Chapter 4

Covered Aquatic Species

Contents

4 COVERED AQUATIC SPECIES

4.1 Introduction

Chapter 4 includes species accounts for each aquatic species covered by our HCP/NCCP. A species account is a brief description and history of the species from the scientific literature. Species accounts are the starting point for conservation planning—a process that must begin with awareness and understanding. The general topics developed for each species account in this chapter are geographic distribution, local distribution, life history, habitat requirements, ecological interactions, sensitivity to disturbance, and key uncertainties. Key uncertainties address gaps in MRC knowledge, such as the historical or current abundance of a species in the plan area, factors that might limit its recovery, impacts of land management upon species populations, and what exactly "species friendly" silviculture might be. MRC has italicized and flagged (▶) the key uncertainties that become hypotheses in the validation monitoring programs; Chapter 13, *Monitoring and Adaptive Management*, discusses these programs in detail.

4.1.1 🔥 Mendocino lighting complex (2008)

MRC assessed the impacts to the habitat and population of covered aquatic species as a result of the lightning fires in 2008 (see section 1.18). Under the individual species accounts, we indicate the apparent damage.

4.2 Coho Salmon

4.2.1 Geographic distribution

Coho salmon (*Oncorhynchus kisutch*) are distributed throughout the northern temperate latitudes in North America and Asia. In North America, they spawn in coastal streams from Monterey Bay, California to Point Hope, Alaska (Kostow 1995). Populations occur in smaller coastal streams as far south as the San Lorenzo River, Santa Cruz County (Moyle et al. 1989).

Illustration by Ron Pittard

Populations of coho salmon in the HCP/NCCP assessment area belong to 2 evolutionarily significant units (ESUs), as currently recognized by NMFS. Populations in river basins from Punta Gorda (Humboldt County) and north to Cape Blanco (Oregon) belong to the Southern Oregon/Northern California Coasts ESU. Populations in river basins from Punta Gorda (Humboldt County) south to and including the San Lorenzo River (Santa Cruz County), belong to the Central California Coast ESU. NMFS listed the Central California Coast ESU as threatened in 1996 and the Southern Oregon/Northern California Coasts ESU as threatened in 1997. In 2005 NMFS upgraded the listing of the Central Coast ESU from threatened to endangered. In 2002, CDFG listed the Southern Oregon/Northern California Coasts ESU as threatened and the Central California Coast ESU as endangered.

4.2.2 Local distribution

Timberlands included in our HCP/NCCP extend over a large area in Mendocino County, from Hollow Tree Creek in the north to the Garcia River in the south. The entire plan area lies within the range of coho salmon and includes the watersheds of numerous small coastal streams and large rivers, such as the Eel River, Big River, Noyo River, Navarro River, Garcia River, and Russian River.

Historically coho salmon were present in many streams and tributaries in the plan area, although populations of coho salmon have decreased in the past few decades. Researchers have recorded fewer coho salmon, especially in streams within the southern watersheds, such as the Gualala River and Russian River (NCWAP 2002, Brown et al. 1994). Coho salmon are currently present in Hollow Tree Creek, Cottoneva Creek, Jack of Hearts Creek, Noyo River, Big River, Albion River, Elk Creek, South Fork Garcia River, and Navarro River (MRC 2002a).

Re-establishment of coho salmon in streams where they were historically present has proved difficult. Weldon "Wendy" Jones, currently retired from CDFG, reported that between 1995 and 1998 CDFG released juvenile coho salmon in Howard Creek in an attempt to re-establish the species in this watershed (Jones 2000, 9). Preliminary data indicates that coho salmon have not yet returned to Howard Creek (MRC 2002a).

4.2.3 Population trends

The most recent information on population trends of coho salmon is quite bleak. Of 13 streams between the Smith River and Scott Creek where surveys were conducted in 2007-2008, there has been a 73% decline in returning coho salmon, compared to the same year-class lineage returns in 2004-2005. No stream had an increase or level returns. One stream, Redwood Creek in Marin County, had a complete failure, with no returns for the first time on record. Scott Creek had only four jack returns, compared to 329 adults in 2004-2005. There is a slight trend of greater declines toward the south, but for the most part, the data shows large reductions in returning adults throughout the California coast. This phenomenon extends beyond California; preliminary and incomplete surveys of 22 streams through January in the Oregon Coast ESU found a mean decline in coho returns of 70% relative to returns in 2004-2005. The survey estimate of 51,000 returning adults to the Oregon Coast ESU in 2007-2008 is the lowest since 1999.

Ocean conditions were poor for salmon growth and survival during spring and summer of both 2005 and 2006. The Wells Ocean Productivity Index (WOPI)[1] accurately predicts zooplankton, juvenile shortbelly rockfish, and common murre production along the California coast. It is, therefore, a valid indicator of ocean productivity. Index values for spring and summer of 2005 and 2006 were low, indicating poor conditions for growth and survival. In fact, only the El Niño years (1982-83, 1992-93, and 1999) had lower WOPI values.

The WOPI assesses conditions on a local scale for California, but has tracked another index, the Northern Oscillation Index (NOI), which describes climate variability in the broader region of the North Pacific Ocean. In 2005 and 2006, the WOPI decoupled from the NOI, suggesting local conditions on the California coast were worse than in the larger North Pacific region. These results indicate that ocean conditions in spring and summer, when juvenile coho and Chinook salmon enter the ocean, were unfavorable to growth and survival.   This may explain the poor returns of both coho in 2007 and 2008 and Chinook salmon in 2007.

Coho salmon have undergone substantial population declines and no longer inhabit many of the streams in California where they historically occurred. Brown et al. (1994) estimated that populations of coho salmon in California have decreased to less than 6% of 1940 levels. Populations of coho salmon in the southern part of the range have shown the greatest declines (Hassler et al. 1991, Brown et al. 1994), with few coho salmon occupying coastal streams near or south of San Francisco Bay. In the Central California Coast ESU, where historical populations

---

[1] WOPI is a composite index of 13 oceanographic variables and indices, weighted heavily by sea level height, sea surface temperature, upwelling index, and surface wind stress.

are estimated to have been between 50,000 and 125,000 naturally spawning fish, current abundance is estimated to be less than 5000, most of which likely originated in hatcheries (Brown and Moyle 1991, Bryant 1994, CDFG 1994).

In the Southern Oregon/Northern California Coasts ESU, where historical abundance of coho salmon is estimated to have ranged from 150,000 to 400,000, abundance is currently estimated to be about 30,000, of which roughly 20,000 originated in hatcheries (Brown and Moyle 1991, Bryant 1994, CDFG 1994, Weitkamp et al. 1995). CDFG conducted annual counts of spawning escapement from 1938 to 1975 at Benbow Dam on the South Fork Eel River; the results fluctuated considerably. The overall trend, however, was a substantial decline in the population of coho salmon during this time period (Taylor 1978). The average annual count during the first 10 years of operation (1938–1947) was 14,327 coho salmon. During the last 10 years of operation (1966–1975), the average annual count was 1846.

NMFS compiled the most recent data showing population trends for coho salmon of Central California Coast and Southern Oregon/Northern California Coasts ESUs (2001). The NMFS review only included streams with consistent data, spanning at least 6 continuous years. Data from the 3 streams in the Central California Coast ESU that fit the time criteria (Caspar Creek, Little River, and Pudding Creek—all in Mendocino County) show that abundance of coho salmon in the 1990s was clearly lower than in the mid-to-late 1980s, with a continual decrease during the 1990s (NMFS 2001). Based on currently available data, NMFS concluded that the Central California Coast ESU is presently in danger of extinction (NMFS 2001).

NMFS observed a general pattern of decline in the California portion of the Southern Oregon/Northern California coho salmon ESU (NMFS 2001). Since there are few datasets going back as far as the 1980s, however, NMFS suggests caution in interpreting the trend data. Data from Hollow Tree Creek, a South Fork Eel River tributary in the plan area, indicates an overall decline in abundance of coho salmon since the late 1980s (NMFS 2001). This contrasts with data recorded at other locations in the South Fork Eel basin, which shows no obvious trend in abundance of coho salmon from 1994–2000 (NMFS 2001). NMFS concluded that the Southern Oregon/Northern California Coasts ESU is presently not at risk of extinction, but is likely to become endangered in the near future.

4.2.3.1 Hatchery production
Most coho salmon produced in California are from hatcheries (Greenley 1985, Baker and Reynolds 1986). Brown et al. (1994) estimated that less than 5000 native coho salmon now spawn in California streams. Those river basins still supporting runs of wild native coho salmon have annual spawning populations of fewer than 100-1300 adults; the higher range of that estimate is from the South Fork Eel River.

The Noyo River Egg Collecting Station, operated by CDFG, is in Jackson Demonstration State Forest on the South Fork Noyo River; this station is the only fish culture facility in California that has focused exclusively on coho salmon (CDF et al. 2000). The number of coho salmon eggs collected between 1973 and 1999 ranged between 4500 and 1,381,755; female coho salmon have had an average of about 2200 eggs (CDF et al. 2000). Eggs collected at the Noyo facility are shipped to Mad River Hatchery for hatching and rearing and then back to the Noyo River for planting. In the 1970s, the Noyo fish culture station supplemented its broodstock of coho salmon, during years of low escapement, with eggs from the Alsea River in Oregon and the Washougal River in Washington; CDFG transfer permits document this. Noyo coho salmon have been transplanted to almost all of the coastal streams of Mendocino County as well as to the Mad River

and Prairie Creek Hatcheries; they are also the origin of the broodstock at Warm Springs
Hatchery on the Russian River (Higgins et al. 1992, CDF et al. 2000).

4.2.3.2 Commercial and recreational harvest

Ocean harvest rates, which were likely high in the 1970s, have declined since adoption of harvest
limits in the early 1980s. Ocean harvest of wild coho salmon is prohibited. In California, harvest
of hatchery coho salmon is prohibited as well. Fisheries outside of California—from Oregon on
north—can only target coho salmon from hatcheries when escapement of wild fish will be high.

4.2.4 Life history

4.2.4.1 Overview

Coho salmon are 1 of 5 species of Pacific salmon distributed along the coast of North America,
the others being Chinook salmon, sockeye, pink, and chum salmon. Lengths and weights of adult
coho salmon range from 45 to 60 cm (18 to 24 in. and 3 to 6 kg (7 to 13 lb.), respectively
(Shapovalov and Taft 1954, Godfrey 1965, Scott and Crossman 1973, Meehan and Bjornn 1991).
Coho salmon weighing over 12 kg (26.5 lb.), however, have been caught (Meehan and Bjornn
1991). Adult coho salmon are generally smaller than Chinook salmon; moreover, a coho salmon
has a white gum line, while a Chinook salmon has a black gum line. Dark spots on the back,
dorsal fin, and upper lobe of the tail distinguish coho salmon from chum and sockeye salmon
(Hassler 1987). Lack of oval blotches on the lower lobe of the tail distinguishes them from pink
salmon (Moyle 1976).

Unlike Chinook salmon and steelhead, coho salmon do not appear to have genetically distinct,
temporally segregated runs (Moyle et al. 1989). However, the strong homing abilities of coho
salmon make it likely that each coastal stream has a distinctive strain adapted to local
environmental conditions (Moyle et al. 1989).

Populations of coho salmon can be divided into 2 types of fish: big-river and short-run. Big-river
coho salmon migrate from 97 to 194 km (60 to 120 mi) or more in main river systems to spawn in
mainstems or tributaries (Moyle et al. 1989). Short-run coho salmon live in smaller coastal
streams and tributaries of the lower reaches of big rivers and rarely migrate more than 97 km (60
mi) miles upstream. While much of the natural production of big-river coho salmon has now been
supplanted by hatchery populations in California, short-run coho salmon are more dependent on
natural reproduction (Moyle et al. 1989).

4.2.4.2 Adult upstream migration and spawning

Coho salmon spend about 3 to 4 months within spawning gravels as eggs and alevins, up to 15
months rearing in fresh water, and approximately 16 months in the ocean (Sandercock 1991).
Over 95% of coho salmon in Washington, Oregon, and California mature and return from the
ocean to fresh water in their third year of life (NMFS 1995). These 3-year-old coho salmon
adults are considered to be age 1.1, having spent one winter in fresh water and one winter in the
ocean. In addition, spawning runs of coho salmon usually include males that return after
spending only 6 months at sea; these males are commonly referred to as jacks (age 1.0). Female
coho salmon may return at 2 years of age as well (Nielsen et al. 1991).

Following 1 or 2 winters at sea, coho salmon attain sexual maturity in the summer. Adult coho
salmon migrate to the vicinity of their natal stream during late summer and fall (Sandercock
1991). They do not enter the stream system all at one time. Instead they arrive throughout the
spawning season in a pattern that reflects the occurrence of storms that increase flow in the
spawning streams (Shapovalov and Taft 1954). There may be some selective advantage to

spawning later in the season. Coho salmon often dig up or disturb eggs deposited in the gravel by females which arrived earlier and then died after spawning; this is called redd superimposition.

Males generally dominate the early part of a coho salmon run, while females return in greater numbers later during the run (Shapovalov and Taft 1954, Moyle et al. 1989). Typically, the number of females slightly exceeds the number of males among 3-year-old spawners; however, jacks and 3-year-old males combined are more numerous than females (Shapovalov and Taft 1954, Salo and Bayliff 1958, Fraser et al. 1983). Jacks probably have higher ocean survival rates than 3-year-old males because of their shorter exposure to ocean fisheries (Gross 1991). The sex ratio for all of the spawning adult coho salmon deriving from a single brood year is close to 50:50.

Most adult coho salmon migrate upstream during daylight hours (Sandercock 1991, Neave 1943), with peak activity usually occurring at dawn and sunset (Fraser et al. 1983). Coho salmon move through shallow riffles quickly, perhaps to reduce predation risk; they seek deeper, quieter pools in which to rest before migrating further upstream (Sandercock 1991).

Briggs (1953) observed coho salmon moving across riffles where the water depth was only 5 cm (2 in.). Under proper conditions, vertical leaps of more than 1.83 m (6 ft) are possible to surmount instream barriers (Reiser and Peacock 1985). Cruising speeds, which are used for long-distance travel, are up to 1 m/s (3.3 ft/s). Sustained speeds, which may last several minutes and are used to surpass rapids or other barriers, range from 1–3.2 m/s (3.3–10.6 ft/s). Darting speeds, which are brief bursts used in feeding and escape, range from 3.2–6.6 m/s (10.6–21.5 ft/s) (Bell 1986).

Homing of coho salmon and other Pacific salmon to their natal streams is well-documented in the literature. Coded-wire tag (CWT) recoveries of wild adult coho salmon indicate that straying rates may be in the range of 0.1%–2.0% (Sandercock 1991, Labelle 1992). Salmon that are blocked from access to their natal tributary may die before choosing an alternate location to spawn (Ricker and Robertson 1935, Hartman and Raleigh 1964).

Females select a nest site after arriving on the spawning ground, and defend the area against other females. Redd construction behavior is similar to that displayed by other anadromous salmonid species; the female excavates a depression in the gravel by turning on her side and using her body and tail to displace the gravel downstream. Digging activity may last as long as 5 days, during which time the female will dig up to 7 egg pockets in succession, progressing in an upstream direction (Shapovalov and Taft 1954, Tautz 1977, van den Berghe and Gross 1984, Sandercock 1991). Egg pockets range from 50–70 cm (19.5–27.3 in.) in diameter and from 9–24 cm (3.5–9.4 in.) in depth (Tautz 1977, van den Berghe and Gross 1989). The average number of eggs deposited per egg pocket ranges from 300 to 1200, with the most frequent number being 800 to 900 (Sandercock 1991).

Fecundity and egg size generally increase with the size of the female (Allen 1958, Ivankov and Andreyev 1969, Stauffer 1976, Beacham 1982). Fecundity can range from 1000 to 7600 eggs, and is commonly 2500–5000 eggs per female (Beacham 1982, Sandercock 1991). Beacham (1982) found significant regional and annual variability in the fecundity of coho salmon. Beacham and Murray (1993) examined data for 40 coho salmon populations and found only a weak association between fecundity and latitude. Results from their investigations also suggest that female coho salmon that spawn in upper river locations after long migrations were smaller

and less fecund than females spawning in coastal streams. Perhaps the smaller females allocate more energy reserves to migration than to gonadal development.

Standardized fecundities for females of 520 mm (20.8 in.) postorbital-hypural length were found to be lowest in California (2610 eggs/female), Vancouver Island, British Columbia (2470 eggs/female), and Queen Charlotte Island, Alaska. Ocean conditions may also influence fecundity. Johnson (1988) reported reduced fecundity of Oregon coho salmon related to the 1982–1983 El Niño episode that affected ocean conditions in the northeast Pacific. Because female coho salmon exhibit little variation in their period of marine residency, ocean conditions favoring increased growth and higher fecundity (van den Berghe and Gross 1989, Fleming and Gross 1990) may result in greater densities in rearing areas and increased competition among fry (Quinn 1994).

Two separate tactics may be used by male coho salmon in fertilizing the eggs laid by a female, depending on whether the male is a 2-year-old jack or a 3-year-old adult (Gross 1985). Adult, or hooknose males, are adapted for fighting other males to achieve dominance and to fertilize a defended female's eggs. Jacks employ a sneaking tactic, using refuges, such as rocks, LWD, or shallow areas near females that are constructing redds. They rush in to fertilize some of the eggs during the act of spawning between a female and a dominant hooknose male. Because jacks have a shorter marine residence, they do not spawn with coho salmon from their same brood year; this increases genetic variation in the population (Young 1999).

The eggs of a female may often be fertilized by more than 1 male (Gross 1984, Sargent et al. 1987). Gross (1991) estimated that there was an equal probability of jack and hooknose reaching the spawning grounds and successfully breeding. This may be due in part to selective harvesting of hooknose males (Gross 1991). Among coho salmon, larger juveniles or smolts tend to return as jacks (Wallis 1968, Hager and Noble 1976, Gross 1991). Factors that may influence juvenile growth and body size include genetics, egg size, time of hatching, water flow, temperature, quality of feeding territory, stream productivity, predation pressure, and population density (Gross 1991). Young (1999) found that the proportion of jacks in a population was positively correlated with spawner density, which, in turn, relates to environmental conditions.

Fertilization rates under natural conditions appear to be very high, close to the percentages of eggs found fertilized under ideal conditions in hatcheries (Shapovalov and Taft 1954). Hobbs (1937) found that various species of salmonids had average fertility rates of over 99% under natural conditions.

Females continue to guard the spawning redd and prevent superimposition by other females until they are too weak to maintain position in the current (Briggs 1953). Both males and females die soon after spawning. The post-spawning survival time for females averages from 8 to 13 days (Briggs 1953, Willis 1954, Crone and Bond 1976, van den Berghe and Gross 1986). Larger females appear to guard redds for longer periods, perhaps because of their greater energy reserves (van den Berghe and Gross 1986, 1989).

Van den Berghe and Gross (1984) found that larger females among a wild population of coho salmon in Washington dug significantly more egg pockets than smaller females. In addition, larger females dug deeper redds, burying their eggs as much as 2.5 times deeper than small females. Females compete aggressively on the spawning grounds for suitable redd sites. Those that arrive late may dig up eggs laid by previous spawners. Broods of larger females are probably less susceptible to redd superimposition. There are fewer potential competitors that can dig as

deep; as a result, there is likely a selective advantage to large-bodied females (van den Berghe and Gross 1984). Holtby and Healey (1986) concluded, however, that large size was not necessarily an advantage; this explains the range of adult female sizes that they observed in the spawning run. Small females may display a tendency to spawn in sites with lower velocity flows and smaller gravels; these sites may be less susceptible to scour than sites selected by the larger females (Holtby and Healey 1986).

4.2.4.3 Egg incubation, alevin development, and fry emergence

Following deposition in the gravel, coho salmon eggs incubate for 35–50 days at temperatures of approximately 9 to 11°C (48.2 to 51.8°F) (Shapovalov and Taft 1954); incubation time is inversely related to water temperature. After hatching, salmon larvae (alevins) remain in the gravel while undergoing further development and absorption of the yolk sac. Emergence begins 2–3 weeks after hatching and may continue for an additional 2–7 weeks (Shapovalov and Berrian 1940). Many alevins emerge at night (Shapovalov and Taft 1954, Mason 1976a), which presumably decreases vulnerability to predation (Shapovalov and Berrian 1940, Bams 1969, Mason 1976a, Godin 1980).

Estimates for egg-to-emergence survival rates of coho salmon range from approximately 10% (Tagart 1984) to 85% (Shapovalov and Taft 1954). Factors which affect egg survival include fine sediments, temperature, permeability, dissolved oxygen, gravel mobilization during high flow events, low flows, freezing, bird and insect predation, and fungal infections (Sandercock 1991).

4.2.4.4 Juvenile freshwater rearing

Upon emergence from the gravels, coho salmon fry seek low velocity areas along shallow stream margins (Shapovalov and Taft 1954). As they grow, juvenile coho salmon move to deeper habitats, although they continue to prefer low-velocity habitat throughout the rearing period. Juvenile coho salmon establish territories or form hierarchical groups in pools based on optimal foraging positions (Dolloff and Reeves 1990, Fausch 1993). During the winter, when water temperatures are low (< 7°C or 44.6°F), aggressive territorial behavior and feeding is reduced (Shapovalov and Taft 1954). Coho salmon generally rear in freshwater for 1 or 2 years prior to out-migration. In more northern parts of their range, such as Alaska, coho salmon may rear in freshwater for up to 4 years before out-migrating (Sandercock 1991, Drucker 1972).

Fry begin to display territorial behavior within a week after emergence (Mason 1966). Fry that emerge early tend to have an ecological advantage over those emerging later. They control better feeding territories, grow at higher rates, and have lower emigration rates (Mason and Chapman 1965).

Juveniles form territories and hierarchies in areas with water current. When food is repeatedly delivered from specific locations, larger individuals generally occupy the best feeding stations (Sandercock 1991, Fausch 1984). In higher velocities, where invertebrates drift in the same direction, coho salmon exhibit the strongest defense of territory. In lower velocity areas, such as off-channel habitats, coho salmon are less territorial, aggregating loosely and scrambling for food (Mundie 1969, Fausch and White 1986, Puckett and Dill 1985, Nielsen 1992). In pools, hierarchies may form into groups, with larger individuals stationed in upstream segments of the pool and smaller individuals in downstream sections (Mundie 1969, as cited in Sandercock 1991).

Individuals unable to defend territories are known as floaters (Puckett and Dill 1985). They occupy space between and around the territories of the dominant fish in glides or pools. Fry that are unable to defend territories may also emigrate downstream to the ocean (Hartman et al. 1982)

or to other areas within the stream system (Shapolalov and Taft 1954). Coho salmon that maintain dominant positions in hierarchies show the highest summer growth rates. Subdominant members of hierarchies grow slower, and floaters grow the slowest of all (Nielsen 1992, Fausch 1984). Allen (1969) estimated the territory size of juvenile coho salmon at 0.34 m^2 (3 ft^2) for fry 49 mm (1.93 in.) in length; 0.79 m^2 (8 ft^2) for juveniles at 4 months of age; and 3.5–5.5 m^2 (39–61 ft^2) for yearling size fish of about 110 mm (4.33 in.) length.

Populations of coho salmon are generally believed to be limited by density-dependent mechanisms operating within the freshwater environment (Allen 1969, Chapman 1962, 1966, McFadden 1969, Marshall and Britton 1990, Meehan and Bjornn 1991). "Habitat conditions are a key factor regulating the production of salmonid smolts. Streams have finite carrying capacities for salmonids, producing similar numbers of young annually if spawning and rearing habitat is not degraded and stocks overfished" (Allen 1969 and McFadden 1969, as cited in House and Boehne 1985, p. 283).

This density dependence has been demonstrated for coho salmon in Pacific Northwest streams (Wickett 1951, Salo and Bayliff 1958, Hunter 1959, Chapman 1965). The availability of suitable rearing habitat is the limiting factor that usually governs the number of juvenile coho salmon produced from a stream system (Larkin 1988, Chapman 1962, 1966).

4.2.4.5 Smolt out-migration and estuarine rearing

Following winter peak flows, juvenile coho salmon emerge from winter hiding areas. In preparation for downstream migration, they feed heavily and grow in size. Outmigration of coho salmon smolt generally occurs in the spring, approximately 1 year after they emerge from gravels at an age termed 1+. Occasionally, coho salmon outmigrate a year later at age 2+. A smaller portion of the out-migration is made up of age 0+ fish migrating downstream as fry, but these fish likely have low probability of surviving to adulthood (Otto 1971, Crone and Bond 1976, Hartman et al. 1982). The proportion of out-migrants that are age 1+ or older generally increases further north in the range of coho salmon. From 40% to 58% of out-migrants in some British Columbia and Alaska rivers are age 2+ or older (Andersen and Narver 1975, Armstrong 1970, both as cited in Sandercock 1991; Mehann and Siniff 1962, Drucker 1972).

Coho salmon appear to have a more protracted out-migration period than other salmon species. Holtby et al. (1989) found that out of 17 years of trapping smolts in Carnation Creek, British Columbia, there was only 1 year in which 50% of the smolts emigrated in less than 14 days. During their study, the period during which 50% of smolts emigrated usually lasted at least 21 days. Unlike other salmon species that migrate as fry or that reach smolt stage at only a few months of age, coho salmon enter smolt stage while migrating downstream. Larger yearlings may be less vulnerable to predation and, therefore, may not need to employ synchronous migration as a predator-swamping tactic. The out-migration of coho salmon generally has been observed at night (Shapovalov and Taft 1954) as with other anadromous salmonids (Mace 1983). This is likely a predator avoidance tactic as piscivorous fish, birds, and mammals generally rely on sight to capture prey.

In general, juvenile coho salmon must reach a fork length of about 100 mm (3.94 in.) for successful smoltification (Drucker 1972, Crone and Bond 1976). Conte et al. (1966) found that maximum salinity tolerance in coho salmon corresponded to a 90 mm (3.55 in.) threshold size. Shapovalov and Taft (1954) found the average size of out-migrating smolts in the range of 103 to 116 mm (4.06 to 4.57 in.) in Waddell Creek, California.

Photoperiod,[2] temperature, water flow, and perhaps lunar or tidal phase are among the environmental factors that may influence smoltification and downstream movement (Parry 1960; Hoar 1965; Wagner 1974; Clarke et al. 1978, 1981; Wedemeyer et al. 1980; Clarke and Shelbourn 1981). McMahon and Holtby (1992) found that the number of coho salmon smolts out-migrating in Carnation Creek, British Columbia increased during temperature declines, but was not strongly influenced by stream discharge or lunar phase. Local factors that signal favorable conditions for rearing downstream within estuaries may influence out-migration of coho salmon smolts. Holtby et al. (1989) found that variability in stream temperatures during spring accounted for 60% of the variability in the median date of coho salmon emigration in Carnation Creek, British Columbia. They hypothesized that this timing/temperature relationship and the timing of adult spawning were adaptations for synchronizing out-migration with windows of opportunity or advantageous conditions in the ocean or estuarine rearing environment. Low flows or high temperatures can accelerate smoltification and advance the timing of out-migration (Shapovalov and Taft 1954, Wedemeyer et al. 1980).

During smoltification, coho salmon may defend their territories less vigorously and migrate downstream in small schools; this is presumably an adaptation for life in the sea (Hoar 1951, Shapovalov and Taft 1954, McMahon and Holtby 1992). In Carnation Creek, McMahon and Holtby (1992) found 95% of smolts aggregated in groups of 5 or more fish. During the smolt out-migration, aggregation size increased and was significantly greater in the estuary than in stream sections. Large smolt aggregations of coho salmon have been observed beneath docks in Oregon estuaries (McAllister 1988) and in low water areas of a Washington estuary with log pilings (Moser et al. 1991). McMahon and Holtby (1992) believed that schooling and cover-seeking behavior of smolts was a continuation of winter parr behavior. When water temperatures in the spring increased above 7°C (44.6°F), they did not observe the return to territorial behavior which was typical in juvenile coho salmon the previous summer.

After reaching the estuary, coho salmon may remain for a few months of residency prior to entering the ocean environment. McMahon and Holtby (1992) found that coho salmon smolts remained in the Carnation Creek estuary for about 2 months (April and May). They noted a similar period of residency for coho salmon in an Oregon estuary (Myers and Horton 1982). Even a short period of estuary rearing may result in enhanced ocean survival, since growth appears to be very rapid in this habitat. Rapid growth during estuary rearing may reduce the vulnerability of coho salmon to predators near shore; predators are a major source of ocean mortality for coho salmon (Holtby et al. 1990).

