Section 3.5 Contents

3.5 Species of Greatest Concern

3.5.1 Introduction to Species of Greatest Concern

Species of greatest concern are those species for which current population status indicates that immediate measures need to be taken in order to halt or reverse serious regional population declines, all species currently listed as threatened or endangered under the ESA and potentially present in the watershed, species considered to be at greatest risk of listing in the near future, and at-risk species with the most uncertainty regarding effects of City operations. The list not only includes all species currently listed under the ESA as threatened or endangered but also some state listed species, and unlisted species identified as being at high risk in the region by taxonomic experts. Some species that are listed as threatened or endangered or that are candidates for listing by the state or federal government occur commonly or abundantly in the watershed, while others are not known to occur at all in the watershed, although some have been reported in areas not far from the watershed.

Additional background information regarding the status of the species of greatest concern is presented below in Sections 3.5.2 - 3.5.15. The status within the municipal watershed and the state and federal status for each species of greatest concern is given in Table 3.5-1 below.

Table 3.5-1. Status of fish and wildlife species of greatest concern that are known to occur or that could potentially occur in the Cedar River Watershed. Species are listed in the order in which they are presented in the text.

Species	Status in the Cedar River Watershed		Designated Status	
	status	notes	State	Federal
Northern Spotted Owl *Strix occidentalis caurina*	present, breeding	one breeding pair in upper watershed in CHU, plus singles	Endangered	Threatened
Marbled Murrelet *Brachyramphus marmoratus*	unknown	aural detection in 1992	Threatened	Threatened
Northern Goshawk *Accipiter gentilis*	present, breeding	one pair in upper watershed in CHU in 1992	Candidate	Species of Concern

Species	Status in the Cedar River Watershed		Designated Status	
	status	notes	State	Federal
Common Loon *Gavia immer*	present, breeding	three nesting pairs; some use of artificial nest platforms	Candidate	
Bull Trout *Salvelinus confluentus*	present, breeding	present in reservoir, major and some minor tributaries		Threatened
Pygmy Whitefish *Prosopium coulteri*	present, breeding	present in reservoir and major tributaries	Sensitive	
Sockeye Salmon *Oncorhynchus nerka*	not currently present above Landsburg Dam	present in Cedar River below Landsburg Dam		
Coho Salmon *Oncorhynchus kisutch*	present, breeding	present in Walsh Lake system; present in Cedar River below Landsburg Dam		Candidate
Chinook Salmon *Oncorhynchus tshawytscha*	not currently present above Landsburg Dam	present in Cedar River below Landsburg Dam	Candidate	Threatened
Steelhead Trout *Oncorhynchus mykiss*	not currently present above Landsburg Dam	present in Cedar River below Landsburg Dam, previously released above Landsburg		
Bald Eagle *Haliaeetus leucocephalus*	present	present regularly as transients and migrants	Threatened	Threatened
Peregrine Falcon *Falco peregrinus*	unknown	closest verified nest 4.5 miles away	Endangered	
Grizzly Bear *Ursus arctos*	not known to be present	watershed at southern periphery of current range in Washington State	Endangered	Threatened
Gray Wolf *Canis lupus*	not known to be present	watershed at southern periphery of current range in Washington State	Endangered	Endangered

3.5.2 Northern Spotted Owl

INTRODUCTION

The spotted owl (*Strix occidentalis*) (Figure 3.5-1) is a forest-dwelling owl that occurs in three sub-species within its range in North America. The sub-species found in the Pacific Northwest and present in the municipal watershed is the northern spotted owl (*S. o. caurina*). Extensive reviews of the general ecology and literature pertaining to this sub-species are included in the Draft Recovery Plan for the Northern Spotted Owl (USDI 1992b) and Habitat Conservation Plans developed by the WDNR (WDNR 1997) and Plum Creek Timber Company, L.P. (Plum Creek 1996). Because both of these plans include lands contiguous with the Cedar River Municipal Watershed, the basic information contained in those plans also applies to this HCP. Much of the following discussion of the northern spotted owl in the Western Cascade Physiographic Province represents a summarization of information contained in those two HCP documents, supplemented with information specific to the Cedar River Watershed.

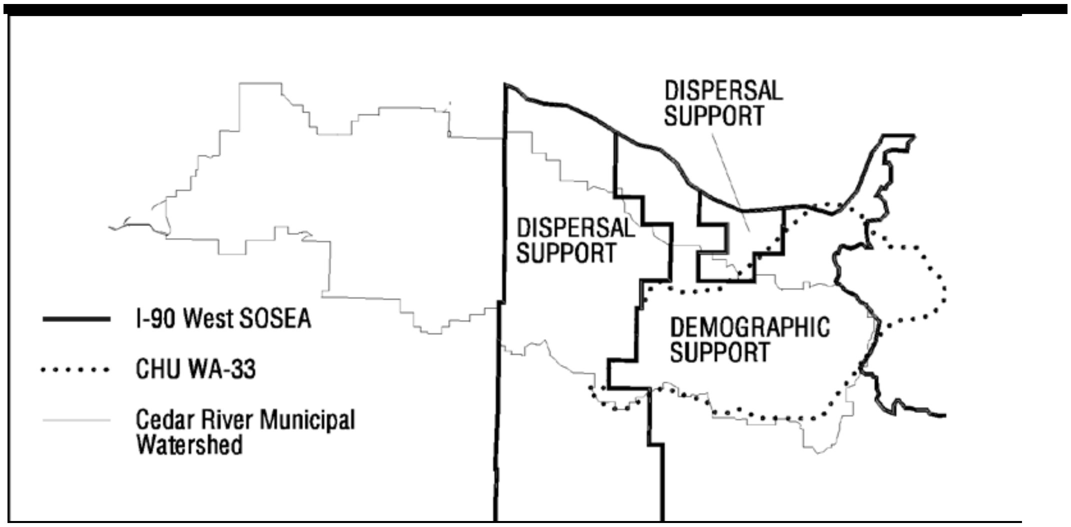
Figure 3.5-1. The northern spotted owl.

RANGE AND STATUS

Northern spotted owls presently occur from southern British Columbia, Canada, south to Marin County in central California. Although numbers vary substantially, they are most strongly associated with old-growth and late-successional coniferous forest habitats throughout their range. As a result of the progressive loss of preferred habitat throughout its range, the spotted owl was federally listed as a threatened species in 1990 (Fed. Reg. Vol. 55, Pp. 26114-26194). It is listed as an endangered species by Washington State (WAC 232-12-014).

In order to protect remaining critical late-successional and old-growth forest habitat for the northern spotted owl and to reduce fragmentation, the USFWS designated several Critical Habitat Units (CHUs) in 1991 (Fed. Reg. Vol. 57, Pp. 1796-1838). One of these, Northern Spotted Owl CHU WA-33, overlaps 22,845 acres of the municipal watershed (Figure 3.5-2). The Washington State Forest Practices Board developed a series of Spotted Owl Special Emphasis Areas (SOSEA) to complement the federal CHUs. One of these, the I-90 West SOSEA, incorporates 48,877 acres of the municipal watershed, and overlaps all of the CHU (Figure 3.5-2). SOSEA land in the municipal watershed is designated either for *demographic support* or *dispersal support*. Demographic support lands offer appropriate habitat for roosting, nesting, and foraging. Dispersal support lands offer minimum necessary habitat for young to cross from the natal stand to a new territory and for non-breeding adults to move across the landscape. There are 25,501 acres of demographic support lands in the municipal watershed, 22,167 acres of which overlap the CHU. An additional 23,367 acres of the municipal watershed within the SOSEA are designated dispersal support, 668 acres of which are in the CHU.

Figure 3.5-2. Relationship of Northern Spotted Owl Critical Habitat Unit WA-33 and I-90 West SOSEA to the Cedar River Municipal Watershed.



A northern spotted owl site center is defined as the location of status 1, 2, or 3 northern spotted owls, based upon the following definitions: (1) pair or reproductive status, (2) two birds, pair status unknown, and (3) resident territorial single status (WAC 222-16-010). Determination of the existence, location, and status of spotted owl site centers are documented in compliance with guidelines established by and available from WDFW. For management purposes, a median home range circle (called an owl circle) with a radius of 1.8 miles is established around each spotted owl site center (WAC 222-16-010). The owl circle incorporates the approximate median annual home range of northern spotted owls in the western Cascade Mountains of Washington. Within the municipal watershed there are two northern spotted owl reproductive site centers (one of which has not been active since 1981), two single resident centers (one of which has not been active since 1987), and one single owl with an unknown status (WDFW 1997d). In addition, there are two reproductive owl circles that partially overlap the municipal watershed (WDFW 1997d).

LIFE HISTORY

Spotted owls live an average of 8 years, and reach reproductive maturity during their third winter (Thomas et al. 1990). Pair bonds are usually formed at the end of winter and are typically of long-term duration. Commitment by a pair to nesting in any particular year depends on whether prey is available and in sufficient abundance, the male's hunting effectiveness, and the physiological condition of the female (sub-adults often breed). Forty to 60 percent of spotted owl pairs initiate nesting in a given year (Fed. Reg., Vol. 55) and nesting success can vary from 1 to 100 percent within a population (USDI 1992b). In March or April, successful pairs typically lay one or two eggs, and rarely three eggs. Owlets hatch in approximately 30 days, remain as nestlings for 3-5 weeks, and depend on their parents for food until they fledge and disperse in September or October. The adults then separate for the winter. Mean survival rates of juveniles, summarized from 11 study sites in California, Oregon, and Washington, have been estimated at 0.258 (range 0 to 0.418), and for adults at 0.844 (range 0.821 to 0.868) (Burnham et al. 1994).

Spotted owls typically hunt by perching on a relatively low branch, locating prey by sight or sound, and then pouncing on and capturing the prey with their talons. The diet of spotted owls is composed of predominately two or three species of small mammals (Solis 1983; Forsman et al. 1984), although it can include small birds and insects. The actual forage species that are preferred are consistent within a particular region, but vary across the sub-species range. Based on biomass and frequency of capture, significant prey species are flying squirrels (*Glaucomys sabrinus*), wood rats (*Neotoma fuscipies* and *N. cinerea*), deer mice (*Peromyscus* spp.), red tree voles (*Arborimus longicadus*), and rabbits (*Sylvilagus* spp.).

HABITAT REQUIREMENTS

Habitat Area

The median annual home range for individual spotted owls in the Western Washington Cascades Physiographic Province has been variously reported as 6,657 acres (USDI 1992b) and 8,205 acres (Hanson et al. 1993), with the smallest home range size reported being 2,969 acres (Hanson et al. 1993). Despite low sample sizes, it can be generalized that individual spotted owl home ranges are relatively large and generally increase as the amount of old-growth forest decreases (Carey 1985; Forsman et al. 1984; Thrailkill and Meslow 1990). It may also be generalized that there is a substantial degree of overlap of home range between members of a mated pair (Forsman et al. 1984; Solis and Gutierrez 1990) and less overlap among adjacent pairs (Forsman et al. 1984). Additionally, there is considerable variation in home range size across the geographic range of the species.

Habitat Structure

Old-growth forests occupied by spotted owls are typically characterized by moderate to high numbers of old trees with structural damage and decay (Brown 1985a). Such trees are important as nest sites for the owls. The distribution of old-growth forest especially, but mature coniferous stands as well, strongly correlates with the known range of the northern spotted owl in the Western Cascades. Suitable habitat for spotted owls consists mainly of older forest stands with large live trees having an average canopy cover greater than 70 percent and containing relatively high densities of logs and snags (Thomas et al. 1990; Buchanan 1991; Hanson et al. 1993; North 1993). Conifers dominate stands used by spotted owls.

Correlation studies indicate that spotted owls prefer old-growth forest for both nesting and roosting, with early-successional, pole, and young stands being consistently avoided. Multilayered, old-growth forests are the preferred nesting habitat of spotted owls in Oregon and Washington (Brown 1985a). Most nests found on public land have been in mature and old-growth forest (Forsman et al. 1984; LaHaye 1988). Nest site locations are typically associated with structures particularly characteristic of old-growth trees (broken tree-top cavities, lateral tree cavities, abandoned raptor stick nests, and debris platforms including mistletoe clumps). Most studies have shown that reproductive success is higher for pairs that have more old growth in their home ranges. Also, adult persistence, defined as adult occupation of the same stand over several years, and spotted owl density both correlate positively with increasing amounts of old-growth forest in contiguous stands.

Radiotelemetry studies in western Oregon have shown that spotted owls forage primarily in old-growth and mature forests, and avoid clearcuts and young second growth (Forsman 1980, 1981, both as cited in Brown 1985a). However, in relative contrast to preferred nesting and roosting habitat, foraging habitat may consist of a slightly wider range of structural conditions. Young stands and pole stands were consistently avoided during foraging in studies reviewed by Thomas et al. (1990), and forests less than 80 years old were avoided by 55 percent of spotted owls and selected by 3 percent. In contrast, Blakesley et al. (1992) found no tendency for owls to avoid stands in the 11-21 inch diameter breast height (dbh) size class, but these young stands did develop after natural disturbances and had diverse composition and complex structure. Stands used most often by foraging owls had relatively closed canopies and more complex understory structure (USDI 1992b).

Dispersal Habitat

Dispersal habitat is defined for the spotted owl because of the extent to which preferred, old-growth forest habitat has been depleted and fragmented throughout most of its original range. After leaving the relative security of their natal territory, dispersing juveniles must be able to forage and at the same time avoid the greater predation risk in typically less optimal habitat (e.g., clearcuts) if they are going to be successful in establishing a territory as an adult. It is also likely that dispersal habitat would be used by many adult owls moving among habitat patches suitable for foraging, roosting, or nesting.

