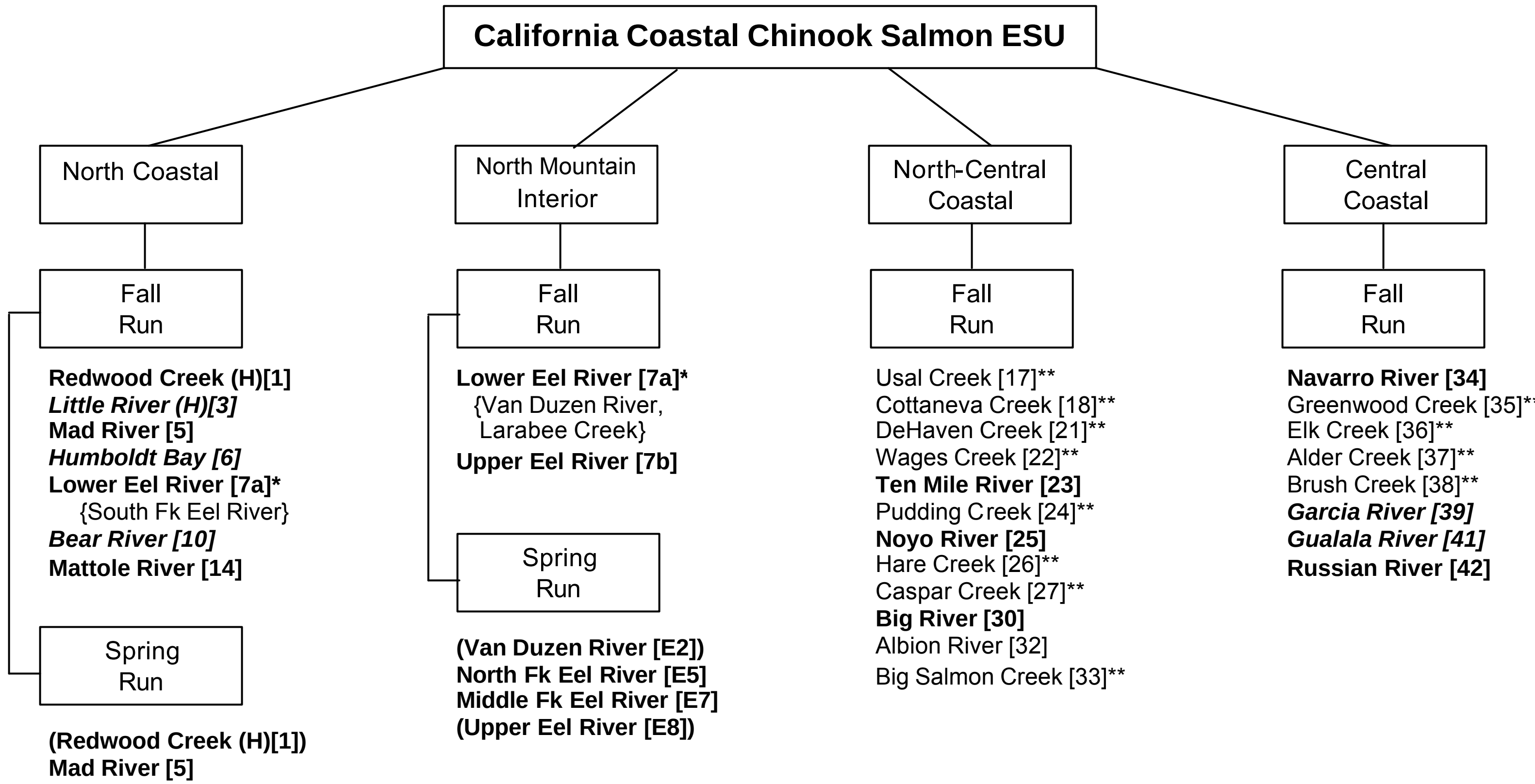


Figure A.2. Historical population structure of the CC-Chinook Salmon ESU, arranged by diversity strata. Functionally independent populations are listed in **bold font**. Potentially independent populations are listed in ***bold-italic*** font. Dependent populations are listed in regular font. Populations indicated by single asterisk are listed under multiple diversity strata and occupy environmentally diverse basins; subpopulations that occur within these different strata are shown in squiggly brackets. Populations indicated by a double asterisk are dependent populations in small watersheds, and are expected to be critically dependent on dispersal for occupancy. Spring-run Chinook salmon populations listed parenthetically are those for which potential historical existence is tentatively inferred from environmental correlates.

Northern California Steelhead Diversity Strata

As with Chinook salmon, the TRT's original proposal for diversity strata for steelhead posited two major groupings based on life-history type: winter versus summer run (Bjorkstedt et al. 2005). Winter-run fish were further divided into five diversity strata (Northern Klamath Mountains, Southern Klamath Mountains, Northern Coastal, Central Coastal, and Southern Coastal) based on environmental characteristics. Summer-run fish were placed into two diversity strata (Interior and Coastal), also based on environmental characteristics (Figure 4.18 in Bjorkstedt et al. 2005). Upon further consideration, we have revised this structure to more accurately reflect what we believe to be the likely evolutionary relationship between winter-run and summer-run steelhead occupying the same watershed—specifically, that summer-run steelhead populations in the DPS likely represent independently evolved life-history types within each watershed rather than a single monophyletic group. Our reasoning parallels that for modifications to the Chinook salmon diversity strata. Although there are no data from which to compare summer steelhead populations within the domain (or within the Eel River basin), microsatellite data indicate that summer steelhead from the Middle Fork Eel River group more closely with winter steelhead from the Middle Fork Eel than to other winter steelhead in the either the South Fork or upper mainstem Eel River (Anthony Clemento and J. Carlos Garza, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data, cited in Bjorkstedt et al. 2005). The strong genetic affinity between summer and winter steelhead in the Middle Fork Eel River suggest a recent divergence, and we hypothesize that this pattern is likely to hold for other summer steelhead populations as well.

To reduce confusion, we have also renamed the steelhead diversity strata so they correspond more closely with those defined for Chinook salmon. The “Southern Klamath Mountains” stratum of Bjorkstedt et al. (2005) is now called the Lower Interior stratum; the “Northern Klamath Mountains” is now the North-Mountain Interior stratum; the “Central Coastal” stratum is renamed the North-Central Coastal stratum; and the “Southern Coastal” stratum is now the “Central Coastal” stratum. The Northern Coastal stratum remains as such.

Several other changes were made in the placement of populations into these diversity strata. First, we consider the Mattole River and South Fork Eel River populations to fall entirely within the Northern Coastal stratum. These two populations were originally considered to span two diversity strata (Northern Coastal and Lower Interior) based on east-west gradients in environmental conditions across these two basins (Bjorkstedt et al. 2005). However, the entire Mattole River basin and the vast majority of the South Fork Eel River fall within the Coast Range ecoregion (see Plate 2 of Bjorkstedt et al. 2005). Further, examination of environmental data indicates that precipitation and temperature regimes in these basins are generally more similar to the more coastal region than they are to the interior portions of the Eel River basin, though they are intermediate to the coastal and interior regions for certain variables. Nevertheless, while environmental gradients do occur across these basins, we believe they are comparable to gradients observed across other coastal basins where we did not assign populations to multiple strata. We do note, however, that in assessing viability of populations, recovery planners should consider the spatial structure of populations across these basins, as environmental gradients may be a source of phenotypic diversity that could contribute to population viability.

We reaffirm our conclusion (Bjorkstedt et al. 2005) that the Mad River steelhead populations (both winter- and summer-run) each span two diversity strata: the Northern Coastal and North Mountain Interior strata. In this case, the east-west environmental gradient is sufficiently large that it spans the boundary between the Coast Range and Klamath Mountains ecoregions (EPA 2006; see Plate 2 of Bjorkstedt et al. 2005). Further consideration of the Redwood Creek populations (winter- and summer-run) suggests that it likewise is more appropriately placed in both the Northern Coastal and North Mountain Interior strata, as approximately half of this basin falls into each of the aforementioned ecoregions. This departs from Bjorkstedt et al. (2005), who placed the population exclusively into the Northern Coastal stratum. The TRT notes that spawning distribution of summer-run steelhead in both Mad River and Redwood Creek is not well known. In general, summer steelhead tend to penetrate farther into watersheds than do winter steelhead, which raises the possibility that the summer-run populations might spawn primarily in the headwater portions of Mad River and Redwood Creek. However, data from summer surveys of adult steelhead in holding pools indicates that they use both the upper and lower portions of the watershed for summer rearing. As we cannot determine whether fish holding in the lower portions of these basins ultimately spawn in the lower or upper reaches, we tentatively conclude that, like winter-run steelhead, summer steelhead span both strata.

Several other changes to population designations warrant discussion. First, within the Lower Interior stratum, the Outlet Creek and Tomki Creek winter steelhead populations have been changed from potentially independent to functionally independent populations, as has been the Larabee Creek winter steelhead population in the North Mountain Interior stratum. Each of these watersheds contain substantial steelhead habitat ($IPkm > 100$ in all cases), and for all three populations, estimates of self-recruitment are well above our threshold of 95%, even assuming a higher rate of straying (10%) for within-Eel River basin populations. In the case of Tomki Creek, some uncertainty remains at to whether this population is most appropriately characterized as functionally or potentially independent. In recent years, significant portions of Tomki Creek have gone dry during the summer (Weldon Jones, CDFG retired, personal observations). However, it is unclear whether this phenomenon is natural or is the result of water diversions, channel aggradation, modification of riparian vegetation, or other anthropogenic factors (Scott Harris, CDFG, Willits, pers. comm.). In the event that our estimate of intrinsic potential for steelhead in this basin is biased high, then predicted self-recruitment may also be biased high, which would suggest that it might be more appropriate to categorize the Tomki Creek population as potentially independent. Finally, upon th recommendation of reviewers, we classified Soda Creek steelhead in the upper Eel River as a potentially independent population; this population had previously been assumed to be part of the Upper Mainstem Eel River population.

The historical population structure for winter steelhead in the NC Steelhead DPS is shown in Tables A.4 (coastal region) and A.5 (Eel River basin), and summer steelhead population structure is shown in Table A.6. The arrangement of winter and summer steelhead populations is illustrated in Figure A.3 and Plates A.4 and A.5.

Table A.4. Historical population structure of winter steelhead in the NC-Steelhead DPS. This table supercedes Table 4.4 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005). Not all dependent populations are shown.

Population	*IPkm*	IP bias index	Self-recruitment	Historical Population Status
Butler Creek [a]	2.0	low	0.747	dependent
Boat Creek [a]	1.6	low	0.536	dependent
Fern Canyon [a]	6.0	low	0.933	dependent
Squashan Creek [a]	4.0	low	0.720	dependent
Gold Bluff [a]	4.4	low	0.574	dependent
Redwood Creek (H) [1]	301.1	low	0.992	**Functionally Independent**
McDonald Creek [a]	6.4	low	0.528	dependent
Maple Creek/Big Lagoon [2]	94.7	low	0.913	***Potentially Independent***
Little River (H) [3]	76.2	low	0.864	***Potentially Independent***
Strawberry Creek [a]	6.1	low	0.498	dependent
Widow White Creek [4]	9.1	low	0.641	dependent
Mad River [5]	553.2*	low	0.980	**Functionally Independent**
Humboldt Bay [6]	283.0	low	0.877	**Functionally Independent**
Eel River - Full [7]	4029.4		0.995	See Table 4.5
Fleener Creek [a]	4.1	low	0.243	dependent
Guthrie Creek [8]	10.9	low	0.623	dependent
Oil Creek [9]	11.7	low	0.551	dependent
Bear River [10]	114.8	low	0.928	***Potentially Independent***
Singley Creek [11]	11.8	low	0.563	dependent
Davis Creek [12]	8.1	low	0.591	dependent
Domingo Creek [a]	3.4	low	0.578	dependent
McNutt Gulch [13]	14.1	low	0.772	dependent
Peter Gulch [a]	2.3	low	0.326	dependent
Mattole River [14]	613.9	low	0.996	**Functionally Independent**
Fourmile Creek [15]	8.8	low	0.569	dependent
Cooskie Creek [16]	8.0	low	0.677	dependent
Randall Creek [b]	2.0	low	0.436	dependent
Spanish Creek [b]	1. 9	low	0.585	dependent
Oat Creek [b]	1.8	low	0.477	dependent
Big Creek [b]	3.8	low	0.625	dependent
Big Flat Creek [b]	6.1	low	0.776	dependent
Shipman Creek [b]	2.3	low	0.565	dependent
Gitchell Creek [b]	2.5	low	0.641	dependent
Horse Mountain Creek [b]	3.2	low	0.782	dependent
Telegraph Creek [b]	5.6	low	0.944	dependent
Humboldt Creek [b]	1.6	low	0.456	dependent
Whale Gulch [b]	5.1	low	0.681	dependent
Jackass Creek [b]	3.6	low	0.801	dependent
Little Jackass Creek [b]	6.3	low	0.777	dependent
Usal Creek [17]	19.0	low	0.905	***Potentially Independent***
Cottaneva Creek [18]	26.1	low	0.912	***Potentially Independent***
Hardy Creek [19]	10.0	low	0.904	dependent
Juan Creek [20]	11.3	low	0.935	dependent
Howard Creek [c]	6.6	moderate	0.832	dependent

Table A.4. (continued)

Population	***IPkm***	**IP Bias Index**	**Self-recruitment**	**Historical Population Status**
DeHaven Creek [21]	13.0	moderate	0.936	dependent
Wages Creek [22]	19.9	low	0.947	***Potentially Independent***
Chadbourne Gulch [c]	3.7	moderate	0.562	dependent
Abalobadiah Creek [c]	6.9	moderate	0.714	dependent
Seaside Creek [c]	2.8	moderate	0.844	dependent
Ten Mile River [23]	204.7	moderate	0.996	**Functionally Independent**
Inglenook Creek [c]	3.2	moderate	0.520	dependent
Mill Creek [c]	5.6	moderate	0.631	dependent
Virgin Creek [c]	4.4	moderate	0.698	dependent
Pudding Creek [24]	32.0	moderate	0.939	***Potentially Independent***
Noyo River [25]	199.1	moderate	0.990	**Functionally Independent**
Hare Creek [26]	18.1	moderate	0.939	***Potentially Independent***
Digger Creek [c]	2.0	moderate	0.569	dependent
Mitchell Creek [c]	5.5	moderate	0.740	dependent
Jug Handle Creek [c]	5.4	moderate	0.743	dependent
Caspar Creek [27]	16.0	moderate	0.928	***Potentially Independent***
Doyle Creek [c]	2.4	moderate	0.547	dependent
Russian Gulch (Me) [28]	19.2	moderate	0.858	***Potentially Independent***
Jack Peters Creek [29]	8.0	moderate	0.799	dependent
Big River [30]	316.6	high	0.993	**Functionally Independent**
Little River (M) [31]	9.9	moderate	0.754	dependent
Buckhorn Creek [c]	1.7	moderate	0.397	dependent
Dark Gulch [c]	2.0	moderate	0.421	dependent
Albion River [32]	77.1	high	0.976	**Functionally Independent**
Big Salmon Creek [33]	24.8	high	0.910	***Potentially Independent***
Navarro River [34]	458.2	high	0.992	**Functionally Independent**
Greenwood Creek [35]	8.7	high	0.606	dependent
Elk Creek [36]	24.3	high	0.876	***Potentially Independent***
Mallo Pass Creek [c]	7.1	moderate	0.584	dependent
Alder Creek [37]	9.1**	high	0.764	dependent
Brush Creek [38]	28.3	high	0.908	***Potentially Independent***
Garcia River [39]	169.0	high	0.984	**Functionally Independent**
Point Arena Creek [d]	4.4	moderate	0.536	dependent
Moat Creek [d]	5.1	moderate	0.676	dependent
Ross Creek [d]	4.0	moderate	0.796	dependent
Galloway Creek [d]	2.4	moderate	0.747	dependent
Schooner Gulch [40]	9.5	moderate	0.838	dependent
Slick Rock Creek [d]	2.8	moderate	0.509	dependent
Signal Port Creek [d]	3.2	moderate	0.498	dependent
Saint Orres Creek [d]	1.8	moderate	0.254	dependent
Gualala River [41]	478.0	high	0.987	**Functionally Independent**
Miller Creek [d]	3.2	moderate	0.137	dependent
Stockhoff Creek [d]	3.2	moderate	0.283	dependent
Timber Cove Creek [d]	1.7	moderate	0.266	dependent

* Mad River value includes habitat upstream of a partial barrier near the confluence of Bug Creek that may not be accessible in all years.
**The *IPkm* value for Alder Creek differs from that presented in Bjorkstedt et al. (2005) due to the subsequent identification of a long-standing natural barrier on Alder Creek. Two consequences of this error are that the self-recruitment estimate is biased high and that the population is now designated as a dependent population.