4.2.4.6 Ocean phase
Coho salmon occur in the ocean along the Pacific coast from Chamula Bay, Mexico (Miller and Lea 1972), to Point Hope, Alaska, through the Aleutians, and from the Anadyr River, USSR, south to Hokkaido, Japan (Scott and Crossman 1973). The genetic characteristics of individual stocks of coho salmon, along with environmental factors, probably determine the timing and pattern of seaward dispersal (Heard 1991). Seasonal movements of salmon at sea are complex and, so far, not linked with any major oceanographic feature (Burgner 1991). However, coho salmon make oriented, rapid, and precise migration to their natal stream (Cury 1994). Salmon migrating to their natal stream may travel at speeds approaching the maximum for sustainable swimming (Burgner 1991).

---

[2] The photoperiod is the duration of an organism's daily exposure to light, considered for its impact on growth and development.

4.2.5 Habitat requirements

4.2.5.1 Adult upstream migration and spawning

Bell (1986) reported that water temperatures ranging from 7.2 to 15.6°C (45 to 60°F) were suitable for adult migration. However, other researchers report that temperatures over 12.7°C (55°F) may result in a marked increase in disease infection rates of adult coho salmon (Fryer and Pilcher 1974, Holt et al. 1975, Groberg et al. 1978). Wedemeyer stated that water temperatures below 13°C (55.4°F) minimize pre-spawning mortalities of adult coho salmon (as cited in McMahon 1983). Temperatures of 25.5°C (77.9°F) or more are lethal to adult migrants (Bell 1973). Dissolved oxygen levels below about 6.5 mg/l (6.5 ppm) may reduce swimming performance of adult coho salmon or cause avoidance reactions during upstream migration (Davis 1975, Davis et al. 1963). Stream velocities of about 3.05–3.96 m/s (10.0–12.9 ft/s) can retard migration (Reiser and Bjornn 1979). Thompson reports that upstream migration of coho salmon requires a stream velocity less than 2.44 m/s (8.0 ft/s) and water depth of at least 18 cm (7 in.) (1972, as cited in Bjornn and Reiser 1991).

Redds are typically in the transitional area at the downstream end (or tail) of pools as they feed into riffles where the water changes from a smooth to a turbulent flow (Hazzard 1932, Hobbs 1937, Smith 1941, Briggs 1953, Stuart 1953). Flow tends to intrude into gravels in these sites, resulting in good intragravel flow and a smaller proportion of fine sediment (Platts et al. 1979). Spawning sites are usually in areas where there are beds of loose, silt free, coarse gravel and nearby cover for adults (Moyle et al. 1989).

Substrates selected for spawning reflect a balance between the flow and depth of the spawning location, as well as the size of the fish and the availability of suitable habitat. As flow, depth, and fish size increase, salmon are able to displace larger particle sizes of substrate. These substrate sizes range from a D_{50}[3] of 5.4 mm [0.22 in.] to a D_{50} of 35 mm [1.38 in.] (Koski 1966 and Chambers et al.1954, as cited in Kondolf and Wolman 1993).

The hydraulic characteristics of redds aid in keeping the majority of the eggs in the gravel during spawning and prevent their loss due to water current. Shapovalov and Taft (1954) estimated that 97% of the eggs spawned lodge in the pit and are properly buried. Burner (1951) reported that the average area of a coho salmon redd was 2.84 m² (30.6 ft²). Similarly, Crone and Bond (1976) note the average area of gravel disturbed by a spawning coho salmon was 2.6 m² (28.9 ft²). Burner (1951) found that the area required by a spawning pair of coho salmon was 11.7 m² (126 ft²). According to Sandercock (1991), redds were generally separated by an area about 3 times their size.

Bell (1986) states that suitable spawning temperatures for adult coho salmon range from 4.4° to 9.4°C (39.9 to 48.9°F). Researchers have observed adult coho salmon in Oregon spawning at temperatures ranging from 2.5 to 12.0°C (36.5 to 53.6°F) (Burner 1951). Briggs (1953) reported coho salmon spawning in California at temperatures of 5.6 to 13.3°C (42.1 to 55.9°F).

4.2.5.2 Egg incubation, alevin development, and fry emergence

Water temperature affects not only the incubation timing, but also the survival of eggs. Suitable water temperatures for incubation of coho salmon eggs are 4 to 13.3°C (39.2 to 55.9°F), with a slightly narrower range of 6 to 10°C (43 to 50°F) considered optimal (Davidson and Hutchinson

---

[3] D_{50} is median particle diameter. It represents the median grain size at which 50% of the sample is coarser and 50% is finer.

1938, Bell 1973, Reiser and Bjornn 1979). Eggs will develop normally at lower temperatures if initial development has progressed to where the eggs are tolerant of cold (Reiser and Bjornn 1979). Beacham and Murray (1990), compiling data from Velsen (1987) and additional sources, noted that coho salmon are apparently adapted to low water temperatures during incubation; coho have the highest survival rates of all 5 Pacific salmon species at incubation temperatures of 1.5°C (34.7°F). Coho salmon are poorly adapted to survive high incubation temperatures. Data compiled by Beacham and Murray (1990) indicated that constant incubation temperatures of 13.5°C (56.3°F) resulted in 50 % embryo mortality. In addition, although egg size was more important in determining salmon alevin weight, incubation temperature was more important in determining alevin length. Coho salmon alevin and fry are proportionately larger when incubated at temperatures of 4°C (39.2°F) than at 8 or 12°C (46.4 or 53.6°F).

Survival to emergence of anadromous salmonid eggs and alevins is closely related to the permeability of spawning gravels and the availability of dissolved oxygen (Cloern 1976, Mason 1976a). Permeability of gravels, in combination with flow (i.e., hydraulic head) and dissolved oxygen concentration, determine the delivery of dissolved oxygen to the eggs and alevins and thus their survival. Permeability of spawning gravels (their relative resistance to flow through the gravels) may be affected by gravel characteristics such as porosity, size, shape, depth, and arrangement of gravel particles, and, perhaps most importantly, amount of fine sediments (Vaux 1962, McNeil and Ahnell 1964, Cooper 1965, Peters 1965, Moring 1975). Fine sediments (also referred to as fines), such as silt, clay, and sand, reduce gravel permeability by filling in the interstices of the gravel particles. This reduces the amount of dissolved oxygen reaching incubating eggs and developing alevins (Wickett 1954; Coble 1961; McNeil 1962a, 1966; Ringler and Hall 1975; Woods 1980). In the Clearwater River, Washington, Tagart (1976) found that dissolved oxygen in 9 redds was inversely related to the percentage of fines under 0.85 mm (0.034 in.) in diameter.

Eggs and alevins require sufficient amounts of dissolved oxygen for good survival and normal development. Dissolved oxygen requirements appear to be highest in the period during and after hatching of the eggs (Alderdice and Brett 1957, Fast et al. 1982). Phillips and Campbell (1961) observed a positive correlation between intragravel dissolved oxygen within redds and survival of coho salmon embryos. They determined that dissolved oxygen levels must average 8 mg/l (8 ppm) for embryos and alevins to survive. Survival of coho salmon eggs and alevins drops significantly at dissolved oxygen levels at or below 6.5 mg/l (6.5 ppm). Levels below 3 mg/l (3 ppm) cause high rates of mortality (Coble 1961, Shumway et al. 1964, Davis 1975).

Reductions in levels of dissolved oxygen can result in delayed hatching and emergence, smaller size of emerging fry, and increasing incidence of developmental abnormalities (Alderdice et al. 1958, Coble 1961, Silver et al. 1963, Shumway et al. 1964, Mason 1976a). Several studies have shown that Pacific salmon embryos subjected to chronically low concentrations of dissolved oxygen hatch later and result in smaller fry than embryos incubated at higher concentrations of dissolved oxygen (Silver et al. 1963, Shumway et al. 1964, Chapman 1988). Reiser and Bjornn (1979) recommend that dissolved oxygen levels be maintained at or near saturation and that temporary reductions fall no lower than 5 mg/l (ppm) for successful incubation.

Fines are often measured as a percentage of the spawning gravel that is smaller in diameter than some designated size. The proportion of fines that have the potential to result in detrimental effects may vary within and among streams. This is to a large extent because permeability is a function of the full range of sediment particles represented, not just the fines (Moring 1975). Moring (1975) suggested that directly measuring permeability with a standpipe may be more

useful for evaluating the quality of spawning gravel than methods that determine proportion of fines. Permeability is a more direct measure of reduced interstitial flow.

Reiser and Bjornn (1979) provided general habitat guidelines for anadromous salmonid incubation. They recommended that fines ≤6.4 mm (0.26 in.) in diameter account for less than 25% of the volume of the spawning gravel. In unlogged Oregon watersheds, the amount of fine sediments less than 3.3 mm (0.13 in.) in diameter that occurred naturally in gravels used by spawning coho salmon varied from 27% to 55% (Koski 1966, Moring and Lantz 1974). An inverse relationship between the proportion of fines less than 3.3 mm (0.13 in.) in diameter in spawning gravels and the subsequent survival-to-emergence of coho salmon fry has been well-documented (Koski 1966, Hall and Lantz 1969, Cloern 1976). Tagart (1984) documented lower survival-to-emergence of coho salmon at higher concentrations of sand (particle size <0.85 mm or 0.034 in.) and fine sediment. Survival was positively correlated with gravel sizes between 3.35 and 26.9 mm (0.13–1.08 in.) in diameter. Survival-to-emergence of fry was high in laboratory and natural stream investigations where less than 5% of fines were below 3.3 mm (0.13 in.) in diameter (Koski 1966, Hall and Lantz 1969, Cloern 1976). In all of these studies, survival-to-emergence dropped sharply when fines exceeded 15% of spawning gravel.

When female salmon build redds, they generally reduce the amount of fines within the gravels (Burner 1951, Kondolf et al. 1993, Kondolf 2000). Fine sediments, particularly sands transported as bedload, can intrude into the gravels of the completed redd and reduce permeability. Fine sediment may also form a seal or cap in the upper layers of the redd gravel (Einstein 1968), impeding or obstructing the emergence of alevins. This process is known as entombment. Koski (1966) hypothesized that mortality within redds was largely due to the inability of fry to emerge from gravel. Phillips et al. (1975) studied emergence of swim-up[4] coho salmon fry placed in various gravel-sand mixtures. Mean rate of survival-to-emergence was 96% in control groups without sand, but decreased precipitously when the amount of fines was above 20%. Additional studies show that mortality may result from entombment, as well as low levels of dissolved oxygen (White 1942, Cooper 1965, Phillips and Campbell 1961, Phillips et al. 1975, Hausle and Coble 1976, Turnpenny and Williams 1980). Selective mortality of larger fry may occur in substrates with higher amounts of sand or smaller gravels. This is because smaller fry can move through the smaller interstitial spaces present in these substrates (Koski 1975).

In many locations, particularly in coastal California, floods are additional, often significant, sources of mortality. Gravel movement can cause displacement of eggs during incubation. It can also cause mortality by physically injuring embryos in the extremely sensitive stages following fertilization (Gangmark and Bakkala 1960). Several researchers have concluded that movement of spawning gravel can be very detrimental to coho survival during incubation (Neave and Wickett 1953, Wickett 1958, Skud 1958, McNeil 1962b). Management activities, such as the removal of obstructions which reduce flow velocity in streams, may result in more frequent movement of spawning gravels and, thus, increased egg mortality.

4.2.5.3 Juvenile freshwater rearing
Within a few days after emergence, coho salmon fry generally disperse upstream and downstream to find suitable rearing habitat. Fry tend to aggregate in backwaters, side channels, stream margins, and other low velocity areas of the stream, especially areas with low light intensity and overhead cover (Nickelson et al. 1992, Ruggles 1966). Water velocity may be the most important factor in habitat selection of coho salmon fry (Fausch 1993). Researchers have observed coho

---

[4] This is the life stage that begins when the alevin has absorbed its yolk sac and begins to swim upward to emerge from the gravels where eggs were deposited.

salmon fry in water velocities ranging from 0 to 39 cm/s (0 to 1.3 ft/s) and water depths ranging from 0 to 73 cm (0 to 2.4 ft). As fry increase in size, the depth and velocity of the water they inhabit also increases; at the same time, their distance from cover decreases (Dollof and Reeves 1990).

Riffles play an important role in the productivity of streams populated by coho salmon because they are a primary source of aquatic invertebrates. Ruggles (1966) proposes a pool-to-riffle ratio of 1:1 as optimum for providing food and cover for rearing coho salmon. Human disturbances that result in loss of LWD or increased sediment input also tend to increase the amount of riffles in streams, while decreasing the pool area.

The availability of high quality habitat in summer is critical to maximize the growth of coho salmon; it may be the main determinant of their freshwater survival and fitness in some streams (Fausch 1993). Overwinter survival of salmonids increases with larger size (Hartman et al. 1987, Hartman and Scrivener 1990). Smolt size correlates with survival of anadromous salmonids in the ocean (Peterman 1982, Bilton et al. 1982, Ward et al. 1989).

Hassler (1987) indicated that juvenile coho salmon prefer temperatures of 10 to 15°C (50 to 59°F) while Brett (1952) cited a narrower preferred range of 12 to 14°C (53.6 to 57.2°F). Brungs and Jones (1977) reported growth of juvenile coho salmon at temperatures from 5 to 17°C (41 to 62.6°F). Stein et al. (1972) observed high growth rates at temperatures from 9 to 13°C (48.2 to 55.4°F). Growth may increase with higher temperatures as long as food supplies are not limiting. Laboratory streams have demonstrated, however, that temperature increases of only 4°C (7.2°F) decrease productivity for coho salmon where food is limiting (Hughes and Davis 1986).

High water temperatures that are below those considered to be lethal may also result in negative impacts to rearing coho salmon (e.g., low growth rates). Stein et al. (1972) reported that growth rate in juvenile coho salmon slows considerably at 18°C (64.4°F); Bell (1973) stated that growth of juvenile coho salmon ceases at 20.3°C (68.5°F). According to Griffiths and Alderdice (1972), decreases in swimming speed may occur at temperatures over 20°C (68°F). During smolting, juvenile coho salmon are very sensitive to increases in temperature. Wedemeyer et al. (1980) reported that smoltification and the onset of desmoltification may accelerate at temperatures over 10°C (50°F) and recommended that water temperatures during the smolt period remain below 12°C (53.6°F). This is to prevent shortened duration of smoltification, onset of desmoltification, and increased incidence of infection and disease.

Recent field studies to determine appropriate stream temperature regimes for coho salmon have focused on presence or absence of the species. Hines and Ambrose (1998) found that the number of days a site exceeded an MWAT of 17.6°C (63.7°F) was one of the most influential variables predicting presence or absence of coho salmon. This indicates that stream temperature thresholds should incorporate a time-of-exposure limit within a significant range of temperatures rather than a single MWAT limit. The authors also caution that persistence of fish under certain conditions does not necessarily imply health or success. Welsh et al. (2001) conducted a similar study in the Mattole River watershed. They found that coho salmon were not present in any streams which had a MWAT greater than 16.7°C (62.1°F) or a MWMT greater than 18.0°C (64.4°F). Likewise, coho salmon were present in all streams with a MWAT lower than 14.5°C (58.1°F) and a MWMT less than 16.3°C (61.3°F).

4.2.5.4 Winter rearing habitat

In the winter, as water temperatures decline and flows increase, coho salmon reduce their feeding; their growth is negligible (Shapovalov and Taft 1954). During the coldest months, the most important habitat element for juvenile coho salmon are deep pools with substantial cover in the form of LWD (Hartman 1965; Bustard and Narver 1975a, 1975b; Tschaplinski and Hartman 1983; Murphy et al. 1984; Bisson et al. 1985; Everest et al. 1986; Glova 1986; Heifetz et al. 1986; Swales et al. 1986; Hartman and Brown 1987; Cederholm et al. 1988; McMahon and Hartman 1989; Shirvell 1990; Nickelson et al. 1992).

Although juvenile coho salmon use pools in all seasons, they show a preference for certain pool types in the winter that provide cover and the best refuge from high-water velocities. Instream cover and areas of slow water are essential for protection against predators and displacement by high flows (Bustard and Narver 1975a, Mason 1976a, Hartman et al. 1982). Deep (> 45 cm or 18 in.), slow (< 15 cm/s or 0.5 ft/s) areas within or near (< 1 m or 3.3 ft) instream cover of roots, logs, and flooded brush appear to constitute preferred habitat (Hartman 1965, Bustard and Narver 1975a), especially during freshets (Tschaplinski and Hartman 1983, Swales et al. 1986, McMahon and Hartman 1989).

Mid-channel pools or glides without LWD that provide habitat for large numbers of coho salmon fry in the summer may become largely unsuitable in the winter due to increased flows (Grette 1985, Sullivan 1986). Over-wintering coho salmon are typically absent from riffles, glides, rubble, and pools or other stream habitats lacking cover (Heifetz et al. 1986, Bustard and Narver 1975a, Tschaplinski and Hartman 1983, Ruggles 1966). Juvenile coho salmon may migrate into secondary channels on floodplains or terraces (Cederholm and Scarlett 1982, Peterson and Reid 1984) or may travel long distances to over-winter in tributaries of the spawning stream to escape high or fluctuating flows in the main channel (Peterson 1982a, 1982b; Cederholm and Scarlett 1982).

Coho salmon prefer large, deep pools during both summer and winter, although they may use pools as shallow as 0.3 m (1 ft) (Nickelson et al. 1992). There are positive correlations between a standing crop of juvenile coho salmon and pool volume (Nickelson and Reisenbichler 1977, Nickelson et al. 1979). Studies by Nickelson in Oregon indicate that pools the size of 10–80 m^3 (333–2667 ft^3) or 50–250 m^2 (556–2826 ft^2), which have sufficient riparian canopy for shading, are optimum for production of juvenile coho salmon (as cited in McMahon 1983). As juvenile salmonids age and grow, they tend to swim to the deepest available pools (Dolloff 1983, Dolloff and Reeves 1990). The availability of suitable pool habitat in a stream may limit the number of juvenile coho salmon during summer or winter, depending upon local conditions.

Because juvenile coho salmon show narrower preferences for pool habitat types in the winter than in the summer, habitat limitations may be more common in the winter. Lack of suitable winter habitat may result in poor survival and many studies indicate that availability of winter habitat may be the ultimate factor limiting coho salmon in rearing areas (Chapman 1966, Mason 1976a, Chapman and Knudsen 1980, McMahon 1983, Nickelson et al. 1992). Tschaplinski and Hartman (1983) documented substantial decreases in numbers of juvenile coho salmon during fall and winter, particularly in response to seasonal freshets. They found that habitats such as deep pools, logjams, and undercut banks with woody debris lost fewer fish during high flow events and maintained higher juvenile populations over the winter. Rodgers (1986, as cited by House et al. 1991) found only 9% of the juvenile coho salmon that were present during the summer remained as smolts in Knowles Creek, a stream that lacked good over-wintering habitat. Mason (1976b) provides additional evidence that winter habitat may limit the number of coho salmon; in an

experiment, supplemental feeding increased the biomass of summer low-flow juveniles nearly seven-fold, with no observed increase in smolt yield the following spring.

4.2.5.5 Large woody debris (LWD)

LWD is a crucial habitat requirement for juvenile coho salmon during all life-stages of their freshwater residency. Juvenile coho salmon select habitat primarily on the basis of water velocity (Shirvell 1990), preferring low velocity habitats throughout the juvenile rearing period. In coastal streams, low velocity habitat conditions are primarily produced by LWD. Depth and light intensity also influence habitat selection of juvenile coho salmon, but to a lesser degree than velocity (Shirvell 1990). Juvenile coho salmon appear to prefer the cover afforded by woody debris to that provided by rocks and rocky substrate, while juvenile steelhead show the opposite affinity (Bustard and Narver 1975a, Bugert 1985). Therefore, the presence or absence of LWD in streams may have an overwhelming influence on the suitability of these streams for rearing coho salmon. McMahon and Reeves (1989) postulate that LWD could be considered a keystone habitat feature for salmonids because of its overwhelming influence on channel morphology (e.g., pool formation, bank condition), sediment and organic matter retention, water velocities, and cover availability.

Numerous studies have investigated the relationship between the habitat of juvenile coho salmon and LWD. Low velocity stream habitats preferred by juvenile coho salmon include pools, beaver ponds, and off-channel habitats, such as side channels, backwaters, and tributaries on floodplains and terraces (Everest et al. 1986, Glova 1986, Taylor 1988, Bugert and Bjornn 1991). Smaller coast range streams tend to have more confined channels and less availability of off-channel habitats; in these streams, pools constitute the primary rearing habitat for coho salmon. LWD creates pool habitat, provides refuge from high velocities during high flows, and provides shade and overhead cover that offers protection against predation (Bisson et al. 1987; Andrus et al. 1988; Bilby and Ward 1989; Robinson and Beschta 1990a, 1990b; Sedell and Beschta 1991). In coastal streams throughout the range of coho salmon, LWD is the primary factor governing stream habitat diversity (Swanson et al. 1976, Lisle 1981, Bryant 1982, Beschta and Platts 1986). Several field studies have related density of juvenile salmonids to pools or hydraulic conditions associated with LWD (Bisson et al. 1988, Dolloff and Reeves 1990, Bugert et al. 1991, Taylor 1991). Underwater observations by Shirvell (1990) found that 99% of all coho salmon fry observed were occupying positions downstream of natural or artificial rootwads during artificially-created drought, normal streamflows, and flood streamflows.

Juvenile coho salmon have a greater chance of survival in undisturbed streams. Many studies have documented reductions in rearing habitat as a result of the removal of LWD (Tschaplinski and Hartman 1983, Bisson and Sedell 1984, Koski et al. 1984, House and Boehne 1987). The number of over-wintering coho salmon is low in stream reaches where debris abundance has been reduced or removed due to stream clearing, streamside logging (Tschaplinski and Hartman 1983, Murphy et al. 1986), or other disturbances (Martin et al. 1986). Tschaplinski and Hartman (1983) and McMahon and Holtby (1992) found that the estimated capacity of streams for coho salmon smolts was reduced in clear-cut areas; debris volume and smolt abundance were significantly lower than in reaches bordered by a forested buffer strip.

Nickelson et al. (1992) found that pool habitat that had been enhanced by the addition of woody debris supported significantly higher densities of coho salmon than pools without woody debris. The addition of woody debris to artificially dammed pools increased the density of juvenile coho salmon inhabiting the pools to levels found in naturally dammed pools. Larger juvenile coho salmon appear to benefit most from availability of deeper pools and cover. A study in a British

Columbia stream found that the standing crop of coho salmon at age 1+ was significantly greater in pools with complex cover and was positively correlated to pool volume in a given section of stream (Fausch and Northcote 1992). Presence or absence of woody debris may also affect mating strategies of coho salmon. Gross (1991) noted that streams with less LWD may favor hooknose males over jacks while increases in either LWD (which serves as hiding cover) or shallow water (that is less accessible to large males) may favor jacks and their mating strategy.

Since low water velocities are more common throughout the stream channel in summer, associations of juvenile coho salmon with LWD may not be as apparent as in winter. Some studies have indicated that cover, such as that provided by LWD or riparian vegetation, is not as important in influencing juvenile distribution during the summer (Bustard and Narver 1975a, Grette 1985, Murphy et al. 1986, Taylor 1988, Nielsen 1992, Fausch 1993), while other studies show that coho salmon prefer areas of reduced light intensity in summer (Bugert and Bjornn 1991). The type and intensity of predator threats may influence dependence on cover during the summer (Bugert et al. 1991).

In late summer or fall when stream temperatures are highest and flows are lowest, juvenile coho salmon move into deeper pools and feeding rates decrease (Shapovalov and Taft 1954). Low summer flows result in reduced rearing areas, increased vulnerability to predators, increased stranding, and increased water temperatures; availability of pool habitat mitigates these effects and may be particularly important for coho salmon in streams where these influences are strongest (Smoker 1955, Nickelson et al. 1979, Peterson and Quinn 1994). Burns (1971) found that the highest mortality of juvenile coho salmon in summer occurred during periods of lowest flow. Neave (1948, 1949) and Smoker (1953) demonstrated correlations between low summer flow and low catch of adult coho salmon 2 years later. Deep pools may be important for moderating late summer temperature extremes. LWD may be especially important to juvenile coho salmon in areas with marked low-flow periods in summer, since it creates deep pools and shade.

McMahon and Holtby (1992) found that coho salmon smolts tended to be associated with LWD in Carnation Creek study reaches. Over 80% of smolts (1260 observations) were within 1 m (3.3 ft) of LWD, and 95% were within 2 m (6.6 ft) of LWD. The LWD jam sites were characterized by deep (> 1 m or 3.3 ft), slow-moving (< 15 cm/s or 0.5 ft/s) water, overhead shade, and structurally complex cover afforded by rootwads, undercut banks, and submerged logs. Most smolts were associated with LWD volumes > 14 m^3 (494 ft^3). Smolts were noticeably scarce in areas with deep pools or bank cover that lacked LWD. The authors theorized that cover and velocity refuges provided by woody debris were important for avoiding predation and preventing displacement into the ocean prior to completion of smoltification.

4.2.5.6 Ocean phase

There are conflicting opinions on whether density-dependent mechanisms are operating at sea for coho salmon. Emlen et al. (1990) found some evidence that coho salmon smolts interact in the near-shore area in a way that generates density feedback on survival. They reported these data mainly to caution against discounting density-dependent mortality at sea. If density-dependent mortality controls the number of coho salmon at sea, hatchery programs to enhance fishery production may be largely ineffective. McGie (1984) had concluded that density-dependent mortality was important in years of low upwelling, but that it was absent in years of high upwelling, when ocean conditions are more favorable for salmon.

Pearcy and Fisher (1988) report that variation in ocean mortality of coho salmon primarily occurs during the first few weeks of ocean residence. Near-shore conditions during late spring and early summer along the coasts of Washington, Oregon, and California may dramatically affect year-class strength (Scarnecchia 1981). Coho salmon along the California and Oregon coasts may be more sensitive to ocean conditions as these areas do not have the extensive bays, straits, and estuaries found in Washington and Alaska. These features may buffer oceanographic effects (Bottom et al. 1986).

Natural changes in climatic conditions may affect populations of coho salmon. For example, a decline in the populations of coho salmon has been associated with ocean warming trends in the northeast Pacific during the period of 1976–1983 (Nickelson 1986; Lawson 1993; Brown et al. 1994). Generally favorable conditions prevailed during the period from about 1945–1975, but upwelling along the Oregon coast has declined in the past 20 years, and marine survival of coho salmon has declined at a similar rate (Bottom et al. 1986, Pearcy et al. 1992, Lawson 1993). The evidence for the positive effects of upwelling is not always strong, however, and the mechanisms that may create benefits to survival of coho salmon from coastal upwelling are not clear (Scarnecchia 1981; Holtby et al. 1990; reviewed by Pearcy et al. 1992). Upwelling along the Pacific coast may be driven by 40–60 year cycles in wind patterns (Ware and Thomson 1991).

Ocean warming along the British Columbia and California coasts has been reported for the last 50 years (Freeland 1990, Roemmich 1992). During periods of climate change, such as warming trends, El Niño events may become particularly intense (Quinn et al. 1986). El Niño events are associated with unusually warm temperatures at the ocean surface and changes in coastal currents and upwelling (NMFS 1995). These ocean conditions result in reductions in primary and secondary productivity and changes in the distributions of predator and prey species; this, in turn, results in reduced growth, survival, and fecundity of coho salmon (Johnson 1988, NMFS 1995). The 1982–83 El Niño had widespread effects on the northeast Pacific and was associated with an estimated 58% reduction in adult survival and a 24–27% reduction in fecundity in populations that rear off the Oregon coast and southward (Johnson 1988). If climate warming continues as predicted, there may be a progressive deterioration of marine habitat for salmonids that could forestall or curtail the recovery of more favorable ocean conditions (Lawson 1993).