When spotted owl juveniles cross open and fragmented landscapes, they are more vulnerable to their primary predator, the great horned owl (*Bubo virginianus*) (Miller 1989), which occurs more frequently in such landscapes than does the spotted owl (Anthony and Cummins 1989; Hamer et al 1989; Johnson 1993). Ideally, the distance between suitable habitat sites for spotted owls should not exceed the distance that most successfully dispersed juveniles are known to have traveled (Thomas et al. 1990). In Oregon and California these distances averaged 4 miles for males and 12 miles for females (USDI 1992b).

POPULATION STATUS

The federal Northern Spotted Owl Recovery Team identified 10 threats to existing populations of the spotted owl (USDI 1992a). Although the threats varied in severity between physiographic provinces, the single most significant factor contributing to the overall decline of the species was consistently identified as the loss of nesting, roosting, and foraging habitat to clear-cutting and other even-aged harvest methods (Thomas et al. 1990). The most severe future threat to the northern spotted owl was attributed to habitat loss (USDI 1992a).

Other conditions posing moderate threats to the northern spotted owl in the Western Washington Cascades Province, which includes the Cedar River Watershed, were:

(1) Limited habitat in which decreased productivity levels and occupancy occur; the province has between 20 and 60 percent suitable habitat coverage (Bart and Forsman 1992);

(2) Population decline (demographic rates exhibiting a downward trend);

(3) Small populations, which are at greater risk from environmental and demographic variations and loss of genetic diversity;

(4) Sparse populations with lack of habitat distribution (surrounded by over 12 miles of poor habitat), which are subject to the same risks as small populations; and

(5) Province isolation, which prevents immigration and leads to loss of genetic diversity.

Other factors were considered to be of lesser overall significance. Vulnerability to such natural disturbances as fire, insect or disease infestation, and windthrow was considered a low threat in the Western Washington Cascades (USDI 1992b). Predation as a threat was not ranked because information was lacking.

HABITAT AVAILABILITY IN THE CEDAR RIVER WATERSHED

The existing landscape within the Cedar River Municipal Watershed presents a wide array of coniferous forest habitat types over an elevation range of approximately 4,500 ft, extending from the lower-elevation foothills to the crest of the Cascades (Section 3.2.2). Over that span, a substantial portion of the habitat is potentially suitable for varied types and degrees of use by spotted owls. Presumably, the most significant of these habitat types, both on a local and regional scale, is the 13,889 acres of unharvested native conifer forest (old growth) older than 190 years that exists today the watershed, of which 13,155 acres is in the upper watershed. These old-growth forest stands, although varying widely in structural development and habitat quality, can potentially provide at least some high-quality reproductive, roosting, foraging, and dispersal habitat.

Additionally, of the 34,710 acres of second-growth forest within the upper watershed that are in varying stages of development from recent clearcuts (regeneration) to mature and late-successional forest, the oldest and most structurally developed stands are potentially suitable for use at least to some degree as foraging or dispersal habitat by owls. Finally, there is a substantial amount (approximately 26,902 acres) of second-growth forest in the lower watershed that is over 50 years old. This second-growth forest has already developed or will progressively develop vertical and horizontal structure considered to be characteristic of mature, and in some cases late-successional, ecological stages of forest development (sections 3.2.2 and 4.2.2, and Map 5).

Complicating the issue of spotted owl habitat requirements and availability is the fact that there is a substantial degree of variation in habitat structural development, and therefore in habitat quality, within stands essentially equal in chronological age in the watershed. Unharvested native forests are not always of equal habitat quality. Therefore, only a small portion of available habitat as it now exists within the watershed may be adequate to support individuals and be effectively utilized by the species. It is likely that only a small amount of the total habitat area available, even old-growth forests, receives use by spotted owls. No comprehensive studies have been conducted within the watershed to accurately document and compare the probable range of habitat preference and differential use patterns of the spotted owls, although there have been surveys of general habitat use throughout the forest of the upper and lower watershed (Egtvedt and Manuwal 1988).

STATUS IN THE CEDAR RIVER WATERSHED

Calling surveys were conducted throughout both upper and lower sections of the municipal watershed during 1986-87 by University of Washington graduate students and City personnel in order to detect the presence of spotted owls and determine their distribution within the watershed. Detection of other owl species and documentation of ecological interactions among the species present were also objectives of the project. Over the course of the study, six species of forest-dwelling owls, including northern spotted, barred (*Strix varia*), great horned, western screech (*Otus asio*), northern pygmy (*Glaucidium gnoma*), and saw-whet (*Cryptoglaux acadica*), were detected (Egtvedt and Manuwal 1988).

During the survey period, call responses from spotted owls were documented in three distinct areas of the watershed. All three of these areas lie within the upper watershed above 2,500 ft elevation. Two of the areas of detection also fall within the CHU, which was designated to protect spotted owl habitat on federal lands (Fed. Reg. Vol. 57, pp. 1796-1838). Both male and female owls responded within each of the two areas in the CHU, but only one mated, reproductive pair was documented. Responses in the other area were evaluated as coming from a single, transient, or unknown-status male individual. Significantly, no detections of spotted owls were documented in the lower watershed during this study (Egtvedt and Manuwal 1988) and none have been documented at these lower elevations to date.

No further surveys or monitoring studies were executed by City staff until 1990, when calling surveys were re-initiated in selected areas of the CHU. Results of these efforts included the confirmation of a transient male in one section of the CHU. In addition, the adult male that had previously been radio-tagged was relocated in 1990 (see section entitled “Habitat Selection within the Cedar River Watershed”), still within the original territory area. This male had not been observed or located since 1988. Subsequently, City staff worked with USFS personnel to locate, capture, and band both adults and one juvenile. The non-functional transmitter was removed from the radio-tagged male and not replaced. The juvenile that was caught represents one of only two spotted owl offspring known to have been found, up to that time, on the west side of the Washington Cascades between I-90 and Mt. Rainier.

The reproductive pair has been monitored periodically by a private timber company with land ownership immediately adjacent to the municipal watershed boundary that is within the potential territory of the pair. This reproductive site center has presumably been occupied each year to date since 1987, when it was originally discovered. Several specific nest trees appear to have been used. The female originally banded in 1990 was still present when they were last located, but a new male - replacing the male that had been radio-tagged - appeared and is still present. Offspring have been produced periodically throughout the term of monitoring. Only one other reproductive site center, located in a separate section of the CHU and not observed to have been active since 1981, has been verified within the municipal watershed boundaries to date.

Very few other spotted owl activity centers have been identified within the Cedar River Watershed boundaries. Only two single resident site centers and one single, status-unknown activity center have been confirmed. Both of these are located in the upper municipal watershed, and only one is in the CHU.

Several spotted owl activity centers also occur on lands in various ownerships that are adjacent to the municipal watershed. Several reproductive site centers have been documented on lands within 4 miles of the municipal watershed boundary, including four to the north, and two to the east. All of these reproductive site centers are within the designated boundaries of the CHU, which includes lands both inside and beyond the municipal watershed boundaries. The relative proximity of these reproductive site centers to the municipal watershed may either directly or indirectly influence both the number and distribution of calling survey detections within the watershed. Detections may be influenced directly if the municipal watershed is within a pair's territory. Detections may be influenced indirectly if juvenile dispersal patterns or territorial shifts include land within the watershed.

Habitat Selection within the Cedar River Watershed

A major objective of the 1988 spotted owl study (Egtvedt and Manuwal 1988) was to gather basic information regarding spotted owl habitat use and general behavior in the municipal watershed. As a means to collect data for this purpose, the adult male from the mated pair was fitted with a radio transmitter and subsequently located by the use of standard radiotelemetry techniques for a period of 10 months from July 1987 through April 1988, when the transmitter signal terminated. During that period the adult male was located 93 times. The total home range of the radio-tagged male was 8,868 acres, including 3,873 acres (43.7 percent) of old-growth forest within the CHU. A disproportionate 73 percent of radio locations were within these old-growth stands. The remaining locations were relatively evenly distributed among four immature and one other mature coniferous forest habitat type. It was assumed, based on subsequent detections, that the bird was in transition between old-growth stands at the time many of these observations were taken.

Significantly, three out of the four spotted owl activity centers documented within the municipal watershed to date are located in unharvested native forest stands greater than 189 years old, three out of the four are within the CHU, and both reproductive site centers (one not confirmed active) are in stands older than 250 years. Additionally, all verified site centers are located within the upper watershed; none have been found in the predominantly second-growth conifer forests of the lower watershed or below 2,500 ft elevation.

POTENTIAL HABITAT LIMITATIONS AND ECOLOGICAL CONSIDERATIONS

The total amount of suitable reproductive habitat (old-growth and late-successional forest) available within the upper watershed and concentrated within the CHU may be of either inadequate area or insufficient quality to support numerous reproductive spotted owl pairs. The relative amounts, quality, and spatial distribution of habitat types required to successfully support additional reproductive spotted owl pairs may be inadequate.

The relatively large amount and distribution of young successional forest types in the upper watershed tends to favor competitive species (barred owls, great horned owls) which may increase competition for nest sites and prey, and also increase the potential for predation, especially on dispersing juvenile spotted owls. Both barred and great horned owls are relatively numerous throughout the watershed.

The dramatic overall difference in major types of spotted owl habitat between the upper and lower watersheds creates a situation where more potential reproductive habitat is available in upper elevations. However, in the lower watershed to the west the lower elevations may actually provide better quality dispersal and foraging habitat than the upper watershed.

Spotted owls, especially dispersing juveniles, moving westward out of the watershed will encounter an ever-increasing amount of intensive commercial forestry activity and rural residential development. Because juveniles are known to disperse more or less in random directions (Guitierrez and Carey 1985), and because the lower watershed is surrounded on three sides by areas experiencing increasing residential development, recruiting breeding habitat in the lower watershed may result in poor reproductive performance of any pair nesting in that area.

3.5.3 Marbled Murrelet

INTRODUCTION

The marbled murrelet (*Brachyramphus marmoratus*) (Figure 3.5-3) is a Pacific seabird that occurs in two subspecies, the North American race (*B. m. marmoratus*) and the Asian race (*B. m. perdix*), which is commonly known as the long-billed murrelet. Only the North American race occurs in Washington State. Because of the species' unique behavior of selecting inland forest nesting sites (see below), the marbled murrelet was the only North American avian species whose nest remained undiscovered as recently as 1974 (Binford et al. 1975). Even today, relatively little information is known about the bird's distribution within the two distinctively different ecosystems that it inhabits (marine and inland coniferous forest) compared with Washington's other seabirds that primarily utilize marine environments (WDW 1993d).

Extensive literature reviews pertaining to the general ecology of the marbled murrelet have been included in recent Habitat Conservation Plans prepared by WDNR (WDNR 1997) and Plum Creek Timber Company, L.P. (Plum Creek 1996). Both of these plans include lands contiguous with the Cedar River Municipal Watershed. The following discussion of marbled murrelets in Washington State was generally derived from these two HCP documents and supplemented with specific information about potential marbled murrelet habitat in the Cedar River Watershed.

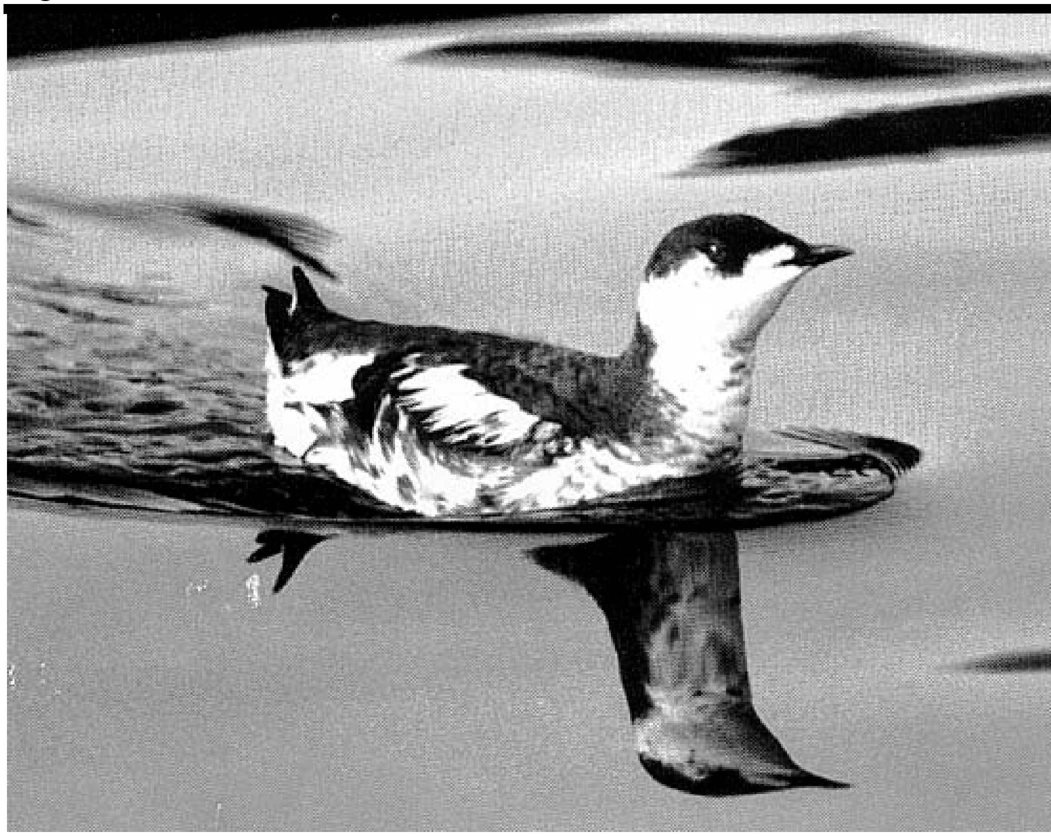
RANGE AND STATUS

The marbled murrelet occurs along the Pacific coastline from the Bering Sea, Alaska, southward to central California. Unlike the relatively continuous occurrence of the species exhibited across the northern extent of its range in Alaska, there are distinct gaps separating the breeding populations throughout Washington, Oregon, and California. The gaps between populations appear to correlate with loss of nesting habitat in late-successional and old-growth forests (Carter and Erickson 1992; Leschner and Cummins 1992; Nelson et al. 1992; Ralph et al. 1995). Observations made along the three states' coastlines document that the largest concentrations of marbled murrelets at sea during the breeding season were adjacent to areas with available inland nesting habitat (old-growth forest) (Sowls et al. 1980, Nelson et al. 1992). This fact supports the theory identifying habitat loss as the major causative agent for gaps in breeding range.