Table A.5. Historical population structure of winter steelhead in the Eel River basin. This table supercedes Table 4.5 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005).

Population	*IPkm*	IP bias index	Self-recruitment	Historical Population Status
Lower Mainstem Eel River*		moderate		dependent populations
Price Creek [A]	20.6	low	0.987	***Potentially Independent***
Howe Creek [B]	15.3	low	0.948	dependent
Van Duzen River [E2]	363.8†	low	0.996	**Functionally Independent**
Larabee Creek [C]	101.0	low	0.971	**Functionally Independent**
South Fork Eel River [E3]	1182.1	low	0.998	**Functionally Independent**
Lower Middle Mainstem Eel River*		low		dependent populations
Dobbyn Creek [D]	52.5	low	0.926	***Potentially Independent***
Jewett Creek [F]	18.2	low	0.874	***Potentially Independent***
Pipe Creek [G]	18.2	low	0.838	***Potentially Independent***
Kekawaka Creek [H]	35.3	low	0.926	***Potentially Independent***
Chamise Creek [J]	38.0	low	0.904	***Potentially Independent***
North Fork Eel River [E5]	372.8	low	0.983	**Functionally Independent**
Upper Middle Mainstem Eel River*		moderate		dependent populations
Bell Springs Creek [K]	18.5	moderate	0.837	***Potentially Independent***
Woodman Creek [L]	39.4	moderate	0.894	***Potentially Independent***
Outlet Creek [N]	313.8	moderate	0.975	**Functionally Independent**
Tomki Creek [P]	131.7	moderate	0.968	**Functionally Independent**
Middle Fork Eel River [E7]	584.3	low	0.989	**Functionally Independent**
Bucknell Creek [R]	21.1	moderate	0.812	***Potentially Independent***
Soda Creek [S]	17.6	moderate	††	***Potentially Independent***
Upper Mainstem Eel River**	387.3	moderate	0.997	**Functionally Independent**

* Indicate the set of small watersheds tributary to each section of the mainstem Eel River that are not listed by name in this table.
** The Upper Mainstem Eel River population occupies the mainstem and tributaries below the confluence of Bucknell Creek (exclusive), and thus differs slightly from the basin designated "Upper Mainstem Eel River" in the multivariate environmental analysis (See Bjorkstedt et al. 2005 for details).
† The *IPkm* value for the Van Duzen River differs from that presented in Bjorkstedt et al. (2005) due to the subsequent identification of a long-standing natural barriers on the river.
†† Soda Creek was previously considered part of the Upper Mainstem Eel Population. Self-recruitment values were not calculated, but are assumed to be similar to Bucknell Creek, which is both nearby and similar in size.

Table A.6. Historical population structure of summer steelhead in the NC-Steelhead DPS. This table supercedes Table 4.6 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005).

Population	Historical Population
Redwood Creek (H)[1])	**Functionally Independent**
Mad River [5]	**Functionally Independent**
Van Duzen River [E2]	**Functionally Independent**
South Fork Eel River [E3]	**Functionally Independent**
Larabee Creek	**Functionally Independent**
North Fork Eel River [E5]	**Functionally Independent**
(Upper Middle Mainstem Eel River [E6])**	**(Functionally Independent)**
Middle Fork Eel River [E7]	**Functionally Independent**
(Upper Mainstem Eel River [8])***	**(Functionally Independent)**
Mattole River [14]	**Functionally Independent**

* All summer steelhead populations are considered functionally independent; see Bjorkstedt et al. 2005 for discussion.
** Summer steelhead have not been documented in this area; however , some of the watersheds that drain the north bank of the Eel River are environmentally similar to Larabee Creek and the major subbasins on the north Side of the Eel River basin and might have harbored historical populations of summer steelhead. Such populations, if shown to exist, would be considered functionally independent, pending further analysis.
*** The extent of habitat suitable for summer steelhead populations in the upper Eel River and its tributaries is unknown, and is likely to be restricted to the northeast corner of the basin (near the Middle Fork Eel River, where annual snowpack occurs).

Northern California Steelhead DPS

Northern Coastal

Winter Run

- Fern Canyon [a]
- Gold Bluff Creek [a]
- **Redwood Cr (H)[1]***
- McDonald Creek [a]
- ***Maple Cr/Big Lagoon [2]***
- ***Little River (H)[3]***
- Strawberry Creek [a]
- Widow White Creek [4]
- **Mad River[5]***
- **Humboldt Bay [6]**
- Lower Eel River tributaries
- ***Price Creek***
- Howe Creek
- **South Fk Eel River [E3]**
- Fleener Creek[a]
- Guthrie Creek [8]
- Oil Creek [9]
- ***Bear River [10]***
- Singley Creek [11]
- Davis Creek [12]
- McNutt Gulch [13]
- **Mattole River [14]**
- Fourmile Creek [15]
- Cooksie Creek [16]
- Lost Coast populations

Summer Run

- **Redwood Creek [1]***
- **Mad River [5]***
- **South Fk Eel River [E3]**
- **Mattole River [14]**

Lower Interior

Winter Run

- ***Jewett Creek***
- ***Pipe Creek***
- ***Chamise Creek***
- ***Bell Springs Creek***
- ***Woodman Creek***
- **Outlet Creek**
- **Tomki Creek**
- ***Bucknell Creek***
- ***Soda Creek***

North Mountain Interior

Winter Run

- **Redwood Creek (H)[1]***
- **Mad River [5]***
- **Van Duzen River [E2]**
- **Larabee Creek**
- ***Dobbyn Creek***
- ***Kekawaka Creek***
- **North Fk Eel River [E5]**
- **Middle Fk Eel river [E7]**
- **Upper Mainstem Eel River [E8]**

Summer Run

- **Redwood Creek (H)[1]***
- **Mad River [5]***
- **Van Duzen River [E2]**
- **Larabee Creek**
- **North Fk Eel River [E5]**
- **(Upper Mid. Mainstem Eel R [E6]**
- **Middle Fk Eel River [E7]**
- **(Upper Mainstem Eel River) [E8]***

North-Central Coastal

Winter Run

- ***Usal Creek [17]***
- ***Cottaneva Creek [18]***
- Hardy Creek [19]
- Juan Creek [20]]
- Howard Creek [c]
- DeHaven Creek [21]
- ***Wages Creek [22]***
- Abalobodiah Creek [c]
- **Ten Mile River [23]**
- Mill Creek [c]
- Virgin Creek [c]
- ***Pudding Creek [24]***
- **Noyo River [25]**
- ***Hare Creek [26]***
- Mitchell Creek [c]
- Jug Handle Creek [c]
- ***Caspar Creek [27]***
- ***Russian Gulch (Me)[28]***
- Jack Peters Creek [29]
- **Big River [30]**
- Little River (Me)[31]
- **Albion River [32]**
- ***Big Salmon Creek [33]***

Central Coastal

Winter Run

- **Navarro River [34]**
- Greenwood Creek [35]
- ***Elk Creek [36]***
- Mallo Pass Creek [c]
- Alder Creek [37]
- ***Brush Creek [38]***
- **Garcia River [39]**
- Point Arena Creek [d]
- Moat Creek [d]
- Ross Creek [d]
- Schooner Gulch [38]
- **Gualala River [41]**

Figure A.3. Historical structure of the NC-steelhead DPS, arranged by diversity strata. Functionally independent populations are listed in **bold** font. Potentially independent populations are listed in ***bold-italic*** font. Dependent populations are listed in regular font (not all dependent populations are shown). Populations indicated by an asterisk are listed under multiple diversity strata and occupy environmentally diverse basins. Populations listed parenthetically are those for which potential historical existence is inferred from environmental correlates.

Central California Coast Steelhead Diversity Strata

Minor modifications have been made to the historical population delineations proposed by Bjorkstedt et al. (2005) for the CCC-Steelhead DPS. First, since Bjorkstedt et al. (2005) was published, the National Marine Fisheries Service has clarified the eastern boundary of the Central California Coast DPS within the San Francisco Bay Region. This DPS was originally defined as including populations in San Francisco Bay east to and including the Napa River (62 FR 43937-43954); however, language defining the Central Valley DPS, which includes steelhead populations in tributaries to the Sacramento and San Joaquin River, was vague as to whether streams entering into the Suisun Bay region were considered part of the Central Valley DPS. The Central Valley Technical Recovery Team (Lindley et al. 2006) considered steelhead in creeks within this region to be part of the Central Valley DPS, proposing that collectively, fish within these tributaries (Green Valley Creek/Suisun Creek, Walnut Creek, Mt Diablo Creek, Arroyo del Hambre, and other smaller watersheds) constituted a single independent population. However, NMFS subsequently concluded that steelhead within the Susiun Bay region from Carquinez Strait to Chipps Island (the confluence of the Sacramento and San Joaquin rivers) are more appropriately considered part of the CCC-Steelhead DPS (71 FR 834-862).

We thus here consider the plausible population structure within this region, and its relation to other populations in the San Francisco Bay region. Based on our IP model, four watersheds within the region are predicted to potentially have had sufficient habitat to support independent populations of steelhead (Table A.9). The smallest of these, Arroyo del Hambre and Mt. Diablo Creek, we conclude likely supported dependent populations. Although the predicted IP exceeds our independence threshold of 16 *IPkm* in both watersheds, the predicted IP bias is “severe,” and we therefore believe it doubtful that these watersheds historically supported populations of sufficient size to be viable in isolation. Green Valley and Suisun creeks both enter into a common slough before reaching Suisun Bay; thus, the exchange of individuals between these two subwatersheds was likely high enough to constitute a single demographically coupled unit. Collectively, these two watersheds contain sufficient potential habitat for an independent population. Likewise, the Walnut Creek watershed also likely contained sufficient habitat to support an independent population. Determining whether these two populations should be classified as functionally independent or potentially independent population is problematic, as not only would these populations have been influenced by strays from other San Francisco Bay tributaries, but they were also undoubtedly influenced by strays from the Sacramento-San Joaquin basin, which historically may have produced as many as 1-2 million fish annually (McEwan 2001)[32]. Because of the potentially large influx

[32] We do not have estimates of intrinsic potential for streams within the Central Valley DPS and thus are unable to run an analysis of self-recruitment.

of strays from neighboring systems, we tentatively conclude that both the Green Valley/Suisun Creek population and Walnut Creek population were most likely potentially independent populations. We do note that it is plausible that the four identified populations (along with other smaller dependent populations in the area) formed a single interdependent unit (as proposed by the Central Valley TRT; Lindley et al. 2006). However, without any direct evidence supporting such aggregations, we opt to consider these populations as separate, as we did elsewhere in the San Francisco Bay area. These populations, along with any other dependent populations that enter into Susiun Bay or Carquinez Strait, we consider to be part of the Interior San Francisco Bay diversity stratum.

Finally, we offer some clarification as to the geographic boundaries of diversity strata as they relate to populations in the Russian River basin. Populations downstream of the confluence of Mark West Creek are considered part of the North Coastal stratum, which also includes coastal watersheds in southern Sonoma and Marin counties. The Interior stratum includes Russian River populations upstream of Mark West Creek (inclusive). Tables A.7, A.8, and A.9 show population structure for the DPS, and Figure A.8 and Plate A.6 show these populations arranged into diversity strata.

Table A.7. Historical population structure of winter steelhead in the CCC-Steelhead DPS. This table supercedes Table 4.7 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005). Not all dependent populations are shown.

Population	*IPkm*	IP bias index	Self-recruitment	Historical Population Status
Kolmer Creek [d]	3.9	moderate	0.517	dependent
Fort Ross Creek [d]	2.1	moderate	0.160	dependent
Russian Gulch (S) [d]	7.1	moderate	0.251	dependent
Russian River [42]	2348.8		0.999	See Table A.8
Scotty Creek [d]	5.8	high	0.243	dependent
Salmon Creek (S) [43]	63.5	high	0.820	***Potentially Independent***
Bodega Harbor [44]	14.1	high	0.535	dependent
Americano Creek [45]	64.2	high	0.887	***Potentially Independent***
Stemple Creek [46]	73.1	high	0.921	***Potentially Independent***
Tomales Bay [47]	294.7	high	0.944	
Walker Creek [TB1]	134.1	high		***Potentially Independent***
Lagunitas Creek [TB2]	170.7[†]	high		***Potentially Independent***
Drakes Bay [48]	10.1	high	0.303	dependent
Pine Gulch [49]	12.9	high	0.302	dependent
Redwood Creek (M) [50]	10.4	high	0.212	dependent
San Francisco Bay [51]	3054.6		0.999	See Table A.9
San Pedro Creek [e]	na	high	na	dependent
Pilarcitos Creek [52]	41.9	high	0.494	***Potentially Independent***
Canada Verde Creek [e]	4.3	high	0.232	dependent
Tunitas Creek [53]	16.4	high	0.668	dependent
San Gregorio Creek [54]	77.6	high	0.953	**Functionally Independent**
Pomponio Creek [55]	11.5	high	0.742	dependent
Pescadero Creek [56]	93.8	high	0.961	**Functionally Independent**
Arroyo de los Frijoles [e]	6.6	high	0.551	dependent
Gazos Creek [57]	16.1	high	0.842	dependent
Whitehouse Creek [e]	7.5	high	0.873	dependent
Cascade Creek [e]	5.9	high	0.898	dependent
Green Oaks Creek [e]	3.3	high	0.720	dependent
Ano Nuevo Creek [e]	4.2	high	0.692	dependent
Waddell Creek [58]*	16.5	high	0.869	***Potentially Independent***
Scott Creek [59]	23.5	high	0.938	***Potentially Independent***
San Vicente Creek [60]	8.0	high	0.859	dependent
Liddell Creek [e]	6.6	high	0.866	dependent
Laguna Creek [61]*	17.4	high	0.923	***Potentially Independent***
Baldwin Creek [e]	7.3	high	0.799	dependent
Wilder Creek [62]	14.1	high	0.850	dependent
San Lorenzo River [63]	225.6[†]	high	0.994	**Functionally Independent**
Rodeo Creek Gulch [e]	6.1	high	0.726	dependent
Soquel Creek [64]$**	66.4	high	0.978	***Potentially Independent***
Aptos Creek [65]	41.0	high	0.919	***Potentially Independent***

* Conclusions for these watersheds reflect the high likelihood that lagoon habitats at least partially offset potential bias in the IP model.
** The historical status of Soquel Creek depends in part on whether substantial immigration from populations in the South-Central California Coast ESU, especially the Pajaro and Salinas rivers, was substantial under historical conditions.
[†] The *IPkm* values for Lagunitas Creek and San Lorenzo River differ from those presented in Bjorkstedt et al. (2005) due to a correction in IP calculations.

Table A.8. Historical population structure of winter steelhead in the Russian River basin. This table supercedes Table 4.8 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005).