4.2.6 Ecological interactions

4.2.6.1 Food web interactions

While in freshwater, juvenile coho salmon feed primarily on aquatic and terrestrial invertebrate drift (Sandercock 1991). Coho salmon depend largely on visual cues for locating food, and rarely feed on non-moving items or benthos. Ideal feeding positions are located in low-velocity areas where energy expenditure is low, adjacent to high velocity water where food delivery rates are high (Mundie 1969). Nielsen (1992) found that dominant individuals, which maintained the optimal feeding positions, fed almost exclusively on aquatic invertebrate drift (chironomids mostly), and that subdominant fish tended to feed on terrestrial invertebrates that dropped into the stream. In most systems coho salmon feed primarily on chironomids, stoneflies, and occasionally crustaceans (Sandercock 1991). Gribanov (1948, as cited in Sandercock 1991) and Mason (1974, as cited in Sandercock 1991) both found that adult insects were the principal life-stage consumed, and secondarily insect larvae. Riffle habitat is the source of much of the aquatic invertebrate production of a stream. Ruggles (1966) proposed a pool-to-riffle ratio of 1:1 as the optimum for providing food and cover for rearing coho salmon. The terrestrial invertebrate component of juvenile coho salmon diet has been correlated with the amount of riparian vegetation adjacent to the streams (Chapman 1965). High flows in winter often present feeding opportunities for juvenile coho salmon. Minakawa and Kraft (1999) found that following these high flows, greater

than 70% of the diet of juvenile coho salmon was composed of springtails; in the periods between high flows, their diet was composed almost exclusively of benthic invertebrates. Yearling coho salmon become piscivorous and often consume other anadromous salmonid fry when available.

4.2.6.2 Diet
During the coho salmon's first few months at sea, they are opportunistic feeders, consuming a wide variety of zooplankton and fish prey (Brodeur 1992). In most northwest Pacific coastal estuaries and bays, coho salmon and Chinook salmon consume mainly invertebrate prey (McCabe et al. 1983; MacDonald et al. 1987, as cited by Brodeur 1991). The most commonly consumed invertebrate prey are euphasiids (krill); in Beacham's (1986) sample of 1364 non-empty stomachs, 54% of total stomach contents were made up of invertebrate prey and 51% of total contents were comprised of euphasiids. In early summer (May–July), while still relatively small, coho salmon tend to feed on small planktonic larvae. Later in the summer they become more piscivorous and the diversity in their diets decreases, consisting mainly of juvenile marine fishes (Brodeur 1992). As coho salmon grow, they may switch from smaller fish, such as sand lance, to larger herring and juvenile rockfish (Beacham 1986); fish prey, in general, constitutes a larger portion of their diet (Brodeur 1991). Prey consumed by coho salmon are generally less than one-fifth their length, but can be up to one-half of the salmon's length (Brodeur 1991).

At sea coho salmon and Chinook salmon are more piscivorous than other Pacific salmon (Beacham 1986). Some differences in diet between coho salmon and Chinook salmon may be attributable to the differences in depths inhabited by the 2 species. Beacham (1986) reported that coho salmon were most abundant at depths of less than 18 m (59 ft), while Chinook salmon were most abundant at depths greater than 18 m.

Changes in the marine food web may result in declines in coho salmon marine survival. Pacific sardine and hake populations, both of which are used as food by coho salmon, have collapsed during the last century and have not recovered in the most recent period of more favorable ocean conditions (Ware and Thomson 1991). Reductions in ocean food supply and intensive hatchery plantings may lead to density-dependent ocean mortality (Brodeur and Pearcy 1990).

4.2.6.3 Predators
Predation is a major component of the mortality suffered by juvenile coho salmon (Sandercock 1991). Predators encountered during the freshwater rearing stage include juvenile steelhead, cutthroat trout, herons, mergansers, otter, and mink. Even where predation is not a major factor affecting coho salmon populations, it may retard recovery when populations are severely depressed (CDFG 1994). Introduced piscivorous fish in some streams may prey on migrating smolts (Kostow 1995). The loss of habitat complexity in streams may make juvenile coho salmon more vulnerable to avian predation by such species as kingfishers, mergansers, and herons. Lonzarich and Quinn (1994) found that predation losses may be up to 50% higher in the simplest habitat type available, indicating the importance of adequate cover and depth in pools. Salmon comprise only a minor part of the diets of harbor seal and sea lions in the Klamath River estuary (Bowlby 1981).

MRC has land in sub-basin tributaries to the South Fork Eel River (Hollow Tree, Mill, and Jack of Hearts Creeks), where Sacramento pikeminnow prey on juvenile salmonids (Nakamoto and Harvey 2002, unpubl. data). Sacramento pikeminnow were first discovered in the South Fork Eel River in 1979. In 1995, over half of the pikeminnow larger than 250 mm (9.8 in.) sampled in the upper South Fork Eel River in August had juvenile salmonids in their stomachs (Nakamoto and Harvey 2002, unpubl. data). Pikeminnow larger than about 100 mm (3.9 in. are potentially

piscivorous; the proportion of fish in their diet generally increases as their length increases. Pikeminnow predation on salmonids appeared to increase after the Potter Valley Project (Lake Pillsbury Reservoir) caused habitat alterations and raised stream temperatures; subsequently, pools filled with sediment from logging (Harvey et al. 2002). However, the pikeminnow's random selection of prey combined with the fact that juvenile coho salmon occur at relatively low densities in the Eel River may prevent adverse effects on the coho salmon population.

During summer, Pikeminnow have occasionally been found in the lower reaches of South Fork Eel River tributaries, such as Hollow Tree Creek, depending on water flow and temperatures (Harvey et al. 2002). Their distribution apparently increases during low-flow years, when water temperatures are warmer, and decreases in high-flow years, when water temperatures are cooler (Harvey et al. 2002). Land management that increases water temperature could further expand their distribution in the South Fork Eel River and its tributaries (Harvey et al. 2002). In addition to pikeminnow predation on coho salmon, the daily interactions between the 2 co-existing species may direct juvenile salmonids to less suitable habitat in the stream and, therefore, decrease their survival rate (Brown and Moyle 1991).

Whether or not Sacramento pikeminnow are present in tributaries, all coho salmon produced in tributaries to the South Fork Eel River must outmigrate through areas with large numbers of pikeminnow during the spring. Pikeminnows in the South Fork Eel River forage on migrating juvenile salmon in the spring (Moyle 2002). However, most of the pikeminnow consumption of salmonids reportedly occur in mid- to late-summer when flows are lower, clearer, and warmer, and the natural ability of salmon to avoid predation may be reduced (Namamoto and Harvey 2002, unpubl. data; Moyle 2002).

Once smolts reach the estuary and the ocean waters near the shore, they may encounter several new predators, including fish, such as spiny dogfish, lamprey, and sharks; piscivorous birds, such as gulls and loons; and marine mammals, such as harbor seals, sea lions, and orcas. Salmon, however, appear to constitute only a small proportion of the diet of most marine mammals (Bowlby 1981, Sandercock 1991).

4.2.7 Sensitivity to anthropogenic disturbances

As a result of their anadromous life history and a full year rearing in freshwater, coho salmon populations are subject to impacts from a variety of sources, including climatic shifts, ocean and freshwater fishing pressure, and natural or man-made habitat changes. Land management has the potential to dramatically affect the suitability of coastal streams for the production of coho salmon. During the freshwater portions of their life cycle, salmon may act as a keystone food resource for terrestrial vertebrate predators and scavengers, and thus they form a critical link between the aquatic and terrestrial ecosystems (Houston 1983, Cederholm et al. 1981, Willson and Halupka 1995). Declines in anadromous fish species, such as coho salmon, may therefore result in cascading effects throughout the trophic system.

Declines in the population of coho salmon in California and Oregon have led to increased regulatory pressure on many activities that potentially affect habitat of coho salmon. On July 25, 1995, NMFS issued a proposed rule to list 3 ESUs of coho salmon as threatened under the ESA (NMFS 1995). The 3 ESUs include the Oregon coast, southern Oregon/northern California, and central California coast populations (NMFS 1995). The present state of these populations is believed to be the result of human-induced conditions (commercial fishing, habitat degradation, and introduction of hatchery fish) exacerbating recent adverse environmental factors, such as drought and poor ocean conditions (NMFS 1995). Degradation of freshwater spawning and

rearing habitat is likely the most important factor causing long-term declines of coho salmon productivity and recent declines in escapement (Pearcy et al. 1992).

Loss of stream habitat is acknowledged as one of the causes of declines of anadromous salmonids in general in the Pacific Northwest and of coho salmon in particular (Nehlsen et al. 1991; Reeves and Sedell 1992; The Wilderness Society 1993). Most of the habitat loss has been the result of watershed disturbances associated with logging, livestock grazing, urbanization, agriculture, mining, and other human activities (Brown et al. 1994, NMFS 1995). These activities typically result in the loss of complex stream habitat that is characteristic of coho salmon streams (Sandercock 1991), particularly a reduction in the amount of LWD and an increase in fine sediment input to the watershed. Other possible factors in the decline of coho salmon are genetic breakdown of native stocks, dam construction, introduced diseases, over-fishing, floods and drought, and climatic change (Brown et al. 1994). In sub-sections 4.2.7.1 through 4.2.7.6, we review several of these causes, which may relate to declines in coho salmon population in the plan area.

4.2.7.1 Changes in hydrologic regimes
Increased peak flows due to logging, grazing, or hydroelectric operations can reduce survival of eggs and alevins through displacement, if gravels are disturbed; juveniles in rearing areas may also be displaced if suitable refuges from high-velocity flows are lacking (Nicholas 1988). Reduced instream flows, due to diversions or reservoir storage, may delay or halt adult and juvenile migrations, limit availability of rearing habitat, and reduce spawning habitat, if minimum water depths are not met (Everest et al. 1985). Droughts may compound oceanic effects by drying up streams, restricting access to some areas, and degrading spawning and rearing conditions (Brown et al. 1994). Lack of sufficient rainfall, resulting in low flows during migration periods, may impede up-stream migration and result in greater vulnerability of the coho salmon to predation (CDFG 1994). Large floods can have deleterious effects as well, scouring stream channels of spawning gravels and woody debris and, thereby, decreasing spawning success and rearing habitat. In recent years, the effects of catastrophic drought and flood have been compounded by stream degradation due to human action.

4.2.7.2 Changes in sediment dynamics
Timber harvesting and road building can increase levels of sediment delivery to channels, which may increase water turbidity, fill pools, and reduce rearing habitat of juvenile coho salmon. Increased water turbidity may have lethal or sub-lethal effects on salmonids. These effects include physiological stress, such as gill trauma and decreased osmoregulatory ability, and behavioral changes, such as delayed migration, decreased feeding rates, and altered prey selection (Bash et al. 2001). Embeddedness of substrates with fine sediments may reduce production (Crouse et al. 1981), primarily by reducing egg-to-emergence survival and aquatic invertebrate production. The sedimentation of coastal estuaries, due to increased upstream erosion, has been documented in rivers of the northern California coast; this reduces good rearing habitat for coho salmon before they migrate to sea (Puckett 1977, Hofstra 1983, Smith 1987). Aggradation of streams from erosion may result in less stable spawning gravels and mortality of eggs and embryos due to gravel movement during freshets (Nawa et al. 1990). Gravel instability may also affect colonization of the streamside area by riparian vegetation. Extremely aggraded streams may have reduced or absent surface flows in the summer (CDFG 1994). Large accumulations of sediment may block juvenile and adult migrations where tributaries join main rivers (Payne and Associates 1989). Intrusion of fine sediments from logging activities into areas, such as pool tails, can cause profound effects on the permeability of spawning gravel and reduce spawning success (McNeil and Ahnell 1964, Peters 1965, Moring 1975). Gravel mining can reduce the

supply of spawning gravels and alter habitat used by rearing coho salmon. Coarse sediment in headwater streams may have particularly persistent and drastic impacts on available habitat.

4.2.7.3 Changes in LWD dynamics

Reduction of LWD in stream channels has been one of the most important long-term effects of forest management on salmonids in North America (Hicks et al. 1991a). Removal of LWD or curtailment of LWD recruitment generally leads to loss of those habitat features most important to rearing juvenile coho salmon and a subsequent decline in anadromous salmonid abundance (Bryant 1980, Toews and Moore 1982, Lestelle and Cederholm 1984, Dolloff 1986, Elliott 1986, Fausch and Northcote 1992). Stream channels tend to become simpler and less stable after the removal of LWD; the habitat complexity that provides substrate diversity, refuges from current velocity, and cover for spawning, feeding, and resting salmonids is also lost (McMahon and Reeves 1989). Several studies have shown that removal of LWD results in wider and shallower channels with little pool volume available at low flows (Bilby 1984, Bisson and Sedell 1984, Heifetz et al. 1986). Cutting of streamside forests and removal of LWD has been observed to decrease frequency and area of pool habitat and increase riffle area (Bryant 1980, Everest and Meehan 1981, Bisson and Sedell 1984). In clearcuts, Bisson and Sedell (1984) observed that the frequency of both pools and riffles appeared to decline, suggesting that the stream profile of gradual steps had changed to a steeper gradient.

Management activities, such as splash damming, stream-cleaning, and timber harvesting, have resulted in the loss of LWD in streams and reduced recruitment from valley slopes (McMahon and Reeves 1989). Sedell and Luchessa (1982) document the extensive removal of LWD and debris jams from streams in the past 150 years. In a basin in which over 50% of the forest had been logged in the past 20 years, House and Boehne (1986) found that the reduction of large conifers in the riparian zone resulted in only 0.4 pieces per 100 m (328 ft) of LWD large enough to influence channel morphology, compared to 18 pieces of LWD per 100 m (328 ft) large enough to influence the channel in a relatively undisturbed section of a small stream in Oregon.

Forests provide LWD to streams that increase aquatic diversity by creating pools and protected backwaters. LWD also provides nutrients and substrate for biological functions, retains sediment, and dissipates energy of the flowing water, which reduces erosion (Maser et al. 1988). Woody debris with a diameter larger than 20 in. is more common in mature and old-growth forests than in young-growth stands where the trees are small (Maser et al. 1988). Forest type and successional stage, disturbance history, and channel size all influence the amount of wood in streams. Streams flowing through young-growth forests and recently harvested areas contain lower numbers of LWD than is found in mature forests (Maser et al. 1988, Meehan et al.1984). The largest trees in old growth or mature forests in the Pacific Northwest, including northern California, are usually conifer species. In some streamside areas where harvest has occurred, hardwood species, such as alder and tanoak, may quickly colonize the cleared areas. Until the conifers eventually out-compete the hardwoods, the only wood available to streams is from the smaller hardwoods.

In general, at any given location in a stream, the larger less mobile, and more rot-resistant conifers will persist longer in streams than the smaller, mobile, and more rot-prone hardwood species. Although instream hardwood logs may function similarly to instream conifer logs, their contributions to aquatic diversity will likely be of smaller magnitude and shorter duration than conifer logs derived from old-growth or mature forest. If recovery of the riparian conifer forest takes an extended period of time before contribution of LWD resumes, there may be a net loss of instream LWD. LWD from smaller hardwood and legacy conifers may decompose, degrade, or

be transported out of the stream before input of LWD resumes; this may result in entire drainage systems devoid of large wood. If this occurs, aquatic habitat can degrade, and populations and diversity of aquatic species may decline.

4.2.7.4 Changes in stream temperatures and water quality

Logging practices that result in increased stream temperatures, such as removing shade over streams, threaten survival and reproduction by coho salmon. Removal of riparian canopy cover exposes more of the stream channel to direct solar radiation. Increased fine sediment inputs can also cause increased stream temperatures by replacing the reflective gravel substrate with darker sediment that could store more solar radiation (Hagans et al. 1986). In addition, sedimentation can reduce intragravel flow, thereby exposing more of the water column to solar radiation (Hagans et al. 1986). Higher temperatures during the incubation period can cause coho salmon to emerge earlier and be displaced by winter freshets (Scrivener and Anderson 1984). High summer water temperatures reduce growth and may cause mortality of juveniles. In addition to its effects on stream temperatures, removal of the riparian canopy also results in reduced input of terrestrial invertebrates and litterfall to the stream, further reducing the allochthonous input that is an important component of stream production. Grazing may also cause degradation of coho salmon habitat through increased upslope and bank erosion and loss of riparian vegetation. Reduction in riparian canopy can also change predation pressures from aerial predators, such as kingfishers.

4.2.7.5 Commercial and recreational harvest

Over-fishing is often mentioned as a major factor contributing to the decline of coho salmon, but its effects are poorly known because catch of wild and hatchery fish are rarely separated (Steward and Bjornn 1990). On the ocean, commercial and sport fishing harvest salmon stocks in a non-discriminatory way. Harvest levels that may be sustainable for hatchery stocks are often too high to be sustained by depressed wild stocks. If conditions within freshwater spawning and rearing areas are poor, salmon cannot rebound quickly from over-fishing; ocean harvest may become an impediment to recovery of populations (CDFG 1994). Coho salmon in areas where females have a strict 3-year life span are particularly vulnerable; these populations lack the resilience to withstand excessive harvest over successive years that might be present in a population of wide-ranging ages. Over-harvest in a single year can decimate the entire population of a stream, if it has only one strong-year class remaining (Brown et al. 1994). Although commercial and recreational fisheries have been severely restricted or closed along much of the west coast for the past few years, these measures have not resulted in increased returns of coho salmon (NMFS 1995). This may be due to continuing habitat degradation and decreasing productivity in streams (Pearcy et al. 1992).

Intensive commercial fishing that preferentially captures the larger and older male coho salmon may enhance jack mating success on the spawning grounds. This would subsequently increase the frequency of jacks in the population, since jack maturity appears to be heritable (Iwamoto et al. 1983, Gross 1991). Reductions of populations may also increase the frequency of coho salmon returning as jacks (Gross 1991), since growth rates could increase at the lower densities found in depressed populations (Bilby and Bisson 1992).

4.2.7.6 Hatcheries

The genetic integrity of native stocks of coho salmon has been affected by the introduction of large numbers of hatchery fish and transplants of different stocks. Nehlsen et al. (1991) suggest that native stocks of coho salmon have high probability of introgression with hatchery stocks. Bartley et al. (1992) noted that transplants of different stocks within California and planting of Oregon and Washington stocks may have effects, such as disruption of locally adapted gene

complexes, swamping and homogenization of native gene pools, and transmittal of non-adaptive traits[5] from hatchery stocks to native stocks. Differences in the genetic structure of native and hatchery stocks can potentially lead to lower survival of subsequent hybrid generations compared with pure wild fish (Steward and Bjornn 1990; Hindar et al. 1991). Studies have shown lower survival of juvenile coho salmon that were offspring of hatchery strays or hybrids of hatchery and wild coho salmon (Smith et al. 1985, Chilcote et al. 1986). Hatcheries have not proven successful in the restoration of native stocks of coho salmon (Withler 1982, Solazzi et al. 1983, Nickelson 1986).

Stocking of juvenile coho salmon may reduce survival of native stocks through increased competition for limited space or food in rearing streams (Nickelson 1986). In streams, survival of juvenile coho salmon from hatcheries can be lower than that of wild coho salmon (Nickelson 1986). Competition for spawning sites can occur when the wild spawners are swamped by larger numbers of hatchery-reared fish. The accelerated growth of fry in hatcheries may result in increased incidence of juvenile coho salmon that return as jacks (Gross 1991). The interaction of hatchery and native stocks also increases disease potential, since native stocks may be exposed to disease organisms originating from hatcheries. For example, some hatchery stocks of coho salmon are known to harbor Bacterial Kidney Disease (BKD), a chronic and slow developing infection that can have adverse effects on coho salmon smolts (CDFG 1994, NMFS 1995).

4.2.8 🔥 Impacts of MLC (2008)

At least 17 CalWater planning watersheds, known to have coho salmon present, experienced some impact from the fires. The fires burned over roughly 15,600 ac of the plan area which were in watersheds with coho salmon. Monitoring efforts in our ASMBs (Annual Salmonid Monitoring Basins) will track cohorts of coho salmon in burned and unburned watersheds.

4.2.9 Key uncertainties

The current and historical abundance of coho salmon in most coastal streams and rivers of Mendocino County is largely unknown. Limited information is available describing historical distribution. The continuation of population estimates will be crucial in providing data to allow more robust estimates of abundance.

Although hypotheses exist, MRC needs more information on factors that may limit the recovery and maintenance of coho salmon populations in the region. Freshwater habitat quality and quantity is likely a limiting factor. Most of the research on habitat relationships of coho salmon has been outside the plan area. Specifics of these relationships may not be the same within the plan area.

MRC management has identified the following key uncertainties, with cross references to validation monitoring programs in parentheses:
- What is the current and historical distribution of coho salmon in coastal streams and rivers of Mendocino County?
- ▶ *Will the abundance of juvenile coho increase as habitat conditions improve over time* (M§13.6.1.2-1)*?*
- How will the Mendocino Fires of 2008 affect cohorts of coho salmon and their habitats?
- What is the contribution of each limiting factor to coho salmon populations?

---

[5] Non-adaptive traits do not promote and may even reduce an organism's fitness to survive or reproduce in an environment.

4.3 Chinook Salmon

4.3.1 Geographic distribution

Chinook salmon (*Oncorhynchus tshawytscha*) are distributed in the Pacific Ocean throughout the northern temperate latitudes in North America and northeast Asia. In North America, they spawn in rivers from Kotzebue Sound, Alaska south to the San Joaquin River in California's Central Valley (Healey 1991). In California, all major rivers of the Central Valley and coastal areas north of San Francisco Bay support Chinook salmon runs (Allen

Illustration by Ron Pittard

and Hassler 1986). They are most abundant in large rivers, and although they occur in many smaller coastal river systems, they are often absent from streams used by coho salmon (Meehan and Bjornn 1991).

Four runs of Chinook salmon occur in California—fall, late fall, winter, and spring (Allen and Hassler 1986, Leet et al. 1992, Mills et al. 1997). Fall-run populations (or "fall Chinook salmon") occur throughout the species' range and are currently the most abundant and widespread salmon runs in California (Mills et al. 1997). Winter-run populations are limited to the Sacramento River basin and were listed as endangered under the federal Endangered Species Act in 1994. Two apparently distinct stocks of spring-run Chinook (or "spring Chinook salmon") occur in California: a Sacramento-San Joaquin population and a Klamath-Trinity population (Moyle et al. 1995). Although other spring Chinook salmon populations may have existed in smaller coastal streams between these 2 basins, such as the Eel River, they have since been extirpated and there is no evidence of recent spawning in these streams (Moyle et al. 1995).

Chinook salmon populations found in the plan area belong to the California Coastal ESU. This ESU, which encompasses all naturally spawning coastal Chinook salmon from Redwood Creek (Humboldt County) through the Russian River (Sonoma County), was formerly included as part of the Southern Oregon and California Coastal ESU. Chinook salmon in the California Coastal ESU were listed as threatened in September 1999 (NMFS 1999).

4.3.2 Local distribution

The timberlands addressed in our HCP/NCCP cover a large area in Mendocino County, extending from Hollow Tree Creek to the Garcia River. Historically, Chinook salmon were present in many of the major rivers within the plan area including the Eel River (Myers et al. 1998), Big River, Albion River (Cherr and Griffin 1979) and Garcia River (Cher and Griffin 1979). Currently, the occurrence and distribution of fall Chinook salmon in the coastal streams and rivers of Mendocino County is largely unknown. Sampling occurs mainly during the summer months. Adult Chinook salmon enter coastal streams in fall. Their progeny typically swim out to the ocean the following spring and early summer. Spring Chinook salmon are not present within the plan area (Moyle et al. 1995).

Spawning ground surveys, out-migrant trapping, and estuary surveys do sample for Chinook salmon, however, such sampling has not been extensive in the region. Prior to 2002, Hollow Tree Creek was the only watershed within MRC forests known to have a spawning population of Chinook salmon. In 2001-2003 and in 2005-2006, CDFG biologists with the Coastal Watershed Planning and Assessment Program observed adult Chinook salmon in our Noyo River tract.[6] In 2002, MRC biologists also observed juvenile Chinook salmon in the Albion River (MRC 2002a).

---

[6] Email to Matt Goldsworthy (MRC) from Sean P. Gallagher (CDFG—Fort Bragg, CA) on 8 December 2006.

4.3.3 Population trends

Recent population trends of Chinook salmon appear quite bleak. The Pacific Fisheries Management Council (PFMC) reported on 29 January 2008 unexpectedly low Chinook salmon returns to California in 2007, particularly to the Central Valley. Adult returns to the Sacramento River, the largest of Central Valley Chinook salmon runs, failed to meet resource management goals (122,000-180,000 spawners) for the first time in 15 years.

Ocean conditions were poor for salmon growth and survival during spring and summer of both 2005 and 2006. The Wells Ocean Productivity Index (WOPI), a composite index of 13 oceanographic variables and indices, weighted heavily by sea level height, sea surface temperature, upwelling index, and surface wind stress, has been used to accurately predict zooplankton, juvenile shortbelly rockfish, and common murre production along the California coast. It is, therefore, a valid indicator of ocean productivity. Index values for spring and summer of 2005 and 2006 were low, indicating poor conditions for growth and survival. In fact, only the El Niño years (1982-83, 1992-93, and 1999) had lower WOPI values.

The WOPI assesses conditions on a local scale for California, but has tracked another index, the Northern Oscillation Index (NOI), which is based on the strength of the North Pacific high pressure cell and describes a broader region of the North Pacific Ocean. In 2005 and 2006, the WOPI decoupled from the NOI, suggesting local conditions on the California coast were worse than for the larger North Pacific region. These results indicate that ocean conditions in spring and summer, when juvenile coho and Chinook salmon enter the ocean, were unfavorable to growth and survival. This may explain the poor returns for coho in 2007 and 2008 and for Chinook salmon in 2007.

Fall Chinook salmon are currently the most abundant and widespread of salmon stocks in California (Mills et al. 1997). However, the abundance of fall Chinook salmon has fluctuated widely over recent decades, with some populations often reaching critically low levels. NMFS reported that trends in abundance of Chinook salmon in the California Coastal ESU were highly variable, with the strongest declines generally occurring in populations that are the furthest south (NMFS 1999).

Data on population trends of Chinook salmon in rivers and streams within the plan area is very limited; most recent data only spans short time periods. In a study of anadromous salmonid resources of Mendocino County, Maahs and Gilleard (1994) and Nielsen et al. (1991) sampled 34 coastal streams and rivers to estimate abundance of Chinook salmon, coho salmon, and steelhead. They found Chinook salmon in Caspar Creek, Ten Mile River, Noyo River, and Hollow Tree Creek. The Ten Mile River population originated from introductions of Chinook salmon in the early 1980s, which resulted in a self-sustaining population.

The MRC trapping program in Hollow Creek provided data on downstream migration of Chinook salmon. Data from the 2000 and 2001 seasons shows extreme variability. MRC biologists captured 2128 juvenile Chinook salmon in spring 2000, but only 46 in 2001 (MRC 2002b). Short-term monitoring of Chinook salmon abundance in Hollow Tree Creek indicates a considerable decline since the late 1980s (NMFS 1999).

For most populations for which sufficient data exists, NMFS reported a general decline in abundance; in addition, almost all coastal populations south of the Klamath River were extremely depressed (NMFS 1998). In the Klamath River basin, fall Chinook salmon escapement has been

estimated annually since 1978; populations have ranged from 113,000 spawning adults in 1986 to approximately 12,000 in 1992 (Mills et al. 1997).

There remains a high degree of uncertainty regarding the status of populations in this ESU. In a NOAA Technical Memorandum (2008), Brian C. Spence et al. concluded the following:

> 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 ESU strongly indicate that this ESU fails to meet low-risk criteria and is therefore at elevated risk of extinction.

4.3.3.1 Hatchery production

Hatchery production of Chinook salmon in California is concentrated in the Central Valley and Upper Klamath-Trinity Rivers ESUs. According to NMFS (1998), the Rogue, Chetco, Redwood Creek, and Eel River basins have received considerable hatchery releases, derived primarily from local sources. There is also a rearing facility for anadromous salmonid operated by the Commercial Salmon Trollers Association on Hollow Tree Creek. On Hollow Tree Creek, 10% of the 4-year-old fish returning in 1991–1992 were hatchery-reared fish that were released in 1987 as fingerlings (Maahs and Gilleard 1994). In addition, the Ten Mile River population may derive from introductions of Chinook salmon in the early 1980s, which have resulted in naturalized Chinook salmon production. However, confirmation of this would likely require genetic work. In the Russian River basin, the Warm Springs Hatchery on Dry Creek (a tributary of the Russian River) rears fall Chinook salmon for release in the Russian River. This facility rears eggs collected at the Warm Springs and Coyote dams. Juveniles are released at both the hatchery and at Coyote Dam. In 1992, Louisiana-Pacific Corporation reared juvenile Chinook salmon in ponds in the upper Russian River watershed near Ukiah.