The marbled murrelet was federally listed as a threatened species in Washington, Oregon, and California September 28, 1992 (Fed. Reg. Vol. 57, No. 191), primarily because of loss of old-growth nesting habitat, and secondarily because of mortality caused by gill nets (USDI 1992a). The species is also listed as threatened by Washington State (WAC 232-12-011).

Washington Department of Natural Resources classifies a stand as an occupied marbled murrelets site (WAC 222-16-010; see Appendix 24 for WAC definition) if an active nest or recent nest site has been discovered, a chick or eggshell fragments have been found on the forest floor, or murrelets have been observed exhibiting subcanopy behaviors (WAC 222-16-010). Occupied marbled murrelet sites are restricted from harvest and road building. Harvesting is also restricted within an average 300-ft managed buffer zone adjacent to an occupied marbled murrelet site (WAC 222-16-080) (Appendix 24). Areas associated with a visual or audible detection of a marbled murrelet, that are not identified as occupied sites, are classed as marbled murrelet detection areas (WAC 222-16-010). Restrictions within marbled murrelet detection areas include timber harvesting, other than removal of down trees outside of the critical nesting season (April 1 to August 31), or road construction within suitable marbled murrelet habitat. The same restrictions apply within suitable marbled murrelet habitat containing seven platforms per acre outside a marbled murrelet detection area (WAC 222-16-080).

Figure 3.5-3. The marbled murrelet.



Distribution in Washington State

Most marbled murrelets found along the Washington coastline are in the northern regions of Puget Sound and the outer coast (Speich et al. 1992). Currently, insufficient information exists on abundance and regional distribution to definitively determine population trends in the western Cascade region. However, the estimate of 5,500 marbled murrelets in Washington (Speich and Wahl 1995; Varoujean and Williams 1995) probably reflects a region-wide decline in population. This regional decline is thought to be caused by a combination of many factors, including a reduction in old-growth forests, oil spills in marine waters, and entanglement in gill nets (Marshall 1988; Leschner and Cummins 1992).

LIFE HISTORY

Marbled murrelets are relatively unique among seabirds because they utilize both marine and coastal environs as well as inland forest habitats within their inclusive range. Most of their activity during fall, winter, and early spring is feeding in the open sea or in coastal habitats. In Washington, they forage mainly in Puget Sound or in near-shore ocean waters. Their predominant prey items in winter and spring are marine invertebrates, with small fish important during the breeding season (Burkett 1995). In addition, they have been observed on large lakes, but foraging behavior has not been documented in fresh water (Marshall 1989).

As stated above, one parameter that defines an occupied marbled murrelet site involves the observation of marbled murrelets exhibiting subcanopy behaviors (WAC 222-16-010). Flying into, through, and out of the tree canopy, and landing in trees are examples of subcanopy behaviors associated with nesting (Nelson and Hamer 1995a). However, marbled murrelet nests are difficult to detect and fairly inaccessible. Because of this, demographic information on the species is limited.

During late spring and summer, reproductive individuals fly substantial distances inland to establish nests and produce offspring. The maximum distance that a marbled murrelet has been found inland was in Oregon at a distance of 66 miles (52.25 miles in Washington), but the majority were located inland within 40 miles of marine waters (Hamer 1995; Miller and Ralph 1995; Ralph et al. 1995). The mated pairs typically nest in old-growth trees, specifically on large branches and usually more than 100 ft above ground. They do not actively build traditional nests, but instead seek out and use naturally formed platforms usually composed of large, wide limbs with thick moss or duff, mistletoe brooms, or other structural deformities that provide a surface of sufficient size to rear a chick. Adult marbled murrelets approach and leave the nest at high speed primarily at dusk and dawn, but also during the night, which makes nest detection difficult for observers (WDW 1993d).

The duration of the breeding period in Washington State has been estimated to be 124 days, from April 26 to August 27, based on 13 breeding records (Hamer and Nelson 1995a). An individual pair produces and incubates a single egg per nesting attempt. Hatching occurs within 27-28 days (Sealy 1974, 1975; Simons 1980; Hirsch et al. 1981; Carter 1984). Each member of a mated pair alternates incubation shifts, one brooding while the other forages, for up to 24 hours per shift. Incubation exchanges typically occur near dawn (Nelson and Hamer 1995a). Chicks fledge at 30-40 days and presumably fly to the ocean during their first flight (Sealy 1975; Quinlan and Hughes 1990; Hamer and Cummins 1991).

REPRODUCTIVE HABITAT REQUIREMENTS

The first marbled murrelet nest was discovered in California in 1974 (Binford et al. 1975), and, as of 1993, only six nests had been verified in Washington State out of the 65 nest trees known in all of North America (Nelson and Hamer 1995b). Of these 65 nest trees, adequate data were compiled for 32 of them to determine nest success, which is defined as a nest that has fledged a chick. Only nine (28 percent) of the nests were successful (Nelson and Hamer 1995b). Predation of an egg or chick resulted in 10 failures (43 percent of the failures). The relatively high rate of nest predation on those nests located near the edge of old-growth forest may be a result of a higher susceptibility to predators relying on visual stimuli. However, there is not enough historical demographic data on marbled murrelets in the western Cascades to distinguish effects of habitat loss from effects of other population-limiting factors (Ralph et al. 1995).

Hamer and Nelson (1995b) conducted an extensive study on marbled murrelet nest stands and individual nest trees from sites in California, Oregon, Washington, and British Columbia. Attributes recorded at Washington nest sites are summarized in Table 3.5-2 below. Of the six nest trees in examined in Washington, all were conifers, including three Douglas-firs, two western hemlocks, and one western red cedar. Nest branches were within the canopy, and an average of 90 percent of the nests were under canopy cover.

Table 3.5-2. Marbled murrelet nest tree and nest stand[1] data from Washington State (after Hamer and Nelson 1995b).

Attribute	Sample Size	Mean	Range
Tree height	5 trees	187 ft	148-213 ft
Tree dbh	5 trees	60 inches	35-87 inches
Branch thickness at nest	4 trees	11 inches	4-18 inches
Nest height	5 trees	121 ft	75-174 ft
Stand size	5 stands	877 acres	12-2,452 acres
Elevation	6 stands	1,142 ft	49-2,001 ft
Slope	6 stands	21%	0-39%
Tree density	5 stands	55 trees/acre	34-65 trees/acre
Canopy height	5 stands	177 ft	144-194 ft
Canopy layers	4 stands	3.4 layers	3-4 layers
Canopy closure	5 stands	69%	36-88%
Stand age	3 stands	879 years	450-1,736 years
Distance from the coast	6 stands	10 miles	3-21 miles

POPULATION THREATS

The principal threat to marbled murrelet populations from land-based activities is loss of nesting habitat as a result of old-growth harvest. Loss of nesting habitat negatively affects reproduction rates in the following four ways:

(1) It reduces the availability of nest sites, thereby reducing the proportion of the population able to reproduce;

(2) It inhibits the ability of displaced adults to find new nest sites after destruction of old ones;

(3) It increases the concentration of nests, thereby increasing use of lower quality nest sites and stand edge nest sites that incur increased predation; and

(4) It results in fragmentation of existing habitat, which increases the number of nests near stand edges and isolates populations, which increases vulnerability to environmental changes and to the loss of genetic vigor (Divoky and Horton 1995; Ralph et al. 1995; Washington Forest Practices Board 1995).

[1] Nest stand as defined by Hamer and Nelson (1995b) as a contiguous group of trees including the nest tree with gaps in the canopy no larger than 330 ft.

HABITAT AVAILABILITY IN THE CEDAR RIVER WATERSHED

In the Pacific Northwest, Marbled murrelets are believed to use forest stands within about 50 miles of marine waters and are reported to nest in forest stands up to 3,600 ft elevation (USDI 1997a). All forested habitat within the municipal watershed is less than about 35 miles from marine waters, but some is above 3,600 feet elevation.

Given the current state of knowledge of the nesting ecology of marbled murrelets on the west slope of the Cascades in central Washington, the greatest potential for suitable habitat to be available within the Cedar River Watershed exists primarily within the 13,889 acres of old-growth forest (greater than 190 years old), some of which is above 3,600 ft elevation. These unharvested native conifer forest stands are the most significant forest type potentially available for use by marbled murrelets within the municipal watershed. These stands (13,155 acres) are almost exclusively located in the upper watershed, in higher elevation subbasins, and most (10,093 acres) are within the spotted owl CHU in the eastern portion of the watershed. The stands range in age from 190 to approximately 850 years, with most of the stands presumed to be in the 200-350 year range and only a few stands as old as 850 years. The oldest stands are essentially remnant islands in the heavily harvested upper Rex River drainage (Map 5).

Most of the remaining old-growth forest is contained in several large, contiguous blocks located in the CHU in the areas of Abiel and Tinkham Peaks, Meadow Mountain, Goat Peak, and Findley Lake. In some cases, these blocks are contiguous with old growth outside the municipal watershed in other land ownership. Other remaining old-growth stands within the watershed are relatively small in comparative size and either widely scattered, isolated, or substantially fragmented (e.g., near Echo Creek), but all are greater than the minimum 7-acre size reportedly used by murrelets (USDI 1997a). Most of these stands are essentially old-growth islands separated from the larger contiguous blocks and either surrounded by stands in early stages of successional development or fragmented by recent harvest units. Most stands of this type are located outside the CHU along the northern border or in the upper reaches of subbasins south of Chester Morse Lake.

Although the chronological age of much of this habitat type is within a similar range, 200-350 years, there is substantial variation in stand structural development (Section 3.2.2). Because climatic conditions, soil conditions, elevation, and chronological age vary widely among this range of old-growth forest stands, the extent of development of both vertical and horizontal structure characteristic of ecologically functional old-growth forest, including structural development required by breeding murrelets, also varies substantially. Therefore, habitat quality may not be assumed to be of equal potential or realized use by murrelets.

The second forest type that represents potential marbled murrelet reproductive habitat is the substantial amount of advanced young and mature stands of second-growth forest in the lower watershed. Ninety-six percent (22,542 acres) of the stands within this total age range are concentrated from 61-80 years old, and very few stands (1,050 acres) fall in the 81-190 year range. Many of these stands, located within the lower Taylor Creek and Walsh Lake subbasins, have already or will soon develop vertical and horizontal structure considered to be characteristic of mature, and in some cases, late-successional stages of forest development (Section 3.2.2). A few stands of this type also exist within the Chester Morse Lake Basin within the upper watershed. Although no comprehensive studies of habitat quality have been made to date, a few preliminary evaluations have indicated that a limited amount of nest platform development is already present or could develop within the next few decades in some of these stands.

STATUS IN THE CEDAR RIVER WATERSHED

In 1991, City staff consulted with WDNR personnel who were actively studying marbled murrelet ecology in other areas of the western slope of the Cascades. Existing habitat conditions within the municipal watershed were reviewed based on topography, relative forest age, and existing cumulative knowledge of forest stand structural development. One area was identified as having the greatest potential for providing nesting habitat for murrelets. The area was surveyed late in the nesting season, and no murrelets were detected. However, in 1992 WDFW surveyed the same area during the nesting season and detected murrelet calls on two occasions (WDFW 1994b). No nest site was located and no additional surveys have been conducted to date.

Several potential habitat limitations for the marbled murrelet exist within the Cedar River Municipal Watershed. These potential limitations are outlined below.

(1) A majority of old-growth forest stands within the municipal watershed have not reached the chronological age, nor presumably the required structural development, of stands preferred by nesting murrelets in regional studies (USDI 1997a). Very few stands within the municipal watershed are over 350 years old or have developed the size or structure of forests commonly utilized by murrelets in Washington.

(2) Much of the available old-growth forest exists at high elevations in the eastern one-third of the municipal watershed, while mature second-growth is mostly in the lower municipal watershed. Suitable nesting habitat is unevenly distributed within and among these stands, and some old growth is above the reported 3,600 ft elevation limit for murrelet nests (USDI 1997a).

3.5.4 Northern Goshawk

INTRODUCTION

The northern goshawk (*Accipiter gentilis*) is a federal species of concern and a state candidate for listing as a threatened species (WAC 232-12-297). The principal reason for decline of goshawk populations is considered to be loss of habitat resulting from intensive timber harvest. A review of the general ecology and literature on the northern goshawk has been included in Habitat Conservation Plans for WDNR (WDNR 1997) and Plum Creek Timber Company, L.P. (Plum Creek 1996). Because both of these plans include lands contiguous with the Cedar River Municipal Watershed, the same general life history and ecology that they detail pertaining to the northern goshawk also applies to the municipal watershed. A portion of the following discussion is based on these two HCP documents.

RANGE

Northern goshawks are found across North America and Eurasia in boreal, temperate, and highland subtropical regions. Goshawks of the Pacific Northwest are associated with late successional coniferous forests, and are most abundant in old-growth forest

(Thomas et al. 1993). They occur as permanent residents, and less commonly as migrants, primarily in old-growth conifer forest habitats. They range widely in the winter, presumably in response to variable climatic conditions and availability of food resources. In Washington State, northern goshawks are less numerous on the west side of the Cascades than on the east side.