Population	***IPkm***	**IP bias index**	**Self-recruitment**	**Historical Population Status**
Lower Russian River*		high		dependent populations
Austin Creek [A]	111.9	high	0.981	***Potentially Independent***
Dutch Bill Creek [B]	17.4	high	0.973	dependent
Green Valley Creek [C]	61.7	high	0.988	***Potentially Independent***
Mark West Creek [D]	366.5	high	0.997	***Potentially Independent***
Middle Russian River**		high		dependent populations
Dry Creek [E]	384.9	high	0.998	***Potentially Independent***
Maacama Creek [F]	106.9	high	0.991	***Potentially Independent***
Sausal Creek [G]	17.3	high	0.957	dependent
Upper Russian River [H] †	892.3	high	>0.999	**Functionally Independent**

* Unnamed and smaller tributaries downstream of the confluence of Mark West Creek. **Unnamed and smaller tributaries between Mark West and Big Sulphur creeks. †The Upper Russian River population occupies the mainstem and tributary habitats upstream from the confluence of Big Sulphur Creek (inclusive).

Table A.9. Historical population structure of winter steelhead in tributaries of San Francisco, San Pablo, and Suisun bays. This table supercedes Table 4.9 in Bjorkstedt et al. (2005). Bracketed codes correspond to watershed delineations defined in Bjorkstedt et al. (2005).

Population	*IPkm*	IP Bias index	Self-recruitment	Historical Population Status
Northwest Bay				
Arroyo Corte Madera del Presidio [S1]	12.8	high	0.294	dependent
Corte Madera Creek [S2]	41.3	high	0.527	***Potentially Independent***
Miller Creek [S3]	44.4	high	0.883	***Potentially Independent***
Novato Creek [S4]	78.6	severe	0.778	***Potentially Independent***
North Bay				
Petaluma River [S5]	225.4	severe	0.939	***Potentially Independent***
Sonoma Creek [S6]	268.7	high	0.955	**Functionally Independent**
Napa River [S7]	593.9	severe	0.978	**Functionally Independent**
Suisun Bay				
Green Valley/Suisun Creek [S17]	164.0	severe	na	***Potentially Independent***
Arroyo del Hambre [S18]	25.5	severe	na	dependent
Walnut Creek [S19]	202.2	severe	na	***Potentially Independent***
Mt. Diablo Creek [S20]	44.9	severe	na	dependent
East Bay				
San Pablo Creek [S8]	67.9	severe	0.754	***Potentially Independent***
San Leandro Creek [S9]	80.5	severe	0.954	**Functionally Independent**
San Lorenzo Creek [S10]	79.8	severe	0.985	**Functionally Independent**
Southeast Bay				
Alameda Creek [S11]	816.6	severe	0.975	**Functionally Independent**
Coyote Creek [S12]	498.3	severe	0.936	**Functionally Independent**
Southwest Bay				
Guadalupe River [S13]	157.3	severe	0.958	**Functionally Independent**
Stevens Creek [S14]	39.6	severe	0.775	***Potentially Independent***
San Francisquito Creek [S15]	59.2	severe	0.655	***Potentially Independent***
San Mateo Creek [S16]	57.6	severe	0.752	***Potentially Independent***
unnamed tributaries				dependent populations

Central California Coast Steelhead DPS

North Coastal

Russian Gulch (S)[d]
Lower Russian tributaries
Austin Creek
Dutch Bill Creek
Green Valley Creek
Scotty Creek [d]
Salmon Creek* (S)*[43]
Bodega Harbor [44]
Americano Creek [45]
Stemple Creek [46]
Walker Creek [TB1]
Lagunitas Creek [TB2]
Drakes Bay [48]
Pine Gulch [49]
Redwood Creek (Ma)[50]

Interior

Mark West Creek
Middle Russian tributaries
Dry Creek
Maacama Creek
Sausal Creek
Upper Russian River [42]

Santa Cruz Mountains

San Pedro Creek [e]
Pilarcitos Creek [52]
TunitasCreek [53]
San Gregorio Creek [54]
Pomponio Creek [55]
Pescadero Creek [56]
Arroyo de los Frijoles [e]
Gazos Creek [57]
Whitehouse Creek [e]
Cascade Creek [e]
Ano Nuevo Creek [e]
Waddell Creek [58]
Scott Creek [59]
San Vicente Creek [60]
Liddell Creek [e]
Laguna Creek [61]
Baldwin Creek [e]
Wilder Creek [62]
San Lorenzo River [63]
Rodeo Creek Gulch [e]
Soquel Creek [64]
Aptos Creek [65]

Coastal S. F. Bay

Arroyo Corte Madera del Presidio [S1]
Corte Madera Creek [S2]
Miller Creek [S3]
Novato Creek [S4]
Guadalupe River [S13]
Stevens Creek [S14]
San Francisquito Creek [S15]
San Mateo Creek [S16]

Interior S. F. Bay

Petaluma River [S5]
Sonoma Creek [S6]
Napa River [S7]
Green Valley/Suisun Cr [S17]
Arroyo del Hambre [18]
Walnut Creek [S19]
Mt. Diablo Creek [S20]
San Pablo Creek [S8]
San Leandro Creek [S9]
San Lorenzo Creek [S10]
Alameda Creek [S11]
Coyote Creek [S12]

Figure A.4. Historical structure of the CCC-steelhead DPS, arranged by diversity strata. Functionally independent populations are listed in **bold** font. Potentially independent populations are listed in ***bold-italic*** font. Dependent populations are listed in regular font. Not all dependent populations have been included in this figure. See table A.4 for complete list.

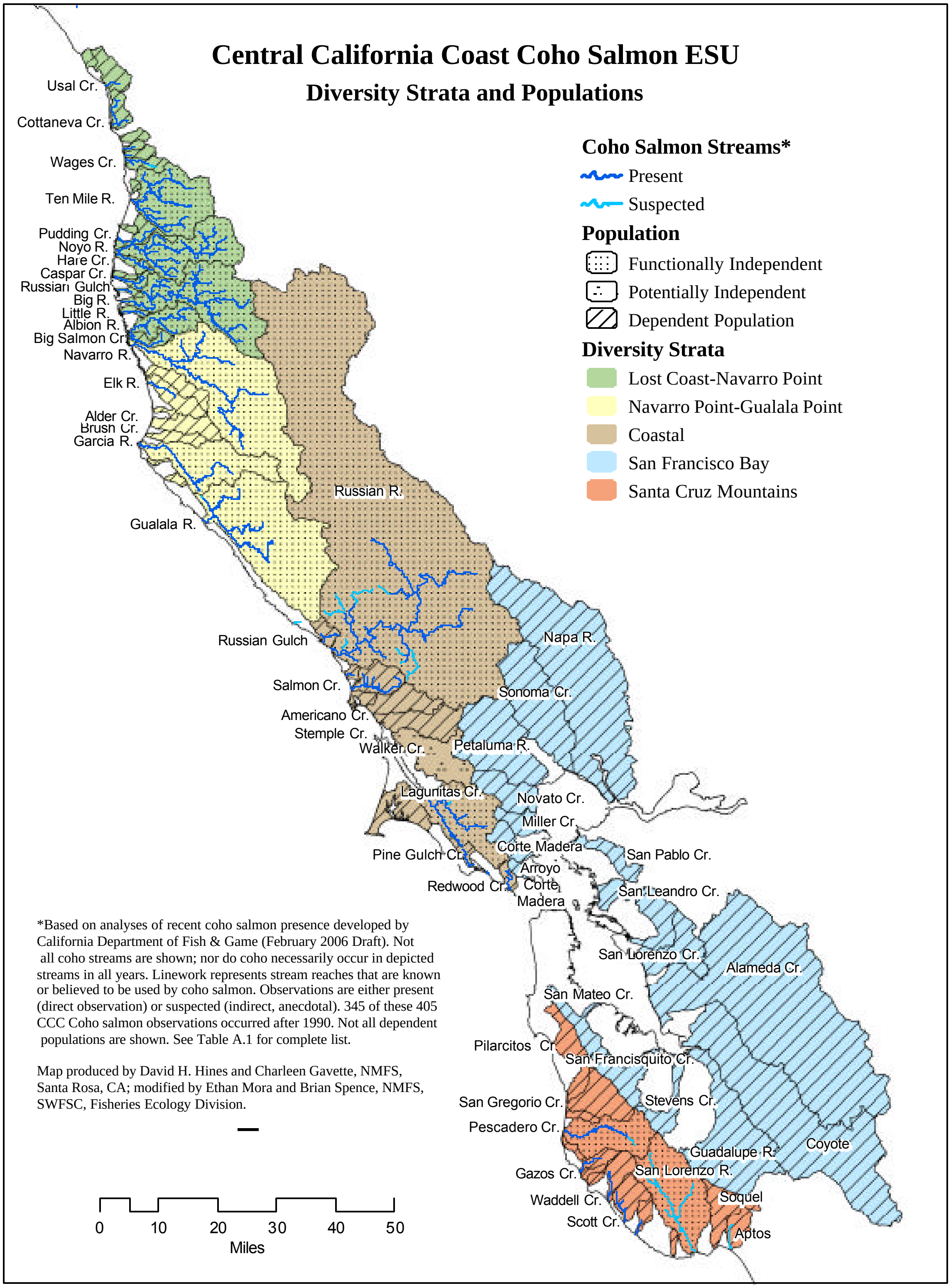


Plate A1. Diversity strata for populations of Central California Coast coho salmon. Based on Bjorkstedt et al. (2005) with modifications described in Appendix A.

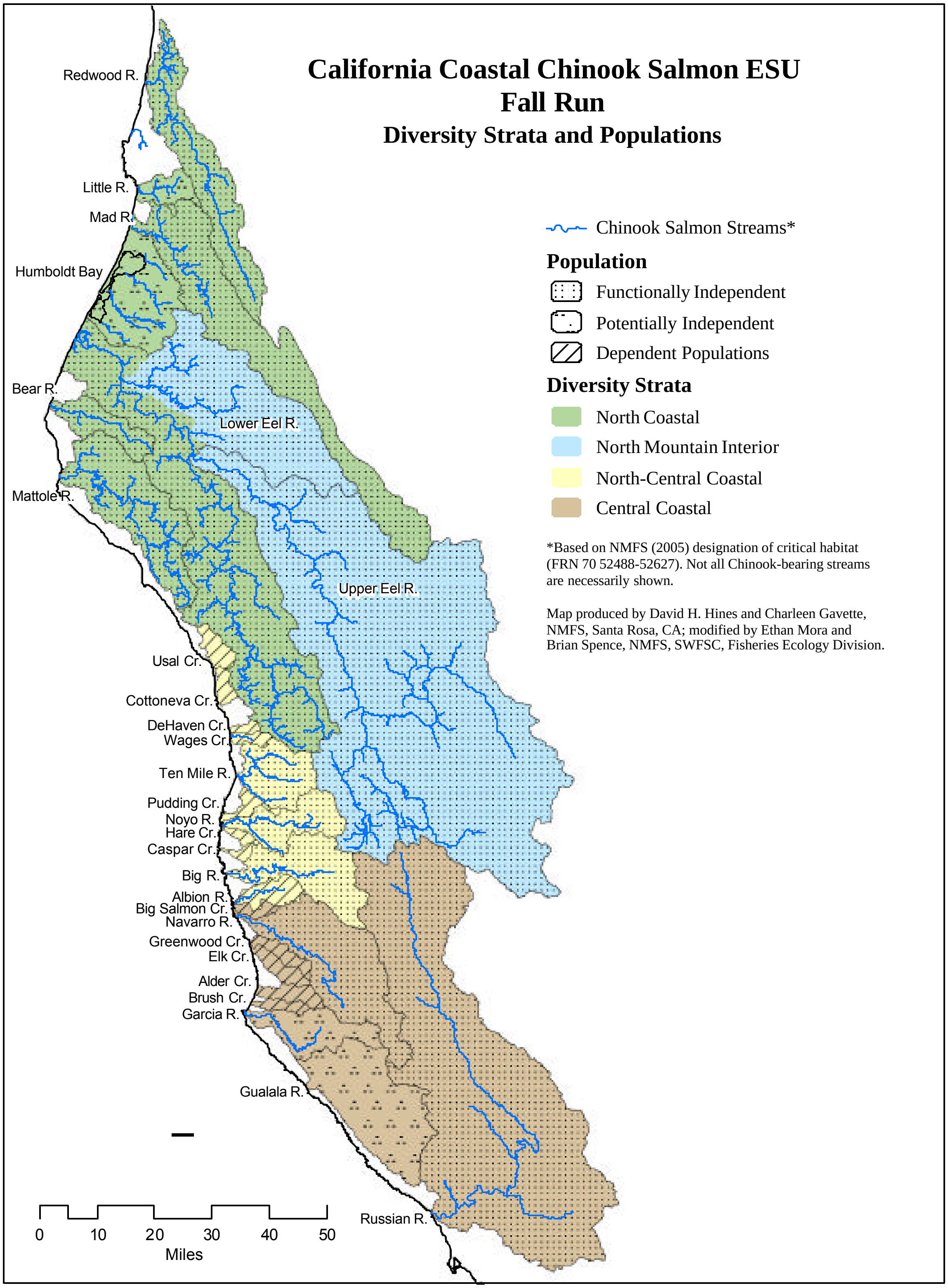


Plate A2. Diversity strata for populations of fall-run California Coastal Chinook salmon. Based on Bjorkstedt et al. (2005) with modifications described in Appendix A.

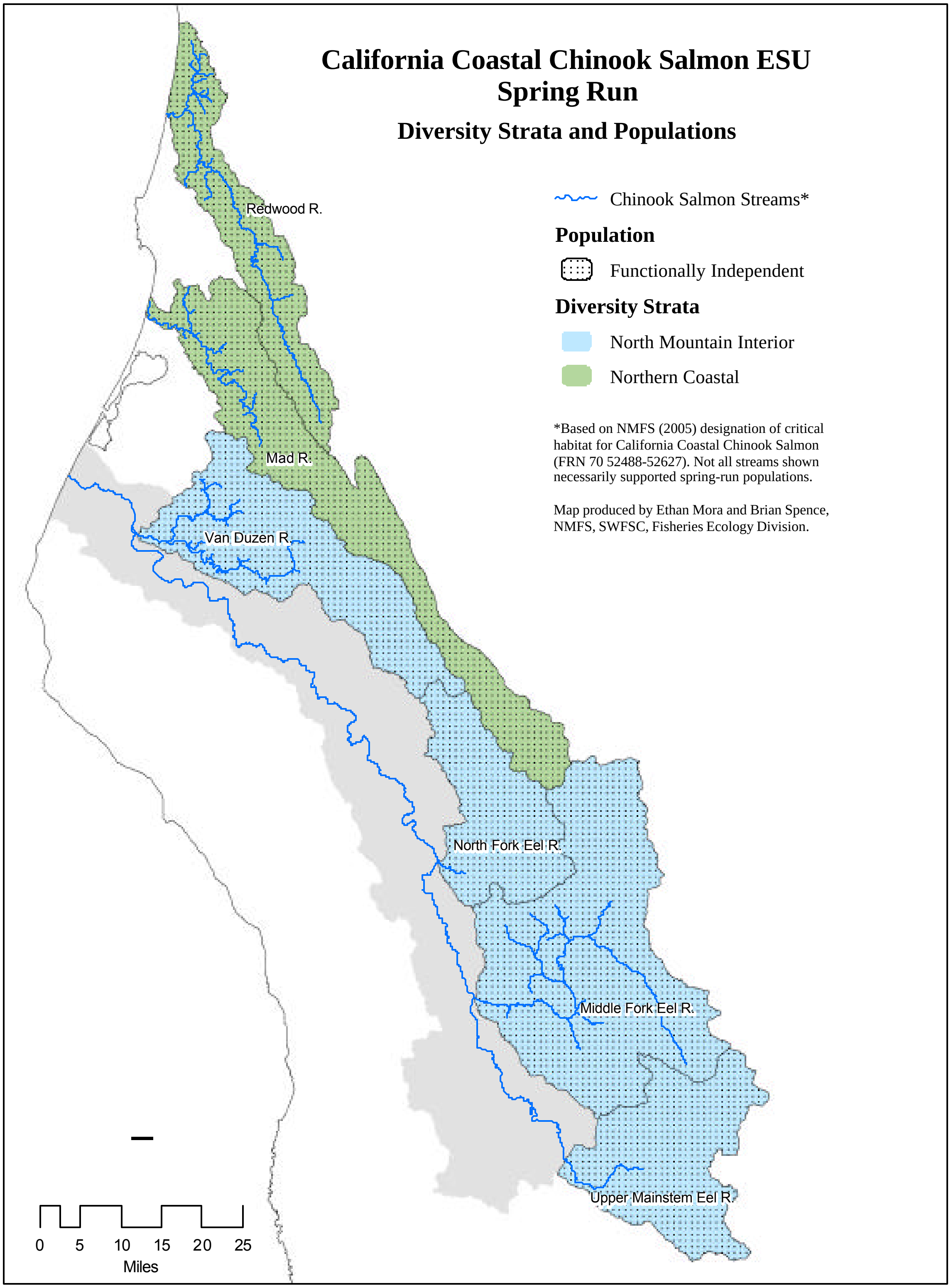


Plate A3. Diversity strata for populations of spring-run California Coastal Chinook salmon. Based on Bjorkstedt et al. (2005) with modifications described in Appendix A.