In the Eel River basin, CDFG is conducting an on-going recovery program for Chinook salmon stock on the main-stem Eel River at the Van Arsdale Fisheries Station. The purpose of this program is to increase the population of fall Chinook salmon to a point where they are self-sustaining. This program is conducted simultaneously with a regime for improved stream flow from the Potter Valley Hydroelectric Project. While the program is aimed at improving Chinook salmon populations in the main-stem Eel River, fish may stray into other tributaries of the Eel River watershed, including Hollow Tree Creek which is within the HCP/NCCP plan area.

4.3.3.2 Commercial and recreational harvest

No known data exists on commercial or recreational harvest of Chinook salmon originating from streams in the plan area. Since 1978, in larger North Coast California river systems, such as the Klamath, sport harvest has varied from 1310 to 22,200 Chinook salmon; it has averaged about 10% of the total number of adult salmon returning to the river (PWA 1994). Sport harvest in rivers within the plan area, however, appears to be virtually nonexistent.

4.3.4 Life history

4.3.4.1 Overview

Chinook salmon are the largest of the Pacific salmon species, reaching weights of up to 99 lb. (45 kg), although most adults weigh from 10 to 40 lb. (4.5 to 18 kg) (Healey 1991, Meehan and Bjornn 1991, Kostow 1995). Chinook salmon have genetically distinct runs differentiated by the timing of spawning migration, the stage of sexual maturity when entering fresh water, the timing of juvenile or smolt out-migration, and other characteristics (Moyle et al. 1989). Fall (or "ocean-type") Chinook salmon migrate to sea during their first year of life—typically within 3 months

after their emergence from spawning gravel. They spend most of their ocean life in coastal waters, and return to their natal river in the fall, a few days or weeks before spawning (Moyle et al. 1989, Healey 1991).

4.3.4.2 Adult upstream migration and spawning

Adult Chinook salmon migrate upstream from the ocean to spawn in their natal streams, although a small percentage may stray into other streams, especially during high water years (Moyle et al. 1989). Throughout their range, adult fall Chinook salmon generally enter estuaries from July to September, remaining in these areas until they become nearly sexually mature; they move upstream as flows increase in the fall. In California, most adult fall-run Chinook salmon enter streams from August through November, with peak arrival usually occurring in October and November (Leet et al. 1992). In the Klamath River basin, upstream migration peaks somewhat earlier, occurring in August and September (Healey 1991). Spawning occurs in river mainstems and tributaries from early October through December.

Adult Chinook salmon appear to be less capable of negotiating fish ladders, culverts, and waterfalls during upstream migration than coho salmon or steelhead (Nicholas and Hankin 1989). This is due in part to their slower swimming speeds and inferior jumping ability compared to steelhead (Reiser and Peacock 1985; Bell 1986, as cited in Bjornn and Reiser 1991). Cruising speeds, which are used primarily for long-distance travel, range from 0 to 1 m/s (0 to 3.3 ft/s) (Bjornn and Reiser 1991). Sustained speeds, which can be maintained for several minutes, range from 1 to 3.3 m/s (3.3 to 10.8 ft/s) (Bjornn and Reiser 1991). Darting speeds, which can only be sustained for a few seconds, range from 3.3 to 6.8 m/s (10.8 to 22.3 ft/s) (Bjornn and Reiser 1991). The maximum jumping height for Chinook salmon is approximately 2.4 m (7.9 ft) (Bjornn and Reiser 1991).

Upon arrival at the spawning grounds, adult females dig shallow depressions or pits in suitably-sized gravels, deposit eggs in the bottom during the act of spawning, and cover them with additional gravel. Over a period of 1 to several days, the females gradually enlarge redds by digging additional pits in an upstream direction (Healey 1991). Redds are typically 10–17 m^2 (108–183 ft^2) in size, although they can range from 0.5 to 45 m^2 (5.4–484 ft^2) (Healey 1991).

Before, during, and after spawning, female Chinook salmon defend the redd area from other potential spawners (Burner 1951). Briggs (1953) observed that the defended area could extend up to 6 m (20 ft) in all directions from the redd. Redds may be defended by the female for up to a month (Hobbs 1937). Males do not defend redds but may exhibit aggressive behavior toward other males while defending spawning females (Shapovalov and Taft 1954). Both male and female adults die within 2 weeks after spawning (Kostow 1995); females defend their redds until they die or become too weak to maintain position over the redd. Adults do not feed while in freshwater, relying instead on fat reserves for maintenance and gonadal maturation.

4.3.4.3 Egg incubation, alevin development, and fry emergence

Egg incubation generally lasts between 40–90 days at water temperatures of 6–12ºC (42.8ºF to 53.6ºF) (Vernier 1969, Bams 1970, Heming 1982, all as cited in Bjornn and Reiser 1991). At temperatures of 2.7ºC (37ºF), it can take up to 159 days for 50% of the eggs to hatch (Alderdice and Velsen 1978, as cited by Healey 1991). The alevins remain in the gravel for 2-3 weeks after hatching; they absorb most of their yolk sac before emerging into the water column.

4.3.4.4 Juvenile freshwater rearing

Following emergence, fry occupy low velocity, shallow areas near stream margins, including backwater eddies and areas associated with bank cover or LWD, where they aggregate in small schools (Lister and Genoe 1970, Everest and Chapman 1972, McCain 1992). The length of time spent rearing in freshwater varies greatly. Juvenile fall Chinook salmon usually migrate to the ocean at 3 to 6 months of age, having reached a size of 2.8–3.5 in. (70–90 mm) (Meehan and Bjornn 1991). In Hollow Tree Creek, downstream migrants of fall Chinook salmon had an average fork length of 1.8–2.0 in. (46–50 mm), when approximately 100 mi (161 km) upstream of the Pacific Ocean (MRC 2002b). Juvenile Chinook salmon feed and grow as they move downstream in spring and summer (Nicholas and Hankin 1989).

Nicholas and Hankin (1989) suggest that the duration of freshwater rearing is tied to water temperatures, with juveniles rearing longer in rivers with cooler water temperatures. Photoperiod or lunar cycles may be important cues for smolt out-migration, although the relative importance of various cues remains unclear (Bjornn 1971, Healey 1991).

Smolt out-migration and estuarine rearing

Chinook salmon may disperse downstream as fry soon after emergence, early in their first summer as fingerlings, in the fall as flows increase, or as yearlings (Healey 1991). Fall fry and fingerlings usually outmigrate from the spawning areas between January and March. Out-migration of larger smolts generally occurs from April through June, with smolts entering the ocean between April and July (Leet et al. 1992). Although fry typically drift or disperse downstream following emergence (Healey 1991), movement upstream or into cooler tributaries has also been observed (Lindsay et al. 1986, Taylor and Larkin 1986). Downstream migrants, especially those migrating at younger ages, typically spend up to several months rearing in estuaries, feeding, and growing before entering the ocean.

Agonistic behavior may decrease in these saline environments; juveniles in estuaries have been observed in aggregations of up to several hundred fish (Reimers 1968). Extended rearing in estuaries results in relatively large juveniles, i.e., 10–16 cm (4–6 in.), entering the ocean in late summer to early fall; this may increase ocean survival (Nicholas and Hankin 1989).

4.3.4.5 Ocean phase

The distribution and migration routes of adult Chinook salmon vary among stocks. In general, most fish migrate north along the Pacific Coast after entering the ocean, but some stocks migrate to the south (Wahle and Vreeland 1978, Meehan and Bjornn 1991). Once in the ocean, Chinook salmon apparently continue to travel in schools (Reimers 1968). Distances traveled from spawning streams to the ocean varies among stocks. Ocean-type Chinook salmon, which includes most fall-runs, migrate to sea as sub-yearlings and tend to remain in nearby coastal waters throughout their ocean lives (Healey 1991). Healey (1991) postulated that ocean-type Chinook salmon are only in the species' southern range (south of latitude 56°N); there they are the most abundant form of Chinook salmon. From the Klamath River, fall Chinook salmon are narrowly distributed in the ocean, ranging between Point Arena (CA) and Cape Blanco (OR) (Leet et al. 1992).

The number of years fish spend at sea also varies by stock. Some stocks are made up of large Chinook salmon that stay in the ocean for 4 to 7 years; other stocks consist largely of 2-year-old fish that have spent only 1 year at sea (Meehan and Bjornn 1991). Ocean conditions are likely an important cause of density-independent mortality and inter-annual fluctuations in escapement sizes.

4.3.5 Habitat requirements

4.3.5.1 Adult upstream migration and spawning

Adult Chinook salmon require water deeper than 24 cm (0.8 ft) and water velocities less than 2.4 m/s (8 ft/s) for successful upstream migration (Thompson 1972, as cited in Bjornn and Reiser 1991). Most Chinook salmon spawn in the mainstem of large rivers and lower reaches of tributaries, although spawning has been observed over a broad range of stream sizes, from small tributaries 2–3 m (6.6–9.8 ft) in width (Vronskiy 1972) to large mainstem rivers (Healey 1991). Chinook salmon prefer low-gradient (<3%) reaches for spawning and rearing, but will occasionally use higher-gradient areas (Kostow 1995). Water temperatures for spawning adult Chinook salmon are reportedly best when <16°C (60°F), and potentially lethal when >23°C (73°F) (Moyle et al. 1995). Spawning site or redd locations are dictated by streambed topography and hydraulic conditions (Burner 1951). Redds are typically located near pool tailouts (i.e., heads of riffles) where concentrations of dissolved oxygen between the layers of gravel are high. Chinook salmon are capable of spawning within a wide range of water depths and velocities, provided that flow between gravel layers is adequate (Healey 1991). Chinook salmon redds are generally reported at depths of 10 to 200 cm (3.9 to 78 in.) and in water velocities of 15 to 100 cm/s (0.5 to 3.3 ft/s), although criteria may vary between races and stream basins. Because of their larger size, for example, fall Chinook salmon are able to spawn in deeper water with higher velocities than are spring Chinook salmon (Healey 1991).

Composition of substrate particle size has significant influence on intragravel flow dynamics (Platts et al. 1979). Chinook salmon may have evolved to select redd sites with specific particle size that will ensure adequate delivery of dissolved oxygen to their incubating eggs and developing alevins. In addition, salmon are limited by the size of substrate that they can physically move during the redd-building process. Selected substrates likely reflect a balance between water depth and velocity, substrate composition and angularity, and fish size. As depth, velocity, and fish size increase, Chinook salmon are able to displace larger substrate particles. Substrates preferred by Chinook salmon consist of sediment sizes ranging from 13–102 mm (0.5–4 in.) in diameter, with less than 25% of fines below 2 mm in diameter (Platts et al. 1979; Bell 1986, Bjornn and Reiser 1991).

4.3.5.2 Egg incubation, alevin development, and fry emergence

Proper embryo development and emergence require suitable water temperatures, dissolved oxygen delivery, and substrate characteristics. Review of the literature suggests that 5.8–14.2 °C (42.5–57.5 °F) is the optimum temperature range for incubating Chinook salmon (Donaldson 1955, Combs and Burrows 1957, Combs 1965, Eddy 1972, Bell 1973, Healey 1979, Reiser and Bjornn 1979, Garling and Masterson 1985). Sub-lethal stress or mortality of incubating eggs from elevated temperatures begins at about 14.4 °C (58 °F) for constant exposures (Combs and Burrows 1957, Combs 1965, Healey 1979).

The deposition of fine sediment in the spawning substrate has a major impact on the delivery of dissolved oxygen to the egg pocket and, consequently, on embryo survival-to-emergence. Several studies have correlated reduced dissolved oxygen levels with mortality, impaired or abnormal development, delayed hatching and emergence, and reduced fry size at emergence in anadromous salmonids (Wickett 1954, Alderdice et al. 1958, Coble 1961, Silver et al. 1963, McNeil 1964, Cooper 1965, Shumway et al. 1964, Koski 1981). Silver et al. (1963) found that low concentrations of dissolved oxygen relate to mortality and reduced size in Chinook salmon and steelhead embryos. Data suggests that growth may be restricted at oxygen levels below saturation (Silver et al. 1963). Fine sediments in the gravel interstices can also physically impair

the fry's ability to emerge through the gravel layer, trapping (or entombing) them within the gravel (Phillips et al. 1975, Hausle and Coble 1976).

4.3.5.3 Juvenile freshwater rearing

Juvenile Chinook salmon tend to use mainstem reaches and estuaries as rearing habitat more extensively than juvenile coho salmon, steelhead, and sea-run coastal cutthroat trout do. Following emergence, fry occupy low-velocity, shallow areas near stream margins, including backwater eddies and areas associated with bank cover, such as LWD (Lister and Genoe 1970, Everest and Chapman 1972, McCain 1992). As fry grow, they move into deeper and faster water further from banks (Hillman et al. 1987, Everest and Chapman 1972, Lister and Genoe 1970). Everest and Chapman (1972) observed at least small numbers of Chinook salmon fry in virtually all habitats sampled in early summer. Because Chinook salmon fry tend to be larger than coho salmon fry upon emergence, they may tend to use areas with higher water velocities than coho salmon (Murphy et al. 1989, Healey 1991). Most researchers have not addressed fry habitat requirements separately from juvenile summer habitat requirements, but there seems to be consensus that Chinook salmon fry prefer quiet, shallow water with cover.

Substantial variability in the depth and velocity preferences of juvenile Chinook salmon has been reported. Juvenile Chinook salmon were in virtually all sampled depths and velocities (Hillman et al. 1987, Murphy et al. 1989). Lister and Genoe (1970) found that juvenile Chinook salmon preferred slow water adjacent to faster water, i.e., 40 cm/s (1.3 ft/s). Everest and Chapman (1972) found emergent Chinook salmon fry using depths less than 60 cm (24 in.) and water velocities less than 15 cm/s (0.5 ft/s).

Juvenile Chinook salmon appear to prefer pools that have cover provided by banks, overhanging vegetation, large substrates, or LWD. As cover increases, so do juvenile densities in pools (Steward and Bjornn, unpubl. data, as cited in Bjornn and Reiser 1991). Water temperature may also influence juvenile habitat use.

Temperatures have a significant effect on juvenile Chinook salmon growth rates. On maximum daily rations, growth rate increases with temperature to a certain point and then declines with further increases. Reduced rations can also result in reduced growth rates. Therefore, declines in juvenile growth rates of anadromous salmonids are a function of both temperature and food availability. Laboratory studies indicate that, with unlimited food supply, growth rates of juvenile Chinook salmon are highest at rearing temperatures of 18.3º to 21.1ºC (65º to 70ºF) (Clarke and Shelbourn 1985, Banks et al. 1971, Brett et al. 1982, Rich 1987). Growth rates decrease at higher temperatures; temperatures above 23.3º C (74º F) are potentially lethal (Hanson 1990).

4.3.6 Ecological interactions

4.3.6.1 Interspecific interactions and competition

A number of studies have attempted to discern the influence of interspecific interactions between juvenile Chinook salmon and other salmonids on habitat preference and migration patterns. Differences in timing of emergence and subsequent growth rates may result in spatial or temporal habitat segregation; this may reduce interspecific competition between species in some areas. For example, in the Big Qualicum River, British Columbia, fall Chinook salmon and coho salmon both occupy stream margin (or lateral) habitats with cover during their first 3 months. Competition for rearing space is reduced by differences in emergence timing and growth rates (Lister and Genoe 1970). Lister and Genoe (1970) observed that juvenile Chinook salmon, perhaps because of their larger size, used higher water velocities in summer than juvenile coho salmon, and moved away from stream margins toward mid-channel areas earlier than coho

salmon. Although coho salmon and Chinook salmon exhibit some degree of habitat segregation on the microhabitat scale, they often inhabit similar regions of streams (Shirvell 1994, Lister and Genoe 1970).

Coho salmon have often been observed to be behaviorally dominant over juvenile Chinook salmon, and thus they may influence habitat use of Chinook salmon. Taylor (1991) found that in streams containing both coho salmon and Chinook salmon, Chinook salmon were more likely to use higher velocity, shallow riffles, while coho salmon formed hierarchies in low-velocity, deep pools. In study streams where coho salmon were absent, Chinook salmon were more likely to inhabit pools, although Chinook salmon appeared to be most abundant in riffles, whether or not coho salmon were present. Coho prefer pools and Chinook salmon prefer riffles. Taylor (1991) suggested that species-specific differences in habitat preferences may be reinforced or exacerbated by the behavioral dominance of coho salmon.

In an experimental stream trough, Stein et al. (1972) found that juvenile coho salmon dominated optimal feeding areas, such as the upstream ends of riffles. In the same study, the presence of juvenile coho salmon reduced growth rates of juvenile Chinook salmon and their access to optimal feeding positions. Stein et al. (1972) suggested that because fall Chinook salmon only rear for a short period in fresh water, the species may only require relatively small territories during this period, moving downstream before late summer flows reduce the amount of habitat available. Stein et al. (1972) concludes that Chinook salmon appear to be better adapted to rearing in mainstem and estuary areas, suggesting a greater tolerance for higher temperatures; coho salmon appear better adapted to rearing in cooler, small tributaries for more extended periods.

Everest and Chapman (1972) observed that timing differences in the spawning and emergence of Chinook salmon and steelhead resulted in size differences and reduced potential for interspecific competition. They found that Chinook salmon and steelhead rearing in sympatry were segregated according to depth, velocity, and substrate characteristics (Everest and Chapman 1972). At age 0+ Chinook salmon typically occupied areas intermediate in depth, velocity, and distance from the stream margin. By comparison, age 0+ steelhead occupied areas shallower, slower, and nearer the bank, while age 1+ steelhead occupied areas deeper, faster, and further from the bank. At age 1+ Chinook salmon occupied similar microhabitats as age 1+ steelhead. Because most Chinook salmon outmigrated earlier in the year, however, competition was reduced through temporal niche segregation (Everest and Chapman 1972). In the Rogue River of Oregon, Reedy (1995) found that juvenile fall Chinook salmon occupied areas closer to the water surface and with lower velocity than age 1+ steelhead, which typically used faster, deeper water. Segregated use of habitat according to depth, velocity, and substrate characteristics has been found to limit competition between rearing Chinook salmon and steelhead in other rivers (Everest and Chapman 1972).

4.3.6.2 Food web interactions
Juvenile Chinook salmon feed on invertebrate drift while rearing in fresh water (Healey 1991). Becker (1973) reported that in the Hanford Reach of the Columbia River, over 95% of their diet consisted of insects, especially adult chironomids. In estuaries, juvenile Chinook salmon may feed on algae, amphipods, fish (e.g., northern anchovy, Pacific herring), and terrestrial insects (e.g., adult Diptera, ants) (Nicholas and Hankin 1989). In the ocean, Chinook salmon feed primarily on small fish such as herring, but also consume amphipods, crab megalopa, squid, and a variety of other organisms (Healey 1980). Adults generally do not feed during their freshwater

spawning migration, relying instead on fat reserves for body maintenance and gonadal maturation.

4.3.6.3 Predators

Predators that feed on eggs, alevins, and juvenile Chinook salmon in rearing habitats include sculpin, trout and other piscivorous fish, as well as river otters, mink, American dippers, mergansers, belted kingfishers, and great blue herons (Healey 1991, Reedy 1995). In addition to these predators, juvenile Chinook salmon migrating downstream are also exposed to predation by birds (e.g., herons, egrets, gulls, and terns), marine mammals, and both native and introduced piscivorous fish. In many systems, juvenile Chinook salmon spend up to several months in estuaries, feeding and growing before entering the ocean (Healey 1991).

4.3.7 Sensitivity to anthropogenic disturbances

During their freshwater residence, Chinook salmon tend to primarily occur in mainstem rivers and larger tributaries. Due to their greater accessibility, these were some of the first areas where human disturbances such as logging, agriculture, and settlement took place. Early valley-bottom logging and the use of splash dams for log transport largely occurred in the mainstems of coastal rivers; this was "particularly devastating" to Chinook salmon habitat (Lichatowich 1989, p. 96). Because Chinook salmon habitat was degraded relatively early on as compared to upper tributary habitats used by coho salmon and steelhead, historical population abundance is particularly difficult to estimate for this species.

Conditions in coastal river habitat used by Chinook salmon appear to have improved since the 1960s, however, and systems have recovered somewhat from certain types of prior damage (Nicholas and Hankin 1989). Mainstem and estuary habitats may actually be recovering from degradation that occurred earlier this century. Smaller streams used by other anadromous salmonids for spawning and rearing have been more recently altered by land-use activities, such as clearcutting and road construction, and remain in degraded condition (Nicholas and Hankin 1989). Because most coastal Chinook salmon are fall-run types, most juveniles leave the rivers during their first summer; watershed disturbances that might increase mortality during winter freshets would not be as important as they are for juvenile coho salmon, steelhead, and cutthroat trout.

4.3.7.1 Changes in hydrologic regimes

Changes in natural flow regimes may impact populations of Chinook salmon through changes in stimuli used for timing of upstream and downstream migrations, dewatering of redds, displacement of fry or juveniles, or scouring of spawning gravels. Flow fluctuations accelerate the rate of downstream migration among Chinook salmon juveniles in laboratory experiments (McPhee and Brusven 1976, as cited in Hunter 1992). Increased peak flows due to logging, grazing, or hydroelectric operations can reduce survival of eggs and alevins through displacement if gravels are mobilized; juveniles may also be displaced if suitable velocity refuges are lacking in rearing areas (Nicholas 1988). Reduced instream flows due to diversions or reservoir storage may delay or halt adult and juvenile migrations, limit availability of rearing habitat, and reduce spawning habitat if minimum water depths are not met (Everest et al. 1985).

4.3.7.2 Changes in sediment dynamics

Timber harvesting and associated road building can cause increased levels of sediment delivery to stream channels, which may cause increased water turbidity, filling of pools, and reduction of rearing habitat for juvenile Chinook salmon. Increased water turbidity may have lethal or sub-lethal effects on salmonids. These effects include physiological stress such as gill trauma and

decreased osmoregulatory ability, and behavioral changes such as delayed migration, decreased feeding rates, and altered prey selection (Bash et el. 2001). Sedimentation and embeddedness of substrates may reduce food availability for salmonids (Crouse et al. 1981), primarily by reducing production of aquatic invertebrate food resources. The sedimentation of coastal estuaries due to increased erosion within the watershed, which has been documented in various coastal northern California rivers, reduces the good rearing habitat available to salmon before they enter the ocean (Puckett 1977, Hofstra 1983, Smith 1987). Increases in fine sediment inputs, that may result from timber harvesting, can substantially reduce spawning gravel permeability and egg-to-emergence survival (McNeil and Ahnell 1964, Peters 1965).

Increased supply of fine sediments to streams can reduce the suitability of spawning and rearing habitats by filling interstitial spaces between sediment particles, reducing intragravel flow and the delivery of dissolved oxygen to incubating eggs and developing alevins (Chapman 1988). Bjornn et al. (1977) found that survival-to-emergence of Chinook salmon declined when percentage of fine sediments (<6.5 mm (0.25 in.)) in spawning substrate was greater than 20–30%.

Sedimentation during the incubation and over-wintering periods may also cause direct mortality by entombing eggs, alevins, fry, and juveniles. Chinook salmon eggs may be more sensitive to reductions in dissolved oxygen than other salmonids, given their large size and small surface-to-volume ratio (Healey 1991). The filling of pools by sediment can reduce the amount of rearing habitat available to juvenile Chinook salmon. Bjornn et al. (1977) found that adding sand and reducing pool volume by 50% reduced abundance of juvenile Chinook salmon by over 66%. Sedimentation may also fill interstitial spaces used as velocity refuge by juvenile salmon during high flow events (Hillman et al. 1987).

4.3.7.3 Changes in LWD dynamics
Reduction of LWD in stream channels has been one of the most pronounced long-term effects of forest management on salmonids in North America (Hicks et al. 1991a); it causes decreased frequency, depth, and complexity of pool habitat used by rearing juvenile salmonids. Removal of LWD from streams throughout coastal Mendocino County, under the direction of CDFG, has taken place since the 1950s (CDFG 1997a, unpubl. data); this has resulted in a general loss of habitat complexity in streams in the plan area. Pool habitat is an important geomorphic feature of channels where Chinook salmon rear; however, such habitat is probably more important to coho salmon (see section 4.2.4). Reduced levels of LWD may limit formation of backwater pools and complex lateral habitat used by emergent Chinook salmon fry (McCain 1992).

4.3.7.4 Changes in stream temperatures and water quality
Logging and grazing practices that reduce riparian vegetation and stream channel shading may increase stream temperature; this, in turn, may reduce survival of adult and juvenile Chinook salmon. In the John Day River (Oregon), high summer water temperatures in mainstem areas appear to reduce usable habitat for juvenile rearing (Lindsay et al. 1986). However, because fall Chinook salmon emigrate to the ocean before the hottest summer months, the effects of elevated water temperatures on juvenile Chinook salmon are generally minimal compared to other salmonids, such as coho salmon and steelhead, that rear in streams during summertime.

Use of fertilizers and pesticides on logging and agricultural areas can reduce water quality and food resources for salmonids (Nicholas 1988).

4.3.7.5 Commercial and recreational harvest
Chinook salmon may be exposed to ocean harvest for several years. Ocean harvest rates of Chinook salmon in Mendocino County are unknown and are difficult to estimate. Restrictions on ocean Chinook salmon harvest in northern California and southern Oregon are likely to have lowered the harvest rate of south-migrating Chinook salmon to levels comparable to north-migrating fish (Nicholas and Hankin 1989). In 1994, the Pacific Fishery Management Council banned commercial salmon harvest off the Washington coast and restricted commercial and sport fishing off the California and Oregon coasts.

4.3.7.6 Hatcheries
Interbreeding between wild and hatchery fish on natural spawning grounds may be a threat to the genetic integrity of wild stocks of Chinook salmon, whether or not the hatchery stock is of native origin. Moyle et al. (1989) point out that hatcheries select for different traits than the natural environment. This is referred to as "domestication selection" because fish are reared under conditions of reduced predation, simplified habitat, artificial feeding, extremely high rearing densities, etc. Hatcheries may also be subject to disease outbreaks, water contamination, and other problems. Hatchery-produced juveniles residing in rivers for an extended period may compete with wild juvenile salmonids for rearing space and food resources. Timing the release of hatchery smolts to avoid competition with juveniles can minimize these effects (Nicholas 1988).

4.3.8 🔥 Impacts of MLC (2008)

At least 5 CalWater planning watersheds, known to have Chinook salmon present, experienced some impact from the fires. The fires burned over 2348 ac of the plan area which were in watersheds with Chinook salmon.

4.3.9 Key uncertainties

The current and historical abundance of Chinook salmon in most coastal streams and rivers of Mendocino County is largely unknown. Limited information is available describing historical distribution. The continuation of population estimates will be crucial in providing data to allow more robust estimates of abundance.

Although hypotheses exist, more information is needed on the factors that may limit the recovery and maintenance of Chinook salmon populations in the region. Freshwater habitat quality and quantity is likely a limiting factor. Most of the details about habitat relationships of Chinook salmon are based upon research outside of the plan area. The specifics of these relationships may not be identical within the plan area.

MRC management has identified the following key uncertainties:
- What is the current and historical distribution of Chinook salmon in coastal streams and rivers of Mendocino County?
- What is the contribution of each limiting factor to Chinook salmon populations?

4.4 Steelhead

4.4.1 Geographic distribution

Steelhead (*Oncorhynchus mykiss*) are distributed throughout the northern Pacific Ocean. Historically they spawned in streams along the west coast of North America from Alaska to northern Baja California. The species is currently known to spawn only as far south as Malibu Creek in southern

Illustration by Ron Pittard

California (Barnhart 1991, NMFS 1996a). Two major genetic groups occur along the west coast of North America: a coastal and an inland group (Schreck et al. 1986, Reisenbichler et al. 1992, NMFS 1996a). Populations in California are believed to belong only to the coastal group (NMFS 1996a, NMFS 2000b).

4.4.2 Local distribution

The steelhead populations found in the assessment area belong to 2 contiguous ESUs (NMFS 1996a). Populations in river basins from Redwood Creek (Humboldt County) south to and including the Gualala River (Mendocino County) belong to the Northern California ESU. Populations in river basins from the Russian River (Mendocino County) south to and including Aptos Creek (Santa Cruz County) belong to the Central California Coast ESU. On August 18, 1997, NMFS listed populations belonging to the Central California Coast ESU as threatened under the federal Endangered Species Act (NMFS 1997). The listing does not include hatchery populations from the Warm Springs Hatchery on Dry Creek, a tributary to the Russian River. On June 7, 2000, NMFS listed the Northern California steelhead ESU, which includes naturally reproducing populations, as well as several natural-origin hatchery stocks (NMFS 2000b).