LIFE HISTORY

Nesting

Northern goshawks build large, stick platform nests, and lay three to five eggs (Peterson 1990). Aerial radio telemetry work in southeast Alaska found adults to be nonmigratory (Titus et al. 1995). Males maintained year-round areas loosely associated with the nest area, while some females vacated the nest area for fall and winter. The same study reported that two out of seven females renested near the previous year's nest, while five selected new mates and nested from 2.4 miles to 26.7 miles away. Reynolds et al. (1991) reported that in the Southwest the northern goshawk typically has two to four alternate nest areas within the home range, using alternate ones in different years. From the time of fledgling until the young disperse in the fall, the northern goshawk family occupies what is called the post fledging-family area. This area includes the nest area and the habitat necessary for the fledglings to learn to hunt while evading predation.

Northern goshawks may be highly sensitive to human disturbance. In Idaho it was found that nest site occupancy was reduced by 75 to 80 percent when timber harvest occurred within 0.25 miles (Patla 1990).

Forage Selection

Remains of prey items collected from 38 northern goshawk territories in western Washington (22 in the western Cascades) have been analyzed (Watson et al. 1996). Douglas squirrels (*Tamiasciurus douglasii*), blue grouse (*Dendragus obscurus*), and ruffed grouse (*Bonasa umbellus*) were the prey species selected with the highest frequency. Prey representing the greatest biomass were snowshoe hare (*Lepus americanus*) (67 percent of the mammalian prey biomass) and grouse (81 percent of the avian prey biomass). The different prey species live in a variety of forest habitats and seral stages, and along forest edges.

HABITAT REQUIREMENTS

Home Range

The home range of the northern goshawk is highly variable, depending primarily on prey density when nest sites are readily available (Reynolds et al. 1992). Based on radiotelemetry data from 51 goshawks (including 24 juveniles) in southeast Alaska, total areas used varied from 1,900 acres to 349,023 acres (Titus et al. 1995). Mean home range size was not considered a usable concept in light of the extreme degree of variability. However, Austin (1994) used minimum convex polygon methods with radio telemetry data to compute a mean home range of 7,657 acres for adults (n = 10) in the southern Cascades.

Habitat Structure

Northern goshawk nests are most often located in old-growth and mature coniferous forest stands that have a closed canopy. Of particular significance is the fact that there is great similarity between the nesting habitats selected by northern goshawks and those selected by northern spotted owls. Their nests have been found less than 100 yards apart (Marshall 1992), and 47 of 85 spotted owl nests on the east slope of the Cascades were on stick nests built by goshawks (Buchanan 1991). Approximately 80 nest sites are known in Washington (Washington Environment 2010 1992). Nest trees on the Olympic Peninsula averaged 28.2 inches dbh for breeding territories (n = 7, range 8.1 to 57.5 inches dbh). These findings also suggest that younger forest stands may not contain enough trees of sufficient size to accommodate the large nests of goshawks, and therefore are not typically selected as nesting habitat.

Goshawk foraging habitat is more varied than that typically selected for nesting and includes deciduous and mixed forests in addition to coniferous forest in all seral stages. However, the usefulness of younger forest stands is limited, because there is insufficient space both within and below the canopy to accommodate the relatively large body size and wingspan of goshawks during hunting flight. Use of such widely varied habitat suggests the northern goshawk may choose foraging habitat based on prey availability, rather than existing habitat structure or composition (Kenward and Widen 1989; Reynolds 1989). Recent observations indicate that goshawks may nest in second-growth forest, at least in some areas (Bogaczyk, B., USFWS, personal communication, April 17, 1998).

HABITAT AVAILABILITY IN THE CEDAR RIVER WATERSHED

No specific studies have been conducted within the Cedar River Watershed to determine habitat preferences of northern goshawks or specifically to evaluate the existing habitat potential. However, because there is a substantial correspondence between goshawk and northern spotted owl habitat preference, especially for nesting habitat, several parallels can be established, and habitat availability and potential can be generally evaluated.

Currently, the municipal watershed landscape exhibits a broad diversity of coniferous forest habitat types in all stages of succession from recently harvested stands to old-growth forest, extending over an elevation range of approximately 5,000 feet (Section 3.2.2). Deciduous stands, although relatively few in number, and mixed species stands are also interspersed throughout the terrain. Over that elevation span and range of diverse forest types, a substantial portion of the habitat is potentially suitable for northern goshawk nesting and foraging activity. However, these habitat types are not distributed evenly over the landscape, nor are they all of equal potential to support use by goshawks.

For northern goshawks, as is the case for spotted owls, the most significant of these habitat types is the 13,889 acres of unharvested, native conifer forest (old growth) that still remains within the municipal watershed. Ninety-five percent of these stands (13,155 acres) are almost exclusively located in the upper watershed, at higher elevations, and along ridge tops, and 10,072 acres is predominately within the spotted owl CHU in the eastern portion of the watershed. The stands range in age from 190 to approximately 850 years, with most of the stands presumed to be in the 200-350 year range, but only a few stands as old as 850 years. The oldest stands are essentially remnant islands in the heavily harvested upper Rex River drainage (Map 5).

The majority of old-growth habitat exists in large contiguous forest blocks located in the CHU in the areas of Abiel and Tinkham Peaks, Meadow Mountain, Goat Peak, and Findley Lake. These areas represent the largest contiguous blocks of old-growth forest still remaining within the watershed. In some cases, these blocks are contiguous with old growth outside the watershed in other land ownerships. Presumably, these large, contiguous blocks of old-growth forest have the greatest potential as both nesting and foraging habitat for goshawks. This type of old-growth habitat includes habitat presently utilized by northern spotted owls (Section 3.5.2).

Other remaining old-growth stands within the municipal watershed are relatively small and either are widely scattered and isolated, or are substantially fragmented (e.g., near Echo Creek). Most of these stands are essentially old-growth islands separated from the larger contiguous blocks and either surrounded by stands in early stages of successional development or fragmented by recent harvest units. Most stands of this fragmented type are located outside the CHU along the northern border or in the upper reaches of subbasins south of Chester Morse Lake.

In contrast to spotted owls, northern goshawks are highly mobile and can utilize both old growth and younger forest types as foraging habitat, and as such may be able to use a broader range of old-growth forest stand types and sizes within the Cedar River Watershed, even fragmented and isolated stands. Such use may entail both nesting and foraging activities.

Significantly, because climatic conditions, soil conditions, elevation, and chronological age vary widely among these old-growth forest stands, the extent of canopy formation (or breakup), tree height and diameter development, as well as the extent of internal structural development characteristic of ecologically functional old-growth forest also varies substantially. As a result, the relative degree of variation in stand development exhibited throughout the old-growth forest available in the upper watershed also presents a similar variation in relative habitat quality for northern goshawks. It is probable that minimum habitat requirements, but not necessarily optimum requirements, can be met for goshawks within the Cedar River Watershed under current habitat conditions, but the number of pairs or individuals that could be supported cannot be determined (see below).

In addition to old-growth forest, a substantial portion (71,525 acres, 79 percent) of the watershed landscape is occupied by second-growth forest that is in varying stages of successional development, from recent clearcuts to mature coniferous forest. These areas represent potential suitable nesting and foraging habitat, but differentially favor one or the other type of activity depending on which particular section of the municipal watershed is examined. As discussed above, the broader age range, growth conditions, and resultant structural development of these second-growth stands exhibits an even greater range of relative habitat quality than within old-growth forest habitat.

Although old-growth forest is conspicuously lacking in the lower watershed, the nearly unbroken canopy of second-growth forest may provide habitat for goshawks. This habitats is perhaps most suitable as foraging, but the oldest young and mature stands may also provide structural development sufficient to support at least some level of nesting activity as well. In particular, there are 26,902 acres of young and mature second-growth forest 50 - 119 years old in the lower watershed, stretching between the lower Taylor and Walsh subbasins (maps 1 and 5). This forest habitat presently has or will soon develop vertical and horizontal structure considered to be characteristic of mature, and in some cases late-successional, ecological stages of forest development. These older and more developed second-growth forests also represent the most substantial area of the municipal watershed in which high quality reproductive and foraging habitat for northern goshawks could be recruited in the near future by protection from harvest.

In addition, the upper watershed also contains a substantial quantity (34,710 acres) of second-growth forest. However, 54 percent of this area is in younger stages (0 - 39 years), with only 10,470 acres in the age classes between 50 and 90 years old. This distribution of younger forest represents a marked contrast both to the lower watershed and to the eastern portion of the upper watershed. In the upper watershed, younger second-growth forests predominate in subbasins both north and south of Chester Morse Lake and throughout low and mid-elevation portions of subbasins within the eastern extent of the watershed and CHU.

These relatively young stands present an especially wide range of habitat structure, and correspondingly, a wide range of foraging habitat quality for goshawks. While these areas are essentially of lower relative reproductive habitat quality than either the old-growth forests of the upper watershed or the older second-growth forests predominating in the lower watershed, they do represent a substantial area of foraging habitat. The considerable extent of young seral stands and associated edge habitat also support substantial populations of preferred prey species (e.g., hare and grouse).

STATUS IN THE CEDAR RIVER WATERSHED

No comprehensive studies of northern goshawk numbers or distribution have been conducted within the Cedar River Watershed to date. Specific knowledge concerning use of existing habitat is very limited.

Presently, only one northern goshawk nesting territory has been documented within the Cedar River Municipal Watershed. Identified in the summer of 1992 in unharvested native forest included within the northern spotted owl CHU during surveys by WDW personnel, the site was occupied, and two offspring were observed. The site was also occupied during 1996, but no offspring were observed (Spencer, R., WDFW, 1997, personal communication). This goshawk nesting territory is within a 1.8 mile spotted owl circle near the reproductive site center. No other information is known to be available on habitat use or activity in this territory.

Several potential habitat limitations for the northern goshawk exist within the Cedar River Municipal Watershed. These potential limitations are outlined below.

(1) The evaluation of northern goshawk habitat requirements and availability is complicated by the fact that there is a substantial degree of variation in habitat structural development, therefore in habitat quality, not only across successional stages but also within stands essentially equal in chronological age. Even unharvested native old-growth forest within the watershed is not of equal habitat quality, and only some of the available old-growth habitat may be adequate to support reproductive individuals.

(2) No studies on the west side of the Cascades have documented thresholds of timber harvest levels, stand age distribution, or extent of disturbance that determine the demographic attributes or limits of goshawk populations in these coniferous forest ecosystems.

3.5.5 Common Loon

INTRODUCTION

The common loon (*Gavia immer*) is one of five species of loons, very specialized diving birds that belong to the small, closely related taxonomic family Gaviidae. The geographic distribution of loons is Holarctic (without tropical species) (McIntyre 1988).

RANGE

The range of the common loon, a migratory species, extends throughout Canada, Alaska and the northern United States, and a much smaller population winters along the European Coast and breeds in Iceland and Greenland (McIntyre 1988). The North American populations breed on freshwater lakes throughout Canada, most of Alaska (except the high Arctic), and the northern tier of the lower 48 United States (Roderick and Milner 1991; McIntyre 1988). The North American populations winter along the Pacific coast from Alaska to northern Mexico, and along the Atlantic coast from Newfoundland to northern Mexico.

Substantial numbers of birds, including an unknown number of nonreproductive adults and subadults, are present temporarily on many lakes throughout sections of western Washington and the Puget Sound region during both spring and fall migrations. However, few of these wintering and migrating loons remain in the region during the summer breeding season (Roderick and Milner 1991). As a result, breeding populations in Washington State are low.

Figure 3.5-4. Common loons on Rattlesnake Lake.

STATUS

Several nest sites have been confirmed on at least five different lakes in King County during the last decade. Nest sites have also been confirmed in five other counties in the state, including several counties in eastern Washington (Roderick and Milner 1991). However, a majority of common loon nest sites in Washington are located west of the Cascade Mountains. In addition, not all of the sites identified have been confirmed active or have been reproductively successful in all years during the decade. Significantly, only 10 or 12 common loon breeding sites are currently known to have been active at any time during the 10-year period in Washington State.

The Pacific Northwest, and particularly the Puget Sound region of Washington, is located at the western and southern edges of the documented breeding range of the common loon. Historic densities of breeding loons in this region may never have been as high as the densities currently found in more central areas of the species' breeding range, particularly in Canada, Alaska, north-central United States, or in some northeastern states. However, because of the impact that human disturbance has on loons (discussed below), it is quite possible that the breeding population of this region was higher than it is today and that it has declined in recent decades.

Several types of human activity may have contributed, either directly or indirectly, to the suspected decline in the common loon breeding population in Washington State. The rising popularity of recreational boating activities on lakes that have or might have supported nesting loons may have been a major influencing factor. These activities are thought to be responsible for a dramatic increase in egg predation, which occurs after incubating loons are frightened off the nest by boats (Titus and Vandruff 1981, as cited in Roderick and Milner 1991). The species is also vulnerable to other human activities such as logging, road construction and traffic, development of shoreline (Vermeer 1973), camping (Ream 1976), removal of large wood from lakes, and fluctuation of water levels such as occurs in reservoir systems (see below). Finally, another human-related environmental factor that has contributed to the decline of loon population numbers in other areas is acid rain (Ream 1976). In areas where acid rain is a problem, loons successfully hatch eggs, but the chicks or young loons starve to death because of a lack of forage fish caused by acidification of lakes.

As a result of a suspected decline resulting from human activities, the common loon has been designated as a state candidate species, under consideration for listing as either a threatened or an endangered species.

LIFE HISTORY

Life Span, Migration, and Pair-bonding

Common loons are relatively long-lived birds that may live 25-30 years. They appear to return to the same breeding areas in successive years throughout their lives (McIntyre 1988). The annual reproductive cycle usually begins in early spring when adult males leave marine wintering areas and migrate inland to breeding territories on freshwater lakes. Pair bonding occurs soon after both members are present and territories are established quickly. Pair-bonding, mating, and egg laying in the Pacific Northwest region typically occurs between April 1 and May 15.

Breeding Territories

Individual loons have been observed to exhibit strong fidelity to nest sites and territories (McIntyre 1974), occupying the same nest sites and utilizing the same territories for many years. Until recently it had been assumed that pairs were monogamous within the breeding season and over their full life span. However, recent evidence suggest that although pair-bonds are maintained within a breeding season, pair membership may change in successive seasons, and then, in some cases, be reestablished.