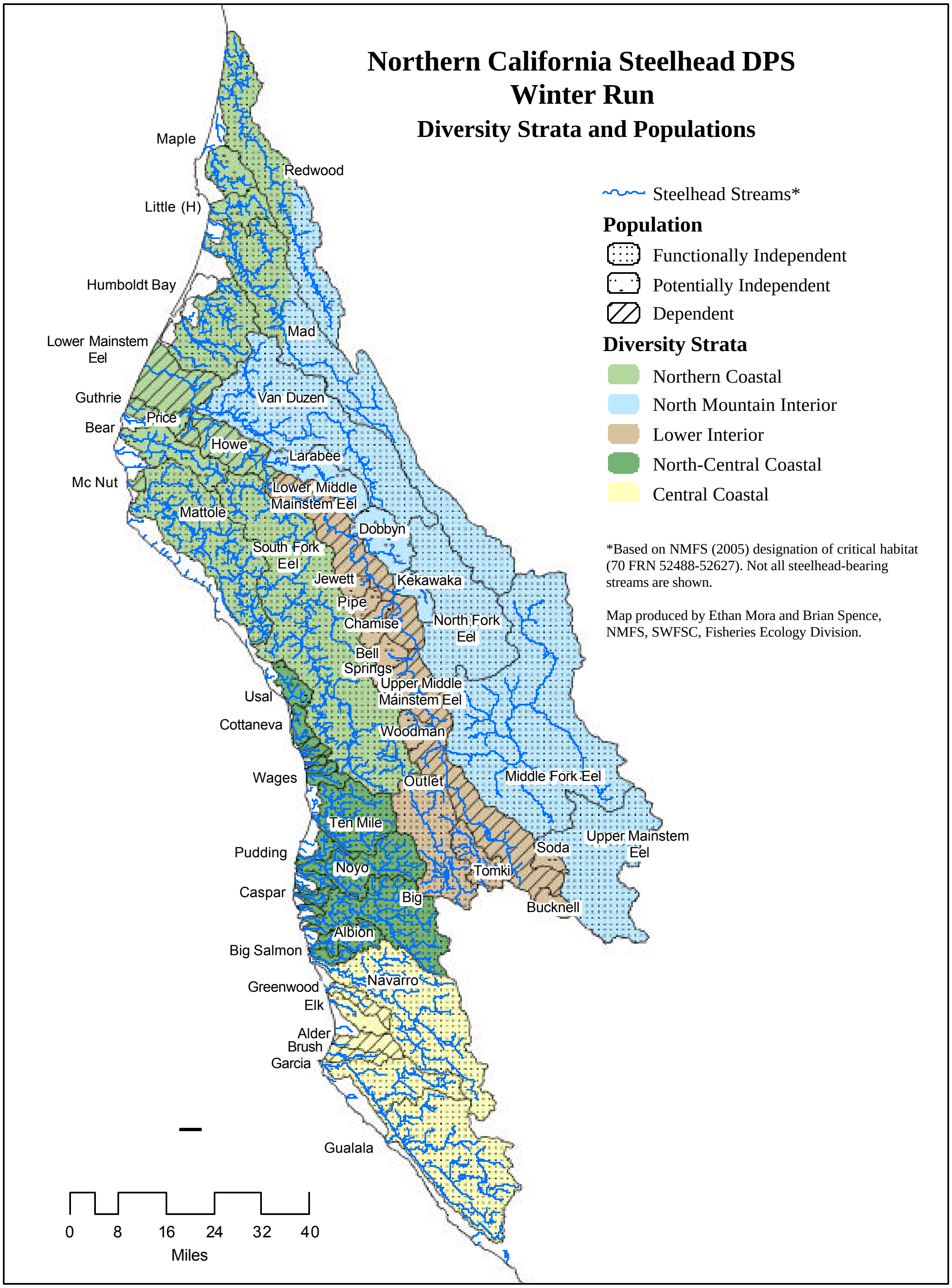


Plate A4. Diversity strata for populations of winter-run Northern California steelhead. Based on Bjorkstedt et al. (2005) with modifications described in Appendix A.

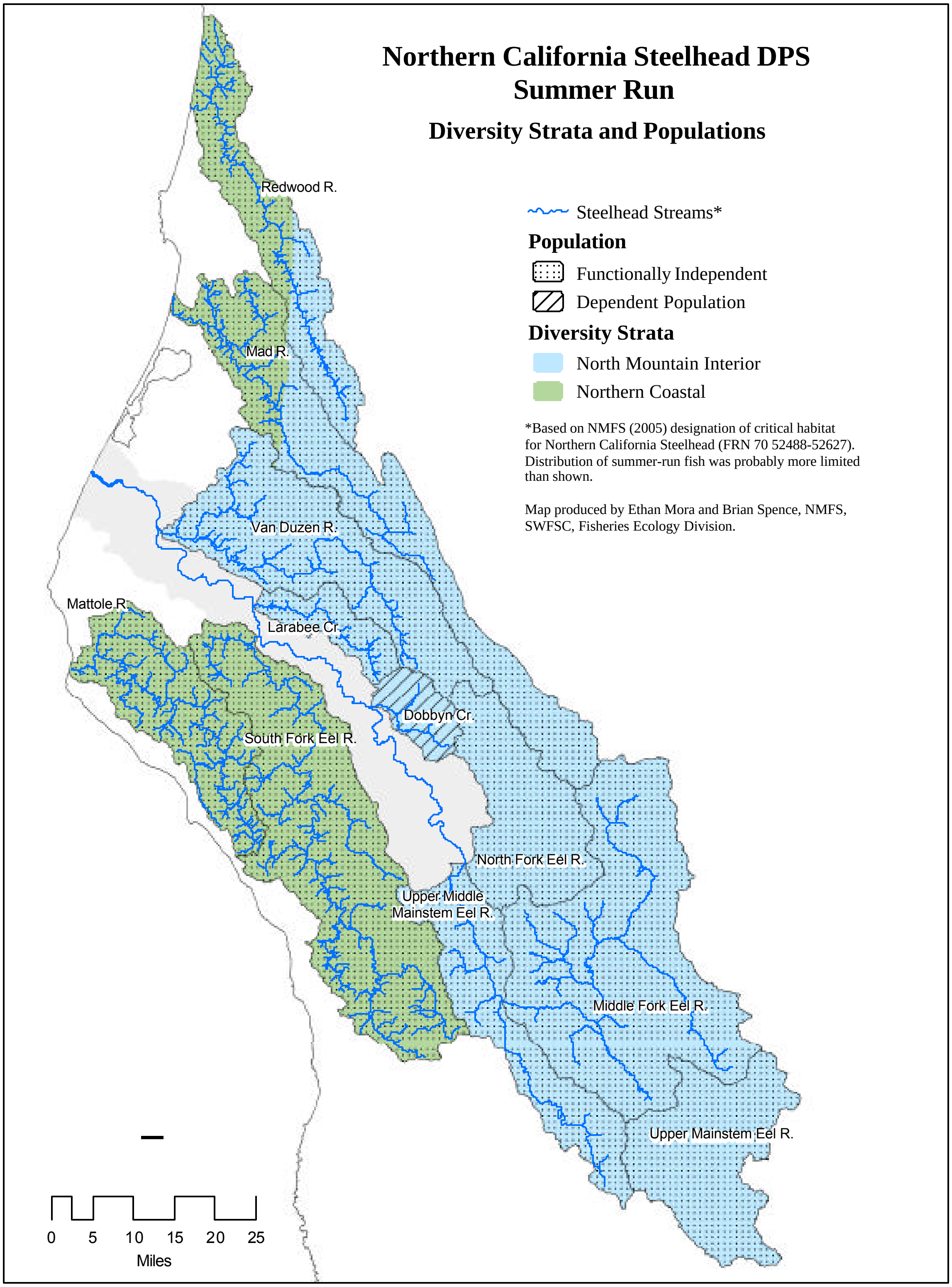


Plate A5. Diversity strata for populations of summer-run Northern California steelhead. Based on Bjorkstedt et al. (2005) with modifications described in Appendix A.

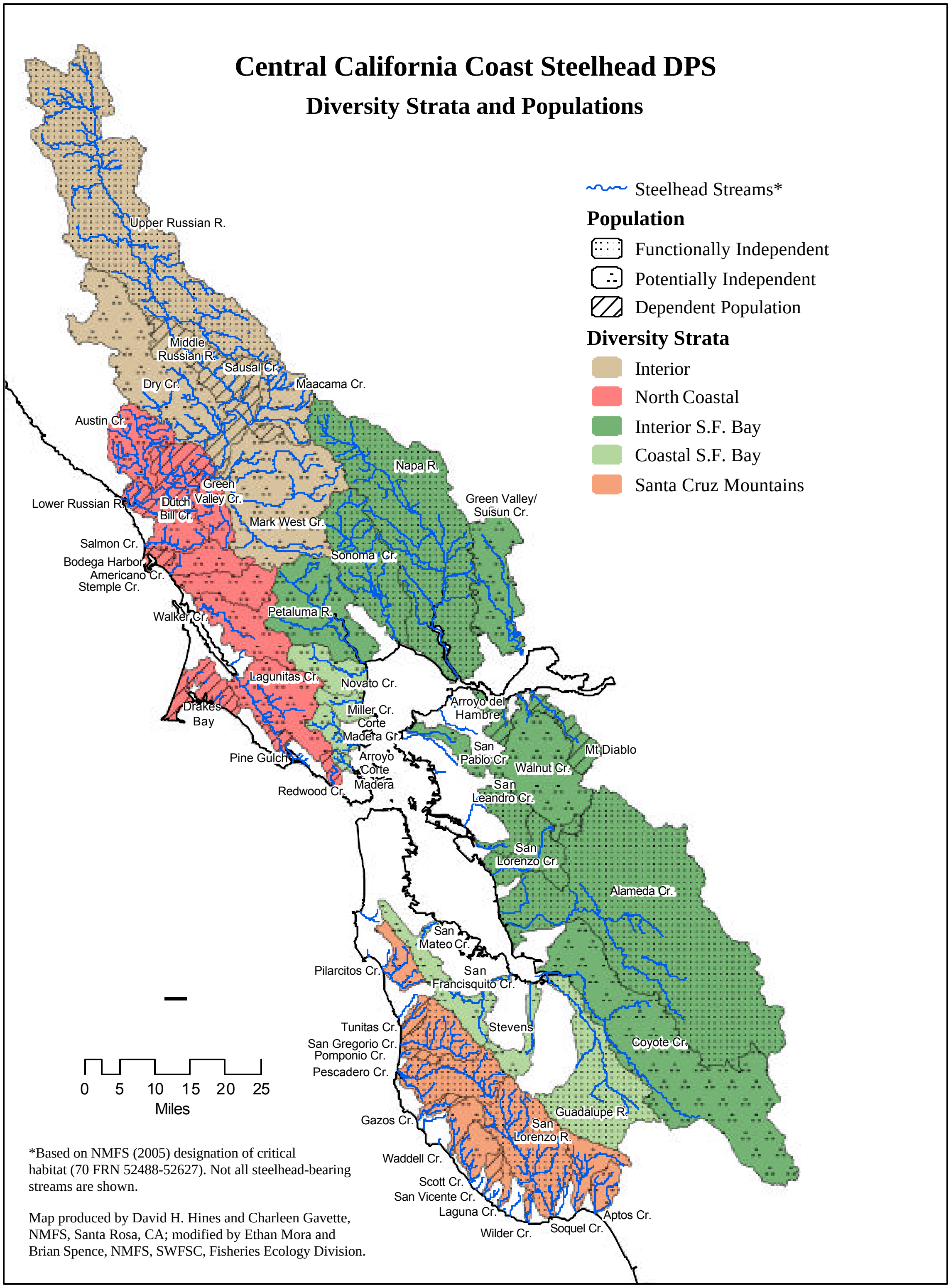


Plate A6. Diversity strata for populations of Central California Coast steelhead. Based on Bjorkstedt et al. (2005) with modifications described in Appendix A.

Appendix B. Discussion of Density Criteria and their Application

As noted in the main body of this report, the NCCC TRT spent substantial time discussing the appropriateness and application of density criteria. Much of the discussion revolved around four central issues: (1) how to estimate density in situations where substantial habitat is no longer accessible due to impassible or so degraded as to preclude use by salmonids; (2) whether the density criteria (or abundance targets dictated by density criteria) for populations at "low risk" were sufficiently precautionary or overly so; (3) whether it was more appropriate to express density criteria in terms of fish per *IPkm* or fish per total accessible kilometers; and (4) whether adjustments to the criteria should be made to account for potential bias in estimates of IP. The first of these issues was covered in the main body of this report. The remaining three issues are treated in the sections that follow.

Are the density criteria sufficiently precautionary or overly so?

During the course of our discussions, some TRT members initially expressed concern that the implementation of low-risk density criteria might result in abundance targets that are unrealistically high for certain watersheds (i.e., they might exceed what was historically possible), particularly in watersheds where the IP bias index (see discussion below) suggests that the IP model may overestimate historical habitat potential. Conversely, other TRT members worried that perhaps the criteria might not be precautionary enough. Ultimately, the TRT concluded that the proposed density criteria—40 spawners per IP-kilometer for watersheds with the minimum amount of potential habitat (*IPkm*) thought to be capable of sustaining an independent population, declining to 20 spawners per kilometers for watershed with 10-fold the habitat potential of the minimum watershed—represented a reasonable "floor" for interim criteria in lieu of more sophisticated population viability analyses.

This conclusion is based on several lines of reasoning. First, recall that for each species, we have defined a minimum threshold of potential habitat (32 *IPkm* for coho salmon, 20 *IPkm* for Chinook salmon, and 16 *IPkm* for steelhead) that was required for the population to be considered viable in isolation when populations were functioning at or near their historical carrying capacity. Thus, estimates of carrying capacity in relatively undisturbed systems might provide a reasonable basis for determining spawner density criteria for these smallest systems. Unfortunately, the scientific literature lacks estimates of carrying capacities for relatively pristine systems. Our estimate of 40 spawners/*IPkm* was based on the analysis of Bradford et al. (2000), who examined inflection points in hockey-stick stock-recruitment curves for 14 coho salmon populations and found that on average full smolt recruitment occurred at spawner densities of 19 female per kilometer (which assuming a sex ratio slightly biased in favor of males translates to roughly 40 spawners/km). In using this value as the basis for spawner density criteria, several things should be kept in mind. First, the watersheds used to estimate spawner densities at full smolt recruitment represented habitats with varying levels of human disturbance, with few in relatively pristine condition. Thus, historical carrying capacities were, in all probability, somewhat higher on average than those suggested by data collected post human disturbance. Additionally, to estimate spawner densities, Bradford et al. (2000) divided adult spawner abundance by an estimate of total accessible kilometers of habitat (although they acknowledge that, in some cases, these estimates may not include all possible habitat). In contrast, the NCCC TRT proposes using $IPkm$ as the denominator in calculating density (see discussion below). Within the NCCC Recovery Domain, the ratio of $IPkm$ to total accessible kilometers typically averages about 0.6 for coho salmon. Assuming that this ratio is similar in other streams in the Pacific Northwest, this would again suggest that densities at carrying capacity may have been higher than suggested by our density criteria.