The plan area extends over a large area in Mendocino County, from Hollow Tree Creek south to the Garcia River. Within this area are watersheds of numerous small coastal streams and large rivers. Major rivers within the plan area include the Eel, Big, Noyo, Navarro, Garcia, and Russian rivers. Winter-run steelhead or "winter steelhead" occur throughout the plan area.

The former landowner (Louisiana Pacific) conducted fish distribution surveys throughout their coastal Sonoma and Mendocino county lands from 1994 through 1996. MRC is currently repeating this effort. NMFS (2000a) has also compiled data on steelhead occurrence for streams in Mendocino and Sonoma counties. Distribution surveys document the consistent presence of steelhead in nearly every stream in the assessment area that is accessible to this species (NMFS 2000a, MRC 2002c). Steelhead are also present within the Albion and Navarro river estuaries. Portions of these estuaries are within the plan area.

4.4.3 Population trends

Recent status reviews (Good et al 2005) indicate that both the Northern California and Central California Coast DPS of steelhead are "likely to become endangered." Nearly 25% of the members of the Biological Review Team who conducted the status review urged that the Central California Coast DPS was "in danger of extinction." The most significant causes for concern for both DPS are the abundance and productivity of steelhead. Although the status review identified a lack of data as a concern, the data available suggests continued declines.

NMFS (1996b) has concluded that populations of naturally reproducing steelhead have been experiencing a long-term decline in abundance throughout their range. Populations in the southern portion of the range have experienced the most severe declines, particularly in streams

from California's Central Valley and southward, where many stocks have been extirpated (NMFS 1996a). Although northern California steelhead populations have also shown downward trends in abundance, these stocks are generally considered to be larger in size and more stable than southern California stocks (Mills et al. 1997). During this century, 23 naturally reproducing populations of steelhead are believed to have been extirpated in the western United States. Many more are thought to be in decline in Washington, Oregon, Idaho, and California. Nehlsen et al. (1991) identified 43 steelhead populations as being of moderate or high risk of extinction. No winter steelhead populations located in the assessment area were identified as at risk.

Based on analyses of dam and weir counts, stream surveys, and angler catches, NMFS (1997) concluded that, of the 160 west coast steelhead stocks for which adequate data were available, 118 (74%) exhibited declining trends in abundance, while the remaining 42 (26%) exhibited increasing trends. In 2000, NMFS again reviewed the most recently available status and trend data and concluded that steelhead in the Northern California ESU were likely to become endangered in the foreseeable future (NMFS 2000b).

 No consistent effort has been made to monitor steelhead populations in California (Mills et al. 1997, Cramer et al. 1995). Population data, therefore, are sparse; methods used and quality of data are inconsistent. Available data include dam and weir counts, hatchery returns, angler data, snorkel surveys, and juvenile rearing surveys. Information on long-term trends in abundance is available for only 3 steelhead populations in the state: (1) Sacramento River above Red Bluff Diversion Dam; (2) Eel River above Cape Horn Dam; and (3) Middle Fork Eel River (Cramer et al. 1995). Additional data on returning adult steelhead exist from counts at Benbow Dam on the South Fork Eel River between 1938 and 1975 (Taylor 1978). Populations in the Sacramento River are outside the plan area and, therefore, outside our discussions.

Counts on the Eel River document steelhead declines beginning in the 1950s (Cramer et al. 1995). At Cape Horn Dam on the upper Eel River, adult steelhead escapement since 1967 (i.e., following the construction of the dam) has ranged from 2200 to less than 100 fish (Mills et al. 1997). The lowest count (approximately 50 fish) occurred in 1993. Counts on the Middle Fork Eel River began in 1966 and significant declines were detected beginning in 1987 (Cramer et al. 1995). Although exhibiting declines, the stock is relatively stable in comparison to other populations. This may be the result of several factors: the fairly remote nature of the watershed; the river not being stocked with hatchery fish; and restrictions on angling harvest. On the South Fork Eel River, steelhead numbers showed a generally declining trend during the period of record. During the first 10 years that steelhead were counted (1938–1947), an average of 18,264 steelhead per year passed Benbow Dam. Between 1966 and 1975, the final 10 years of steelhead counting at Benbow Dam, the average had declined to 3195 fish per year (Taylor 1978).

Steelhead populations in the Central California Coast ESU are believed to have exhibited more substantial declines than those in other ESUs. Definitive abundance trends in the area covered by our HCP/NCCP are difficult to discern, due to the paucity of data on steelhead populations in this ESU. In the 1960s, the California Department of Fish and Game (CDFG 1965, as cited in NMFS 1997) counted 50,000 steelhead in the Russian River. NMFS recently estimated that only 7000 steelhead exist in this drainage (NMFS 1997). Similarly, adult steelhead returning to the Warm Springs Hatchery have been monitored since 1981; they have declined steadily, corroborating information which suggests that naturally reproducing steelhead stocks in this basin are also undergoing declines (Cramer et al. 1995).

4.4.4 Life history

4.4.4.1 Overview

Steelhead is the term used to distinguish anadromous populations of rainbow trout from resident populations. Much life history variability exists among steelhead populations. Populations may be broadly categorized, however, into 2 reproductive groups, most commonly referred to as either winter-run or summer-run, depending on the time at which adults enter rivers at the beginning of their spawning migration. Summer-run steelhead occur within the Northern California steelhead ESU, but are not believed to range further south than the Middle Fork Eel River (NMFS 2000b). Because the assessment area extends only as far north as the South Fork Eel River drainage, only winter-run steelhead are believed to occur in the plan area. The following discussions of life history and habitat requirements, therefore, refer only to winter-run steelhead.

4.4.4.2 Adult upstream migration and spawning

Steelhead return to spawn in their natal stream, usually in their fourth or fifth year of life, with males typically returning to freshwater earlier than females (Shapovalov and Taft 1954, Behnke 1992). A small percentage of steelhead may stray into streams other than those in which they were born. Although the majority of steelhead populations are either primarily winter-run or summer-run, adults may enter spawning streams in almost any month of the year and spawning may occur at any time from January to June (Behnke 1992, NMFS 1996a). Winter-run steelhead populations generally enter spawning streams from fall through spring as sexually mature adults and spawn a few months later in late winter or spring (Roelofs 1985, Meehan and Bjornn 1991, Behnke 1992).

Adult steelhead migrate upstream on both the rising and falling limbs of high flows, but do not appear to move during flood peaks. Some authors have suggested that increased water temperatures trigger movement, but some steelhead ascend into freshwater without any apparent environmental cues (Barnhart 1991). Peak upstream movement appears to occur in the morning and evening, although steelhead have been observed to move at all hours (Barnhart 1991). Steelhead are among the strongest swimmers of freshwater fishes. Cruising speeds, which are used for long-distance travel, are up to 1.5 m/s (5 ft/s); sustained speeds, which may last several minutes and are used to surpass rapids or other barriers, range from 1.5 to 4.6 m/s (5 to 15 ft/s), and darting speeds, which are brief bursts used in feeding and escape, range from 4.3 to 8.2 m/s (14 to 27 ft/s) (Bell 1973, as cited in Everest et al. 1985; Roelofs 1987). Steelhead have been observed making vertical leaps of up to 5.2 m (17 ft) over falls (Roelofs 1987).

During spawning, female steelhead create a depression in streambed gravels by vigorously pumping their body and tail horizontally near the streambed. Steelhead redds are approximately 10–30 cm (4–12 in.) deep, 38 cm (15 in.) in diameter, and oval in shape (Needham and Taft 1934, Shapovalov and Taft 1954). Males do not assist with redd construction, but may fight with other males to defend spawning females (Shapovalov and Taft 1954). Males fertilize the eggs as the females deposit them in the redd. Afterwards the females move to the upstream end of the nest and stir up additional gravel, covering the egg pockets (Orcutt et al. 1968). Females then move 2 to 3 ft upstream and dig another pit, enlarging the redd. Females may dig 6 to 7 egg pockets, moving progressively upstream; spawning may continue for several days to over a week (Needham and Taft 1934). A female approximately 85 cm (33 in.) in length may lay 5000 to 10,000 eggs; fecundity is related to age and length of the adult female and varies between populations (Meehan and Bjornn 1991). In cases where spawning habitat is limited, late-arriving spawners may superimpose their redds atop existing nests (Orcutt et al. 1968).

Although most steelhead die after spawning, adults are capable of returning to the ocean and migrating back upstream to spawn in subsequent years, unlike most other Pacific salmon. Runs may include from 10 to 30% repeat spawners, the majority of which are females (Ward and Slaney 1988, Meehan and Bjornn 1991, Behnke 1992). Repeat spawning is more common in smaller coastal streams than in large drainages requiring a lengthy migration (Meehan and Bjornn 1991). Hatchery steelhead are typically less likely than wild fish to survive to spawn a second time (Leider et al. 1986).

Whereas females spawn only once before returning to the sea, males may spend 2 or more months in spawning areas and may mate with multiple females, incurring higher mortality and reducing their chances of repeat spawning (Shapovalov and Taft 1954). Steelhead may migrate downstream to the ocean immediately following spawning or may spend several weeks holding in pools before migrating (Shapovalov and Taft 1954).

4.4.4.3 Egg incubation, alevin development, and fry emergence
Hatching of eggs follows a 20–100 day incubation period, the length of which depends on water temperature (Shapovalov and Taft 1954, Barnhart 1991). Newly-hatched steelhead alevins remain in the gravel for an additional 14–35 days while being nourished by their yolk sac (Barnhart 1991). Fry emerge from the substrate just before total yolk absorption under optimal conditions; later-emerging fry that have already absorbed their yolk supply are likely to be weaker (Barnhart 1991). Upon emergence, fry inhale air at the stream surface to fill their air bladder, absorb the remains of their yolk, and start to feed actively, often in schools (Barnhart 1991, NMFS 1996c). Survival from egg to emergent fry is typically less than 50% (Meehan and Bjornn 1991), but may be quite variable depending upon local conditions.

4.4.4.4 Juvenile freshwater rearing
Juvenile steelhead (parr) rear in freshwater before migrating to the ocean as smolts. The duration of time parr spend in freshwater appears to be related to growth rate, with larger, faster-growing members of coho from the same brood year smolting earlier (Peven et al. 1994). Steelhead in warmer areas, where feeding and growth are possible throughout the winter, may require a shorter period in freshwater before smolting, while steelhead in colder, more northern, and inland streams may require 3 or 4 years before smolting (Roelofs 1985).

Juveniles typically remain in their natal streams for at least their first summer, dispersing from fry schools and establishing feeding territories (Barnhart 1991). Peak feeding and freshwater growth rates occur in late spring and early summer. In the Smith River of Oregon, Reedy (1995) suggested that rising stream temperatures and reduced food availability occurring in late summer may lead to a decline in steelhead feeding activity and growth rates.

Juveniles either overwinter in their natal streams if adequate cover exists or disperse as pre-smolts to other streams to find more suitable winter habitat (Bjornn 1971, Dambacher 1991). As stream temperatures fall below approximately 7°C (44.6°F) in the late fall to early winter, steelhead enter a period of winter inactivity spent hiding in the substrate or closely associated with instream cover, during which time growth ceases (Everest and Chapman 1972). Age 0+ steelhead (less than 1 year of age) appear to remain active later into the fall than 1+ steelhead (Everest et al. 1986). Winter hiding behavior of juveniles reduces their metabolism and food requirements and reduces their exposure to predation and high flows (Bustard and Narver 1975a), although substantial mortality appears to occur in winter, nonetheless. Winter mortalities ranging from 60 to 86% for 0+ steelhead and from 18 to 60% for 1+ steelhead were reported in Fish Creek in the Clackamas River basin, Oregon (Everest et al. 1988, as cited in Dambacher 1991).

Juveniles appear to compete for food and rearing habitat with other steelhead. Age 0+ and 1+ steelhead exhibit territorial behavior (Everest and Chapman 1972), although this behavior may dissipate in winter as fish reduce feeding activity and congregate in suitable cover habitat (Meehan and Bjornn 1991). Reedy (1995) found that steelhead in the tails of pools did not exhibit territorialism or form dominance hierarchies. Steelhead may overwinter in mainstem reaches, particularly if coarse substrates in which to seek cover from high flows are available (Reedy 1995), or they may return to tributaries for the winter (Everest 1973, as cited in Dambacher 1991).

At the end of the freshwater rearing period, steelhead migrate downstream to the ocean as smolts, typically at a length of 15 to 20 cm (5.9 to 7.8 in.) (Meehan and Bjornn 1991). A length of 14 cm (5.5 in.) is typically cited as the minimum size for smolting (Wagner et al. 1963, Peven et al. 1994). Evidence suggests that photoperiod is the most important environmental variable stimulating the physiological transformation from parr to smolt (Wagner 1974). During smoltification, the spots and parr marks characteristic of juvenile coloration are replaced by a silver and blue-green iridescent body color (Barnhart 1991) and physiological transformations occur that allow them to survive in salt water.

Less is known regarding the use of estuaries by steelhead than for other anadromous salmonid species; however, the available evidence shows that steelhead in many systems use estuaries as rearing habitat. Smith (1990) concluded that even tiny lagoons unsuitable for summer rearing can contribute to the maintenance of steelhead populations by providing feeding areas during smolt migration in winter or spring.

Estuarine rearing may be more important to steelhead populations in the southern half of the species' range. This is due to greater variability in ocean conditions and paucity of high quality near-shore habitats in this portion of their range (NMFS 1996a). Estuaries may also be more important to populations spawning in smaller coastal tributaries due to the more limited availability of rearing habitat in the headwaters of smaller stream systems (McEwan and Jackson 1996). Most marine mortality of steelhead occurs soon after they enter the ocean and predation is believed to be the primary cause of this mortality (Pearcy 1992, as cited in McEwan and Jackson 1996). Because predation mortality and fish size are likely to be inversely related (Pearcy 1992, as cited in McEwan and Jackson 1996), the growth that takes place in estuaries may be very important for increasing the odds of marine survival (Pearcy 1992, as cited in McEwan and Jackson 1996; Simenstad et al. 1982, as cited in NMFS 1996a; Shapovalov and Taft 1954).

Steelhead have variable life histories and may migrate downstream to estuaries as age 0+ juveniles or may rear in streams up to 4 years before migrating to the estuary and ocean (Shapovalov and Taft 1954). Steelhead, migrating downstream as juveniles, may rear for 1–6 months in the estuary before entering the ocean (Barnhart 1991). Shapovalov and Taft (1954) conducted exhaustive life history studies of steelhead and coho salmon in Waddell Creek (Santa Cruz County, CA) and found that coho salmon went to sea almost immediately after migrating downstream, but that some of the steelhead remained for a whole season in Waddell Creek lagoon or the lower portions of the stream before moving out to sea. Some steelhead individuals remained in the lagoon rather than moving out to sea and migrated back upstream and underwent a second downstream migration the following year. In Scott Creek lagoon (Santa Cruz County), Marston (1992, as cited in McEwan and Jackson 1996) found that half of the steelhead rearing in the lagoon in June and July of 1992 were less than 90 mm (3.5 in.) and appeared to be pre-smolts. Coots (1973, as cited in McEwan and Jackson 1996) found that 34% of juvenile steelhead in San Gregorio Creek lagoon captured in summer were juveniles less than 100 mm (3.9 in.) in length.

From these studies and others, it has been shown estuaries provide valuable rearing habitat to juvenile and yearling steelhead and not merely a corridor for smolts migrating to the ocean.

In the Columbia River estuary, McCabe et al. (1983) found that steelhead were one of the 3 most common salmonids present, along with Chinook salmon and coho salmon. In this study, yearling steelhead were an important part of the sampling catch in May in the pelagic areas of the upper Columbia River estuary. Overall, steelhead were captured almost exclusively in the pelagic areas of the upper and lower estuary.

4.4.4.5 Ocean phase
The majority of steelhead spend 1 to 3 years in the ocean, with smaller smolts tending to remain in salt water for a longer period than larger smolts (Chapman 1958, Behnke 1992).   Steelhead grow rapidly in the ocean compared to in freshwater rearing habitats, with growth rates potentially exceeding 2.5 cm (1 in.) per month (Shapovalov and Taft 1954, Barnhart 1991). Steelhead in the ocean for 2 years typically weigh 3.2 to 4.5 kg (7 to 10 lb.) upon return to fresh water (Roelofs 1985). Unlike other salmonids, steelhead do not appear to form schools in the ocean. Steelhead in the southern part of the species' range appear to migrate close to the continental shelf, while more northern populations of steelhead may migrate throughout the northern Pacific Ocean (Barnhart 1991).

4.4.5 Habitat requirements

4.4.5.1 Adult upstream migration and spawning
During their upstream migration, adult steelhead require deep pools for resting and holding (Puckett 1975, Roelofs 1983, as cited in Moyle et al. 1989). Deep pool habitat (> 1.5 m or 4.88 ft) is preferred by summer steelhead during the summer holding period. Steelhead need water with a minimum depth of 18 cm (0.59 ft) and maximum velocity of 240 cm/s (8 ft/s) for successful upstream migration (Thompson 1972, as cited in Everest et al. 1985). Relatively cool water temperatures, between 10° to 15°C (50° to 59°F) are preferred by adults, although they may survive temperatures as high as 27°C (80.6°F) for short periods (Moyle et al. 1989).

Areas of the stream with water depths from about 18 to 137 cm (7 to 53 in.) and velocities from 0.6 to 1.15 m/s (2 to 4 ft/s) are typically preferred for spawning by adult steelhead (Moyle et al. 1989, Barnhart 1991). Pool tailouts or heads of riffles with well-oxygenated gravels are often selected as redd locations (Shapovalov and Taft 1954). The average area encompassed by a redd is 4.4–5.4 m^2 (47–58 ft2) (Orcutt et al. 1968, Hunter 1973, both as cited in Bjornn and Reiser 1991). Gravels ranging in size from 0.64 to 13 cm (0.25 to 5 in.) in diameter are suitable for redd construction (Barnhart 1991). Steelhead pairs have been observed spawning within 1.2 m (4 ft) of each other (Orcutt et al. 1968). Bell (1986) indicates that preferred temperatures for steelhead spawning range from 3.9° to 9.4°C (39° to 48.9°F). Steelhead may spawn in intermittent streams, but juveniles soon move to perennial streams after hatching (Moyle et al. 1989). In the Rogue River drainage, summer steelhead are more likely to spawn in intermittent streams, while winter steelhead typically spawn in permanent streams (Roelofs 1985).

4.4.5.2 Egg incubation, alevin development, and fry emergence
Incubating eggs require dissolved oxygen concentrations, with optimal concentrations at or near saturation. Low dissolved oxygen increases the length of the incubation period and causes emergent fry to be smaller and weaker. Dissolved oxygen levels remaining below 2 mg/l (2 ppm) result in egg mortality (Barnhart 1991). Information available in the literature indicates that preferred incubation temperatures range from 9° to 11°C (48° to 52°F) (McEwan and Jackson 1996, FERC 1993).

4.4.5.3 Juvenile freshwater rearing

After emergence from spawning gravels in spring or early summer, steelhead fry move to shallow-water, low-velocity habitats, such as stream margins and low-gradient riffles, and will forage in open areas lacking instream cover (Hartman 1965, Everest et al. 1986, Fontaine 1988). As fry increase in size in late summer and fall, they increasingly use areas with cover and show a preference for higher-velocity, deeper mid-channel waters near the thalweg (Hartman 1965, Everest and Chapman 1972, Fontaine 1988). In general, age 0+ steelhead occur in a wide range of hydraulic conditions (Bisson et al. 1988), appearing to prefer water less than 50 cm (19.5 in.) deep with velocities below 0.3 m/s (0.98 ft/s) (Everest and Chapman 1972). Age 0+ steelhead have been found to be relatively abundant in backwater pools and often live in the downstream ends of pools in late summer (Bisson et al. 1988, Fontaine 1988).

Older age classes of juvenile steelhead (age 1+ and older) occupy a wide range of hydraulic conditions. They prefer deeper water during the summer and have been observed to use deep pools near the thalweg with ample cover as well as higher-velocity rapid and cascade habitats (Bisson et al. 1982, Bisson et al. 1988). Age 1+ fish typically feed in pools, especially scour and plunge pools, resting and finding escape cover in the interstices of boulders and boulder-log clusters (Fontaine 1988, Bisson et al. 1988). During summer, steelhead parr appear to prefer habitats with rocky substrates, overhead cover, and low light intensities (Hartman 1965, Facchin and Slaney 1977, Ward and Slaney 1979, Fausch 1993). Age 1+ steelhead appear to avoid secondary channel and dammed pools, glides, and low-gradient riffles with mean depths less than 20 cm (7.8 in.) (Fontaine 1988, Bisson et al. 1988, Dambacher 1991). In the Steamboat Creek basin, a tributary to the North Umpqua River in Oregon, Dambacher (1991) found that age 1+ and older juvenile steelhead were most abundant in areas with large boulder substrates compared to other substrate categories.

As steelhead grow larger, they tend to prefer microhabitats with deeper, higher velocity water as locations for focal points. They attempt to find areas with an optimal balance of food supply versus energy expenditure, such as velocity refuge positions associated with boulders or other large rough elements close to swift current with high macroinvertebrate drift rates (Everest and Chapman 1972, Bisson et al. 1988, Fausch 1993). Reedy (1995) indicates that 1+ steelhead especially prefer high-velocity pool heads. Here there are abundant food resources as well as pool tails, which provide optimal feeding conditions in summer due to lower energy expenditure requirements than the more turbulent pool heads. Fast, deep water, in addition to optimizing feeding versus energy expenditure, provides greater protection from avian and terrestrial predators (Everest and Chapman 1972).

Age 1+ steelhead appear to prefer rearing habitats with velocities ranging from 10–30 cm/s (0.3–1.0 ft/s) and depths ranging from 50–75 cm (19.5–29.3 in.) (Everest and Chapman 1972, Hanson 1977, as cited in Bjornn and Reiser 1991). During the juvenile rearing period, steelhead are often observed using habitats with swifter water velocities and shallower depths than coho salmon (Sullivan 1986, Bisson et al. 1988), a species they are often sympatric with. In comparison with juvenile coho salmon, steelhead have a fusiform body shape that is better adapted to holding and feeding in swifter currents (Bisson et al. 1988). Where the 2 species coexist, this generally results in spatial segregation of rearing habitat that becomes most apparent during the summer months. While juvenile coho salmon are strongly associated with low-velocity habitats, such as pools, throughout the rearing period (Shirvell 1990), steelhead will use riffles (age 0+) and higher velocity pool habitats (age 1+), such as scour and plunge pools in the summer (Sullivan 1986, Bisson et al. 1982). In streams within the plan area, coho salmon are sympatric with steelhead in some locations, but patterns of overlapping habitat use are not known.

Preferred rearing temperatures for steelhead range from 7.2° to 14.4°C (45°F to 57.9°F), with optimum temperatures for juveniles occurring from 10° to 12.8°C (50° to 55°F) and lethal temperatures occurring at 23.8°C (74.8°F) (Bell 1991). Preferred migration temperatures are <57°F (<13°C).

4.4.5.4 Lagoon habitat

Lagoon and estuarine habitat are quite important for steelhead and other salmonids. Estuaries serve as important nursery habitat, as they are high in habitat diversity, produce large quantities of low trophic level food, and offer a relatively sheltered environment from predators. Within these dynamic and highly productive environments, juvenile coho salmon and steelhead trout feed and grow while undergoing physiological and behavioral changes associated with smolt transformation. As the juvenile salmonids rear in lagoons and estuaries, additional growth increases their survival rate upon entry into the ocean, especially during years with poor ocean conditions (Cannata 1998).

Smith (1990) found that fish populations utilizing lagoons consisted of freshwater, estuarine, and saltwater species. The number of species increased with lagoon size. Analysis of scales from 27 adult steelhead collected on Pescadero Creek from 1985-1989 showed that 16 fish (i.e., 59.3%) reared in a lagoon their entire lives before entering the ocean. At least 70% of the steelhead sampled reared in the lagoon during some phase of their life. One fish attained 40% of its length while rearing in the lagoon.

Bond (2008) found that during the annual spring emigration in Scott Creek (California), the largest smolts, >150 mm fork length (FL), moved directly to sea, while some smaller smolts remained in the estuary until sandbar formation created a closed freshwater lagoon. High growth rates in the estuary throughout the summer resulted in a near doubling of fork length from the time of estuary entry. The mean FL of spring migrants was 102.2 mm while the mean FL of fall lagoon residents was 195.9 mm. Analysis of the scale morphology of returning adult steelhead indicated that there is strong size-dependent mortality at sea, with estuary reared steelhead showing a large survival advantage.

4.4.5.5 Winter rearing habitat

Steelhead over-winter in pools, especially deep, low-velocity pools with large rocky substrate or woody debris for cover; they also use backwater and dammed pools (Hartman 1965, Swales et al. 1986, Raleigh et al. 1984, Fontaine 1988). Juveniles are known to use the interstices between substrate particles as overwintering cover. Bustard and Narver (1975a) typically found age 0+ steelhead using 10–25 cm (3.9–9.7 in.) diameter cobble substrates in shallow, low-velocity areas near the stream margin. Everest et al. (1986) observed age 1+ steelhead using logs, rootwads, and interstices between assemblages of large boulders (>100 cm or 39 in. diameter) surrounded by small boulders to cobble size (50–100 cm or 19.7–39 in. diameter) materials as winter cover. Age 1+ fish typically stay within the area of the streambed that remains inundated at summer low flows, while age 0+ fish frequently overwinter beyond the summer low flow perimeter along the stream margins (Everest et al. 1986).

In winter, 1+ steelhead prefer water deeper than 45 cm (17.5 in.), while age 0+ steelhead often occupy water less than 15 cm (5.8 in.) deep and are rarely found at depths over about 60 cm (23.4 in.) (Bustard and Narver 1975a). Below 7°C (44.6°F), juvenile steelhead prefer water velocities <15 cm/s (0.5 ft/s) (Bustard and Narver 1975a). Spatial segregation of stream habitat by juvenile coho salmon and steelhead is less pronounced in winter than in summer, although older juvenile steelhead may prefer deeper pools than coho salmon (Bustard and Narver 1975a).

In many coastal California streams, including those in the plan area, the quality of overwintering habitats in tributaries used by steelhead has likely declined due to reductions in in-channel LWD and associated loss of pool habitats and habitat complexity.

4.4.5.6 Ocean phase

Little is known about steelhead use of ocean habitat, although changes in ocean conditions may be important for explaining trends among California coastal steelhead populations (Kostow 1995). Evidence suggests that increased ocean temperatures associated with El Niño events may increase ocean survival as much as 2-fold (Ward and Slaney 1988). The magnitude of upwelling determines the amount of nutrients brought to the ocean surface and relates to wind patterns; it influences ocean productivity and has a significant effect on steelhead growth and survival (Barnhart 1991). Steelhead appear to prefer ocean temperatures of 9°–11.5°C (48.2°–52.7°F) and typically swim in the upper 9–12 m (30–40 ft) of the ocean's surface (Barnhart 1991).

4.4.6 Ecological interactions

4.4.6.1 Food web interactions

Emergent fry initially feed on zooplankton and other microorganisms (Barnhart 1991). Juveniles feed on a wide range of items, primarily those associated with the stream bottom such as aquatic insects, amphipods, aquatic worms, fish eggs, and occasionally smaller fish (Wydoski and Whitney 1979). Juveniles may also feed on spiders, mollusks, and fish, including smaller steelhead (Roelofs 1985). Age 0+ steelhead prefer benthic invertebrates (Johnson and Ringler 1980); larger steelhead, having larger mouths, can consume a broader range of foods (Fausch 1991). In the ocean, steelhead feed on juvenile greenling, squids, amphipods, and other organisms (Barnhart 1991).

Adult summer steelhead do not usually feed in fresh water and can endure long periods without food, during which time their stomachs shrink (Shapovalov and Taft 1954, Roelofs 1987). Summer steelhead, which spend longer periods in fresh water before spawning, may be more likely to feed in freshwater than adult winter steelhead. Food items taken by adults include caddisflies, mayflies, stoneflies, salmon eggs and, infrequently, other fish (Barnhart 1991).