Mated pairs typically occupy their established breeding territory throughout the summer until just prior to fall migration. However, individual pair members may leave territories periodically during the summer on a temporary basis, move to other lakes, and then return, even if chicks are present. This type of shift in habitat use may be related to available food supply within breeding territories and nursery areas (McIntyre 1988).

Nest Site Characteristics

Loons prefer to nest on small islands within lakes and ponds rather than on perimeter shoreline (Roderick and Milner 1991; McIntyre 1988; Titus and Van Druff 1981; Ream 1976; Vermeer 1973). Because loons' walking ability is very poor, nests are usually located at the water's edge or within 4.5 ft of shore (Vermeer 1973). Nests are constructed from a wide range of terrestrial and aquatic plant materials (McIntyre 1988), but may also include other types of materials such as sticks, and in some cases, stones. If nest material is not available, eggs may be placed on bare substrate. Ideally, the nest floats or is adjacent to water deep enough to permit underwater approach and departure by the adults (Ehrlich et al. 1988). Common loons also nest on emergent aquatic vegetation at the edge of shallow water, and are known to successfully use artificial nesting platforms in some areas (McIntyre 1988; McIntyre and Mathisen 1977, including the Chester Morse Lake reservoir complex.

Effects of Water Level Fluctuations on Nest Sites and Behavior

As mentioned above, common loons typically establish nest sites at the waterline to provide both easy and covert access to the nest from the water and immediate escape from the nest to water when threatened (McIntyre 1988). Nests may consist of a minimal amount of material deposited on a relatively flat surface or be of more substantial size, constructed of materials such as sticks, vegetation, and aquatic plants, and attached to a shoreline structure or emergent vegetation (McIntyre 1988). Nests established in either of these ways are relatively secure under most conditions in natural systems, but because of their waterline locations, are always susceptible to any changes in water levels. Relatively small increases in water levels can inundate nests, tilt them, or break them free and cause the nests to float away, resulting in loss of eggs or nest abandonment. Conversely, decreasing water levels may prevent adults from accessing established nest sites in certain situations because of the awkwardness of loons on land (McIntyre 1988; Leahy 1982). Decreasing water levels may also prevent newly hatched chicks from reaching water and thus expose them to abandonment or excessive threat from predators.

During the nesting season, loons are capable of behaviorally compensating for small changes in water levels by adding material to nests to keep eggs above water, or in rare cases traversing dewatered substrates to access nests. On most natural bodies of water, changes in water levels tend to be gradual and of relatively small magnitude. However, in many regulated systems such as reservoirs, operational constraints imposed by water supply, flood control demands, and unpredictable environmental conditions produce both rapid and dramatic fluctuations in surface elevations. Such fluctuations can create adverse conditions for nesting loons.

Clutch Size and Incubation

Common loon females typically lay from one to three large, brown eggs per season, but four-egg clutches have been observed on rare occasions (McIntyre 1988). A clutch size of two eggs is the most common. Pairs do renest and deposit additional eggs, especially if initial nest sites are disturbed or eggs are lost early in incubation, but subsequent clutches may not contain more than a single egg. Both pair members alternately incubate eggs over a period of time that is approximately 26-31 days in duration. Nests are normally occupied by an adult more than 99 percent of the time (McIntyre 1988).

Chick Behavior

Chicks are usually coaxed from the nest onto open water by offers of small food items and communication calls within a few hours of hatching, but may return and leave the nest several times if hatching of a second egg is delayed. If hatching is synchronized, the chance of more than one chick surviving is significantly increased. Mated pairs are not known to raise more than one brood in a single breeding season (Ehrlich et al. 1988). Once all viable chicks have been taken from the nest, they move to an open water nursery pool where the chicks are alternately carried on the backs of adults (up to 65 percent of their first week) or encouraged to swim on their own. In the nursery area, both adults feed the chicks small aquatic invertebrates at first, but gradually shift the diet to small fish as the chicks develop.

A suitable nest site is important during the egg-laying and incubation period, but a suitable nursery pool is also necessary to ensure chick survival (Ehrlich et al. 1988). The water in the nursery should be clear enough for the birds to spot their prey, shallow enough to limit the size of predatory fishes and turtles, and reasonably free of predatory eagles and gulls, and it should be rich enough to furnish an 11-week supply of food for two chicks. The nursery pool should also provide a good view of neighboring territories and should be protected from wind and wave action that could separate the adults from the chicks as they swim from nest to nursery. An adequate food supply of larger fish and aquatic invertebrates must also be available for adults either in the nursery area or within accessible distance during the remainder of the summer season prior to fall migration.

STATUS IN THE CEDAR RIVER WATERSHED

Historic Perspective

Relatively little is known about the historic presence or reproductive success of common loons within the Cedar River Watershed prior to the last 20-25 years. Despite the lack of information before that period, a general knowledge does exist of (1) the historic uses of the watershed, (2) the major habitat changes through time, and (3) the degree of protection that has been afforded Chester Morse Lake over the last 100 years. The City assumes that loons have nested on the shores of Chester Morse Lake (reservoir) for many decades, and possibly on the original natural lake (Cedar Lake) for hundreds of years. In the period of the mid-1970s to late-1980s, loons were frequently sighted on Chester Morse Lake, and young chicks were observed by City staff on the Masonry Pool at least once in each of the years 1979, 1982, and 1988.

Common Loon Ecology Project in the Cedar River Watershed

Beginning in 1989, City biologists have been conducting an ongoing research project investigating the ecology of common loons in the Cedar River Watershed, focusing primarily on the Chester Morse Lake/Masonry Pool reservoir complex. In addition to annual surveys of the extent of loon utilization on watershed lakes, the reproductive success of nesting pairs has also been monitored. Since 1990, a third component of the project has been the construction and experimental deployment of floating nest platforms to enable nesting pairs to deal more effectively and consistently with fluctuating reservoir levels (Section 4.5.5). Loons have consistently utilized several bodies of water within the watershed, and individual pairs have been reproductively successful on the reservoir complex in each of the years that the research and monitoring project has been conducted (see below).

Lakes and Ponds Used by Common Loons in the Cedar River Municipal Watershed

Several bodies of open water are present in the Cedar River Watershed below 2,000 ft elevation that are of sufficient size (greater than 30 acres) and provide a food supply of mostly salmonid fishes to adequately support common loons (Map 8). Both Chester Morse Lake (1,537 acres) and Masonry Pool (187 acres) are frequented by loons during spring and fall migrations and are also utilized on a regular basis during the nesting season. The use of these nesting areas is likely increased by the minimal human activity adjacent to the lake, the absence of artificial structures from almost all of the shoreline, and very limited boat traffic. Boat traffic is limited to essential operational activities.

Such activities are minimized whenever possible, especially during the nesting season, to avoid adverse impacts on loons.

Only supervised public access to these two areas is allowed, providing substantial protection form disturbance. However, both Chester Morse Lake and Masonry Pool are subject to substantial water level fluctuations of 10-20 ft during the spring to fall period, when nesting loons are present on the reservoir complex. Such fluctuations may produce habitat conditions that can negatively impact loon reproductive success in the reservoir system.

Walsh Lake – 69 acres in size and at elevation of 725 ft – is protected and of adequate size to support loons, yet receives a relatively low level of foraging use and has had no documented nesting activity. The lack of use may be a result of an inadequate food supply, insufficient water quality or shoreline habitat, or high levels of predation. Rattlesnake Lake (111 acres), near the boundary of the watershed and on a major vehicle access route, is open for public recreation (boating, fishing, swimming) and supports heavy human recreational use, especially during summer months. Despite the high level of human activity, common loons use Rattlesnake Lake during migration periods, especially for foraging. However, loons do not presently nest there, presumably because of the consistently high level of intrusive human disturbance.

Only one other water body over 25 acres exists within the municipal watershed: Findley Lake, at 3,701 ft elevation. However, Findley Lake, a high-elevation lake of relatively low productivity, probably does not contain a food supply adequate to sustain use by loons on a regular basis. No known observations of loons have been documented at Findley Lake. Several small bodies of water (less than 5 acres) are scattered throughout the watershed, mostly at elevations above 3,000 ft, that may receive occasional limited use by loons, but there have been no observations of loons on these water bodies.

Migration

Annual field surveys from 1990 to 1997 have confirmed that common loons consistently use the reservoir complex during spring and fall migrations. In smaller numbers, loons also use the reservoir as foraging and nesting habitat during summer months. Loons typically begin arriving in the watershed by mid-March, but individuals may arrive or travel through the watershed as soon as late February. At the peak of migratory activity from mid-March to mid-April as many as 25 birds may be using bodies of water within the watershed system. The greatest numbers of loons have been observed on the Chester Morse Lake/Masonry Pool complex (up to 18 individuals), followed by Rattlesnake Lake (up to 7), and Walsh Lake (up to 2). It is not unusual to observe birds moving back and forth between Chester Morse Lake and Masonry Pool or Rattlesnake Lake, especially during periods of peak migration.

Although many loons are present on the reservoir complex during the peak of spring migration, numbers decline by late April to three nesting pairs and one or two additional birds that may remain for some portion of the summer. Several loons are present on Rattlesnake Lake during the migration period and prior to the seasonal increase in human activity coinciding with the opening of fishing season. One or two birds may be sighted occasionally during the summer months, but typically not on a sustained basis, suggesting only sporadic or transient use. Substantially less use of Walsh Lake occurs both during late spring migration and during the summer period. Use of this lake appears to be far less consistent than use of other bodies of water in the watershed.

Fall migration, including possibly inter-regional movement, is less well defined with varying periods of activity extending from late August through September and sometimes into early October. Adults, and occasionally juveniles reared on other lakes, move through the watershed system during this time. Most adults, including birds that nested on Chester Morse Lake and Masonry Pool, have typically left the reservoir complex by mid-October, but individual juveniles may remain as late as mid-November. Fewer loons are in the watershed system at any one time during the fall season.

Nest Site Availability

Because a steep, rocky slope dominates most of the perimeter of Chester Morse Lake, only a small percentage of its shoreline appears to constitute suitable loon nesting habitat. These areas are concentrated in the delta regions of the Cedar and Rex rivers described by Raedeke Associates, Inc. (1997). Additional nesting habitat exists adjacent to the Masonry Pool, but habitat along most of its shoreline presents is poor. Loons nesting in this system typically select nest sites that are protected by both adjacent and overhead deciduous cover, usually willow (*Salix* spp.) shrub, alder (*Alnus rubra*), or black cottonwood (*Populus trichocarpa*). Natural nests on Chester Morse Lake and Masonry Pool are rarely located completely in the open. Although loons are capable of constructing nests attached to the shoreline, the loons nesting in this system tend to select floating logs as nest sites. Both of these types of nests are susceptible to adverse impacts resulting from widely fluctuating water levels. Nests can be trapped or tilted by overhanging vegetation as water rises, or tilted or left far from water as levels drop.

Experimental Nest Platforms

Artificial nest platforms have been made available within each of the three loon nesting territories each year since 1990. The floating platform nests ideally provide nesting loons with an alternative, more stable nest site that can more effectively adjust for most rising water conditions, but only some degree of falling water level conditions. Platforms have been used in at least one, and typically in two, of the three nesting territories in each of the 8 project years during which platforms have been deployed. Nesting platforms have been used exclusively whenever birds have nested in two of the three territories since the first year platforms were utilized in those territories, including 7 consecutive years in one territory and 6 of 8 years in the second territory. A platform has been used in only 2 of 8 years in the third territory; different natural nest sites were selected instead.

Nesting Activity on Chester Morse Lake and Masonry Pool

Three mated pairs of common loons have been present on Chester Morse Lake and Masonry Pool during each pair-bonding and nesting season for the last 9 years (1989-1997). These pairs have consistently occupied very similar, widely separated territories at the two opposite ends and on the south side of the reservoir system. All three areas are partially isolated both visually and audibly from each other. Each territory encompasses approximately 25-30 percent of the reservoir complex area. Two of the nesting territories have been occupied by reproductive pairs in each of the 9 years of the study. However, although a pair of loons has been present in a third territory in each of the 9 years of the study, the pair has failed to establish a nest during 3 of those years. Low water levels, pair member changes, or other factors may have prevented reproductive efforts during these 3 years.

Nests are typically established in mid-April, but may be delayed until early to mid-May by individual pairs in some years. Clutch size is consistently two. Hatching usually occurs in mid-May after an approximate 28-30 day period of incubation, but may not occur until mid-June if egg-laying is delayed. Hatching success varies widely between pairs from year to year, as some eggs are lost during the incubation period to predation, mainly by river otters and birds, or are addled and abandoned, which results in a single chick or no chicks being hatched. Survival to fledging is high for chicks, which live 3-5 days after hatching. Between one and four chicks have fledged on the reservoir system each year since 1989.

Common loons have established a total of 21 nests on Chester Morse Lake and the Masonry Pool during the period 1990-1997, since experimental nest platforms were first deployed in 1990. Of the 21 nests established during that 8-year period, 7 have been on natural nest sites and 14 have been on experimental platforms. A total of 24 chicks have hatched: 6 on natural nests (5 of which survived to fledging) and 18 on platforms (16 of which survived to fledging). Four chicks hatched and survived to fledging from 3 natural nests in 1989, before any experimental platforms were deployed.

POTENTIAL HABITAT LIMITATIONS AND ECOLOGICAL CONSIDERATIONS

Effects of Fluctuating Water Levels on Reproductive Success

Water levels are one of the major environmental elements determining reproductive success of common loons in the Cedar River Watershed. Cumulative observations of loon nesting behavior during the 9 years of study strongly indicate that water level fluctuations in the Chester Morse Lake/Masonry Pool reservoir complex during the nest establishment and incubation periods are the most significant factor that determines (1) the timing of loon nest establishment, (2) the general areas of nest locations, and (3) the ultimate stability of natural nest sites once established. While water levels are of importance in all natural bodies of water, their potential impact on loon nesting success is magnified within reservoir systems that present widely fluctuating water levels on a day-to-day or week-to-week basis.