Ideally, information on historical population abundance prior to extensive human disturbance could provide a means of validating the proposed density criteria. Unfortunately, data on historical adult abundance of salmon and steelhead are extremely scarce in the NCCC Recovery Domain, and where such estimates are available, they are for time periods during and after substantial human-caused impacts had already occurred. The only published comprehensive (in geographic scope) coastwide estimates of historical abundance are contained in a report prepared by CDFG (1965). Additionally, there are historical counts of salmon and steelhead at two dams in the domain (Benbow Dam on the South Fork Eel River and Sweasy Dam on Mad River), as well as of coho salmon and steelhead at Waddell Creek. In the sections below, we compare these historical estimates with our abundance targets. Further, we apply our density criteria to populations in nine coastal watersheds of Oregon and compare projected abundance targets with estimates derived from cannery pack records from the late 1800s and early 1900s.

Comparison of population abundance targets with historical estimates of abundance

1965 CDFG coastwide estimates

A report published by CDFG (1965) provides estimates of abundance of Chinook salmon, coho salmon and steelhead for most major watersheds in California. For coastal watersheds, these estimates are based primarily on the professional judgment of local biologists working in the area, who “made comparisons with better-studied streams” and, in a few instances, had some additional data to assist them, such as dam counts (e.g. Mad and Eel rivers) or harvest information. Though there is very high uncertainty surrounding these estimates, they nevertheless provide the only basis for assessing whether the abundance projections produced by application of the density criteria fall within or outside a plausible range across the recovery domain.

Comparison of the NCCC TRT density-based population projections and the 1965 CDFG estimates indicates that, for many systems, there is reasonably good concordance between the two values (Table B.1). For most populations on the Mendocino and Humboldt county coasts, the projected low-risk abundances tend to be somewhat lower than the CDFG estimates, whereas in more southern populations, the projected abundances tend to be somewhat higher than the CDFG estimates (particularly for coho and Chinook salmon). Part of this pattern almost certainly reflects the fact that in the 1960s, while all populations in the domain had likely experienced significant declines due to a variety of human impacts (CDFG 1965), the southern portion of the domain was more severely disturbed. However, it may also

Table B.1. Comparison of projected spawner abundances satisfying the NCCC TRT "low risk" density criteria with population estimates taken from CDFG (1965).

Population	Projected Low-risk Abundance	CDFG 1965 Estimate	Population	Projected Low-risk Abundance	CDFG 1965 Estimate
CCC-Coho salmon			*NCC Steelhead*		
Ten Mile River [23]	3,700	6,000	Redwood Creek (H) [1]	6,000	10,000
Noyo River [25]	4,000	6,000	Mad River [5]	11,200	6,000
Big River [30]	5,600	6,000	Eel River - Full [7]		
Navarro River [34]	5,700	7,000	Van Duzen River [E2]	10,900	10,000
Garcia River [39]	2,800	2,000	South Fork Eel River [E3]	23,600	34,000
Gualala River [41]	6,200	4,000	North Fork Eel River [E5]	7,500	5,000
Russian River [42]	15,600	5,000	Middle Fork Eel River [E7]	11,700	23,000
San Lorenzo River [63]	4,400	1,600	Mattole River [14]	12,300	12,000
			Ten Mile River [23]	4,100	9,000
CC-Chinook salmon			Noyo River [25]	4,000	8,000
Redwood Creek (H)	3,400	5,000	Big River [30]	6,300	12,000
Mad River [5]	3,000	5,000	Navarro River [34]	9,200	16,000
Eel River	22,100*	55,000	Garcia River [39]	3,400	4,000
Mattole River [14]	4,000	5,000	Gualala River [41]	9,600	16,000
Ten Mile River [23]	2,300	0			
Noyo River [25]	2,200	<50	*CCC Steelhead*		
Big River [30]	3,200	0	Russian River	40,800**	50,000
Navarro River [34]	3,600	0	San Lorenzo River [63]	4,900	19,000
Garcia River [39]	2,000	0			
Gualala River [41]	4,000	0			
Russian River [42]	11,700	500			

* denotes aggregate abundance for Upper and Lower Eel River independent populations

** denotes aggregate abundance of independent steelhead populations in the Russian River; excludes dependent populations reflect a north-south gradient in the degree of IP-bias (discussed below). Overall, however, comparison with the 1965 estimates strengthens the argument that the projected abundances are within a plausible range. We do note, however, that if the 1965 abundance estimates, made at a time when habitat degradation from land and water use were already widespread, are even somewhat close to true abundances, then the density-based low-risk abundances suggested by our criteria are more appropriately viewed as minimum "floors," rather than indicative of historical carrying capacities.

Waddell Creek coho salmon and steelhead estimates

Adult population abundance estimates are available for both coho salmon and steelhead in Waddell Creek from the study of Shapovalov and Taft (1954). Adult salmon and steelhead were counted at a weir placed about 2.5 km upstream of the ocean and 1 km above the uppermost extent of tidewater. During the nine-year period covering spawning seasons 1933-34 to 1941-42, the average annual adult (including jacks) run size for coho salmon was estimated to be 313 (range 111-748). During the same period, the estimated abundance of adult steelhead was 481 (range 428-554) [33].

Bjorkstedt et al. (2005) concluded that Waddell Creek likely supported a dependent population of coho salmon, as total *IPkm* in the basin (9.12 *IPkm*) was only about 29% of that deemed necessary to support an independent population. Nevertheless, if we were to apply the density of spawners used to produce abundance targets for the smallest independent populations (i.e., 40 spawners per *IPkm*), we would arrive at an estimated abundance of about 365 spawners for coho salmon. For steelhead, we estimated a total of 16.24 *IPkm* for the Waddell Creek basin, which translated to a target abundance of 649 spawners (which we rounded to 600) for this independent population. Consequently, the estimated historical abundance between 1933 and 1942 averaged about 86% and 80% of the projected abundance targets for coho salmon and steelhead, respectively, based on a spawner density of 40 spawners per *IPkm*.

Although the density-based abundance targets are slightly higher than abundances recorded in the 1930s and 1940s, it is important to consider the historical context. Foremost, the condition of the Waddell Creek watershed at the time of the Shapovalov and Taft study was far from pristine. Shapovalov and Taft (1954) describe Waddell Creek in the following terms:

> *" Some changes from the primitive condition of the area have taken place as a result of human usage. The redwood forest of the watershed below Big Basin was logged off by 1870 and is now*

[33] Estimated run sizes include weir counts plus estimates of numbers of adults that spawned below the weir or that jumped over the weir during high flows. Coho salmon and steelhead totals from Table 9 (pg. 47) and Table 35 (pg. 138), respectively, in Shapovalov and Taft (1954).

covered by a second growth. The early lumbering operations have resulted in the creation of several semipermanent log jams and temporary accumulations of logs, which have hastened erosion of the stream banks, with consequent increase in silting during flood stage."

The statements of Shapovalov and Taft likely understate the degree to which Waddell Creek had been affected by clearing of the redwood forests. The first steam sawmill in Santa Cruz County was built near the confluence of the East and West forks of Waddell Creek in 1862, and the basin was heavily logged between 1862 and 1875. Big Basin Redwoods State Park was established in 1902 to protect the last significant stand of old-growth redwoods in the Santa Cruz Mountains[34]. At the time Shapovalov and Taft conducted their research, Big Basin State Park covered an area of fewer than 10,000 acres, all of which was in the headwater regions of Waddell Creek basin, upstream of the two known natural barriers to anadromy on the East and West branches of Waddell Creek. (Major additions to the park, including the middle and lower reaches containing most of the coho salmon and steelhead habitat, came between the late 1950s and 1980s). Consequently, virtually all portions of the watershed accessible to coho salmon had been extensively disturbed prior to the onset of Shapovalov and Taft's study. We do not believe it unreasonable to think that such disturbance would have resulted in at least a 20%-25% reduction in productive capacity for coho salmon and steelhead. Consequently, we do not believe that density-based criteria produce predictions of capacity that are unrealistic for either species. This is encouraging because Waddell Creek lies near the southern edge of the coho salmon's historical range, where bias associated with the IP model is expected to be greatest.

We note that there were two active hatcheries in Santa Cruz County during the period Shapovalov and Taft conducted their study. However, our review of historical records indicate that coho salmon and steelhead were planted into Waddell Creek on only a few occasions and in small numbers during the Shapovalov and Taft years[35]. Specifically, Waddell Creek received a planting of 15,000 coho salmon fry in 1933 and plantings of steelhead fry totaling 36,000 fish in 1930, 34,000 fish 1932, and 1,005 fish 1933. We conclude that the potential influence of stocking on the adult counts was likely small for the following five reasons: (1) the total numbers of fish stocked were small; (2) the stocked fish were primarily fry (except perhaps the 1,005 steelhead released in 1933), which typically have very low survival rates; (3) the duration of stocking was limited to one of eight years for coho salmon and three of eight years for steelhead (with only 1,005 fish released in one of those years); (4) the majority of steelhead were released

[34] A second smaller old-growth redwood stand (about 40 acres) remained unharvested near Felton.

[35] Source: State of California, Department of Natural Resources, Division of Fish and Game, Record of Fish Distributions. Compiled by Dayes (1987).

in the headwaters of Big Basin State Park, upstream of barriers to anadromy; and (5) adult counts in years following stocking are not obviously higher or lower than in years without planting. Therefore, we consider the counts to be a reasonable indicator of the natural carrying capacity for this period.

Eel River coho salmon, Chinook salmon, and steelhead.

Counts of coho salmon, Chinook salmon, and steelhead were made at Benbow Dam on the South Fork Eek River from 1938 to 1975. Benbow Dam was located about 133 km upstream of the ocean, and about 67 km upstream of where South Fork Eel River enters the mainstem. Counts at this dam, consequently, represent only a portion of the independent populations of Chinook salmon, coho salmon, and steelhead delineated in the population structure report.

To compare historical abundance estimates with density-based projections for coho salmon, Chinook salmon, and steelhead, we estimated the fraction of total *IPkm* for each population that occurred upstream of the Benbow Dam and then multiplied this fraction by the overall abundance targets to obtain estimates of the contribution of above-dam habitats to the total population targets. We then compared these estimates to historical counts from 1938 to 1950 at the dam. This time period was presumed to be when the influence of human impacts was lowest (for the period of record), as evidenced by the fact that counts during these periods were generally higher on average than in the decades that followed. We note that the period 1938 to 1950 does not represent a particularly favorable period with respect to oceanic conditions. Data presented in Hare et al. (1999) indicates that commercial catch of coho salmon in California and Oregon was relatively low from 1938 through the mid-1950s, and then increased substantially from the late 1950s into the mid-1970s. This contrasts with the Benbow Dam coho counts, which averaged only about 30% of the 1938-1950 counts from 1951 to 1975. The continued decline of coho in the South Fork Eel after 1950, when production was increasing elsewhere in the California Current system, indicates that the high counts recorded in the 1930s and 1940s were not the result of unusually favorable ocean conditions. In fact, the first half of this period occurred during a positive phase of the Pacific Decadal Oscillation, conditions that typically result in lower salmon production in Oregon, Washington, and California (Hare et al. 1999).

For the South Fork Eel River, our density-based abundance projections for populations upstream of Benbow Dam were 6,836 for coho salmon, 4,415 for Chinook salmon, and 15,732 for steelhead[36]. In all three cases, these projections are well below the recorded average abundances for these three species

[36] Because the total *IPkm* for coho salmon, Chinook salmon, and steelhead populations that include the South Fork Eel River basin are 10 times the minimum *IPkm* required for an independent population, we assume a spawner density of 20 spawners per *IPkm* for all three species. Data on historical counts from StreamNet (Available online at: www.streamnet.org).

during the 1938-1950 period (Table B.2): projected abundances were about 51%, 37%, and 91% of the dam counts for coho salmon, Chinook salmon, and steelhead during the period. Thus, there is strong evidence that our methods do not overestimate the historical carrying capacities of these three species in the South Fork Eel River basin upstream of Benbow Dam (see further discussion below).

Our conclusion gains strength when we consider that, for a number of reasons, the counts at Benbow Dam underestimate the total population sizes for the South Fork Eel River. First, the fish counts at Benbow Dam do not take into account harvest of salmon in ocean and in-river fisheries downstream of the dams, which was considerable during the late 1930s to 1950s. Although commercial catch statistics for California are generally not available for this period (INPFC 1979), local newspaper accounts indicate that recreational fishers were deeply concerned that ocean troll fisheries were severely depleting Eel River salmon populations during this time. One article in the Ferndale Enterprise from September 1937 reports that commercial troll fishers harvested about 100,000 lbs of salmon in a single day in the waters off of the Eel River mouth. They protested that this equated to about 5,000 20-lb Chinook salmon, which was more than the total take in sport fisheries for an entire season (Van Kirk 1996d).

Second, the counts at Benbow Dam were likely influenced by the legacy of historical commercial net (seine and gill-net) fisheries that operated in the lower Eel River from the 1850s into the 1920s. By the 1890s, these fisheries had caused a precipitous decline in the number of salmon returning to the Eel River. Between 1877 and 1889, canneries in the lower Eel River basin processed in the neighborhood of three-quarters of a million pounds of salmon annually. Increasing public concern resulted in prohibitions on seining in 1913 and gill-netting in 1922 (Lufkin 1996). Commercial troll fishing was initiated in 1916 and soon replaced the net fisheries as the dominant Eel River fishery. Newspaper accounts in the 1930s and 1940s periodically make reference to the devastating impact that net fisheries had on Eel River salmon populations, from which the populations apparently never fully rebounded (Van Kirk 1996a,b,c).

Table B.2. Comparison of average historical (1938-1952) counts of adult migrant coho salmon, Chinook salmon, and steelhead at Benbow Dam, South Fork Eel River, with density-based abundance targets developed by the TRT.

Population	Historical counts of adult migrants: Mean (range)	Years	Total *IPkm* above dam	Projected number of spawners above dam based on density criteria
S. Fk. Eel River Coho salmon	13,514 (7,370-25,289)	1938-1950	341.8	6,836
S. Fk. Eel River Chinook salmon	11,782 (3,424-21,011)	1938-1950	220.8	4,415
S. Fk. Eel River steelhead	17,343 (12,995-25,032)	1938-1950	786.6	15,732

Third, a significant amount of habitat degradation had likely already occurred in the South Fork Eel River by the late 1930s, when the counts began. Logging of the coastal redwood forests, which began in the 1800s throughout much of the North Coast, began somewhat later in the South Fork Eel River basin, due to the fact that much of the drainage was not easily accessible by road (BLM et al. 1996). However, completion of the Redwood Highway (Hwy 101) in the late 1920s, which runs along the South Fork Eel River, allowed rapid expansion of logging in the South Fork Eel River basin.