4.4.6.2 Predators

Major predators of adult steelhead include humans, marine mammals, and large pelagic fish. Eggs may be eaten by macroinvertebrates, crayfish, and other fish. Juvenile steelhead may be preyed upon by garter snakes, piscivorous fish, such as older salmonids (including steelhead), freshwater sculpins, introduced piscivorous fish (e.g., smallmouth bass, striped bass), mammals (e.g., river otter, mink), and piscivorous birds (e.g., mergansers, kingfishers, herons, ospreys, loons). Juvenile steelhead have been observed feeding on emergent fry (Shapovalov and Taft 1954).

MRC has land in sub-basin tributaries to the South Fork Eel River (Hollow Tree, Mill, and Jack of Hearts creeks), where Sacramento pikeminnow prey on juvenile steelhead (Nakamoto and Harvey 2002, unpubl. data). Sacramento pikeminnow were first discovered in the South Fork Eel River in 1979. In 1995, over half of the pikeminnow larger than 250 mm sampled in the upper South Fork Eel River in August had juvenile salmonids in their stomachs (Nakamoto and Harvey 2002, unpubl. data). Pikeminnow, larger than about 100 mm, are potentially piscivorous; the proportion of fish in their diet generally increases with length. Habitat alterations and increases in stream temperature from the Potter Valley Project, Lake Pillsbury Reservoir, and pool filling from logging-related sediment appear to increase pikeminnow predation on salmonids (Harvey et al. 2002). However, the non-selective prey selection strategy of pikeminnow, and the fact that

juvenile steelhead occur at relatively low densities in the Eel River, may prevent them from having strong effects on the steelhead population. In addition, because they outmigrate at a relatively large size, steelhead may be less susceptible during migration than other anadromous species.

Pikeminnow sometimes occur in the lower reaches of some South Fork Eel River tributaries, such as Hollow Tree Creek, during summers, depending on flow and water temperatures (Harvey et al. 2002). Their distribution appears to expand during low-flow years when water temperatures in these streams are warmer, and decrease in years with higher flows (Harvey et al. 2002). Land management activities that result in water temperature increases could result in further range expansion in the South Fork Eel River and its tributaries (Harvey et al. 2002). In addition to causing direct mortality through predation on juvenile steelhead, interspecific interactions between the 2 species may force juvenile steelhead to use less optimal habitat in streams where they coexist (Brown and Moyle 1991). Laboratory experiments conducted by Reese and Harvey (2002) indicated that juvenile steelhead growth decreased at temperatures over 20-23°C (68–73°F) in the presence of pikeminnow, likely as a result of competition for food and optimal habitat.

Whether or not Sacramento pikeminnow are present in tributaries, all steelhead produced in tributaries to the South Fork Eel must outmigrate through areas with large numbers of pikeminnow during the spring. Pikeminnows in the Eel River forage on migrating juvenile salmon in the spring (Moyle 2002). However, most of the pikeminnow consumption of salmonids reportedly occurred in mid- to late-summer when flows were lower, clearer, and warmer, and the natural ability of salmon to avoid predation was reduced (Nakamamoto and Harvey 2002, unpubl. data; Moyle 2002).

4.4.7 Sensitivity to anthropogenic disturbances

Because of their anadromous life history and changes in habitat requirements at different life stages, steelhead are vulnerable to a wide range of watershed disturbances, including dams, timber harvest, road construction, recreational use, and other human-related disturbances. The relative importance of anthropogenic and natural disturbances as well as ocean conditions for controlling steelhead populations is uncertain. Coastal steelhead habitats, which historically consisted of old-growth temperate moist conifer forests with streams having high structural complexity, have been significantly altered (Kostow 1995). Production of steelhead in coastal northern California appears to be significantly lower than historically, due at least in part to habitat degradation (NMFS 1996b).

4.4.7.1 Physical barriers to migration and movement

There are no major dams or diversions in the plan area that are known to act as barriers to steelhead migration or movement. However, improperly designed or constructed road crossings and culverts may act as barriers to upstream-migrating steelhead and reduce the amount of habitat available for spawning and rearing.

4.4.7.2 Changes in hydrologic regimes

Changes in natural flow regimes may impact steelhead populations through changes in stimuli used for timing of upstream and downstream migrations; dewatering of redds; displacement of fry or juveniles; scouring of spawning gravels; and changes in the quality and quantity of habitat for different life stages. The effects of increased magnitude and altered timing of peak flows due to logging, grazing, or urbanization may include reduced survival of eggs and alevins through displacement if gravels are mobilized. Juveniles may also be displaced if suitable velocity

refuges are lacking in rearing areas (Nicholas 1988). Droughts may compound oceanic effects by drying up streams, restricting access to some areas, and degrading spawning and rearing conditions (Brown et al. 1994). Lack of sufficient rainfall, resulting in low flows during migration periods, may impede upstream migration and result in greater vulnerability to predation (CDFG 1994). Large floods can have deleterious effects as well, scouring stream channels of spawning gravels and woody debris and thereby decreasing spawning success and rearing habitat. The effects of catastrophic drought and flood events have been compounded in recent years by human-induced stream degradation.

4.4.7.3 Changes in sediment dynamics

Sedimentation of streams resulting from increased erosion may reduce spawning success of steelhead and the carrying capacity of juvenile rearing areas. Sedimentation due to land use activities is a primary cause of habitat degradation for steelhead populations on the west coast (NMFS 1996a). Coarse sediment in headwater streams may have particularly persistent and drastic impacts on available habitat. Increased input of fine sediment resulting from natural or anthropogenic disturbance may be the principle cause of egg and alevin mortality in some areas (Shapovalov and Taft 1954). Filling of interstitial spaces with fine sediments reduces intragravel flow through redds, thereby diminishing concentrations of dissolved oxygen and the removal rate of metabolic wastes (Everest et al. 1985). Alevins that develop in oxygen-deficient gravels are smaller at emergence, placing them at a competitive disadvantage (Doudoroff and Warren 1965, as cited in Everest et al. 1985). Sedimentation also reduces the amount of interstitial habitat available for use as a refuge by juvenile salmonids during high-flow events or low temperatures (Hillman et al. 1987). Bjornn et al. (1977) observed reduced juvenile steelhead abundance in Idaho streams characterized by a high degree of substrate embeddedness. Sedimentation of interstices may also reduce aquatic invertebrate production and, therefore, reduce production of juvenile anadromous salmonids (Suttle et al. 2004). Filling of pools with fine sediments can reduce carrying capacity of rearing habitats for juvenile salmonids (Bjornn et al. 1977). Accumulation of fine organic material in gravel, which may occur following logging or other land use disturbances, can also reduce the amount of dissolved oxygen available to incubating eggs, since the decay of this material consumes oxygen (Barnhart 1991).

Aggradation of streams from erosion may result in less stable spawning gravels and mortality of eggs and embryos due to gravel mobilization during freshets (Nawa et al. 1990). Gravel instability may also affect colonization of the streamside area by riparian vegetation. Extremely aggraded streams may have reduced or absent surface flows in the summer (CDFG 1994). Large accumulations of sediment may block juvenile and adult migrations where tributaries join main rivers (Payne and Associates 1989). Gravel mining can reduce the supply of spawning gravels and alter habitat used by rearing steelhead.

Erosion may result in increased water turbidity. Increased water turbidity may have lethal or sub-lethal effects on salmonids. These effects include physiological stress, such as gill trauma and decreased osmoregulatory ability, and behavioral changes, such as delayed migration, decreased feeding rates, and altered prey selection (Bash et al. 2001). The sedimentation of coastal estuaries due to increased upstream erosion, which has been documented in north coastal California rivers, reduces the area of this very high quality rearing habitat available to salmonids before going to sea (Puckett 1977, Hofstra 1983, Smith 1987).

4.4.7.4 Changes in LWD dynamics

Reduction of LWD in stream channels has been one of the most important long-term effects of forest management on salmonids in North America (Hicks et al. 1991a). Stream channels tend to

become simpler and less stable after the removal of LWD. The habitat complexity that provides substrate diversity, refuges from current velocity, and cover used by spawning, feeding, and resting salmonids is also lost (McMahon and Reeves 1989). Reduced LWD may also limit formation of backwater pools and the complex stream margin habitat used by emergent fry (McCain 1992). Reductions in the amount of LWD in stream channels due to either past removal (stream cleaning) efforts or harvest of streamside trees may reduce (a) the carrying capacity of these streams for juvenile anadromous salmonids, especially of the older age classes which may prefer deeper habitats, and (b) the occurrence of deep pools used by adults during migration and holding (NMFS 1996a). Murphy et al. (1985, 1986) found that higher juvenile steelhead densities occurred in reaches with buffer strips adjacent to clearcuts than in reaches without buffer strips where LWD had been removed. Reduced LWD may also result in decreased retention of (a) spawning gravels, (b) fine and coarse particulate organic matter, and (c) carcasses of anadromous salmonids. All of these are important for nutrient cycling and maintenance of macroinvertebrate communities.

In assessment area streams, past removal of in-channel LWD (Holman and Evans 1964, CDFG 1997, unpubl. data) and logging in forest stands adjacent to streams have reduced the occurrence and recruitment of LWD in many areas used by steelhead. Reductions in the frequency of deeper pools required by older age classes of juvenile steelhead due to LWD removal may result in early migration to mainstem habitats, particularly of age 1+ juveniles. Removal of LWD may also be affecting sediment transport processes by increasing stream power and reducing the amount of alluvial material stored in tributaries. This, in turn, reduces the availability of substrate for spawning and overwintering in these tributaries.

4.4.7.5 Changes in stream temperatures and water quality

Factors that result in increased stream temperatures, such as large-scale clearcutting, removal of riparian vegetation, and changes in natural flow regimes, may reduce steelhead populations both directly through increased mortality and indirectly through such factors as changes in growth rates or timing of emergence and downstream migration. Logging practices that result in increased stream temperatures, such as removing shade over streams, threaten survival and reproduction by steelhead. Removal of riparian canopy cover exposes more of the stream channel to direct solar radiation. Increased fine sediment inputs can also cause increased stream temperatures by replacing the reflective gravel substrate with darker sediment that could store more solar radiation (Hagans et al. 1986). In addition, sedimentation can reduce intragravel flow, thereby exposing more of the water column to solar radiation (Hagans et al. 1986). High summer water temperatures reduce growth and may cause mortality of juveniles. In addition to its effects on stream temperatures, removal of the riparian canopy also results in reduced input of terrestrial invertebrates and litterfall to the stream, thereby reducing the allochthonous input that is an important component of stream production. Grazing may also cause degradation of steelhead habitat through increased upslope and bank erosion as well as loss of riparian vegetation.

4.4.7.6 Commercial and recreational harvest

Although sport, tribal, and commercial harvest of steelhead has not occurred on the same scale as other species of Pacific salmon, harvest may be a factor in the decline of some steelhead populations (NMFS 1996b). Estimated sport catch of steelhead in California in the early 1960s was 120,000 adult fish (CDFG 1965, as cited in NMFS 1996b). In the late 1970s and early 1980s, the recreational steelhead harvest rates (percent of population harvested) were estimated to range from 12.5–28% in the Trinity River (CDFG, unpubl. data, as cited in NMFS 1996b) and 5.9–20.2% in the South Fork Trinity River (Mills and Wilson 1991, Wilson and Mills 1992, Wilson and Collins 1992, all as cited in NMFS 1996b). In 1991, CDFG (1991) estimates there

were 99,700 steelhead anglers in California, and steelhead catch in 1993 was in the vicinity of 40,000 fish (NMFS 1996b). However, recreational harvest of wild (non-hatchery) steelhead is currently prohibited in streams draining the assessment area.

Although there is generally little or no commercial or recreational harvest of steelhead in the ocean, some evidence suggests that driftnet fishing may play a role in the decline of Pacific Coast steelhead populations (Light et al. 1988, as cited in NMFS 1996b). In California, gillnet scars have been observed on adult steelhead returning to the Smith River (Higgins et al. 1992) and several streams in Santa Cruz County (NMFS 1996b). Steelhead may also be caught incidentally by large-scale commercial driftnet operations targeting salmon or squid, as well as by illegal high seas driftnet fishing.

4.4.7.7 Hatcheries

Interactions between hatchery and wild steelhead stocks may have contributed to population declines of Pacific Coast steelhead (NMFS 1996b). Wild stocks of steelhead may be adversely impacted by hatchery supplementation through increased competition during the juvenile rearing period and genetic changes resulting from interbreeding between wild steelhead and hatchery steelhead not native to the basin (Moyle et al. 1989). Widespread use of hatchery fish has been cited as contributing to declines in wild steelhead stocks on the Oregon coast (Nehlsen et al. 1991). Differences in the genetic structure of native and hatchery stocks can potentially lead to lower survival of subsequent hybrid generations compared with pure wild fish (Steward and Bjornn 1990, Hindar et al. 1991). The interaction of hatchery and native stocks also increases disease potential, since native stocks may be exposed to disease organisms originating from hatcheries that they would not be exposed to under normal conditions.

Poaching and other impacts on adult holding habitat

Summer steelhead adults are vulnerable to human disturbance during their holding period. Holding steelhead are vulnerable to poaching, because they typically congregate in large numbers in a relatively small number of suitable pools. Steelhead fishing has been restricted in many areas in response to population declines, but the species remains vulnerable to poaching. Adult summer steelhead are especially vulnerable to poaching during summer low flows. Roelofs (1983, as cited by Moyle et al. 1989) has indicated that steelhead populations showing signs of severe declines tend to be in areas that are more accessible to people, while stable populations tend to be found in the most inaccessible streams. Poachers may capture adult steelhead by snagging, spearing, netting, trapping, shooting, or blasting (Roelofs 1987). In both mainstem and tributary streams, increased human disturbance associated with recreational activities, such as boating, swimming, or fishing may affect adult holding habitat. Moyle et al. (1989) indicate that these types of activities may stress adult fish and result in increased mortality in streams heavily used for recreation. These impacts would not affect winter steelhead, which do not require extended use of holding areas prior to spawning.

4.4.7.8 Estuary impacts

Estuary conditions may have an important influence on anadromous fish survival, since anadromous fish must pass through these areas during upstream and downstream migration and since estuarine rearing prior to ocean entry is a life history strategy used by many juvenile anadromous fish to increase marine survival (Giger 1972, Healey 1991, McMahon and Holtby 1992). Degradation of estuary habitats due to diking and filling, increased temperatures, introduction of piscivorous fish, sedimentation due to upstream impacts, and other human activities may have contributed to anadromous fish declines in coastal northern California.

4.4.8 🔥 Impacts of MLC (2008)

There were 28 CalWater planning watersheds, known to have steelhead present, which experienced some impact from the 2008 fires. Nearly all the acres burnt over by the fires were in watersheds with steelhead. Monitoring efforts in our ASMBs (Annual Salmonid Monitoring Basins) will track steelhead in burned and unburned watersheds.

4.4.9 Key uncertainties

The current and historical abundance of steelhead in most coastal streams and rivers of Mendocino County is largely unknown. Limited information is available describing historical distribution. The continuation of population estimates will be crucial in providing data to allow more robust estimates of abundance.

Although hypotheses exist, more information is needed on the factors that may limit the recovery and maintenance of steelhead populations in the region. Freshwater habitat quality and quantity is likely a limiting factor. Most of the details about habitat relationships of steelhead are based upon research outside of the plan area. The specifics of these relationships may not be identical within the plan area.

MRC management has identified the following key uncertainties, with cross references to validation monitoring programs in parentheses:

- What is the current and historical distribution of steelhead in coastal streams and rivers of Mendocino County?
- ▶ *Will the abundance of juvenile steehead increase as habitat conditions improve over time* (M§13.6.1.2-1)*?*
- What is the contribution of each limiting factor to steelhead populations?

4.5 Northern Red-legged Frog and California Red-legged Frog

4.5.1 Dual focus

There are 2 subspecies of red-legged frog in the plan area. The California red-legged frog (*Rana draytonii)*, listed by USFWS, occurs in the southern portions, while the northern red-legged frog (*Rana aurora*), unlisted, occupies the northern portions. An overlap between species occurs near the Elk Creek Watershed. Our HCP/NCCP provides consistent conservation measures for each.

Photo by Dr. Mark Jennings

4.5.2 Systematics and taxonomy

The taxonomic relationship of the 2 subspecies of red-legged frog is unclear. Confusion in identification occurs where the distributions of the 2 subspecies overlap in northern California (Green 1985, Hayes and Krempels 1986). Significant differences in behavior and morphology of the 2 subspecies suggests that they may be separate species in secondary contact following previous isolation (Hayes and Krempels 1986). Recent investigations by Dr. H. Bradley Shaffer suggest that the species may soon be split into 2 full species: the northern red-legged frog and the California red-legged frog.

The 2 subspecies are difficult to distinguish even in the hand, although California red-legged frogs tend to have more numerous dorsal spots with light centers, rougher skin, shorter limbs, and smaller eyes than northern red-legged frogs (Stebbins 1985).

4.5.3 Geographic distribution

Red-legged frogs occur west of the Cascade Range and Sierra Nevada from British Columbia, Canada south to northern Baja California, Mexico (Stebbins 1985). Northern red-legged frogs occur from Sullivan Bay, British Columbia south to northern Humboldt County, California (Stebbins 1985). Populations of red-legged frogs along the west coast of California from southern Humboldt County south to Point Reyes, Marin County exhibit some characteristics that are intermediate between northern and California red-legged frogs (Hayes and Miyamoto 1984, USFWS 1994a), but appear more closely affiliated with the northern subspecies (Jennings and Hayes 1994).

USFWS considers the California red-legged frog as *threatened* wherever they are found to occur. The historical range of the California red-legged frog included Pacific Slope drainages from Point Reyes in Marin County, California inland to the vicinity of Redding in Shasta County and south to Baja California, Mexico. Currently, they are found primarily in small coastal drainages between Point Reyes south to Santa Barbara County.

The known range of elevations for northern red-legged frogs and intermediate populations extends from near sea level to 1160 m (3800 ft) (Dunlap 1955, as cited in Jennings and Hayes 1994). In California, northern red-legged frogs and intermediate populations occur from near sea level to about 300 m (Jennings and Hayes 1994).

4.5.4 Local distribution

Northern red-legged frogs were detected adjacent to MRC land in the Jackson Demonstration State Forest (JDSF) during baseline wildlife surveys (Kitchen 1992), and one record for JDSF was listed in the California Natural Diversity Database (CDFG 1996a). MRC biologists and foresters have collected information on amphibian presence in the plan area from (1) electrofishing and snorkeling surveys for fish (MRC 2002a); (2) diurnal amphibian surveys on Class II streams conducted in summer 2001 (MRC 2002d); (3) incidental observations during forestry and wildlife surveys (MRC 2002c) and (4) baseline distribution surveys.

MRC has conducted surveys to identify the baseline distribution of red-legged frog breeding sites since 2002. To date, we have detected occupied breeding sites in the following basins: Doyle Creek, Albion River, Railroad Gulch (tributary to Albion River), Ray Gulch (tributary to Navarro River), Greenwood Creek, South Fork Elk Creek, and Mallo Pass Creek. We also observed adult red-legged frogs in Hollow Tree Creek, Juan Creek, Big River, and Navarro River, although we do not have confirmed evidence of reproduction.

MRC has not detected any life stages of red-legged frogs during surveys in Cottaneva Creek, North Fork Noyo River, Russell Brook Creek, East Branch North Fork Big River, Mettick Creek, Ackerman Creek, South Branch North Fork Navarro, North Fork Navarro, Little North Fork Navarro, John Smith Creek, Rolling Brook, and the South Fork Garcia River planning watersheds.

In 2003 Dr. H Bradley Shaffer conducted genetic analyses on larval red-legged frogs from breeding ponds in Doyle Creek, Albion River, Greenwood Creek, and Mallo Pass. Larval frogs from ponds in Doyle Creek and Albion River were determined to be 100% northern red-legged frogs. Larval frogs from ponds in Greenwood Creek and Mallo Pass Creek were determined to be hybrid frogs. These hybrid frogs possessed mitochondrial and nuclear DNA of both northern red-legged frogs and California red-legged frogs. The results of Dr. Shaffer's work conclude that red-

legged frogs residing south of Point Arena may possess 100% California red-legged frog genes (Shaffer et al. 2004).

Red-legged Frog Egg Mass
Photo by Jen Cary

4.5.5 Population trends

◆ Northern red-legged frog

Although northern red-legged frogs are declining in British Columbia, Oregon, and Washington, there have not been systematic surveys in northern California.

◆ California red-legged frog

Populations of red-legged frogs in California have declined drastically statewide (Fisher and Shaffer 1996, Hayes and Jennings 1986, Moyle 1973). They are believed to have disappeared from 75% of their historical range (Jennings et al. 1993). The species was extirpated from the Central Valley by 1960 and is currently extremely rare in the foothills of the Sierra Nevada (Jennings et al. 1993). Declines in the species' abundance began in the mid-1800s when they were hunted for food by humans. Over half of the reduction in their range, however, has likely taken place in the past 25 years (Jennings et al. 1993, as cited in Davidson 1993). Habitat loss and alteration are probably the primary causes of decline. Small coastal drainages between Point Reyes National Seashore in Marin County and the town of Carpenteria in Santa Barbara County are some of the only remaining areas with significant numbers of California red-legged frogs (Davidson 1993). Recent investigations into the current population status of this species by Davidson et al. (2001) led him to suggest that declines show the strongest association with upwind agricultural land use. Consequently, wind-borne agricultural pesticides are an important factor in declines. For the examined species, patterns of decline were marginally consistent with ultraviolet radiation and inconsistent with hypotheses on climate change.

4.5.6 Life history

4.5.6.1 Reproduction

◆ Northern red-legged frog

Male northern red-legged frogs assemble at breeding sites as early as mid-December in warm winters and vocalize beneath the water's surface (Nussbaum et al. 1983). Males have been observed at breeding sites up to a month before females, in water temperatures as low as 2°C (35.6°F). Oviposition occurs early in the year, during a relatively restricted time period from January to March, in ponds and in intermittent and permanent streams with slow or still water. The timing of egg-laying may be related to water temperature. A study conducted in Willamette Valley, Oregon found that egg-laying did not occur until temperatures were above 7.5°C (45°F), although lower minimum temperatures for egg-laying have been documented in populations located further north (Nussbaum et al. 1983). Spawning occurs only at night (Licht 1971). Eggs are typically deposited as a grapefruit-sized mass consisting of 200 to 1100 eggs, which is attached to emergent vegetation, such as cattails and bulrushes in ponds or other still water habitats (USFWS 1997a). Egg masses of northern red-legged frogs are deposited at least 46 cm (18 in. below the water surface and at least 61 to 92 cm (2 to 3 ft) from the water's edge; some egg masses may break free and float to the surface after a few days (Licht 1971). Once egg-laying is complete, adults leave breeding sites and disperse into moist areas with dense, thick vegetation, where they may be found throughout late spring and summer (Jennings and Hayes 1994).

🔹 California red-legged frog

California red-legged frogs breed from late November to early April (Jennings and Hayes 1989, as cited in Davidson 1993). Unlike northern red-legged frogs, California red-legged frogs vocalize in the air rather than underwater. Their egg masses are laid attached to emergent vegetation in a vertical orientation in contact with the water surface rather than submerged (Hayes and Miyamoto 1984, Hayes and Krempels 1986).

4.5.6.2 Growth and development

Northern red-legged frog eggs are relatively large, averaging about 3.03 mm (0.12 in.) in diameter (Licht 1971). Depending on water temperatures, hatching can occur after less than a week (at 20°C) or take more than 8 weeks (at 4.5°C). Hatching typically occurs in 4-5 weeks at temperatures between 8° to 12°C (47° and 53°F) (Nussbaum et al. 1983). Storm (1960) and Licht (1971), on the other hand, specify 6-9°C (43-48°F) for the hatching temperatures (as cited in Jennings and Hayes 1994). Metamorphosis occurs approximately 11 to 14 weeks after hatching (Licht 1974, Brown 1975b; both as cited in Jennings and Hayes 1994), but may take more than a year (Cochram and Goin 1970, as cited in USFWS 1980). Jennings and Hayes (1994) comment that there is a lack of information on variation in the length of larval development.
Male northern red-legged frogs reach sexual maturity in the breeding season following metamorphosis, but most probably do not reproduce until their second breeding season. Females apparently do not attain sexual maturity until the second breeding season after metamorphosis, and most probably do not breed until their third breeding season (Licht 1971).

California red-legged frogs generally take from 6-14 days to hatch and larvae metamorphose between July and September, 3.5 to 7 months after egg laying (Jennings et al. 1993, as cited in Davidson 1993). Males likely reach sexual maturity at 3 years of age and females after 4 years of age (Jennings and Hayes 1985, as cited in Davidson 1993). Most mortality of red-legged frogs is believed to occur in the larval stage (USFWS 1996a).

California red-legged frog tadpoles have been documented overwintering at 11 sites in 4 central coastal California counties (Fellers et al. 2001). Tadpoles in this study were located in spring-fed pools and ponds with cooler temperatures than typical for the red-legged frog. Fellers et al. (2001) suggested that overwintering is not likely to be common in larval red-legged frogs, but should be considered when monitoring for this species.

4.5.6.3 Movements and dispersal

Haggard (2000) found that the average distance that northern red-legged frogs moved from their breeding sites was 149 m. Recent movement studies indicate that the percentage of California red-legged frogs moving away from breeding habitat varies: 66% of females and 25% of males dispersed in one study (Fellers and Kleeman 2007) and less than half in another (Tartarian 2008). Bulger et al. (2003) found that 90% of non-dispersing California red-legged frogs stayed within 60 m of their aquatic sites, with a maximum distance of up to 130 m recorded after summer rain. Fellers and Kleeman (2007) found that the median distance California red-legged frogs dispersed was 150 m, generally moving to the nearest available non-breeding habitat. Dispersing frogs can move great distances; at least 1 record indicates a straight-line map distance of 2.8 km in a single season (Bulger et al. 2003).

Dispersal capabilities of this species are thought to be substantial. Northern red-legged frogs have been found at distances up to 900 m (2953 ft) away from the nearest source of water (Hayes et al. 2001). A radio-tagged female was recorded moving a distance of 2.4 km (1.49 mi) and 3 males from 1.1 to 1.3 km (0.68 mi to 0.81 mi) away from original capture points (Hayes et al.

2001). There is no data available on home range size of northern red-legged frogs, and the species is not known to be territorial.

California red-legged frogs have been found to move distances of up to 1.6 km (1 mi) up or down a stream drainage and may be found up to 1.6 mi (1 mi) from aquatic sites on rainy nights (USFWS 1997a). They will often move away from the water after the first winter rains, causing sites where they were easily observed in the summer to appear devoid of the species (USFWS 1997a).

4.5.7 Habitat requirements

4.5.7.1 General habitat types used

Northern red-legged frogs use a variety of habitat types, from aquatic sites for breeding to riparian and mesic upland forests during the post-breeding season and upland habitats for overwintering at low elevations (Gomez and Anthony 1996, Nussbaum et al. 1983, Licht 1969). In the southern Washington Cascade Range, Aubry and Hall (1991) found adult northern red-legged frogs to be more abundant in mature forest stands, but not significantly so. Gomez and Anthony (1996) found northern red-legged frogs in Oregon to be most abundant in deciduous forest types and reported no captures in mature coniferous forests.

Deep pools are important for several aspects of red-legged frog life history. Licht (1969) reported that northern red-legged frogs usually call underwater from a depth of at least 92 cm (3 ft). The behavioral response to predators by both subspecies of red-legged frogs includes fleeing directly into water and into the deepest portion of the channel or pool (Gregory 1979, as cited in Davidson 1993). Hayes and Jennings (1988) found California red-legged frogs in Central Valley drainages almost exclusively (99%) at sites with some water at least 70 cm (27.5 in.) deep.

4.5.7.2 Reproductive habitat

Red-legged frogs breed in coastal lagoons, permanent or temporary pools, marshes, ponds or backwater portions of permanent or intermittent streams, and artificial impoundments, such as stock ponds and irrigation ponds (Stebbins 1985, Jennings and Hayes 1994, USFWS 1997a). Ponds with emergent vegetation, undercut banks, semi-submerged rootwads, or dense cover from woody riparian vegetation may offer cover from predation and higher quality breeding habitat (USFWS 1997b). The absence of emergent and riparian vegetation, however, does not rule out the possibility that a site can be used for breeding (USFWS 1997a). When breeding occurs in temporary ponds or intermittent streams, water is usually available for a period of at least 4–6 months. Eggs are deposited on strong stems of emergent or aquatic vegetation (Nussbaum et al. 1983), particularly cattails and bulrushes, but also on the stems of sedges, willows, spiraea, and pondweed (USFWS 1980, USFWS 1997a). Eggs have also been found in ponds without vegetation (USFWS 1980).