Experimental nest platforms in the watershed have proven adequate in stabilizing the environment of active nests by compensating for relatively large increases (5-8 ft) in water levels. Nest platforms have also compensated for some decreases in water levels, but are limited in this capacity, just as natural nests are, when water depths adjacent to and under floating nest structures (logs, platforms) reach very low levels (less than 2 ft depth) or critical levels (less than 1 ft depth to being stranded). Observations of common loons' sensitivity to disturbance of nest sites, especially early in the nesting period, strongly indicate that attempts to physically move platforms with nests already established, possibly depending on the stage of incubation, would probably cause abandonment.

The timing of nest platform deployment would be inconsequential in natural systems that have relatively stable water levels, because platforms can remain in place year round without substantial change in the nest site environment or unusual risk to nesting birds. However, in reservoir systems where water levels fluctuate irregularly, the timing of nest platform deployment and exact placement are both critical factors directly affecting nest success. Unless placement and timing of platform deployment is handled carefully, birds may establish nests on platforms that will eventually expose the nests to risk. This risk may occur where early-season high water allows access to higher elevation sites, including platforms, that will later become stranded as water level drops. Risks may also result from the opposite situation in which early-season low water levels cause the birds to use initially exposed, marginal sites and platforms, which will become progressively more precarious as water levels rise.

Although nest platforms have been used with some success to compensate for extremely low reservoir levels, such as those created by drought conditions when all known traditional habitat was inaccessible, the nesting environment created for the incubating birds subjects them to risk. When positioned in open water on either Chester Morse Lake or Masonry Pool, as is necessary under existing low reservoir conditions, nests on platforms are exposed to the severe winds and wave action and are constantly exposed to an increased risk of egg predation. Incubation is much less consistent because birds leave the nest platforms more often under such severe conditions, which ultimately may adversely affect nest success.

These observations indicate that loon nesting platforms in the Chester Morse Lake/Masonry Pool reservoir complex work very effectively within a certain range in the level and timing of reservoir conditions to provide more stable nesting environments than presented at many natural sites affected by water level fluctuations. However, these same observations strongly indicate that nest platforms should not be considered as capable of offsetting the effects of *widely fluctuating* reservoir levels, especially in the case of prolonged, early-season low surface elevations characteristic of a drought.

Water Level Effects on Nest Site Habitat Structure

Common loons nesting in the Cedar River Watershed on Chester Morse Lake and the Masonry Pool during the past 9 years have consistently selected nest sites that are located within the willow scrub-shrub vegetation communities of the Cedar and Rex river deltas or under a deciduous canopy of red alders and black cottonwoods at the shoreline perimeter of the lake. Both of these habitat types exist within an elevation zone that is directly influenced by reservoir levels during some period of time during the annual fill and drawdown cycle of the lake. These vegetation communities are typically exposed for much of the annual cycle and not appreciably affected during periods of drawdown and low lake levels. However, each habitat is impacted during the late winter and early spring period of reservoir refill when the elevation zones occupied by these deciduous species are inundated by varying depths of water. Duration of inundation also varies from year to year.

The direct effect of inundation is to reduce oxygen and light reaching roots and other growth tissues of the willow, red alder, and scattered coniferous species present in the flooded zone. The duration of inundation is also critical, because most of the shrub and tree species in these areas can withstand some level of inundation for relatively short periods of time but are progressively impacted as periods of inundation are extended. Conifer species are most sensitive, alder is moderately sensitive, and willow is the most resistant to such inundation.

Reservoir levels and the duration of periods of inundation during the critical spring season have increased in recent years. Raedeke Associates, Inc. (1997) have identified a trend in lake level increases and correlated the observed die back of conifer, red alder, and willow in mid-delta zones and lake perimeter areas with decreases in growing season length over the last 3-4 years. These findings suggest that if current reservoir fill regimes are continued, die back of deciduous vegetation may continue to some extent, with a potential for additional loss of nesting habitat. The extent to which these same habitats would recover and regenerate if reservoir levels were lower and periods of spring inundation were shorter is unknown.

3.5.6 Bull Trout

INTRODUCTION

Bull trout (*Salvelinus confluentus*) (Figure 3.5-5) is a western North American char in the family Salmonidae. The USFWS listed the Puget Sound Distinct Population Segment as threatened on November 1, 1999 (Fed. Reg. Vol. 64, No. 210). Currently, Washington State classifies bull trout as a priority species because it is considered to be vulnerable to significant population declines (WDFW 1996a).

The bull trout is declining in numbers throughout its range, especially along its southern limits (McPhail and Baxter 1996). Over-fishing, human-made migration barriers, increased siltation, changes in temperature and flow regimes, and competition and hybridization with introduced salmonids are the predominant causes of population declines (McPhail and Baxter 1996). The factors affecting Puget Sound bull trout include (Fed. Reg. Vol. 63, No. 111):

- Habitat degradation, with a variety of causes, including reservoir operations, dams, agricultural practices, urbanization, forest management, and human-constructed barriers in streams;
- Over-fishing and poaching;
- Inadequacy of existing regulatory mechanisms; and
- Impacts from introductions of non-native species.

Bull trout in the Cedar River Watershed have been insulated from some of these pressures. Three factors have contributed to maintaining the viability of the native bull trout population in this system: (1) Since 1908, the municipal watershed has been closed to public access, and the Chester Morse Lake population has not been harvested; (2) habitat modifications from land use have been reduced because the municipal watershed is relatively undeveloped; and (3) non-native fish species detrimental to the health of stocks in other systems are not present in the Chester Morse Lake system.

STATUS OF STOCK

In a 1998 study, WDFW assessed the status of individual populations of bull trout and Dolly Varden (*Salvelinus malma*) in Washington State, and found that of the 80 identified stocks placed into five rating categories (healthy, depressed, critical, unknown, or extinct), the status of 72.5 percent are unknown and 17.5 percent are categorized as healthy. The Chester Morse Lake stock status is classified as unknown. However, the assessment states "there are no data suggesting a chronically low condition, or short-term severe decline" in the population. (WDFW 1998a). One of the reasons for this unknown status is the discrepancy between the number of redds expected and the number counted. By applying information from bull trout spawning studies in the Flathead Lake system to the Cedar River Municipal Watershed, an estimate can be derived for the number of redds that may be expected. R2 Resource Consultants (in preparation) predict over half of the estimated 3,100 or more bull trout in Chester Morse Lake are adults. An average of 57 percent (38-69 percent) of adult-size bull trout are believed to leave Flathead Lake to spawn, and there were an estimated 3.2 spawners per redd in the spawning tributaries (Fraley and Shepard 1989). Using these data as a basis for estimation, of the approximately 1,550 mature fish in Chester Morse Lake, 884 (589-1070) could be expected to leave to spawn, potentially creating 276 (184-334) redds. However the annual number of bull trout redds counted from 1992-1997 in the municipal watershed has averaged 38 (6-109).

Figure 3.5-5. Bull trout.

Most of the information regarding the status of bull trout in the Cedar River Watershed discussed in this section was collected during a resident fisheries study that was conducted between 1992 and 1995 by R2 Resource Consultants with assistance by Fisheries Consultants, Herrera Environmental Consultants, and Taylor and Associates. This study was initiated in response to concerns about the possible effects of current and future water management operations on resident fish in the reservoir complex. The purpose of the study was to obtain baseline information regarding bull trout, rainbow trout, and pygmy whitefish in Chester Morse Lake, Masonry Pool, and their primary tributaries, the upper Cedar and Rex Rivers. Hydroacoustic surveys, gill net sampling, and stream surveys were combined to determine fish abundance distribution; spawning locations; habitat; fry emergence and migration; feeding habits and food availability; age and growth of basin bull trout; and basin conditions that influence this stock (R2 Resource Consultants, in preparation). Additional information on the bull trout population was gathered from other fisheries studies in the Cedar River Watershed (Wyman 1975; EVS Consultants 1984), published literature, and studies and field observations conducted by City biologists.

No substantive evidence to date indicates that either a self-sustaining population of bull trout or any significant number of individuals exists in the approximate 14 miles of the mainstem Cedar River, or its tributaries, between the Masonry Dam and the Landsburg Diversion Dam. Although passage over the Masonry Dam, and subsequent downstream movement, of a limited number of bull trout is expected to occasionally occur during seasonal spillway releases of water from the Masonry Pool, it apparently has not been sufficient to support establishment of bull trout populations under the ecological conditions existing in downstream reaches. Only recently has an observation been documented from this reach, an incidental sighting of a single adult bull trout near the powerhouse at Cedar Falls during September, 1997 (Binkley, K., Seattle City Light, Personal communication, 1997).

Until recently, only limited sampling of the Cedar River and its tributaries between the Masonry Dam and the Landsburg Diversion Dam had been conducted. Casne (1975) reported that rainbow trout were predominant in the river, and did not report capturing bull trout. Similarly, no bull trout have been documented by Water Department, Seattle Public Utilities, or other state agency personnel during subsequent, periodic sampling efforts.

Most recently (1994), City personnel, with Taylor Associates, conducted systematic snorkel surveys of four, 1-mile reaches and two, 100-meter reaches of the 12.5-mile section of the mainstem Cedar River between Landsburg Diversion Dam and the natural passage barrier approximately 0.75 mile upstream of Cedar Falls. All sample reaches were sampled during daylight hours and two, 1-mile reaches were sampled at night. Of the total 5,250 salmonids observed, none were identified as bull trout.

LIFE HISTORY

In 1978, bull trout were differentiated from Dolly Varden as a separate species (Cavender 1978). This original work was supported by the further investigations of Haas and McPhail (1991). In 1995, detailed physical analyses (meristics) were performed on the char population in Chester Morse Lake. The analyses determined that these fish are bull trout and not Dolly Varden (R2 Resource Consultants, in preparation). The bull trout in the Cedar River Watershed are considered to be native.

Bull trout are known to exhibit four types of life history strategies. The three freshwater forms include adfluvial forms, which migrate between lakes and streams; fluvial forms, which migrate within river systems; and resident forms, which are non-migratory. The fourth strategy, anadromy, occurs when the fish spawn in fresh water after rearing for some portion of their life in the ocean. Data indicate that the bull trout in the Chester Morse Lake system are adfluvial. Adfluvial forms rear as juveniles in tributaries, migrate to lakes where most growth occurs, then return to the tributaries as adults to spawn.

Although it is possible that bull trout with a resident life strategy exist in the watershed, ongoing studies have not provided clear evidence that confirms this scenario. In the municipal watershed, bull trout are found only within the Chester Morse Lake/Masonry Pool system and their major tributaries, above the natural migration barrier of Lower Cedar Falls (Map 8).

Age and Size

Survey results indicate that most of the bull trout in Chester Morse Lake are 3 years of age or older, although survey techniques used in Chester Morse Lake did not target fish smaller than 200 mm. Bull trout in Chester Morse Lake become mature at approximately 5 years of age and may live to at least 12 years (R2 Resource Consultants, in preparation). This life span and age at maturity is consistent with observations of bull trout in other systems (McPhail and Baxter 1996).

The bull trout captured in Chester Morse Lake and Masonry Pool ranged in size from 250 to 581 mm and were 2-12 years of age (R2 Resource Consultants, in preparation). Although the sampling gear of gill nets and rod and reel limited the size range of fish captured, the lower size limit of bull trout in other lakes is usually around 200 mm (McPhail and Baxter 1996). The growth rate of bull trout in Chester Morse Lake appears to be typical of the species in other lakes in the Pacific Northwest, although as the bull trout become larger, their growth rate in Chester Morse Lake becomes somewhat slower than in other systems. This slower growth rate of the larger fish may be a consequence of lower lake productivity or a reduced prey base relative to other systems.

In the Cedar and Rex rivers, bull trout sizes were all less than 200 mm in the age classes 0+ to 2+ (R2 Resource Consultants, in preparation). The average size of age 1+ fish caught in the rivers was approximately 140 mm. This size was similar to those reported for other northwestern char and trout, but less than the theoretical size in the river back-calculated from larger fish caught in Chester Morse Lake (R2 Resource Consultants, in preparation). Back calculations of fish size at earlier ages from analysis of scale annuli showed two peaks in distribution for years 2-5 (R2 Resource Consultants, in preparation). These were 180 and 220 mm for age 2, 270 and 325 mm for age 3, 340 and 380 mm for age 4, and 440 and 520 mm for age 5.

The two size groups (called a bimodal distribution) probably represent fish that follow different life history strategies. Those in the larger group likely enter the lake before reaching age 1+, or rear in particularly favorable stream or off-channel habitat. Populations with a similar bimodal size distribution were found in coho fry that overwintered in off-channel ponds and the mainstem Coldwater River, B.C. (Swales and Levings 1989), and between two off-channel ponds with different characteristics on the Clearwater River, Olympic Peninsula (Peterson 1982). Cedar River bull trout juveniles have been found residing in wall-base channels, a type of stream noted for low velocity flows and moderate temperatures (Peterson and Reid 1984). The absence of bull trout older than 2+ in the streams indicates that the stock is adfluvial.

ABUNDANCE

The population of Chester Morse Lake bull trout was estimated to be approximately 3,100 fish in 1995 (R2 Resource Consultants, in preparation). This is probably an underestimate, as it was based on hydroacoustic analysis. As a method of censusing fish populations, hydroacoustic analysis is generally incapable of providing data that can clearly distinguish between the reflected acoustic signals of fish near a lake bottom and background noise. Bull trout were most abundant between 75 and 125 ft, and they typically remained within 10 ft of the bottom. Although bull trout appear to prefer deeper waters, they use a wide range of lake habitat as discussed below in the subsection entitled "Rearing Distribution and Habitat Use."