Fourth, for a number of reasons, counts at Benbow Dam almost certainly underestimate the total number of fish that passed upstream of the dam. The weekly reports prepared by those operating the Benbow Dam facilities indicate that there were two ladders (south and north) around the dam. During the 1937-38 and 1938-39 seasons, both ladders were monitored on a regular basis. However, frequent landslides plagued the north ladder, and by the 1940-41 season, counts were made almost exclusively at the south ladder. The degree to which rocks and soil deposited into the north ladder precluded use by salmon and steelhead is uncertain. However, various notes from the weekly reports indicate that, under certain flow conditions, the number of fish using the north ladder was substantial and even exceeded numbers using the south ladder[37]. Indeed, a memo written by Shapovalov (1946) indicates that the ladder operator during the 1944-45 and 1945-46 seasons estimated that 900 steelhead passed through the north fishway during the 1945-1946 season (about 7% of the number of steelhead counted at the south ladder that year), and that 1,000 salmon and steelhead passed through the north ladder in the 1945-1946 season (about 2% of the south ladder count). These estimates are not included in the published annual totals. The same operator made a note on March 19, 1945 that he saw *"a few fish hurdling No*. [north] *ladder. Same condition has been going on for 3 years, so absurd to change tallies now.*" (Coons 1945). Thus, it appears safe to assume that passage of uncounted fish through the north ladder was a fairly regular occurrence. Additionally, notes on water clarity were routinely made in the weekly reports, and they frequently describe the water a muddy, murky, or cloudy. In some cases, the observers make reference to "difficult conditions" for census work. Under such conditions, it seems likely that some fish were missed by observers. And finally, there were many instances where flows were so high that the station had to be closed. Collectively, these pieces of evidence indicate that the counts should be viewed as partial counts, Although there is no means for estimating what fraction of the total run was sampled in any given year, suffice it to say that total escapement likely exceeded the recorded counts.

[37] All indications are that the north ladder effectively passed fish under a narrower range of flow conditions than did the south ladder.

Finally, it is well documented that in the first years of operation (1932-1937), the fish ladders at Benbow Dam functioned poorly, which prompted considerable public concern and outrage (Van Kirk 1996d). On February 28, 1936, the Ferndale Enterprise wrote:

> *"The soul-sickening spectacle of thousands of splendid steelhead and salmon—all heavy with spawn—sentenced to a miserable death without completing their life cycle because the Department of Natural Resources State of California has failed to provide adequate fish ladders at Benbow Dam, on the Eel River, has aroused sportsmen of that district"*

It is unclear how problems with fish passage in the mid-1930s may have affected populations in subsequent years, but it seems safe to assume that any effect was negative.

All of these pieces of evidence would suggest 1) that carrying capacities during the period 1938-1950 were substantially higher than counts at Benbow Dam would indicate, and 2) that historical capacities prior to arrival of Euro-Americans were likely higher still by a good margin.

Conversely, there was some hatchery activity during the 1930s and 1940s on the Eel River, which potentially could artificially inflate adult counts at Benbow Dam. A few hatcheries operated in Humboldt County during this period. The most likely candidate for plants into the Eel River was the Fort Seward Hatchery. Fort Seward hatchery, which operated from 1916 to 1941 was located on the Eel River mainstem approximately 36 km upstream of the confluence of the South Fork. Between 1935 and 1941, the hatchery distributed an average of about 579,000 steelhead and 480,000 Chinook salmon to streams and rivers of Humboldt County, with an additional 170,000 steelhead on average going to streams in Mendocino County from 1938 to 1941. Coho salmon were also released from 1935-1938, with an average annual total of about 693,000. Unfortunately, the distribution locations of these fish are not known; thus, it is unclear if any of these fish (and if so, how many) were released into the South Fork Eel River and so may have influenced counts at Benbow Dam.

We do note that Benbow Dam counts before and after the "plausible" stocking periods indicate no clear changes in abundance. Counts of Chinook salmon were slightly lower (~11,000) during the period potentially affected by stocking (1938-1944) than in 1945-1950, the period following stocking, when the average count was about 12,700 adults. Likewise, counts of coho salmon from 1938-1940 (the years that would have been directly affected by plantings if they occurred in the South Fork) are lower on average (~9,400) than those in the period from 1941-1950 (~14,900) when no planting occurred. Only for steelhead were counts at Benbow dam slightly lower (~15,600) in the years after stocking (1945-1950) than in the years potentially affected by stocking (1938-1944; average ~18,800). Again, we have no direct evidence that stocking actually took place in the South Fork Eel. But the lack of evidence of substantial population declines when Fort Seward hatchery ended production indicates that any effects of stocking were either small or swamped out by other factors.

Mad River coho salmon, Chinook salmon, and steelhead.

Counts of coho salmon, Chinook salmon, and steelhead were made at Sweasy Dam on the Mad River from 1938 to 1964. Sweasy Dam was located some 15 km upstream of the river mouth. Thus, counts at the dam represent only a portion of the total population sizes for the Mad River basin. Density-based projections for coho salmon, Chinook salmon, and steelhead were made by estimating the percentage of total *IPkm* for each population that occurred upstream of Sweasy Dam (27%, 51%, and 76% for coho salmon, Chinook salmon, and steelhead, respectively) and then multiplying this fraction by the overall abundance targets to obtain estimates of the contribution of above-dam habitats to the total population targets. These estimates were then compared to historical counts from 1938 to 1950 at the dam, as again, this period likely was the least affected by human activities.

For the Mad River, comparison of projected abundances versus historical counts produces more equivocal results. Abundance projections for populations upstream of the Sweasy Dam were 1,334 for coho salmon, 953 for Chinook salmon, and 8,430 for steelhead (Table B.3)[38]. For Chinook salmon, the average count from 1938 to 1950 exceeds projected abundance by about 38%. Conversely, for coho salmon and

Table B.3. Comparison of average historical counts of adult migrant coho salmon, Chinook salmon, and steelhead at Sweasy Dam, Mad River from 1938-1950 compared with density-based abundance targets developed by the NCCC TRT.

Population	Historical counts of adult migrants: Mean (range)	Years	Total IP above dam (% of basin total)	Projected number of spawners above dam based on density criteria
Mad River coho salmon	395 (73-515)	1938-1950	41.7	1,334
Mad River Chinook salmon	1,312 (484-3,139)	1938-1950	47.7	953
Mad River steelhead	4,401 (3,110-6,650)	1938-1950	421.5	8,430

[38] For Chinook salmon and steelhead, total *IPkm* for the Mad River basin exceed 10 times the minimum *IPkm* required for an independent population; thus, we assume a spawner density of 20 spawners per *IPkm* for these two species. For coho salmon, the minimum required spawner density for a basin with 152.9 *IPkm* is 32 spawners/*IPkm*.

steelhead, the projected abundances exceed the average historical dam counts. Thus, while the historical data indicate that the abundance projections do not over-predict historical carrying capacity for Chinook salmon, the same cannot be said for coho salmon and steelhead at first glance. We do note that the projected abundance for steelhead is subject to substantial uncertainty, as a considerable amount of predicted *IPkm* lies upstream of a partial natural barrier near Bug Creek that apparently can limit access to a substantial amount of habitat in some years.

There remains uncertainty as to operating procedures at the fish ladder and whether there existed the capability to block fish passage during periods when counts were not made. We attempted to obtain information from California Department of Fish and Game regarding dam and counting operations, but thus far no one has come forth with definitive information that would enable us to ascertain whether the counts represent full or partial counts, though obtaining full counts at any such facilities under all flow conditions is usually quite difficult.

A second potential reason that dam counts for coho salmon and steelhead were lower than predicted by our model likely relates to the condition of the Mad River watershed at the time counts were made. Extensive clearing of the redwood forests along the Mad River downstream of Bug Creek (the apparent upper distributional limit coho and Chinook salmon) had occurred by the end of the 1800s (Carranco 1982; HBMWD 2004). Undoubtedly, substantial modification of habitat, including removal of large wood, loss of riparian canopy, increased sedimentation, and other impacts of logging had substantially reduced carrying capacity of the Mad River and its tributaries at the time the dam counts were made.

Additionally, the Mad River was subject to splash and crib dams, along with log drives during the early logging period (Carranco 1982). These activities would have resulted in substantial modification of habitat. Because roads and other transportation mechanisms were lacking, logs were typically moved downstream using several different types of dams. Splash dams were constructed across the stream channel to impound the river. Logs were dragged into the impoundment behind the dam or the stream channel below the dam. Water was then released suddenly by opening flood gates or blasting with explosives, and the water, logs, and anything their path was carried down the river until they were hung up on the next obstruction, where the splash-damming process was repeated. In other cases, semi-permanent crib or frame dams were built to impound water so that logs could be floated down from upstream or, when released, could transport logs downstream. Sometimes, release of water from multiple dams was carefully timed to facilitated transport of logs downstream. Often times, crews cut out any accumulations of wood downstream of a splash or crib dam to facilitate passage of logs when the dams were blasted or water was released. Cutaway dams were dams that were used only once, often to "float" logs that had accumulated in massive log jams resulting from splash and crib dam operations. Collectively, dam and log drive activities would have severely scoured stream channels, resulting in highly simplified habitats, reductions in the gravel remaining for spawning, and decreased stability of gravels during high flow conditions. Such impacts would have been particularly harmful to Chinook and coho salmon upstream of Sweasy Dam (particularly above Blue Slide Creek), as most of the potential habitat in this reach lies in the mainstem, rather than the steep tributaries that characterize this reach.

Density-based targets compared with historical abundance estimates for Oregon coho salmon

In addition to comparing TRT abundance targets with historical records from within the NCCC recovery domain, we also compared projected target abundances that would result if we applied our IP-based density criteria to populations with estimates of historical adult abundance for nine coastal watersheds in Oregon. The Oregon abundance estimates were based on cannery records from 1892 to 1915 (from Meengs and Lackey 2005). Meengs and Lackey (2005) estimated historical run sizes from cannery pack records through a series of steps including 1) converting salmon pack data (in cases) into pounds of salmon caught (by assuming a certain constant "waste" in processing); 2) converting pounds of salmon captured into numbers of adult fish (by assuming an average weight for adult fish of 4.46 kg); 3) converting numbers of harvested salmon into an estimate of total population sizes (assuming a specific catch efficiency rate); and 4) using abundance estimates from the five years of highest cannery pack in each watershed as indicative of run size[39]. Several other authors have estimated run sizes from cannery pack records using slightly different methods and assumptions (see e.g., Mullen 1981, Lichatowich 1989, Lawson et al. 2007), but overall the estimates derived by the various methods are generally fairly similar. We therefore present only the results of Meengs and Lackey (2005).

Estimation of projected target abundances using the NCCC TRT density criteria was straightforward. We obtained estimates of total coho salmon *IPkm* for each of the nine watersheds for which cannery records were available. Intrinsic potential coverages were provided by the CLAMS project (Kelly Burnett and Kelly Christiansen, US Forest Service, Pacific Northwest Research Station, Corvallis, Oregon). In calculating *IPkm*, we considered only reaches downstream of natural barriers (including barriers that have since been removed) so that the *IPkm* reflects those reaches historically available to coho salmon at the turn of the 20th century. For all nine populations, the estimated *IPkm* exceeded 320, or ten times the amount of *IPkm* required for population independence. Consequently, the target spawner density was

[39] Cannery pack is a function not only of numbers of fish, but also market forces. Consequently, years of highest cannery pack are not necessarily the years of highest abundance.

assumed to be 20 spawners per *IPkm*, and the target abundance 20 times the total *IPkm* for the watershed (see Table B.4).

A plot of *IPkm* versus historical estimates of abundance derived from cannery records (Figure 1) shows that there is a reasonably strong correlation between *IPkm* and historical abundance in these watersheds (R^2 = 0.51). When abundance is regressed against estimates of stream miles accessible to coho salmon unadjusted for IP, the relationship is slightly weaker (R^2 = 0.48)[40]. These results contributed to the NCCC TRT's confidence that the IP model provides a reasonable basis for scaling habitat.

Much more importantly, the data in Table B.4 indicate that for Oregon Coast coho salmon populations the abundance targets that would result from application of our density-based criteria are well below—by an order of magnitude—historical estimates of abundance. In all cases, the target abundance expressed as a percent of the historical estimates of abundance fall between about 3% and 12%. Thus, during the late 1800s and early 1900s, a period during which logging (and splash damming) was already well underway (Seddell and Luchessa 1982), spawner densities of coho salmon in coastal watersheds of Oregon were generally 10-fold to 20-fold higher than those required by our viability criteria. Even if we assume substantial bias in the IP model for the southern portion of the range, which lies in the NCCC Recovery Domain, it seems very unlikely that historical densities were lower than those the TRT has proposed for viability.

Table B.4. Comparison of historical abundance estimates and hypothetical density-based abundance targets for coastal watersheds in Oregon.

Population	Historical estimates of abundance derived from cannery records (Meengs & Lackey 2005)	*IPkm*	Estimated historical spawner density (Spawners/*IPkm*)	Projected abundance target based on MRD (20 spawners/*IPkm*)	Projected abundance target as percent of historical estimate
Nehalam	236,000	1,116	211	22,300	9.3%
Tillamook	234,000	537	436	10,700	4.7%
Nestucca	107,000	299	358	6,000	5.6%
Siletz	122,000	310	394	6,200	4.9%
Siuslaw	547,000	902	607	18,000	3.3%
Yaquina	65,000	385	169	7,700	12.3%
Alsea	153,000	466	328	9,300	5.9%
Coquille	342,000	883	387	17,700	5.3%
Coos	161,000	552	292	11,000	6.8%

[40] One might have expected IP to predict more of the variability; however, average IP scores are fairly constant across the nine coastal watersheds (range 0.56 to .67). Thus, the ability to evaluate whether *IPkm* is a better predictor of abundance than unadjusted stream kilometers is limited.

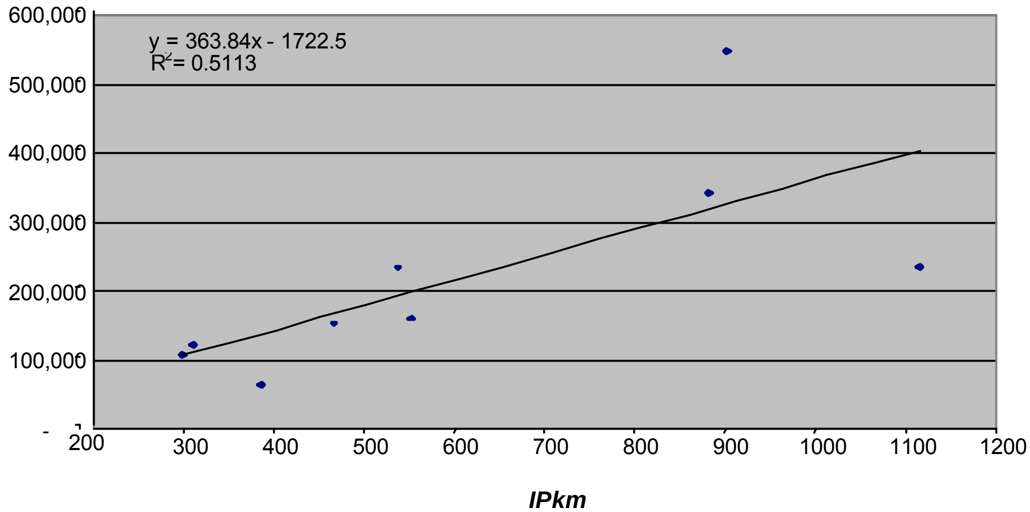


Figure B.1. Relationship between historical abundance, as estimated from cannery records (Meengs and Lackey 2005), and *IPkm* for nine coastal watersheds in Oregon.

Should density criteria be expressed in terms of *IPkm* or total accessible km?