Limited information exists on movement patterns and site fidelity of red-legged frogs. One study showed that males had a tendency to return to the same breeding site (Calef 1973) and that individuals may show fidelity to particular breeding sites from year to year. Pechman et al. (2001) observed frogs and toads returning to former breeding sites even after the ponds had been drained and filled with soil. Whether adult frogs return to the same terrestrial foraging area following breeding is unknown.

4.5.7.3 Larval habitat

Larval red-legged frogs use both mud and vegetation for cover (Calef 1973, as cited in USFWS 1980). Wiens (1970) characterized optimal habitat as including emergent willow, stems, grasses,

cattails, submerged weed stems, and filamentous algae. Older larvae may be less closely associated with vegetation (Calef 1973, as cited in USFWS 1980).

4.5.7.4 Post-metamorphic and foraging habitat

During the non-breeding season, red-legged frogs may be found far from water, especially in moist or humid habitats (Nussbaum et al. 1983). Northern red-legged frogs forage primarily on land (Licht 1986). California red-legged frogs are frequently encountered in upland settings, including open grasslands with seeps and springs (USFWS 1997a). Young California red-legged frogs may occur in low-velocity, shallow riffles in streams or along the margins of ponds (USFWS 1997a).

In western Oregon, northern red-legged frogs were captured more frequently in riparian than in upslope habitats (Gomez and Anthony 1996). Post-metamorphic individuals have been found to inhabit dense patches of grass and shrubs, such as willow thickets and sedges with a moist substrate (Stebbins 1951, Storm 1960, Twedt 1993; all as cited in Jennings and Hayes 1994). In northwestern California, Twedt (1993) often observed red-legged frogs in dense undergrowth of ferns and sedges along streamside flats in redwood forests. A dense understory may provide both cover from predation as well as create a more humid microclimate. Downed wood may be important as cover in upland areas (Dunlap 1955, Porter 1961; both as cited in USFWS 1980). Aubry and Hall (1991) reported that abundance of northern red-legged frogs in the southern Washington Cascades was significantly associated with downed wood. Under drier conditions, red-legged frogs may remain closer to riparian areas near streams or ponds (Nussbaum et al. 1983). Beavers may create habitat conditions that are particularly favorable for northern red-legged frogs (Jennings and Hayes 1994). Northern red-legged frogs have been observed frequently in association with beavers (Stebbins 1951, Brown 1975b; both as cited in Jennings and Hayes 1994).

During the non-breeding season, California red-legged frogs have been reported to use small mammal burrows and moist leaf litter up to 25.9 m (85 ft) from water in dense riparian vegetation for estivation (USFWS 1997a).

4.5.7.5 Water temperature

Northern red-legged frogs have the lowest embryonic critical thermal maximum of any native North American ranid frog (Licht 1971). Embryos in the early stages of development have a lower lethal temperature tolerance of 4°C (39°F) and an upper lethal temperature tolerance of 21°C (70°F), which are the lowest and highest known lethal limits among North American ranid frogs (Nussbaum et al. 1983). Adult California red-legged frogs exhibit stress when exposed to water temperatures at or above 29°C (Jennings and Hayes 1989, as cited in Davidson 1993). Chronic exposure to such temperatures may result in mortality (Davidson 1993).

4.5.8 Ecological interactions

4.5.8.1 Diet

Red-legged frog tadpoles are herbivorous grazers on algae.   The prey of post-metamorph red-legged frogs includes a variety of terrestrial and aquatic invertebrates, such as beetles, caterpillars, spiders, crustaceans, and mollusks (Blaustein et al. 1995). Other amphibians, small fish, and even small mammals may also be eaten (Hayes and Tennant 1985, as cited in Davidson 1993).

4.5.8.2 Predators

The rough-skinned newt and northwestern salamander are likely among the most important predators of larval northern red-legged frogs throughout their range (Calef 1973, Licht 1974; both as cited in Jennings and Hayes 1994). Certain species of garter snakes are likely the most important predators of metamorphic and recent post-metamorphic individuals (Fitch 1941, Licht 1974; both as cited in Jennings and Hayes 1994). Rainbow trout, giant diving bugs, damsel and dragonflies, and garter snakes have also been found to be major predators of red-legged frog tadpoles (Licht 1986, Calef 1973). Wading birds may be significant predators on adult red-legged frogs (Jennings and Hayes 1988).

Introduced bullfrogs may be major predators of red-legged frogs. In laboratory studies, the presence of bullfrog adults and tadpoles was found to significantly reduce mass at metamorphosis, increase time to metamorphosis, and decrease survival to metamorphosis of red-legged frog tadpoles (Kiesecker and Blaustein 1998). Adult bullfrogs also significantly decreased the survival of red-legged frog post-metamorphs, and the presence of both bullfrog tadpoles and smallmouth bass appeared to contribute to negative developmental effects on larval red-legged frogs (Kiesecker and Blaustein 1998). A study conducted in coastal Washington wetlands, however, revealed that neither bullfrog nor exotic fish presence was related to the relative abundance of northern red-legged frogs (Adams 1999).

Doubledee et al. (2003) found a positive correlation between the absence of California red-legged frogs and the presence of introduced bullfrogs. These researchers modeled bullfrog predation rates upon red-legged frogs after various management treatments (shooting, pond draining, etc). The model indicated that killing or removing bullfrogs required an extreme effort to be effective. A combination of shooting bullfrogs and draining ponds proved the most effective approach to control bullfrog populations and facilitate conservation of red-legged frogs (Doubledee et al. 2003).

4.5.9 Sensitivity to anthropogenic disturbances

Jennings and Hayes (1994) identified habitat alteration from timber harvest, urban development, livestock grazing, and predation from introduced fish species and bullfrogs as potential threats to red-legged frogs. Little is known about the sensitivity of red-legged frogs to forest management activities. Logging and herbicide spraying were not found by Cole et al. (1997) to significantly alter capture rates of red-legged frogs in Oregon. In this study, capture rates were higher in uncut red alder stands than in Douglas-fir stands (Cole et al. 1997).

4.5.9.1 Physical barriers to movement and habitat fragmentation

Red-legged frogs likely exist in metapopulations on the landscape and are dependent on the existence of numerous breeding sites across the landscape for long-term viability of the metapopulation (Welsh et al. 1998). Because of this, suitable breeding locations (i.e., still water habitats, such as ponds and lakes) should be managed as red-legged frog habitat even if reproduction by the species at a certain location is not observed in any given year (Welsh et al. 1998). Because bullfrogs may eliminate red-legged frogs (Hayes and Jennings 1986), breeding locations should be free of this introduced species where possible.

Suitable dispersal habitat is necessary for maintaining populations of red-legged frogs across the landscape. Because red-legged frogs are able to use a variety of habitat types, especially during wet seasons, and because they are relatively mobile as adults, many habitat types appear to be suitable as movement corridors (Bulger et al. 2003). Bulger et al. (2003) suggest that specific protection of dispersal habitat for California red-legged frogs in forested habitat is likely

unnecessary because high-quality dispersal habitat is nearly ubiquitous in these landscapes. Methods for ensuring that suitable dispersal habitat for this species is maintained across the landscape could include protecting riparian habitats, maintaining downed wood in upland habitats, uneven-age management of forest stands, and maintaining understory vegetation.

Ann Allaye Chan-McLeod (2003) examined potential barrier effects posed by clearcuts on Vancouver Island, British Columbia, Canada by radio tracking 120 red-legged frogs. She found that young clearcuts (<12 years old) were significant barriers to movement. However, these results were highly dependent upon the weather (rain) and to a lesser extent the body mass of the frog. Perhaps barrier effects from clearcuts may be diminished in regenerating stands as young as 11-years old.

4.5.9.2 Loss or degradation of breeding habitats
Timber harvesting and road building adjacent to red-legged frog breeding sites may reduce habitat quality through sedimentation, reduction in hiding and thermal cover, changes to predator-prey dynamics, and changes to microclimate. Semlitsch (1998, as cited in Welsh et al. 1998) estimated that buffers around Ambystomid salamander breeding ponds of 164 m (534 ft) would protect 95% of the population at any given site. Welsh et al. (1998) believe that this buffer size represents a good "first cut" in proposing buffers to protect a high percentage of red-legged frogs at any particular breeding site, and would likely maintain red-legged frogs over the long term.

4.5.9.3 Artificial breeding habitats
Man-made ponds are an important habitat for red-legged frogs during breeding. The majority of the red-legged frog breeding sites discovered in the plan area have been in man-made ponds. In most cases, the ponds are a water source for dust abatement or road compaction and grading; drafting from the ponds leaves fish-bearing waters undisturbed. Our HCP/NCCP proposes conservation measures for water drafting (C§10.2.2.3-4 and C§10.2.2.3-5) which minimize impact to embryonic or larval frogs that rear in man-made ponds and manage for quality habitat in ponds. The demonstrated success of these man-made ponds as breeding habitat prompts MRC to construct new ponds for red-legged frogs.

4.5.9.4 Changes in water quality
Amphibians may be particularly sensitive to developmental disruption in the egg and early larval stages (Berrill et al. 1994, 1997; both as cited in Welsh et al. 1998). Developmental abnormalities that can cause mortality may be triggered by some herbicides at very low concentrations. Herbicides used on forest lands in northern California (e.g., 2,4-D; 2,4,5-T; and atrazine) mimic the female hormone estrogen (Colborn et al. 1993). They have been linked to deformities and mortalities in birds, mammals, amphibians, reptiles, and fish (Hall and Henry 1992; Colborn et al. 1993; Berrill et al. 1994, 1997). Adverse effects may occur at miniscule concentrations (parts per trillion) of some of these chemicals (Colborn and Clement 1992, Colborn et al. 1993, USEPA 1997). Welsh et al. (1998) believe that, apart from direct habitat destruction, the single greatest threat to red-legged frogs on managed forestlands may be the use of forest herbicides and pesticides that can contaminate breeding sites. Bettaso et al. (2000) sampled blood from male and sub-adult northern red-legged frogs from 15 populations in northwest California in 1999 and 2000 to determine if the female protein vitellogenin could be detected in quantities sufficient to use as an indicator of contamination by estrogen-mimicking compounds. The results of blood serum analysis showed that 4 of 7 populations analyzed had male frogs producing vitellogenin, indicating that an exogenous source of estrogen was present in north coast California.

4.5.10 🔥 Impacts of MLC (2008)

There were 5 CalWater planning watersheds, known to have red-legged frogs present, which experienced some impacts from the 2008 fires. The fires impacted only foraging and dispersal habitats of red-legged frogs. The fires did not directly affect any documented breeding sites. There were 10 documented red-legged frog breeding sites in planning watersheds impacted by the fires. Of these 10 sites, only 2 sites were in close proximity to a fire—Marsh Mallo Pond and Upper Railroad Pond. The fire line was roughly 100 meters east of the documented breeding site at Marsh Mallo Pond. Monitoring the number of egg masses deposited in each documented breeding site provides some evidence of the fire impacts to the population.

4.5.11 Key uncertainties

MRC management has identified the following key uncertainties:
- What is the current and historical distribution of red-legged frogs in the plan area?
- What past management activities have affected red-legged frog populations in the plan area?
- Are ponds and still water better breeding habitats than rivers?
- What effect will cooler water temperatures and increased shade have on survival of the red-legged frog?

4.6 Coastal Tailed Frog

4.6.1 Systematics and taxonomy

The coastal tailed frog (*Ascaphus truei*) is one of only 2 living representatives of the genus *Ascaphus* (Jennings and Hayes 1994), the sole genus in the primitive frog family Ascaphidae. The second species is the Rocky Mountain coastal tailed frog, *Ascaphus montanus* (Neilson et al. 2001). Historically, the coastal tailed frog was included with a small group of similarly primitive New Zealand frogs in the genus Liopelma of the family Liopelmatidae. Recent phylogenetic analysis has identified a suite of unique features, and the weight of evidence led to placing coastal tailed frogs in their own family (Green and Cannatella 1994). Historically considered to belong to the most primitive group of living frogs (Nussbaum et al. 1983), this analysis also revealed

Photo by Brad Moon

that coastal tailed frogs exhibit a curious mix of primitive (e.g., possession of ribs) and derived characters (e.g., internal fertilization). Moreover, recent worldwide phylogenetic analysis of frog families indicates that coastal tailed frogs are the most likely sister group of all living anurans (Cannatella and Hillis 1993, Ford and Cannatella 1993). Interestingly, genetic material from coelacanths is very similar to that of coastal tailed frogs (Bogart et al. 1994). While only 2 species of coastal tailed frog are now recognized, biochemical and morphological variation across the range of the coastal tailed frog may conceal additional cryptic species (Stillwater Sciences 2001).
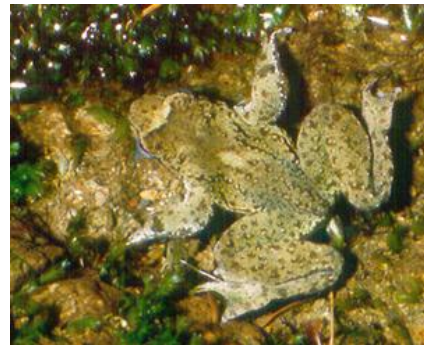
4.6.2 Geographic distribution

Metter and Pauken (1969) believed that coastal tailed frogs were more or less continuously distributed across the Pacific Northwest (coastal to inland) as recently as 25,000 years ago. As

the climate became drier at the end of the Pleistocene, inland system clusters were isolated from the coastal ones.

Coastal tailed frogs are known to occur from the central coast of British Columbia (Dupuis et al. 2000), through western Oregon and Washington, and south to northern California. Within California, coastal tailed frogs occur in the northwestern portion of the state from Del Norte County south to central Sonoma County and as far east as the southwest portion of Shasta County (Bury 1968, Stebbins 1985, Jennings and Hayes 1994). From the Rogue River system in Oregon south, this species has not been recorded outside of coastal forest areas, presumably due to its narrow habitat requirements (Bury 1968, Welsh et al. 1993). Coastal tailed frog distribution in California is patchy and restricted to areas having a very specific set of habitat conditions.

At present, little data exists to define an elevational cline with latitude (i.e., decreasing upper elevation limit with increasing latitude). Some investigators suggest that elevational limits may be more closely related to regional climate than latitude (Wahbe et al. 2001).

4.6.3 Local distribution

MRC biologists and foresters have collected information on amphibian presence in the plan area from (1) electrofishing and snorkeling surveys for fish (MRC 2002a); (2) diurnal amphibian surveys on Class II streams conducted in summer 2001 (MRC 2002d); (3) incidental observations during forestry and wildlife surveys (MRC 2002c, Incidental Sighting Database) and (4) baseline distribution surveys.

MRC has conducted baseline distribution surveys for coastal tailed frogs since 2003. As of 2010, we have surveyed 356 sites throughout the plan area and found over 75 occupied sites. We observed coastal tailed frogs in the following basins: Cottaneva Creek, Hardy Creek, Juan Creek, Howard Creek, Doyle Creek, Buckhorn Creek, Albion River, Navarro River, Greenwood Creek, Elk Creek, Mallo Pass Creek, Alder Creek, and Garcia River.

MRC has not detected coastal tailed frogs during surveys in the Hollow Tree, Noyo River, Big River, South Fork Albion River, Middle Albion River, South Branch Navarro, North Fork Navarro, Little North Fork Navarro, John Smith Creek, Ackerman Creek, South Fork Garcia and Rolling Brook planning watersheds.

Some information on coastal tailed frog distributions is available from surveys conducted in nearby areas by other timber companies. Information for Georgia-Pacific property (Jones & Stokes Associates, Inc. 1997) indicates that coastal tailed frogs were found to be present in the Ten Mile River and Usal Creek watersheds north of Fort Bragg. In the Jackson Demonstration State Forest, coastal tailed frogs have been documented from the South Fork Noyo River and Caspar Creek watersheds (CDFG 1996b).

Habitat for coastal tailed frogs in the HCP/NCCP plan area likely exists primarily in higher-gradient, perennial Class II streams and adjacent forest stands in both redwood and montane hardwood habitats. Coastal tailed frogs may also be found in some Class I (fish-bearing) streams, as the species has been found to co-occur with various fish species (Metter 1964a).

4.6.4 Population trends

Coastal tailed frog population trends have not been studied, although the species is thought to be declining by some and may now be at least regionally rare (Marshall et al. 1996). Neither

historical surveys nor population-level demographic analyses have been conducted on this species. Jennings and Hayes (1994) considered this species to be declining in California, especially in the upper Sacramento River system.

4.6.5 Life history

4.6.5.1 Reproduction

Tailed frogs, including the coastal tailed frog, reproduce with a mating strategy unique among frogs (Stebbins 1985). The male's tail-like sex organ, which can attain 10 mm (0.4 in.) in length when fully engorged, facilitates copulation and internal fertilization (Nussbaum et al. 1983, Stebbins 1985). Internal fertilization and subsequent laying of large, relatively scour-resistant eggs allows coastal tailed frogs to reproduce in high-gradient, high-velocity reaches. Males search underwater for females (Noble and Putnam 1931, Brown 1975a). Copulation, which also occurs underwater, may last 30 hours (Wernz 1969, Blaustein et al. 1995). The lung morphology and function of coastal tailed frogs facilitate the ability of males to stay underwater; the lungs are reduced in size and serve as hydrostatic organs (Noble and Putnam 1931).

The scientific literature describes few nests of coastal tailed frogs and the timing of egg laying is not well known (Karraker and Beyersdorf 1997). Coastal tailed frogs reportedly breed from late August to September, although pairs have been found in amplexus from March through October (Noble and Putnam 1931, Slater 1931, Wernz 1969, Brown 1975a, Nussbaum et al. 1983, Stebbins 1985). More recent reproductive studies suggest that June and July mark the peak of mating season in northwestern California (Sever et al. 2001). Metter (1964a) suspected that coastal populations (coastal tailed frogs) have more prolonged breeding seasons than inland populations (Rocky Mountain coastal tailed frogs). Female coastal tailed frogs store sperm in the oviducts for several months or possibly up to 2 years (Metter 1964b, Nussbaum et al. 1983, Hayes 1996). Data from a study conducted in northwest California indicated that a maximum of 1-year sperm storage appeared most likely (Sever et al. 2001). Female coastal tailed frogs likely retain sperm through the winter and lay eggs in the following spring or summer after spring runoff (Gaige 1920, Franz 1970a, Brown 1975b, Adams 1993, Karraker and Beyersdorf 1997). Some evidence suggests that females of inland populations (Rocky Mountain coastal tailed frogs) may lay eggs only every other year (Metter 1964b, Metter 1967, Daugherty and Sheldon 1982a). There are few studies on oviposition frequency in the coastal tailed frog; based on small clutch sizes, oviposition appears to be annual (Noble and Putnam 1931).

Larval Tailed Frog
Photo by Matt Goldsworthy

The eggs of coastal tailed frogs are among the largest of North American frogs (Wright and Wright 1949, as cited in Wahbe et al. 2001), averaging about 4 mm (0.16 in.) in diameter. The number of eggs laid per female ranges from 33 to 98 (Metter 1964b, Nussbaum et al. 1983, Daugherty and Sheldon 1982a). Coastal tailed frogs apparently have smaller clutch sizes than inland populations. Eggs are deposited in double strands attached to the underside of rocks (Gaige 1920, Metter 1964a, Franz 1970a, Brown 1975a, Adams 1993, Brown 1989, Capula 1989, Daugherty and Sheldon 1982a, Leonard et al. 1993, Noble and Putnam 1931, Nussbaum et al. 1983; all as cited in Wahbe et al. 2001). At one California site, eggs were attached to the underside of a boulder in a pool of a second-order stream (Karraker and Beyersdorf 1997). In August 2005, an MRC biologist discovered a nest site in a seep within Mendocino County (Goldsworthy 2007). The nest contained 25 eggs which further supports the

hypothesis that coastal populations produce smaller clutch sizes than inland populations. This was also the first observation of coastal tailed frogs using seep habitats for reproductive purposes.

4.6.5.2 Growth and development

Tailed frog embryos develop more slowly than any other North American species of frog (Daugherty and Sheldon 1982b, Blaustein et al. 1995, Hayes 1996). Eggs may take up to 9 months to hatch once they have been deposited, a time interval that may relate to water temperature (Brown 1977, Brown 1989, Hayes 1996). Hatching often occurs in August and September (Franz 1970a, Nussbaum et al. 1983, Adams 1993), after which larvae tend to remain in hatching areas until their yolk supplies are depleted the following spring (Metter 1964a). During this period, the mouthparts of tadpoles are not developed enough to allow them to leave their protected nest site. They would be carried downstream, unable to attach to the substrate and brace themselves against the swift-moving current (Brown 1975a). By the time yolk stores are fully absorbed, the tadpoles develop an adhesive oral sucker, by which they are able to attach to the substrate (Gradwell 1973, Nussbaum et al. 1983, Altig and Brodie 1972). These specialized mouthparts are a unique adaptation among Pacific Northwest frogs that allow feeding in high-velocity streams. Tadpoles of coastal tailed frogs do not often completely detach from the substrate and swim freely (Altig and Brodie 1972). Tadpoles of this species have been observed climbing on rocks 10–20 cm (4–8 in.) out of the water, possibly a means to obtain a richer source of food (Noble and Putnam 1931).

Tadpoles of coastal tailed frogs metamorphose into juvenile frogs 1 to 5 years after hatching, with development time generally increasing with elevation, latitude, and distance inland (Brown 1990, Bury and Corn 1991, Gray 1992, Wahbe 1996, Wallace and Diller 1998, Bury and Adams 1999). Recent work has demonstrated that coastal tailed frogs can frequently metamorphose in 1 year in the southern portion of its geographic range (i.e., southern Oregon and northern California), but may require 4 or 5 years at higher elevations and latitudes (Wallace and Diller 1998, Bury and Adams 1999). Shifts in the number of years required to reach metamorphosis can occur in the same stream as a function of inter-annual variations in climate (Bury and Adams 1999). Rates of development may relate to length of the growing season; aspect, gradient, elevation, snowpack, and number of frost-free days may also influence development (Dupuis 1999). Streams that Wallace and Diller (1998) reported as containing coastal tailed frogs that metamorphosed after 2 years at the southern end of their geographic range were at higher elevations and had higher flows. They hypothesized that the 2-year larval period at higher elevations was an adaptation to lower water temperatures. The lower elevation sites are more likely to have intermittent flows in late summer and early fall, thereby selecting for a 1-year larval period. Sites where there was a switch from a 1-year to a 2-year larval stage may have been the result of instream disturbance (Wallace and Diller 1998). Variables that were not statistically correlated with the length of the larval cycle included mean late summer water temperature, mean maximum water temperature, stream aspect, and cover type. Recently metamorphosed frogs are approximately 25 mm (1 in.) long (Nussbaum et al. 1983).

4.6.5.3 Movements and dispersal

Tadpoles in captivity preferentially take cover under rocks in turbulent water rather than in calm water and tend to seek cover in interstitial spaces of the stream substrate during the day (Altig and Brodie 1972).

Adult coastal tailed frogs generally emerge from the water at dusk and feed during the night on small terrestrial arthropods found along streams and in adjacent forest stands (Capula 1989, Leonard et al. 1993). In wet weather, coastal tailed frogs have been found to forage up to 90 m

(295 ft) or more from water (Metter 1967, Noble and Putnam 1931) During drier periods they tend to forage nearer to the stream; for the most part, Rocky Mountain coastal tailed frogs reportedly stray less than 1 m from streams (Metter 1964a, 1967). Daylight hours are usually spent resting under rocks in streams or on moist stream banks (Daugherty and Sheldon 1982a, Stebbins 1985). Studies in Washington and Oregon revealed that adult coastal tailed frogs were most often found under large rocks in turbulent areas of the stream (Bury et al 1991a). Lower humidity and higher ambient air temperatures over its geographic range likely influences the nocturnal behavior of the Rocky Mountain coastal tailed frog (Daugherty and Sheldon 1982b, Metter 1964a).

Female coastal tailed frogs make local movements during the mating season. Researchers have observed movement downstream from tributaries to larger mainstem streams during the mating season of Rocky Mountain coastal tailed frogs in the Wallowa Mountains of northeastern Oregon (Landreth and Ferguson 1967, as cited by Daugherty and Sheldon 1982a). There are also observations of upstream movement of females to egg deposition sites in June and July in western Washington (Brown 1975a, as cited by Daugherty and Sheldon 1982a).

Coastal tailed frogs are reportedly poor re-colonizers of streams from which they have disappeared, because they exhibit fidelity to natal streams (Bury et al. 1991a, Daugherty and Sheldon 1982a). However, some research suggests that substantial overland movements may occur during favorable (wet) conditions, so the potential for re-colonizing disturbed sites needs to be reconsidered (Aubry 2000; see also Hawkins et al. 1997, Crisafulli and Hawkins 1998). Moist, cool conditions, as occur in the Coast Range and Cascade Mountains, may facilitate movement to upland areas and migration between drainages (Daugherty and Sheldon 1982a). A recorded observation indicates a maximum dispersal distance of 360 m per year for a single juvenile female (Daugherty and Sheldon 1982b, as cited in Wahbe et al. 2001).

Recovery of streams following some channel disturbances can occur within 2 to 3 years by the undisplaced (terrestrial-bound) adults (Dupuis and Friele 1996). Following the 1980 eruption of Mt. St. Helens, important components of both plant and animal communities recovered to pre-disturbance levels within 3 to 5 years (Hawkins et al. 1997, Crisafulli and Hawkins 1998). Within 5 years of the disturbance, abundance of coastal tailed frog tadpoles in some streams within the blast zone represented more than 90% of the herbivore biomass and were at densities higher than had ever previously been reported for the area (Crisafulli and Hawkins 1998). Colonists likely came from epicenters of survival within the blast zone rather than from more distant, unaffected populations (Crisafulli and Hawkins 1998). Snow-pack over many high-elevations streams may have acted as a buffer from the effects of the eruption, allowing some life stages of coastal tailed frogs located in water or under snow to survive. While there is no distance data to confirm it, these individuals may have colonized streams at lower elevations, where aquatic biota appeared to have been completely extirpated (Crisafulli and Hawkins 1998).

Like many amphibians, coastal tailed frogs either reduce their activity or become completely inactive in areas with cold winters (i.e., inland, high altitudes, or high latitudes). During the cold period, Rocky Mountain coastal tailed frogs seek cover under large downed logs and boulders (Daugherty and Sheldon 1982a). In milder coastal areas, coastal tailed frogs may remain active and continue feeding all year (Daugherty and Sheldon 1982b). Gradwell's (1973) study implied that coastal tailed frog larvae select overwintering sites among the more protected interstices of a streambed.

4.6.5.4 Life span and age at sexual maturity

Little is known regarding coastal tailed frog survival rates and longevity (Wahbe et al. 2001). Daugherty and Sheldon (1982b) speculated that in a Montana population, Rocky Mountain coastal tailed frog females may live 15 to 20 years. Longevity estimates for coastal tailed frogs are unavailable, but Brown (1990) speculated that they might live up to 40 to 50 years. Bury and Adams (1999) pointed out that some coastal tailed frog larvae from high-elevation populations may require 5 years to metamorphose. Brown (1990) extrapolated that an additional 10 years may be needed for them to reach sexual maturity. This is longer than the maturation period in a Montana population of the Rocky Mountain coastal tailed frog, where males and females were estimated to begin reproducing at ages 7 and 8, respectively (Daugherty and Sheldon 1982b).