The density of bull trout in Masonry Pool is much less than, and possibly half, the density observed in Chester Morse Lake. The overall bull trout population in Masonry Pool is estimated to be less than 5 percent of the total Chester Morse Lake bull trout population, or approximately 150 fish.

In the upper Cedar River, bull trout density ranged from 69 to 543 fish per acre. Bull trout density in the Rex River ranged from 89 to 348 fish per acre. These densities are within the range of bull trout densities reported for other Pacific Northwest rivers. The highest densities found at certain sites in the Cedar and Rex rivers are characteristic of high densities found in critical rearing habitat in other systems. However, the variety of methods used to estimate abundance that are reported in the literature makes comparisons among systems difficult (McPhail and Baxter 1996).

REARING DISTRIBUTION AND HABITAT USE

Chester Morse Lake

As mentioned above, hydroacoustic analysis in Chester Morse Lake revealed that bull trout typically distribute themselves in deep water within 10 ft of the bottom. However, angling surveys indicated that the fish are also abundant in the littoral areas at certain times of year, particularly near the river deltas (Wyman 1975). Because bull trout are cold-water fish and prefer temperatures between 7.8º and 13.9º C (46º and 57º F) (Shepard 1985), their distribution in the lake is probably influenced by temperature. In the summer, as surface water temperatures increase, bull trout move out of shallow, littoral areas and into deeper water (Shepard 1985).

Bull trout that are rearing in the lake depend for food on benthic insects and fish, and rely less on food that is carried in from the rivers. As bull trout grow, their diet shifts from predominantly benthic insects to fish. An examination of bull trout stomachs revealed that 29 percent of all stomachs contained dipterans (primarily chironomids), 25 percent contained sculpins, and 25 percent contained pygmy whitefish (R2 Resource Consultants, in preparation). These food items indicate that bull trout rely on all major regions of the lake, including the near-shore (littoral), deep (profundal), and mid-column (pelagic) waters. In the spring, bull trout rely more heavily on benthic insects and pygmy whitefish. In the fall, bull trout rely more on sculpins. Crayfish were also a commonly consumed prey item in the spring and fall. Research in other systems similarly has shown that sculpins and whitefish are important food items for bull trout (McPhail and Baxter 1996).

Streams

In streams, bull trout are primarily bottom dwellers, occupying positions in contact with, and often within, the substrate. Although rearing bull trout use a wide range of habitats, they are most commonly associated with a large cobble and boulder substrate, woody debris, and deep scour and plunge pools (Shepard et al. 1984; Fraley and Shepard 1989;

Heifetz et al. 1986; Goetz 1991; R2 Resource Consultants, in preparation). Newly emerged fry are often found in shallow, low-velocity side-channels or alcove pools, often in association with woody debris and fine substrates (Goetz 1991).

In the Rex and Cedar rivers, the highest densities of juvenile bull trout were found in pools with high hydraulic complexity and large woody debris or boulders (R2 Resource Consultants, in preparation). Additional data on bull trout distribution in smaller tributaries to Chester Morse Lake and Masonry Pool were collected during continuing field investigations by City biologists. In 1995 and 1996, minnow traps were deployed in several streams not included in the R2 Resource Consultants study. The 1995-1996 effort was conducted as part of a larger fish distribution study throughout the municipal watershed. Results confirmed that bull trout juveniles were present in seven additional streams: four small groundwater-fed channels (wall-base channels) to the Cedar River near Camp 18, and in Eagle Ridge Creek, Morse Creek, and Cabin Creek (Map 7).

In 1997, a preliminary study in the municipal watershed was initiated in which visual observations of juvenile bull trout were made from streambanks during daylight hours. Although during the day juvenile bull trout may react to observers by seeking cover more readily than at night (Bonneau et al. 1995), daylight observations in the municipal watershed seemed an efficient method of spotting recently emerged bull trout. Results from this preliminary study reaffirmed the reliance of newly emerged bull trout on low velocity, shallow side-channels, alcove pools, and woody debris. These foot surveys also confirmed the presence of juvenile bull trout in an eighth stream, Rack Creek.

In addition to hydraulic conditions, water temperature and food supply may be important factors to juvenile fish distribution. Juvenile bull trout were observed actively feeding in the daytime at temperatures above 9º C (48.2º F), but were not observed feeding in the water column of streams below this temperature (R2 Resource Consultants, in preparation). The diet of juveniles in the Rex and Cedar rivers was not examined directly, but newly emerged fry are believed to feed predominantly on benthic insects. As they grow larger their diet shifts to drifting insects and small fish such as sculpins and juvenile trout (McPhail and Baxter 1996).

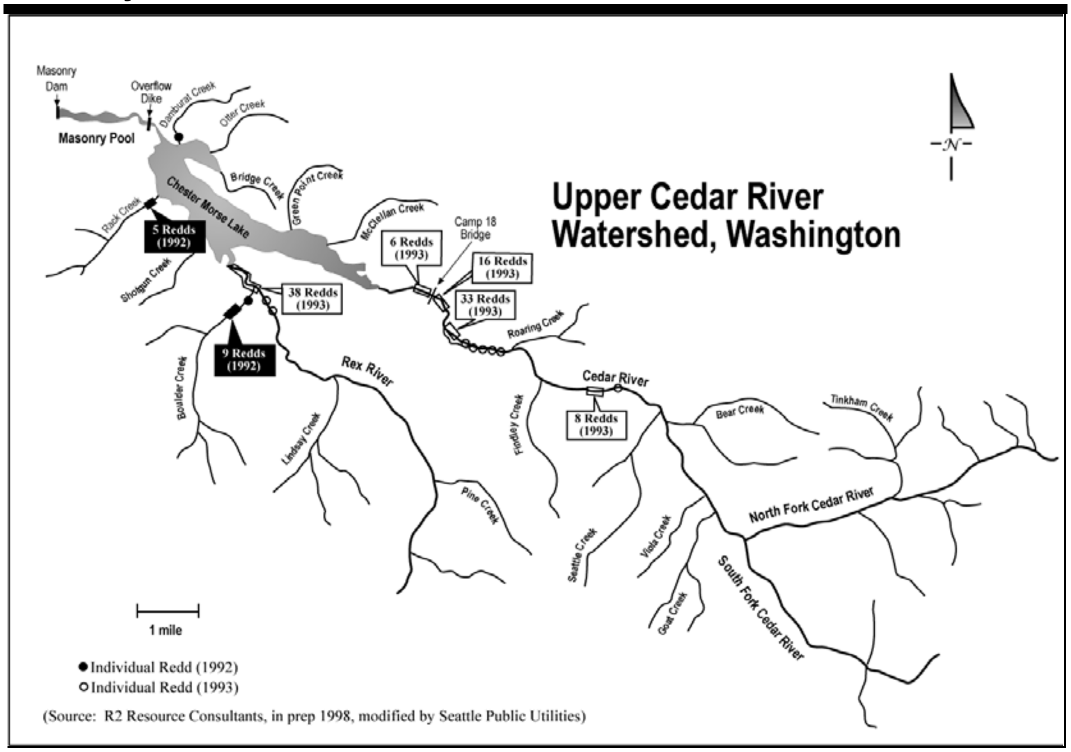
SPAWNING DISTRIBUTION AND HABITAT USE

Bull trout spawn in the Cedar and Rex rivers, as well as in some of the smaller tributaries of Chester Morse Lake. Spawning surveys of Chester Morse Lake bull trout were conducted over 3 years (1992-1994) as part of the R2 Resource Consultants study (in preparation). Fourteen tributaries to Chester Morse Lake were surveyed during this study, and redds were observed in five tributaries. A total count from all streams yielded 38, 109, and 27 redds during the years 1992, 1993, and 1994, respectively (Figure 3.5-6). While it is unknown if lake spawning occurs along the shores of Chester Morse Lake, lake spawning is known to sustain the populations of at least one bull trout stock in Washington State (Middle Hidden Lake, Okanogan County) and possibly one other (Lower Hidden Lake, Okanogan County) (WDFW 1997a).

City biologists have continued spawning surveys in the Cedar and Rex rivers since 1995. The total number of redds observed in 1995, 1996, and 1997 were 6, 8, and 41 redds, respectively. Redd surveys in 1995 and 1996 were seriously hampered by repeated high flows throughout the spawning period, and may not reflect the actual number of redds constructed. In 1997, some surveys were extended to upstream reaches in the Cedar and

Rex rivers and Boulder and Rack creeks in order to gain further insight into spawning habitat distribution.

Figure 3.5-6. Distribution of bull trout redds, showing year of highest count by stream, 1993-1997.



Chester Morse Lake bull trout spawn later in the year than any other known bull trout population. Some other bull trout stocks spawn as early as mid-July, and all populations complete their spawning by mid-November (WDFW 1997a; McPhail and Baxter 1996). The start and finish dates of the bull trout spawning period in the municipal watershed varied among the study years, but generally ranged from early October to early December. Peak spawning varied from mid-October to mid-November, and seemed to be somewhat correlated with flow conditions.

Table 3.5-3. Bull trout fry trapping summary in the Cedar and Rex rivers, 1994–1997.

Trap Site Location	Year	Distance From Lake	Trapping Period	Trap Hours	Days Set	Fry Captured
Cedar River Camp 18	1994	0.96 mi	3/24 - 6/12	811	46	193
Cedar River Camp 18	1995	0.96 mi	3/16 - 6/06	1364	68	163
Cedar River Camp 18	1996	0.96 mi	3/23 - 6/21	963	54	0
Cedar River Camp 18	1997	0.96 mi	3/11 - 6/13	743	38	15
Cedar River down-stream of Bear Ck.	1995	7.06 mi	3/23 - 6/23	1051	60	2
Rex River	1994	0.58 mi	3/24 - 6/10	454	25	4
Rex River	1995	0.58 mi	4/23 - 6/07	657	33	17

It is unknown why the Chester Morse Lake population exhibits such a unique spawning period, but it is probably because the stock is adapted to the specific flow conditions and species assemblage present in the watershed. The fish apparently migrate into the streams and rivers to spawn following high flows. Water temperatures may also be a stimulus for bull trout spawning. Spawning surveys conducted in the Wenatchee National Forest over an 8-year period indicate that redd construction begins as temperatures drop below 9-11º C (Brown 1992a). These observations are consistent with the water temperatures recorded in the Cedar River near Camp 18 at the beginning of the spawning period (Figure 3.5-4). Unlike with the direct observations of bull trout on redds in the nearby Skykomish River (Pfeiffer, B., WDFW, 1997, personal communication), bull trout in the municipal watershed have not been seen on their redds during daylight hours.

In the municipal watershed, the highest spawning densities occurred in the lower reaches of the rivers where gravels were most concentrated, and in areas with low gradients. Spawning densities decreased as the gradient increased up each river. In the Cedar River, the preferred habitat was between the Camp 18 bridge and 1 mile upstream, and in the Rex River the preferred spawning habitat was downstream of Boulder Creek (Figure 3.5-6). In other watersheds, spawning sites are commonly associated with upwelling groundwater and nearby cover (Goetz 1989; Shepard 1985; McPhail and Baxter 1996). In the municipal watershed, temperature differentials indicative of upwelling were not found at redd sites during the 1992-1994 spawning seasons, and nearby cover did not appear to influence redd site selection (R2 Resource Consultants, in preparation).

In some years access to spawning habitat in the smaller tributaries was limited during fall low-flow conditions by subsurface flows near the outlets. Additionally, some Cedar River redds below the Camp 18 bridge and redds in lower Rex River were inundated by rising reservoir levels during late winter and early spring (see subsection entitled "Redd Inundation" below, and sections 3.2.4 and 4.5.5).

FRY EMERGENCE AND DOWNSTREAM MIGRATION

Trapping of newly emerged fry in the Rex and Cedar rivers was initiated during 1994 and 1995 (R2 Resource Consultants, in preparation), and was continued by City biologists during 1996 and 1997. The purpose of this study was to determine the timing of fry emergence and movement in the river system, and to contribute to the development of an index of reproductive success and population condition. A fyke net (8 ft long by 30 inches square), placed mid-stream, was used to collect downstream-moving fry at three sampling locations. One sampling location was in the Cedar River at Camp 18 and has been used in each of the sampling years. This site is located downstream of the majority of bull trout redds in the river. A second site in the Cedar River was just downstream of Bear Creek and was used only in 1995. It was located upstream of all observed bull trout redds. The third sampling site was used in 1994 and 1995 and was located in the mainstem of the Rex River. This site was located upstream of some observed redds and downstream of other observed redds, but it was above the area of the river subject to inundation from rising reservoir levels in the spring (Section 3.2.4).

The number of fry trapped in the nets varied among locations and years (Table 3.5-3). The greatest catch variability was between different years at a single location. In 1994, 193 fish were trapped at the Camp 18 sampling site on the Cedar River, but no fish were trapped there in 1996. Based on daily catch of bull trout fry per unit effort (the number of fry captured per hour of fishing multiplied by the average proportion of trap submerged), the period of most substantial fry emergence and movement in the Cedar River extends from late March through mid-May, with peak activity during April and early May (Figure 3.5-7). These observations are similar to those of fry movement in other systems (McPhail and Baxter 1996). Limited data from the Rex River indicates that the timing is somewhat later in the Rex River than in the Cedar, as fry emergence occurred from late April through early June, with peak activity during early May (Figure 3.5-8). However, results from the Rex River are less reliable than data from the Cedar River, because the trap was placed upstream of many redds inundated by spring reservoir levels.

The trap in the Cedar River located just downstream of Bear Creek was established to help clarify the upstream extent of spawning in the Cedar River (Figure 3.5-6). Only two newly emerged fry were captured at this location. This confirmed that bull trout spawning occurs upstream of this area, but suggests that the level of spawning activity may be relatively low compared to that the lower reaches of the river.