Another issue that faced the TRT was whether density criteria should be expressed in terms of spawners per *IPkm* or total accessible kilometers within a watershed. In the literature, spawner densities (including those in Bradford et al. 2000) are obviously most commonly expressed in terms of spawners per kilometer of stream. However, productive capacity within and among watersheds may be highly variable depending on the nature and quality of habitats. For example, for coho salmon, certain habitat types such as alcoves and dam pools typically found in unconstrained, low-gradient reaches of small-to-moderate-sized streams often account for a disproportionate portion of the total smolt production in a system (Nickelson et al. 1992). Thus, streams with comparable numbers of total accessible miles may produce substantially different numbers of fish. The IP models seek to predict such differences in the potential for different stream reaches (and watersheds) to express habitat characteristics that are likely to be favorable to each species, and thus we chose to use the aggregate *IPkm* in each watershed as the basis for density calculations. Doing so assumes that, in general, density increases in direct proportion to the IP value for a reach, which may not be entirely true (and is difficult to validate in the absence of reference streams that have not been altered by human activities). However, the fact that estimates of *IPkm* were correlated with historical estimates of total abundance in coastal watersheds or Oregon (Figure 1) and provided some improvement in explanatory power over an unadjusted estimate of accessible stream kilometers suggests that *IPkm* provides a reasonable basis for scaling differences in density criteria (and resulting abundance projections) among watersheds.

Should adjustments to density criteria be made to account for potential IP Bias?

In our population structure report, the TRT acknowledged potential bias in the IP model that may arise due to regional differences in precipitation-runoff relationships or other local factors that are not accounted for in this relatively simple model (Bjorkstedt et al. 2005). The most likely source of bias relates to the use of mean annual discharge as a variable in the IP model. Stream hydrology is strongly influenced by complex interactions among a variety of factors including the amount and timing of precipitation, seasonal temperature patterns, and topographic and geomorphic characteristics of watershed that affect water routing and groundwater storage. All of these attributes vary across the NCCC Recovery Domain, some in systematic fashion. Consequently, while we used appropriate regional precipitation and runoff data to develop estimates of mean annual discharge, how stream discharge is distributed through time is likely to vary across the region. This potentially may result in the IP model over-predicting potential habitat in areas with the strongest seasonal patterns in precipitation, the warmest summer temperatures, or the least water storage capacity. For example, preliminary field investigations in San Mateo and Santa Cruz counties suggest that in some small headwater streams where the IP-model predicts potential habitat for coho salmon, summer low flows may be insufficient to support the species in most years (Brian Spence, NMFS, Southwest Fisheries Science Center, Santa Cruz, unpublished data). Bjorkstedt et al. 2005 (pg. 55) characterized this potential bias using an index of IP bias, specifically the ratio of mean annual precipitation to mean annual air temperature. Potential IP bias was qualitatively considered when assigning populations into the categories of functionally independent, potentially independent, and dependent. Where predicted habitat potential for populations fell near the minimum thresholds we used for discriminating between independent and dependent populations, the index of potential bias was used to adjust our final independence categorizations.

A second potential source of IP-bias may arise in areas where summer water temperatures are marginal for the species. For coho salmon, the issue of temperature is dealt with in a very general way through the use of the temperature mask (see Bjorkstedt et al. 2005, pages 54-55), which uses August air temperatures to eliminate from consideration areas where coho salmon occurrence was highly unlikely due to high water temperatures[41]. However, there may be instances where local conditions historically were such that water temperatures may have exceeded the tolerable range for coho salmon. Examples may be where the natural levels of canopy closure were relatively low and allowed for greater stream heating through direct solar radiation. Conversely, there may be some instances where the relatively simple temperature mask

[41] Temperature masks were not used for Chinook salmon or steelhead. Chinook salmon juveniles typically emigrate to sea as juveniles in spring, before waters get excessively warm, and warm temperatures do not appear to have limited historical steelhead distribution within the NCCC Recovery Domain.

does not account for localized effects, such as areas with substantial cold groundwater inputs; thus, some areas "masked out" may have been cool enough to support coho salmon.

A third potential source of bias is the potential role that seasonal access played in historical population viability. Specifically, sandbars form across the mouths of many streams and rivers on the north-central California coast during summer, such that entry by salmon in fall or early winter is dependent on storm events that both produce stream runoff and coastal wave erosion sufficient to breach these bars. In years where sandbars are not breached until late in the spawning season, average population abundance over many years could potentially be lower than that projected based on *IPkm*. There does not appear to be any reliable information on periods of sandbar formation and breaching for most coastal streams from which to assess whether access may be a significant factor regulating population abundance or persistence. Additionally, it is difficult to tell whether current sandbar dynamics represent historical conditions, since most watersheds have experienced some changes in hydrology, sediment regimes, or physical structure (e.g., levees, breakwaters, etc) of estuaries, lagoons, and nearshore areas that that could affect sandbar formation and erosion.

And finally, the IP model does not account for the potential influence of unique rearing habitats such as lagoons and their potential contribution to productive capacity of individual watersheds. For example, recent evidence suggest that steelhead that rear in lagoons are larger at time of ocean entry and experience higher survival rates at sea than steelhead that migrate directly to sea and do not spend significant time in a lagoon (Bond 2006). In such circumstances, target abundances based on *IPkm* alone may underestimate the historical productive capacity of these systems.

In recognizing that such biases may exist, the TRT was then faced with the question of whether the density criteria should be adjusted to account for these potential biases. More specifically, the TRT debated three interrelated questions. First, if there are regional differences in the degree of IP bias, is it reasonable to assume that the densities required for viability should be consistent among populations across an ESU? Second, because the practical outcome of density criteria (based on a prediction of *IPkm*) is to produce a population size target (i.e., the density threshold multiplied by the predicted *IPkm*), is it reasonable to have two basins with similar predicted *IPkm* but different IP bias to have comparable target population size requirements? And third, if some adjustment for IP-bias is deemed necessary, can the IP bias be quantitatively incorporated into the density criteria?

After considerable discussion, the TRT concluded that the density criteria should not be adjusted to accommodate IP bias for two primary reasons. First, we could find no satisfactory way to quantitatively relate the density criteria to various potential sources of IP bias. The IP model is a very coarse-scale model intended to predict the potential for development of habitat suitable for a particular species across large geographic areas. We felt it inappropriate to further adjust IP values based on a relatively simple indicator of IP-bias without any empirical basis for doing so. Second, while from a conceptual basis it may seem reasonable to expect that population density would, on average, be lower per unit $IPkm$ near the edge of the species' distributions, the same cannot be said for total population abundance for a viable population. Extinction risk in a population increases with decreasing intrinsic productivity and increasing variability in abundance and vital rates. Populations near the periphery of a species range, where IP-bias may be strongest, would be expected to exhibit lower productivity and greater variability than populations more toward the center of the species distribution. In this context, it is likely that abundance in southern or more interior populations needs to be larger than more northern populations to attain comparable viability. Because these two factors oppose one another, we concluded that no immediate adjustment should be made for IP bias.

That said, the TRT is not averse to the density-based criteria being revised on a population-by-population basis provided that credible evidence can be brought forth indicating that intrinsic potential is truly overestimated or underestimated through some bias in the IP model. As noted above, NMFS Southwest Fisheries Science Center is gathering information that may allow us to adjust for potential hydrologic bias in the southern portion of the coho salmon's range. Similar adjustment may be appropriate if it can be demonstrated that warm water temperatures historically precluded coho salmon from using certain watersheds or stream reaches. Where potential bias associated with water temperature is proposed, it should be demonstrated that water temperatures were historically above tolerable levels for coho salmon before any adjustments to population targets are made. Identifying areas where temperatures are currently unsuitable for coho salmon would not, by itself, constitute sufficient evidence of IP bias since current temperatures may reflect anthropogenic disturbances such as loss of riparian canopy, diminished stream flows (due to diversions or alteration of hydrologic processes), or any of the other many anthropogenic changes that could result in increased water temperatures.

Summary and conclusions regarding the density criteria

In summary, we believe that the density criteria and the IP-models provide a reasonable basis for scaling expected historical spawner densities within a watershed. Where historical data are available, they indicate that, in the majority of cases, adult abundances projected by the TRT as viable are lower than those observed during the 1930s into the 1950s. In the few instances where projected targets exceed the reported fish counts, there is reasonable grounds for expecting that the historical counts substantially underestimate historical carrying capacities, both because the dam and weir counts represent partial counts (incomplete census at the counting facilities) and because the counts do not take into account the effects of harvest or land-use practices. Thus, we believe that the projected abundance targets do not overestimate natural carrying capacity for the majority of populations within the domain, and in some cases may substantially underestimate historical abundances. Achieving these criteria would substantially reduce risk in most populations and thus be a useful part of a precautionary strategy; however, a highly precautionary approach might call for even higher numbers of spawners.

Finally, we believe that while there may be some uncertainties associated with our approach for establishing preliminary viability targets, these uncertainties should pose few impediments to recovery planning. The TRT has offered its best recommendations regarding recovery criteria with full acknowledgement that these should be considered preliminary and subject to change on a population-by-population basis if credible evidence suggests that they are too conservative or not conservative enough. However, the reality is that the vast majority of independent populations within the NCCC Recovery Domain are so far from reaching the proposed targets that resolving whether a recovery target should be 2,000 or 3,000 fish does little to advance recovery planning. Regardless of the specific targets, the critical actions needed for recovery will, in the majority of cases, be the same[42]. Should we ever get to the point where (a) we have sufficient data to estimated population abundances with reasonable precision, and (b) we begin to approach the proposed viability targets, the questions about the uncertainties can and undoubtedly will be reassessed.

[42] Occasional exceptions may occur when resolution of these uncertainties might help to focus recovery efforts in certain portions of a watershed where the likelihood of success is greatest.

Appendix C. Guidance for Evaluating Hatchery Risks

The types of risks associated with hatcheries, and hence the approaches to evaluating such risk, depend to a substantial degree on the specific type of hatchery program. In this appendix, we provide general guidance for evaluating various risks. We begin by distinguishing two broad classes of hatchery program, based on program goals and protocols for broodstock selection: *integrated* and *segregated* programs. We then provide an overview of the factors that need to be considered when evaluating genetic, demographic, and ecological risks associated with each of these hatchery program types. We draw on several recent and thoughtful treatments of hatchery programs and reform in the scientific literature. The Hatchery Scientific Review Group (HSRG 2004; Mobrand et al. 2005) provided a range of principles and recommendations for the management of both integrated and segregated hatchery programs. Several recent publications discuss specific "best management practices" for integrated supplementation programs (see e.g., IMST 2001; ISAB 2003; Flagg et al. 2004; Olson et al. 2004; Reisenbichler 2004; Mobrand et al. 2005; Williams et al. 2003). Other published studies present a variety of methods for examining ecological and genetic risks associated with hatcheries (Currens and Busack 1995, 2004; Pearsons and Hopley 1999; Ford 2002; Goodman 2004, 2005). The reader is referred to these publications for more detailed discussion of hatchery risks and management practices.

Fundamentally, there are two primary purposes of hatchery programs: 1) to help conserve naturally spawning populations and their inherent genetic composition, and 2) to provide fish for harvest[43]. The HSRG (2004) suggests that, for the purpose of assessing risks and benefits, hatchery programs can be further categorized into two types based on the management goals and protocols for propagating the hatchery broodstock. *Integrated* programs are those in which a primary goal is to minimize genetic divergence between the hatchery broodstock and a naturally spawning wild population by systematically incorporating wild fish into the hatchery broodstock. Integrated programs potentially include several distinct types of hatchery programs including "augmentation" programs intended to increase the number of fish available for harvest; "supplementation" programs, which are hatcheries designed to *"maintain or increase natural production, while maintaining the long-term fitness of the target population and keeping the ecological and genetic impacts on non-target populations within specified biological limits*" (ISAB

[43] Other general purposes of hatcheries may include research, education, and providing cultural benefits, but there are no such hatcheries currently operating within the NCCC Recovery Domain. Mitigation for habitat loss is often mentioned as a "purpose" of hatchery programs; however, under the framework presented here, mitigation programs could fall into the category of either segregated or integrated programs.

2003); and conservation programs, such as captive broodstock programs, which are intended to prevent extinction of specific populations while other recovery efforts are conducted[44].

Segregated programs, in contrast, strive to maintain hatchery broodstock that are distinct from their wild counterparts by using predominately or exclusively hatchery-origin adults returning to the hatchery in subsequent broodstock. Ideally, segregated programs seek to minimize (to the extent possible) gene flow between hatchery and wild populations, both to minimize adverse effects on wild populations and to maintain variation in characteristics such as adult run timing, which may allow directed harvest on the hatchery stock. Segregated programs are generally production or augmentation programs intended to increase opportunities for harvest of stocks that are not at risk. Restoration hatcheries, defined as those intended to re-introduce fish into watersheds where they have been extirpated, might initially be considered segregated programs, though they can evolve into integrated programs if reintroduction is successful and broodstock eventually come from the naturalized population.

Approaches for meeting genetic, demographic, and ecological goals—including minimizing potential adverse effects on wild populations—will often be substantially different for integrated and segregated hatchery programs. In the discussion below, we highlight key issues related to potential effects of integrated and segregated programs, as well as information needs for evaluating whether or not goals are being met. Without thorough evaluation of these issues, populations affected by hatcheries should generally be considered at risk because of the high uncertainty surrounding these potential effects.

Genetic Risks

Before discussing specific issues associated with the evaluation of genetic risks of integrated and segregated hatchery programs, there are several general principles germane to both types of programs. These principles form the conceptual basis for quantitative criteria put forth by the Interior Columbia and Central Valley TRTs (ICTRT 2005; Lindley et al. 2007):

- **Genetic risks associated with hatcheries generally increase with increasing genetic dissimilarity between hatchery and natural populations.** Genetic dissimilarity may be a function of hatchery stock origin or artificial selection. Assuming that hatchery and wild fish freely interbreed, relative risks will follow the following order with respect to the source of hatchery

[44] Captive broodstock programs are, in principle, a form of supplementation program. The distinction is that in supplementation programs, broodstock are generally collected to proportionally represent the genetic composition of the wild population, whereas in a conservation hatchery program, populations are typically so depressed that strict mating protocols are needed to avoid adverse genetic effects that are likely to occur when closely related individuals interbreed.

populations: out of ESU > out of basin > within basin > within basin with best management practices[45]. This general ranking of relative risks can be confounded if there are differences in the relative reproductive success of hatchery-origin fish versus wild fish, or if there is divergence in traits such as run timing or maturation schedule.

- **Genetic risks associated with hatcheries increase with the percentage of successful natural spawners (i.e., those spawning naturally, outside of the hatchery) that are of hatchery origin.** The higher the percentage of effective spawners that are of hatchery origin, the greater the risk to wild populations.

- **Genetic risks associated with hatcheries increase with time for a wild population exposed to a given level of interaction with hatchery fish.** Genetic effects on wild populations are cumulative; thus, long-term programs pose greater risks than short-term programs.

- **Genetic risks associated with hatcheries can be reduced if "best management practices" (BMPs) are followed.** Best management practices depend on the specific goals of the program; thus, generalizing about genetic BMPs is difficult, as discussed below.

Integrated hatcheries — Fundamental goals of most integrated hatcheries are 1) to minimize genetic differences between hatchery broodstock and the wild population that the program seeks to conserve or augment, and 2) to minimize change in genetic composition of the composite hatchery-wild population resulting from hatchery practices (HSRG 2004). Achieving these goals requires incorporating local-origin wild fish into the hatchery broodstock in sufficient numbers such that the genetic composition of the hatchery broodstock represents that of the wild population and avoids inadvertent effects of genetic drift, domestication, and selection in natural and hatchery environments. Typically, it is assumed that genetic representation can be achieved by proportionally representing various phenotypes found in the wild population in the hatchery broodstock, an assumption that can be evaluated using modern molecular genetic techniques. For an integrated program, the proportion of natural-origin broodstock that is needed to avoid genetic divergence remains a subject of substantial scientific uncertainty and debate and will depend on the specific goals of the hatchery program and the status of the wild stock. For example, the HSRG (2004) recommended that 10%–20% of hatchery broodstock be composed of natural-origin adults

[45] Best management practices for integrated supplementation programs remain an area of active research and scientific discussion. For further elaboration, see HSRG 2004; Mobrand et al. 2005; ISAB 2003; Flagg et al. 2004; IMST 2004; Olson et al. 2004; Reisenbichler 2004; Mobrand et al. 2005; Williams et al. 2003.

each year to avoid genetic divergence between the hatchery and wild populations. In contrast, the ISAB (2003) suggests that for supplemental programs (i.e., programs intended to provide a "demographic boost" to rebuild a depressed natural population[46]), 100% of hatchery broodstock should be drawn from the products of natural spawning. However, for conservation hatcheries where the natural populations are very small, it may be more appropriate to cross wild fish with hatchery or captive fish.