4.6.5.5 Population densities

Coastal tailed frogs are patchily distributed, but are often locally abundant in appropriate habitat. Bury (1988) found average densities of $4.5/m^2$ ($0.41/ft^2$) in the southern Washington Cascade Range, including one 10-m (33-ft) reach with 109 individuals, 95% of which were larvae. Corn and Bury (1989) found coastal tailed frog densities of $0.76/m^2$ ($0.07/ft^2$) in streams in unmanaged, old-growth forests and $0.37/m^2$ ($0.03/ft^2$) in harvested areas of the Coast Range in western Oregon. Diller and Wallace (1999) reported that larval density varied from 0.04 to $0.73/m^2$ (0.004 to $0.07/ft^2$) and averaged $0.24/m^2$ ($0.02/ft^2$) in managed landscapes at the southern end of coastal tailed frog range, which is less than the densities (0.58 to $4.40/m^2$ or 0.05 to $0.41/ft^2$) observed by Corn and Bury (1989) and Hawkins et al. (1988). Dupuis and Friele (1996) found coastal tailed frog densities averaged between $0.40/m^2$ ($0.04/ft^2$) and $2.40/m^2$ ($0.22/ft^2$) in managed landscapes in British Columbia, but averaged $1.80/m^2$ ($0.17/ft^2$) in unmanaged landscapes in British Columbia. Similarly, coastal tailed frog densities averaged $2.37/m^2$ ($0.22/ft^2$) in the unmanaged landscape of the Central Cascades, but densities averaged $1.01/m^2$ ($0.09/ft^2$) in the managed landscape (Kelsey 1995). Adams and Bury (2000) found relatively low densities ($0.37/m^2$ or $0.03/ft^2$) in the unmanaged forest landscape of the Olympic Peninsula. Unfortunately, due to differences in sampling methods, results from these studies cannot be compared directly.

4.6.6 Habitat requirements

4.6.6.1 General vegetational, climactic, and geological conditions

Streams with coastal tailed frogs typically flow through areas forested by Douglas-fir, coast redwood, Sitka spruce, western hemlock, and to a lesser extent ponderosa pine (Hayes 1996). Metter (1964a) found that the Rocky Mountain coastal tailed frog typically inhabited streams with close to 100% canopy cover; coastal tailed frogs are substantially more abundant in closed-canopy forests, including regenerated forests, than in clearcuts (Bury and Corn 1988, Bury et al. 1991b, Bury et al. 1991a). In some instances, adults may also use dense, moist, forests upslope from streams (Bury et al. 1991b).

Coastal tailed frogs are generally found in areas of high annual precipitation, but Metter (1967) listed several localities from the Pacific Northwest that receive less than 75 cm (30 in.) of annual precipitation. Most California coastal tailed frogs are known from areas with over 100 cm (40 in.) of annual precipitation, indicating that heavy precipitation may be important to the distribution of the frog in the southern part of its range by sustaining perennial flow in streams.

Diller and Wallace (1999) looked at relationships of various physical features on distribution of coastal tailed frogs in the redwood zone of northwestern California. Related variables included (1) landscape level—geologic formation; (2) reach level—percent fines, stream gradient, water temperature; and (3) microhabitat. In this study, coastal tailed frog larvae occurred in 75% of 72 streams surveyed. Out of 21 habitat variables measured, only 3 adequately predicted the presence

of coastal tailed frog larvae. Negative associations were found for both percent fine substrate particles and water temperature, and a positive association was found for stream gradient. Occurrence of larvae was positively associated with cobble, boulder, and gravel substrates with low embeddedness, and negatively associated with fine substrates. Larvae were found more often than expected in high gradient riffles and less frequently than expected in pools and runs (Diller and Wallace 1999). Aspect (northerly versus southerly), as measured on a landscape scale, was not significantly related to presence of coastal tailed frogs in this coastal study. Although the association of coastal tailed frog distribution with stream temperature was minimal, it was presumably because temperature variation was small due to the cool coastal climate. No significant correlation between water temperature and canopy closure or aspect was noted. Diller and Wallace concluded that coastal tailed frog distribution is not tied to old growth forest *per se*, but to specific microhabitats that are more likely to occur in undisturbed areas.

In Douglas-fir/hardwood forests of northwestern California and southwestern Oregon, Welsh and Lind (1991) found that mean relative abundances of coastal tailed frogs differed significantly between forest age-classes (p=0.001) and were 0.111, 0.555, and 0.810 for young (<100 yrs), mature (100-200 yrs), and old (>200 yrs) forests, respectively. Welsh (1990) and Welsh et al. (1993) hypothesized that forest structure rather than stand age determined coastal tailed frog occurrence. Important attributes of stand structure for coastal tailed frogs include (a) low ambient temperatures resulting from high canopy closure; (b) downed woody debris, particularly in and around streams, which contribute to greater habitat diversity, trap fine sediments, prevent cementation of aquatic substrates, and provide hiding and thermal cover for adults; and (c) a deep duff layer that filters clearer, cooler water and maintains cool, moist streamside microclimates. Welsh et al. (1993) reported a positive correlation between coastal tailed frog distribution and abundance and several attributes of late-seral stage forests, including high canopy cover, downed woody material, ground cover consisting of ferns and herbs (grass showed a negative correlation), litter depth, stream width, and flow rate. Coastal tailed frogs also exhibit a preference for extremely low-ambient light levels, a finding that suggests selection for darker microsites (Hailman 1982, Welsh et al. 1993).

Hawkins et al. (1988) compared larval densities of coastal tailed frogs in 3 types of sites in the Mt. St. Helens area shortly after the volcanic eruption: (1) both sample sites and headwaters were deforested due to the eruption; (2) sample sites were deforested but headwaters remained forested; and (3) both sample sites and headwaters remained forested (watershed received only heavy ash fall). They found that distribution of coastal tailed frog tadpoles correlated with high water velocity, low embeddedness of interstitial spaces, and large substrate (Hawkins et al. 1988). In this study, deforested sites with forested headwaters contained the highest larval densities (even greater than in forested sites with forested headwaters), while deforested sites with deforested headwaters had the lowest densities. The abundance of coastal tailed frogs at sites where only the headwaters were forested may have been due to the availability of suitable adult habitat upstream in the headwater forests, and the availability of abundant food for tadpoles on site (due to increases in primary productivity from reduced canopy cover). However, the authors caution that the adults produced from larvae in these sites may not fare well. The low densities of coastal tailed frogs in deforested watersheds may be due to high water temperatures and harsh terrestrial environments limiting adult survival and reproduction.

Gomez and Anthony (1996) found that coastal tailed frogs were more often recorded in riparian than upslope transects in their Lincoln County (OR) study sites. The coastal tailed frog was most abundant in large sawtimber and old-growth stands, compared with earlier successional stands and deciduous stands. In contrast, McComb et al. (1993) reported that within mature forests,

similar numbers of coastal tailed frogs were captured in stream and upslope transects (as cited in Gomez and Anthony 1996).

Diller and Wallace (1999) noted a correlation between coastal tailed frogs and younger forests, but believed that this resulted from a relationship with consolidated geological formations, which were subject to timber harvesting later than forest types on unconsolidated substrate.

At 3 survey locations in central Oregon, coastal tailed frogs selected streams within old-growth forests at the warmest location, mature stands at the location with intermediate temperatures, and young forest at the coolest location (Gilbert and Allwine 1991c). This finding further suggests that coastal tailed frogs may occur within a wider range of timber age classes than previously believed, particularly in the cool temperatures of the coastal redwood forest belt. However, Corn and Bury (1989) found significantly higher densities and biomass of coastal tailed frogs in old-growth versus logged forest stands, despite similar temperature regimes and a higher canopy closure in logged stands. The only measured difference between logged and unlogged stands other than canopy closure was size of the aquatic substrate resulting from increased sedimentation in logged sites. Nevertheless, logging activities appeared to be less detrimental in their coastal study plots than in their interior plots, presumably because of the maritime climate and more favorable temperature regime.

4.6.6.2 Stream habitat
Coastal tailed frogs are uniquely adapted to life in cold, fast-flowing, high-gradient, perennial mountain streams (Nussbaum et al. 1983). Coastal tailed frogs are not typically associated with seeps, although their post-metamorphic life stage may sometimes be found in seeps (Adams and Bury 2000) and, unlike the southern seep salamander, will often use trout-bearing streams to some extent (Nussbaum et al. 1983). Coastal tailed frogs prefer streams in mature and old-growth stands with dense, shaded habitat, low sediment loads, cool temperatures, high ambient humidity levels, and high water quality over much of their range (Bury and Corn 1988, Corn and Bury 1989, Welsh 1990, Hayes 1996). Perennial streams in which the substrate changes infrequently are likely to provide higher-quality habitat for coastal tailed frogs (Wahbe et al. 2001).

M.G. Hunter studied habitat use by coastal tailed frog larvae and post-metamorphs in the west-central Cascades of Oregon. Larvae were not detected in the smallest streams and only rarely in the largest streams, but they were common in moderately sized streams. Larvae were not detected in basins less than 37 ha (92 ac). Most larvae were found downstream of where surface water was intermittent. Habitat with high likelihood of occurrence of coastal tailed frog larvae consisted of "a moderate-sized stream, with a basin size of a few hundred ha (approximately 500 ac), with only very small amounts of particles as small as or smaller than pebbles, but rather a predominance of boulder and cobble, with wetted channel dimension of about 13 cm (5 in.) average depth (and an average maximum depth of about 25 to 30 cm (10 to 12 in.) and 3.5 to 4.0 m (11.5 to 13.2 ft) average width, moderate mid-level vegetation cover within a harvested stand, but only about 100 to 150 m (330 to 50 ft) away from unharvested forest" (Hunter 1998, p. 53).

Hunter found post-metamorphic coastal tailed frogs in all basin sizes, but at greater frequencies at higher elevations, above about 850 m (2788 ft). The elevation of the study area ranged from about 450 m (1476 ft) to 1300 m (4264 ft). The likelihood that they would occur in south-facing streams was 33% of that in north-facing streams. Hunter concluded that habitat likely to contain coastal tailed frogs consists of "a high elevation stream in a region of relatively steep topography, with at least a moderate percent of the glide channel unit, and a small

proportion of cascades, a moderate presence of boulders in the wetted channel, within or very near to unharvested forest" (Hunter 1998, p. 56). Because larvae must spend one or more winters in a stream, coastal tailed frogs cannot occupy ephemeral streams (Brown 1990). However, they can persist in streams in which some reaches are de-watered.

4.6.6.3 Stream substrate particle size

Optimal habitat for coastal tailed frogs has instream substrate with numerous interstitial spaces. In a study of Oregon and Washington streams, adult coastal tailed frogs were most often found under large rocks and boulders in riffle areas (Bury et al. 1991a). During winter, Rocky Mountain coastal tailed frog adults appear to use larger substrate materials for shelter than during summer (Metter 1964a).

Laboratory experiments by Altig and Brodie (1972) showed that coastal tailed frog larvae preferred substrates with smooth rocks greater than 55 mm (2.2 in.) in diameter and turbulent water and that they avoided high light intensities. They found that tadpoles preferred larger substrate particles (85 to 125 mm or 3.3 to 5 in.) slightly over gravel (55 to 96 mm or 2.2 to 3.8 in.), and strongly avoided sand and pebble substrates. Altig and Brodie (1972) indicated that, in general, tadpoles may require smooth-surfaced stones with a minimum diameter of about 55 mm (2.2 in.) because of the larvae's oral disk; substrates that are too small may not allow the larvae to become attached. Additionally, the substrate must be large enough to provide interstitial spaces for daytime retreats. Bury et al. (1991a) found that tadpoles appeared to prefer deeper water and smaller rocks for cover than adults and to occupy gravel, pebble, and cobble beds. Larvae also avoid mossy rocks and silt deposits (Nussbaum et al. 1983).

Hawkins et al. (1988) found tadpoles on large substrates (>30 cm or >12 in.) at night, but smaller substrates during the day. This may be because larger rocks are preferred for grazing, but because they were heavily embedded, interstitial spaces for cover were a limiting factor. During the day, larvae retreated to smaller, less embedded substrates. Corn and Bury (1989) noted that the mean size of rocks used by coastal tailed frogs for cover was 433 cm^2 (67 in^2), representing cobble-sized rock.

In one southwestern Washington study conducted in second-growth forests, coastal tailed frog larvae were found only in basalt sediment streams at elevations above 300 m (984 ft) with low levels of fine sediments, and not at all in marine sediment habitats (Wilkins and Peterson 2000). Even though appropriate-sized substrate might be available in marine sediment streams, the rocks and cobbles present were possibly too soft and porous for successful larval attachment (Wilkins and Peterson 2000).

Stream channels characterized by coarse cobble and boulder substrates and bedrock that breaks down into coarse particles (e.g., intrusive or highly metamorphic rock) maintains higher densities of larval coastal tailed frogs by providing interstitial habitat during natural and anthropogenic disturbances (Wahbe et al. 2001). Coarse substrates also provide stable sites for depositing eggs and cover from predation.

4.6.6.4 Stream gradient

Coastal tailed frog populations are found in streams across a wide range of gradients from 2% to 60% (Wahbe et al. 2001). They appear, however, to be less common in very steep (e.g., > 50% gradient) channels where bedload is likely to be mobilized more frequently (Dupuis et al. 2000). Coastal tailed frogs may be more abundant in higher gradient channels because these streams have a greater capacity for flushing fine sediments that may otherwise fill interstitial spaces

important to coastal tailed frogs. Dupuis et al. (2000) found coastal populations to be generally absent from lower-gradient (<10%) streams in British Columbia coastal lowlands. Rocky Mountain coastal tailed frog populations in British Columbia occur in streams with an average gradient of 4%, likely because the steeper headwater streams in this region were ephemeral or were subject to high bedload transport (Dupuis and Wilson 1999).

In the north coastal California redwood region, Wallace and Diller (1998) found coastal tailed frogs in reaches averaging 9.1% gradient. However, Welsh et al. (1998) found no correlation between stream gradient and the presence of coastal tailed frogs in the pristine streams of Prairie Creek Redwoods State Park in northern California. Likewise, Adams and Bury (2000) did not find a statistically significant association between gradient at the sample site and coastal tailed frog density in the Olympic peninsula.

4.6.6.5 LWD

Welsh et al. (1993) noted positive associations between the presence and abundance of coastal tailed frog larvae and LWD in northern California, both in the stream channel and on the surrounding forest floor. The authors believed that this relationship could be due to (1) increased flushing of sediment in the general area created by increased turbulence from the LWD; (2) entrapment of sediment by LWD; and (3) presence of increased densities of adult coastal tailed frogs, which use the downed wood for terrestrial cover. No association was found between the presence of coastal tailed frogs and LWD at sites in the southern Washington Cascade Range (Aubry and Hall 1991).

4.6.6.6 Water temperature

The coastal tailed frog is restricted to a narrow range of habitat types, in part because it has the lowest known temperature needs and one of the narrowest ranges of temperature tolerances of any frog species. Coastal tailed frogs typically live in waters between 5° and 16°C (41° and 61°F) (Marshall et al. 1996). Cool water temperature, especially during the warm egg-laying season, is critical to reproduction of coastal tailed frogs. Embryos of coastal tailed frog have the narrowest temperature tolerance range (5° to 18°C or 41° to 64°F) and the lowest lethal temperature limit among North American frogs (Brown 1975a). Streams with water temperatures above 15°C (59°F) for extended periods are not suitable for reproduction (Hayes 1996).

Laboratory tests indicate that young (first year) larvae of coastal tailed frogs prefer water temperatures less than 10°C (50°F), especially 5 to 8°C (41 to 46°F). Older larvae prefer waters above 10°C (50°F), especially 12° to 16°C (54° to 61°F), although they do not avoid colder waters and generally have less precise temperature preferences than first-year tadpoles (de Vlaming and Bury 1970). Tadpoles strongly avoid waters above 22°C (72°F) (de Vlaming and Bury 1970), and post-metamorphs also avoid high temperatures (Welsh 1990, Hayes 1996). Claussen (1973) suggested that mortality of post-metamorphs may occur at 23° to 24°C (73° to 75°F), and de Vlaming and Bury (1970) indicated a lethal maximum of approximately 30°C (86°F) for tadpoles.

Welsh (1990) found stream temperature to be an excellent predictor of coastal tailed frog abundance, accounting for 37.3% (P = 0.00002) of the variation observed, with higher numbers of coastal tailed frogs occurring in streams with lower temperatures. The highest stream temperature Welsh observed with the coastal tailed frog was 14.3^{o}C.

4.6.7 Ecological interactions

4.6.7.1 Diet

Larvae of coastal tailed frogs are primarily dependent upon diatoms, which they scrape off of rocks (Franz 1970b, Altig and Brodie 1972, Nussbaum et al. 1983), but small amounts of filamentous algae, tiny insects, and sand are incidentally ingested during feeding. The stomachs of Rocky Mountain tailed tadpoles collected from interior sites contain large amounts of conifer pollen (Nussbaum et al. 1983). Larvae of coastal tailed frogs are classic *aufwuchs* gleaners, although they take advantage of other temporarily available food sources, such as pulses of conifer pollen (Jennings and Hayes 1994). Although some researchers have speculated that larvae of coastal tailed frogs may filter feed, Altig and Brodie (1972) determined in laboratory studies that tadpoles feed entirely on material scraped from rocks (see also Gradwell 1973).

Adult coastal tailed frogs feed on a variety of aquatic and terrestrial larval and adult insects, arthropods, and snails (Metter 1964a). Bury (1970) compared diet data of metamorphosed coastal tailed frogs with that of the southern seep salamander and concluded that the 2 species depend on similar food resources. At their study site in the fall and winter, the food supply seemed abundant and no competition was observed between them.

Rocky Mountain coastal tailed frog adults primarily forage along stream banks, but will also feed underwater, where they are less efficient at capturing prey (Metter 1964a). Whereas Metter (1964a) noted that Rocky Mountain coastal tailed frog adults consumed a variety of aquatic and terrestrial larval and adult insects, arthropods, and snails, Bury (1970) reported that stomach samples of coastal tailed frogs contained primarily terrestrial arthropods.

4.6.7.2 Predators

Predators of coastal tailed frogs include garter snakes, fish (those documented include sculpins, brook and cutthroat trout), giant salamanders, American dippers, and dobsonfly larvae (Metter 1963, Daugherty and Sheldon 1982a, Feminella and Hawkins 1994, Jones and Raphael 1998). Tadpoles and juveniles are likely more vulnerable than adults to predation (Daugherty and Sheldon 1982b). Pacific giant salamander larvae eat coastal tailed frog tadpoles, and trout take Rocky Mountain coastal tailed frog tadpoles and adults (Metter 1964a). Hunter (1998) found that metamorphosed coastal tailed frogs did not coexist with crayfish at his Oregon Cascades study site, perhaps due to predation by crayfish.

Feminella and Hawkins studied how tailed frog tadpoles adjusted their feeding behavior in response to 4 of their known predators:

> In absence of predators, tadpoles usually emerged from under cobbles to feed at night (2000 to 0100 hours), and spent the remainder of the 24-hour interval hidden in crevices. In the presence of giant salamanders, cutthroat trout, and brook trout that were confined within separate *in situ* enclosures immediately upstream of tadpoles, tadpole activity was reduced two-, three-, and six-fold, respectively. In contrast, tadpoles appeared unable to detect upstream sculpins. (Feminella and Hawkins 1994, p. 310)

4.6.8 Sensitivity to anthropogenic disturbances

Narrow habitat requirements, limited mobility, an extensive time period required for development, and an apparent preference for conditions associated with old-growth forests is thought to make populations of coastal tailed frogs extremely vulnerable to extirpation (Welsh 1990, Bury and Corn 1988). These factors may restrict gene flow (Daugherty and Sheldon 1982a) and retard recovery after habitat alteration (Marshall et al. 1996). Diller and Wallace (1999)

believed that because coastal tailed frogs can occur lower in a watershed than seep salamanders, they were more likely to experience the indirect, cumulative effects of habitat disturbance.

4.6.8.1 Physical barriers to movement and habitat fragmentation

Given the dependence of coastal tailed frogs on stream corridors or continuous moist habitat for terrestrial movement, any of the following may prevent or limit its movements: (a) habitat fragmentation due to timber harvesting; (b) road construction; (c) drying of stream reaches due to flow diversions; and (d) physical barriers, such as dams, waterways, culverts, and private residences.

4.6.8.2 Changes in hydrologic regimes

Changes in timing, frequency, and duration of high and low flows may affect survival of coastal tailed frogs. Flushing flows are important because they remove sediment and maintain interstitial spaces between rocks and gravels, thereby providing both diurnal cover for larvae and adults, and egg-laying substrate. However, exceptionally high spring runoff may cause larval mortality (Metter 1968, as cited in Daugherty and Sheldon 1982b). Adults may be able to escape high flows by leaving streams (Daugherty and Sheldon 1982b). Mortality can occur when eggs and larvae are stranded as flows diminish. Clearcut logging and roads may alter hydrologic regimes within watersheds and accentuate peak flows and low summer flows (Jones and Grant 1996, as cited in Wahbe et al. 2001).

4.6.8.3 Changes in sediment dynamics

Increased fine sediment inputs to streams associated with timber harvesting and roads can reduce distribution and abundance of coastal tailed frogs (Corn and Bury 1989, Marshall et al. 1996). The filling of substrate interstices is likely detrimental for coastal tailed frogs because it reduces cover from predation and high flow events (Corn and Bury 1989, Hayes 1996). Direct tests (e.g., stream substrate and population measurements taken before and after harvest) have not been done, however, to confirm this hypothesis. If true, the extent of these impacts is likely dependent on the sensitivity of the channel habitat to disturbance (Dupuis and Steventon 1999). More specifically, areas with weak rocks or thick surficial material deposits may produce more landslides and sedimentation than areas with thin surficial cover and more competent bedrock. Welsh and Ollivier (1998) found reduced abundance of coastal tailed frog larvae in streams impacted by an erosion event even in high velocity habitats that were likely to have lower sediment loads, suggesting that something other than the filling of interstitial spaces was affecting their abundance in these streams. Welsh and Ollivier (1998) suggested that perhaps a thin layer of sediment may be sufficient to block light and inhibit the growth of the diatoms on which coastal tailed frog larvae feed. In addition, greater amounts of sediment in a stream may scour algae from substrates (Alabaster and Lloyd 1982, as cited by Welsh and Ollivier 1998) and reduce periphyton biomass available to coastal tailed frogs even when not contributing to embeddedness (Welsh and Ollivier 1998).

4.6.8.4 Changes in LWD

Logging in riparian areas and removal of in-channel LWD has substantially reduced this habitat element in many northern California streams. Reduced LWD results in reduced structural complexity in the channel and riparian zone and reduced storage of fine and coarse organic material that supports food webs upon which the species depends.

4.6.8.5 Changes related to timber harvesting

Timber harvesting can affect coastal tailed frogs through increased water and ambient temperatures, removal of the overstory canopy, increased fine sediment inputs resulting from

logging activities, and reduced recruitment of LWD to stream channels and riparian foraging areas (Nussbaum et al. 1983, Corn and Bury 1989). Numerous researchers have documented the negative effects of timber harvesting on population densities and relative abundance of coastal tailed frogs (see Wahbe et al. 2001).

Of all Pacific Northwest frogs, coastal tailed frogs may be the species most vulnerable to degradation or loss of old-growth forest habitat (Blaustein et al. 1995). A number of studies have documented negative effects of timber harvesting on coastal tailed frogs. In western Oregon, Corn and Bury (1989) found that coastal tailed frogs were present in only 35% of stream reaches in logged areas, compared with 96% of reaches in unlogged areas; mean densities were twice as high in uncut reaches as in logged areas. Welsh (1990) documented relationships between the frequency of coastal tailed frog detections and forest age class; significant differences were found between young and mature, young and old, and mature and old age classes. Welsh did not find a significant relationship between coastal tailed frog densities and stand age when all study sites were pooled; however, he noted that his study sites in southwestern Oregon tended to occur at higher elevations in forests dominated by true fir, compared with sites in California, which tended to be at lower elevations and dominated by Douglas-fir. When higher elevation sites (>1000 m or 3280 ft) were excluded from the analysis, the relationship between coastal tailed frog density and forest age was significant.

Metter (1964a) thought that timber harvesting led to the disappearance of a Rocky Mountain coastal tailed frog population in Oregon's Blue Mountains. Coastal tailed frogs may be less sensitive to deforestation in coastal streams where maritime influences can maintain cool temperatures during the summer (Bury 1968, Corn and Bury 1989). In the drier climate of northeastern Oregon, however, Bull and Carter (1996) found no significant difference in the numbers of larval or post-metamorphic Rocky Mountain coastal tailed frogs in streams with low, medium, or heavy timber harvest, although coastal tailed frog numbers decreased with increasing harvest intensity. Based on a very general conceptual model, Bury and Corn (1988) predicted that timber harvesting reduces abundance of coastal tailed frogs in the Cascades of Oregon and Washington, and may extirpate the more vulnerable populations of the Coast Range of Oregon and northern California.

P.S. Corn and R.B. Bury compared Pacific giant salamander, southern torrent salamander, coastal tailed frog, and Dunn's salamander populations in harvested (14- to 40-year-old) and uncut (60- to 400+-year-old) stands in central coastal Oregon. "Physical comparisons between types of streams were similar," they wrote "except that streams in logged stands had generally smaller substrate, resulting from increased sedimentation" (Corn and Bury 1989, p. 39). The likelihood of occurrence, density, and biomass of coastal tailed frogs were significantly greater in uncut than in previously logged forest. Although the densities of Pacific giant salamanders and southern torrent salamanders were higher in steeper stream reaches within previously logged stands, the density of coastal tailed frogs was unrelated to stream gradient in both uncut and previously logged sites. At previously logged sites, coastal tailed frogs were more likely to be found where uncut timber was present upstream. Timber harvesting upstream from uncut forests had no effect on the presence, density or biomass of any of the study species.

The coastal tailed frog's sensitivity to timber harvesting may reflect its dependence on cold waters for breeding and larval development and on forested, late-seral riparian habitats for shade and fine sediment retention (Bury and Corn 1988, Corn and Bury 1989, Bury et al. 1991b). Timber harvesting of riparian canopy cover may increase stream temperatures and fine sediment inputs, and may lower streamside humidity. Increased insolation following timber harvesting

appears to cause shifts in periphyton away from diatoms, which are the preferred food of larval coastal tailed frogs, towards filamentous green algae, which is less palatable (see Kupferberg 1996) and may inhibit the ability of tadpoles to attach to rocks (Bury and Corn 1988, Beschta et al. 1987). Reductions in LWD in stream channels following timber harvesting may harm coastal tailed frog populations by reducing cover availability and increasing water temperatures. Diller and Wallace (1999) believed that the larval stage of coastal tailed frogs, which is restricted to streams, is the most sensitive to the impacts of timber harvest.

In areas with more stable streams (streams that provide suitable microhabitats during channel disturbances), timber harvesting may have beneficial short-term effects due to temporary increases in light penetration and subsequent stream productivity (Richardson and Neill 1998, Kim 1999). This is particularly true in areas where metamorphosis occurs more rapidly (one to 2 years) and where summer temperatures do not become lethal, or where periphyton abundance is nutrient limited, as suggested by Kiffney and Richardson (2001). Hawkins et al. (1988) noted that some environmental parameters of their study sites that were deforested from the Mt. St. Helens eruptions were comparable to clear-cut logged sites. These parameters include higher temperatures, high primary production by algae, and low input of terrestrial litter. The authors found high densities of coastal tailed frog larvae in the deforested sites where the headwaters of the study streams remained forested. However, Bury and Corn stated that "The bloom of productivity following clear-cutting is short-lived, and once shade is reestablished over a stream, invertebrate, salmonid, and salamander populations decline (Murphy et al. 1981, Hawkins et al. 1982, 1983)." The latter likely also applies to the coastal tailed frog. In the long-term (25–80 years), coastal tailed frog populations can persist in managed forests, but their numbers may be reduced (Corn and Bury 1989, Aubry and Hall 1991, Corn and Bury 1991a, Richardson and Neill 1998, Aubry 2000, Welsh and Lind 2002).

4.6.9 Impacts of MLC (2008)

There were 10 CalWater planning watersheds, known to have coastal tailed frogs present, which experienced some impacts from the 2008 fires. Of the 79 sites known to be occupied by coastal tailed frogs, 38 sites (roughly 48% of all occupied sites) experienced some impacts from the fires. Of the 38 sites affected, 20 sites had over 50% of their drainage acres burnt over by the fires. There were 8 sites with coastal tailed frogs present which had 100% of their drainage acres burnt over by fires. Monitoring the presence or absence of coastal tailed frogs throughout all of the burnt sites may provide more evidence of the fire impacts to the population.

4.6.10 Key uncertainties

MRC management has identified the following key uncertainties:
- What is the current and historical abundance and distribution of coastal tailed frogs in the plan area?
- What past management activities have affected populations of coastal tailed frogs in the plan area?