Hatchery practices should also seek to minimize intentional or unintentional domestication selection by employing appropriate mating protocols, rearing environments (i.e., environmental conditions that follow natural pattern of temperature, photoperiod, etc.), and release strategies. Additionally, collection of wild broodstock should be done in a manner that leaves sufficient numbers of individuals on natural spawning grounds to avoid unintended alteration of the genetic composition of the wild component. The HSRG (2004) concludes that associated natural populations must be "viable and largely self-sustaining if they are to support successful integrated programs.…" Implicit in this statement is recognition that hatcheries are subject to catastrophic losses due to mechanical failures, human error, disease outbreaks, and malicious acts. When such events happen, sufficient numbers of individuals must remain in the wild population to maintain the genetic integrity of the population[47]. And finally, integrated programs should strive to ensure that the rate of gene flow from the natural component into the hatchery broodstock should exceed gene flow in the reverse direction. The long-term goal of an integrated program is to ensure that selection in the natural environment (rather than the hatchery environment) drives the evolution of the integrated population (HSRG 2004).

Evaluating the likelihood of genetic risks of integrated programs requires a substantial amount of information, including the following:

- Estimation of the number and proportion of wild fish that are incorporated into the hatchery broodstock

[46] An objective of supplementation programs is to, at least temporarily, increase the number of spawners on the spawning grounds by having hatchery-origin adults spawn in the wild (ISAB 2003). However, this is not necessarily a goal of all integrated programs. As the HSRG (2004) notes, the goal of an integrated broodstock program is to maintain the genetic characteristics of the natural population in the hatchery-origin fish, not the reverse.

[47] These statements do not imply that integrated "supplementation programs" are not appropriate conservation tools, only that long-term viability of the population should not be dependent on the hatchery component.

- Estimation of the number of hatchery-origin fish that spawn on natural spawning grounds, their proportional contribution to the spawning population, and their effective contribution to reproductive output[48]
- Quantification of changes in the genetic composition of the integrated population through time
- Quantification of phenotypic characteristics (e.g., age and size at maturity, age and size at smoltification, timing of spawning run and smolt outmigration, egg size, fecundity, etc.) of the integrated population through time
- Estimation of effective population size of the integrated population.

For captive broodstock programs, which are a highly specialized form of integrated hatchery program, substantial genetic information at the level of individual fish is required so that spawning matrices that avoid crossing of siblings and other close relatives can be implemented. By their very definition, captive broodstock programs exist because wild populations are perceived to be at high risk of extinction. When captive broodstock programs succeed and population abundance increases to levels that might suggest viability, additional evaluation of potential long-term genetic risks associated with a recent population bottleneck would be required.

Segregated hatcheries — A primary genetic goal of segregated hatcheries is to minimize or eliminate gene flow between the hatchery and wild populations, which entails minimizing the occurrence of hatchery fish spawning in the wild (to avoid outbreeding depression) and excluding or minimizing the contribution of wild fish to the hatchery gene pool (to avoid convergence of genotypic and phenotypic characteristics). Strategies recommended by the HSRG (2004) for achieving this goal include 1) releasing fish in areas where opportunities to capture non-harvested adults are high; 2) rearing and releasing fish in a manner or at a location that minimizes straying and opportunities for natural spawning; 3) ensuring that harvest opportunities are commensurate with adult production from segregated programs; and 4) ensuring that hatchery-origin adults make up no more than 1%–5% of natural spawners (see footnote). Several authors (ISAB 2003; Goodman 2004; Ford 2002) have argued that even where the percentage of hatchery-origin fish on natural spawning grounds is low, the effects on fitness may still be significant over time, especially since many “wild” fish may be progeny of hatchery-origin fish. As with integrated programs, evaluation of genetic risks associated with segregated programs requires estimating the number and fraction of natural spawners that are of hatchery origin and their contribution to the next

[48] Estimating the contribution of hatchery-origin fish to reproductive output is complicated by the fact that, although it is now common to mark hatchery fish upon release, the progeny of hatchery fish are not easily identified. Thus, the potential influence of hatchery fish on the genetic composition of the wild population is not strictly a function of the fraction of identifiable hatchery-origin spawners.

generation, as well as the proportion of wild fish incorporated into hatchery broodstock. Additionally, genetic monitoring is needed to determine whether genetic composition of the wild population is being affected by introgression by genetically divergent hatchery fish.

For both integrated and segregated programs, evaluation of genetic risks may also need to include assessment of potential residual genetic effects associated with historical hatchery practices. Within the NCCC Recovery Domain, there is a substantial history of plantings of out-of-basin and out-of-ESU fish into many river basins (reviewed in Bjorkstedt et al., 2005). Other programs may have used local broodstock but used mating or rearing protocols that, by today's standards, would be considered likely to result in domestication. Furthermore, many long-running programs have only recently been terminated. In most cases within the recovery domain, there is little or no information on parameters important for understanding potential genetic effects (e.g., percentage of wild fish used for broodstock, percentage of hatchery fish on natural spawning grounds, or information on historical genetic composition of wild populations that could be compared with current genetic data). Genetic evidence suggests that among anadromous salmonids, indigenous populations may resist introgression when the introduced stock is genetically strongly divergent (Utter 2001, 2004)[49]. However, when introduced hatchery fish are from geographically proximate watersheds, the probability of introgression likely increases.

Recent genetic data from populations of steelhead, coho salmon, and Chinook salmon from the NCCC Recovery Domain are generally consistent with these patterns (see Bjorkstedt et al. 2005 for summary of available genetic information). There is little evidence to suggest that strongly divergent stocks (primarily from Oregon and Washington) of salmon and steelhead that were introduced into various watersheds in the region have left a lasting genetic signature. However, in some instances, transfer of fish among basins that are relatively close to one another appears to have resulted in some homogenization of genetic composition (e.g., Eel River and Mad River steelhead). Little is known about whether longer-term hatchery programs that used locally-derived broodstock have resulted in loss of diversity through inbreeding or reduced fitness through domestication processes. Unfortunately, there often may be no easy way to evaluate any potential impacts of past hatchery practices. Genetic methods may provide some insight into whether past introductions have affected population genetic composition or structure. For example, occurrence of unique alleles present in the donor stock but previously absent from the recipient population would indicate introgression. Additionally, low genetic diversity in local populations with a

[49] The lack of a lasting genetic signature from such introductions does not necessarily mean that the stocking was without adverse effects when it occurred. Rather, it suggests either failure of hatchery fish to reproduce or strong selection against individuals carrying alleles from the hatchery stock.

long history of artificial propagation could be indicative of hatchery effects, though it could also arise from other processes. In general, we would expect genetic risk to be greatest in populations affected by recent out-of-basin transfers (risks that would be expected to diminish with time since last stocking, assuming strong selection against nonnative stocks) or long-running production programs that released large numbers of fish derived from local or nearby sources. Fish of intermediate divergence are potentially the most problematic, since they are generally expected to be more successful at reproduction and introgression in the recipient basin than highly divergent populations, but less successful at maintaining population fitness than closely related populations.

Demographic Risks

Integrated hatcheries — Goals for minimizing demographic risks of integrated hatcheries should consider several distinct types of risk. Of primary concern is that hatchery-reared progeny of wild adults will fail to replace those progeny that would have been produced in the wild had adults been left to spawn naturally (ISAB 2003). In this regard, assessment of whether an integrated program represents a net benefit to the target stock requires analysis not only of how many juveniles or smolts are produced in the hatchery, but also how well they survive and reproduce in the wild compared to their wild counterparts (ISAB 2003). Such analyses are critical because hatchery programs can increase the number of fish on natural spawning grounds, even if there is a decrease in the productivity of the wild component of the integrated population. In such cases, any potential benefits of an integrated program to population abundance will cease when the program is ended. Where adult broodstock are being taken from small wild populations, an additional concern is that removal of adults for use in hatchery broodstock could potentially lead to depensation in the wild population (e.g., remaining adults may have difficulty locating mates or produce too few juveniles to swamp local predator populations). A third demographic concern is the potential for adverse effects on wild stocks in mixed-stock fisheries. In an integrated program, an abundance of hatchery fish may result in increased harvest pressure while simultaneously masking decreasing productivity of the natural component. These circumstances can lead to incorrect assessment of stock status and drive wild populations toward extinction if escapement drops below replacement levels (NRC 1996).

Evaluation of these potential demographic risks involves the following information:

- Estimates of the adult spawner population size and spawner density on natural spawning grounds
- Estimates of the number and proportion of wild adults captured for broodstock
- Estimates of population growth rate (productivity over the entire life cycle) for both wild and hatchery-origin fish

- Estimates of harvest rates on the integrated stock.

Segregated hatcheries — For segregated hatchery programs, the intent of which is to increase the number of fish available for harvest, goals for minimizing demographic risks focus primarily on minimizing mixed-stock fishery effects on at-risk wild stocks. Evaluation of whether such goals are being met requires estimates of harvest rates on both wild and hatchery stocks in mixed-stock fisheries, which in turn requires estimates of total adult abundance (harvest+escapement) and the proportion of both harvest and escapement that are of hatchery and wild origin.

Ecological Risks

As noted earlier, releases of hatchery fish can influence the success of wild populations through a variety of ecological processes including increased competition, increased predation (direct predation of hatchery fish on wild fish or attraction of predators), transmission of diseases, and through direct effects of hatchery or rearing facilities (e.g., migration barriers, water diversions, and pollutants/pathogens in hatchery effluent). Consequently, conservation goals associated with hatchery programs should seek to minimize these negative interactions; however, the specific goals will differ for integrated and segregated programs.

Integrated hatcheries — For integrated hatcheries, an overarching objective is to produce hatchery fish that mirror their wild counterparts as closely as possible. Achieving this goal requires creating a hatchery rearing environment that yields fish that are similar to wild fish in terms of their physiological disposition, behavior, health status, and nutrition (HSRG 2004). This may entail regulating temperature and photoperiod regimes to match ambient conditions within the river, rearing fish at lower densities than is typical of most hatcheries, feeding fish underwater to reduce surface feeding behaviors, and providing cover and physical structure so that released fish exhibit natural responses to predators and conspecific competitors. Additionally, integrated hatchery programs need to consider the ecological context of receiving waters, such that released fish do not adversely affect the target population (or other at-risk populations with which hatchery fish may eventually intermingle) through competition, predation, or introduction of diseases. Hatchery fish should be released in numbers consistent with productive capacities of the natural systems (both freshwater and marine) that they enter. Because carrying capacities of both the freshwater and marine environments may vary from year-to-year, constant release targets—a standard performance measure for many existing hatcheries—will likely be inappropriate. Hatchery fish should also be released at sizes and times that minimize potential for competitive interactions with wild fish and predation on wild fish. The HSRG (2004) suggests that, in the context of an integrated program, this means mimicking to the degree possible the distribution of sizes and physiological states of wild fish[50]. However, there may be circumstances where release of large numbers of hatchery-reared coho salmon smolts may be an important temporary management tool, because such releases may increase returns of two-year-old females and thereby help re-establish depressed or extirpated year classes (Smith 2006). Hatchery fish should also be released in numbers that do not cause unnatural aggregation of predators. Only hatchery fish free of disease should be released into the wild. And finally, program operations should seek to minimize effects of hatchery and rearing facilities on the wild population (i.e., release of pollutants/pathogens, water diversions for hatchery water supplies, and barriers to migration).

Evaluating whether an integrated hatchery program is achieving ecological goals with respect to conserving the composite hatchery-wild population requires a substantial amount of information not traditionally collected for most hatchery programs, which historically have focused on producing large smolts to be released during a relatively narrow window during the migration period. Among the information needs for evaluating integrated programs are

- Assessment of carrying capacities (including their interannual variation) of the freshwater and marine systems into which fish are being released in order to prevent overstocking
- Estimation of wild fish density in relation to carrying capacity and numbers of hatchery fish released
- Monitoring the size and condition of hatchery and wild populations before release and upon return as adults to ensure that hatchery fish match the wild template
- Monitoring the effect of hatchery releases on predation rates in wild populations
- Monitoring for occurrence of disease in the hatchery population
- Monitoring for facility effects (e.g., water quality downstream of hatcheries; evaluation of fish collection structures/practices on passage by upstream- or downstream-migrating wild fish; potential effects of water withdrawals on stream discharge).

Segregated hatcheries — For segregated hatchery programs, the primary goal should be minimizing interactions with wild fish, but the approaches for achieving these goals will most likely involve creating either temporal or spatial separation between hatchery and wild populations, rather than trying to match the natural template. Practices designed to help achieve these goals include 1) releasing fish at sizes,

[50] There may be instances where the goal of minimizing competitive interactions and that of rearing fish that are similar in their developmental state to wild fish are in conflict with one another, if the carrying capacity of the receiving water is approached. In such cases, some temporal separation between wild fish and hatchery fish may be preferable.

times, or locations that minimize potential for competitive interactions with wild fish during the juvenile and smolt stages; 2) releasing fish in locations where opportunities for adults to stray into streams inhabited by wild fish, where they may compete for mates or spawning habitats, are low; 3) releasing fish at sizes, times, or locations that minimize potential for direct predation on wild fish by hatchery fish or attraction of large numbers of predators during the juvenile or adult phases; and 4) releasing only fish that are free of disease.

In general, information needs for evaluating segregated hatchery programs are similar to those needed for integrated programs, and include

- Assessment of carrying capacities (including their interannual variation) of the freshwater and marine systems into which fish are being released in order to prevent overstocking
- Estimates of density of wild fish in relation to carrying capacity and numbers of wild fish released
- Monitoring the effect of hatchery releases on predation rates in wild populations
- Monitoring for occurrence of disease in the hatchery population
- Assessment of facility effects (e.g., water quality downstream of hatcheries; evaluation of fish collection structures/practices on passage by upstream- or downstream-migrating wild fish; potential effects of water withdrawals on stream discharge).

In evaluating potential risks imposed by hatcheries and developing recovery strategies, recovery planners should recognize that there is a distinction between evaluation of whether a hatchery poses a particular type of risk relative to our viability criteria versus evaluation of whether or not the hatchery program overall provides a *net benefit or risk* with respect to conservation of the population. The former analysis simply seeks to determine whether a given wild population may be at genetic, demographic, or ecological risk due to ongoing or past hatchery operations. The latter analysis, which has substantial bearing on whether a hatchery program should be continued, involves consideration of the various types of risk in the context of one another. For example, within the NCCC Recovery Domain, as well as elsewhere in the Pacific Northwest, there are several captive broodstock programs intended to conserve severely depleted populations of salmon. Without these programs, there may be little chance of recovering these populations and under such circumstances concerns about inbreeding depression and loss of fitness are secondary to the immediate demographic risks of small population size. Likewise, restoration programs intended to reintroduce fish into watersheds from which they have been extirpated will, by virtue of the need to use out-of-basin fish, constitute a plausible risk as assessed through our viability criteria but may be entirely appropriate actions for recovering fish within a diversity stratum, particularly if the available hatchery broodstock are genetically similar to the extirpated population and there is reasonable certainty that the receiving habitat has recovered sufficiently to support fish through their full life cycle. Both captive broodstock and restoration programs exist because populations are perceived to be either extinct or at high risk of extinction. Thus, the question of whether the associated wild population is viable or not has already been resolved.

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