The numbers of fry trapped near Camp 18 in 1994 -1997 during the spring shows a roughly positive relationship to the numbers of redds counted the previous fall. The greatest number of fry were captured in 1994 (Table 3.5-3), and this was preceded in the fall by the largest redd count (see Spawning Distribution and Habitat Use above). In contrast, the lowest redd count was made in 1995, and this was followed in 1996 by the complete absence of captured fry. In addition, scouring flood flows occurred during both the 1995 spawning and 1995-96 incubation periods. These results suggest a highly variable reproductive success from year to year, depending on both the number of spawning fish and the flow conditions during incubation.

Fry size and development were fairly uniform during the fry-sampling period. All fry ranged in size from 24 to 40 mm, with most about 27-28 mm long. All fish had completely absorbed yolks, except for a few captured during March.

Figure 3.5-7. Bull trout fry catch per unit effort (CPUE) in the Cedar River near Camp 18 from 1994 through 1997.

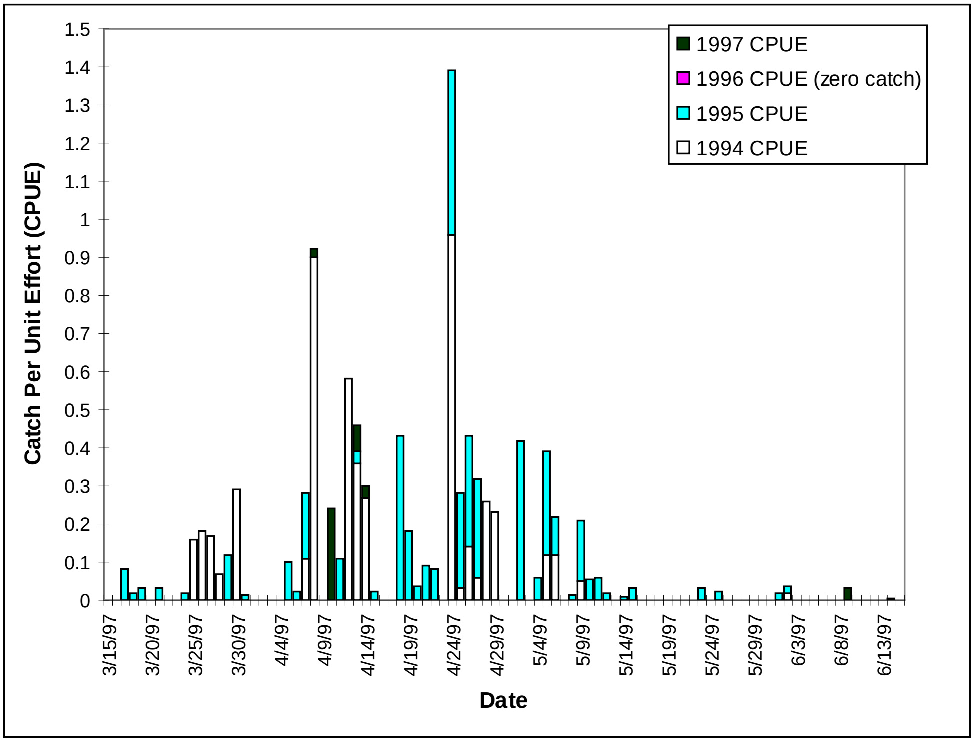
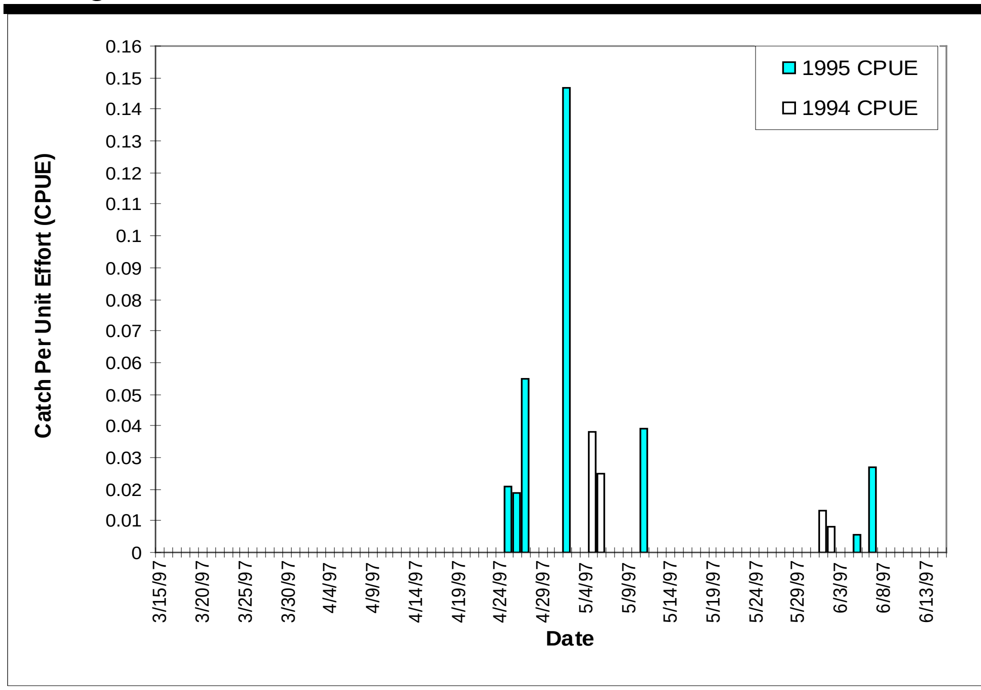


Figure 3.5-8. Bull trout fry CPUE in the Rex River below Cabin Creek during 1994 and 1995.



REDD INUNDATION

Inundation of bull trout redds by rising winter and spring reservoir levels occurs in the lower reaches of the tributaries of Chester Morse Lake. The probable result of this phenomenon is diminished water flow over and through the redds and death of some developing eggs or alevins. R2 Resource Consultants calculated that 22 out of 63 Cedar River bull trout redds (35 percent) in 1993 were at elevations that would be inundated at a reservoir level of 1,565 ft above sea level, the highest reservoir elevation that year (normal high pool elevation is generally 1,563 ft). However, 6 of the 22 bull trout redds were between thalweg elevations 1,554.57 ft and 1,560.96 ft and the other 16 were between thalweg elevations 1,560.96 ft and 1,565.06 ft. The R2 Resource Consultants report (in preparation) states that the 6 bull trout redds in the lower elevation range were all located downstream of the Camp 18 bridge. Independent field observations by City biologists indicate that the most upstream extent of reservoir inundation caused by a high pool reservoir level of about 1,563 ft is still several hundred feet below the Camp 18 Bridge. Thus, a high pool elevation of about 1,563 ft would have flooded 6 of the 63 redds (9.5 percent) in 1993. In the Rex River, at least 38 out of 40 (95 percent) of the observed bull trout redds in 1993 were located below the confluence of Boulder Creek at thalweg elevations less than 1,556.69 ft. Redds below this elevation would be inundated almost every year. The remaining two redds in the Rex River were located above elevation 1,563 ft (R2 Resource Consultants, in preparation).

The extent to which bull trout spawning habitat is inundated varies by different years depending on precipitation and operationally related fluctuations in the reservoir level (Section 2.2.4; Appendix 22, Figure 22-1). Bull trout redds in the Rex River are at greater risk from inundation than those in the Cedar River because many redds in the Rex River are located at lower elevations. Eggs or alevins in the Rex River also may be more at risk from inundation because they appear to emerge later in the spring when reservoir levels are historically highest.

The actual level of mortality caused by inundation of redds in the lower Rex and Cedar rivers is not known. It is particularly puzzling to biologists that such a high percentage of Rex River bull trout redds are built at elevations that have been annually inundated by Chester Morse Lake for almost 85 years. Severe mortality to eggs and alevins usually would be expected to exert a strong selective pressure against those bull trout spawning in the regularly inundated stream reaches. Inundation of salmonid redds is known to cause mortality in some reservoirs (Seattle City Light 1989). In Chester Morse Lake, one hypothesis is that the degree of impact is somewhat reduced by water upwelling through the spawning gravels in the inundated stream reaches. Upwelling in spawning gravels serves to aerate eggs and alevins and remove metabolic wastes. It is not known whether upwelling actually occurs in bull trout spawning areas in the lower Cedar or Rex rivers. However, the fact that regular inundation has been occurring for decades in much of the area in which bull trout spawn, suggests that there has been relatively little selection exerted on bull trout to avoid these areas. Furthermore, even if a high degree of mortality from inundation does occur, it is possible, even likely, that the limiting life stage for bull trout in the watershed is not spawning but juvenile rearing (Section 3.3.4; Foster Wheeler Env. Corp. 1995d).

In short, while the degree of impact from redd inundation is unknown, it is of concern. As a part of the Bull Trout Monitoring and Research Program (Section 4.5.4) a comprehensive research study will be conducted on potential mortality in inundated bull trout redds.

POTENTIAL BLOCKAGE OR IMPEDANCE OF SPAWNING MIGRATIONS

Bull trout spawning migrations may be blocked or impeded at the mouths of the rivers during exceptionally dry years with their concomitant extraordinarily low reservoir water levels (sections 3.2.4 and 4.5.6). Detailed bathymetry conducted at the edge of the Cedar and Rex river deltas determined that the slope of these areas is 14 and 17 percent, respectively. Exposure of several feet of these steeply sloped delta fans may present a barrier to migrating bull trout. These steeply sloped delta areas first begin to be exposed as reservoir levels drop below 1,540 ft in elevation. The degree of potential impact is smallest immediately below 1,540 ft, as only a short distance of steep gradient stream channel is exposed. However as the reservoir level drops below 1,535 ft, the steep channel gradients are believed to extend for sufficient length to potentially impede or block migration (R2 Resource Consultants, in preparation). Actual field observations of this phenomenon with low reservoir levels have never been made. Chester Morse Lake's minimum drawdown level is 1,532 ft, and under emergency conditions it can be lowered to 1,502 ft using the existing pumps (Section 2.2.3). An extended period of 60 consecutive days of lake levels below 1,540 ft occurred in 1987. The lowest level reached during the 1987 spawning period was 1,533 ft, when a portion of the steeply sloped delta fans was theoretically exposed. Chester Morse Lake water levels have not dropped below 1,540 ft since 1991. The potential for blockage or impedance of bull trout spawning migrations during infrequent periods of low reservoir levels will be thoroughly examined under the HCP Monitoring and Research Program as part of the Environmental Evaluation of the Cedar Permanent Dead Storage Project contained in Section 4.5.6.

ENTRAINMENT

There may be some loss to the bull trout population in the Chester Morse Lake/Masonry Pool system resulting from entrainment through the intakes of the Cedar Falls Hydroelectric Project at Masonry Dam and through the Overflow Dike into Masonry Pool. The issue of possible entrainment was raised at a City-sponsored Bull Trout Workshop on November 18, 1994 (Section 3.3.4) (Foster Wheeler Env. Corp. 1995d). To address this question, the City contracted Foster Wheeler Environmental Consultants to develop an estimate of entrainment based on a comparative literature search (Appendix 19). Foster Wheeler's study concluded that any potential loss of fish from the Chester Morse Lake/Masonry Pool system is likely having little effect on the reservoir's population. The study estimated that about 200 bull trout per year may be lost to entrainment through Masonry Dam, with a possible range of 10 fish to several hundred fish (Knutzen 1997).

Foster Wheeler considered the estimated 200 fish lost, or 6.4 percent of the estimated 3,100 bull trout in Chester Morse Lake, to be a sustainable loss, because any such entrainment has been ongoing for most of the twentieth century, and because only about 5 percent of the reservoir's bull trout population occurs in the Masonry Pool. In other systems, trout have been able to maintain stable population levels despite annual exploitation rates greater than 20 percent (Nehring and Anderson 1982). The health and long-term sustainability of the Chester Morse Lake bull trout population, in spite of entrainment, is further supported by the fact that losses to the population above Cedar Falls have always occurred, even before Masonry Dam was constructed. Historically, any trout or char in the upper Cedar River watershed that migrated downstream on its own volition or during storm events would have passed over Cedar Falls.

Entrainment losses from the Overflow Dike between Chester Morse Lake and Masonry Pool can occur whenever the reservoir level drops near or below 1,550 ft (the top of the Overflow Dike spillway), which occurs about 36 percent of the year (Section 2.2.4). At these lake levels the flow from Chester Morse Lake to Masonry Pool is primarily through a 6.5 ft diameter discharge pipe and onto a concrete energy dissipation block. It appears that some fish are likely injured or killed from passing through this Overflow Dike pipe, but solid conclusions cannot be drawn from available information (Knutzen 1997). Knutzen postulated that the fish population probably incurs much less damage from passing through the Overflow Dike than from entrainment at the Masonry Dam.

3.5.7 Pygmy Whitefish

BACKGROUND AND STATUS

The pygmy whitefish (*Prosopium coulteri*) (Figure 3.5-9) is a glacial relict species in the family Salmonidae. During the last ice age it was widely distributed across North America, but currently the species is found in small, isolated populations in deep, high-elevation lakes from the Columbia River basin in Washington, Montana, and British Columbia north to Alaska (Wydoski and Whitney 1979). Pygmy whitefish also occur in northern Asia in rivers flowing to the Arctic Ocean (Chereshnev and Skopets 1992), and a population outside of its typical distribution also exists in Lake Superior (Eschmeyer and Bailey 1955). The number of lakes that support pygmy whitefish is relatively uncertain, however, because the common sampling methods of gillnetting and angling are not efficient at capturing these small, deep-water fish.

The pygmy whitefish is not a federal listed or candidate species, but it is a Washington State Sensitive species (WDFW 1998b). In Washington State, pygmy whitefish are known to have previously occupied 15 lakes, although currently they are found in only 9 (Hallock, M., WDFW Freshwater Fish Division, January 21, 1998, personal communication). The species disappeared from 3 of the 15 lakes as a result of interactions with introduced fish species. Pygmy whitefish were extirpated from the other 3 lakes through fish removal efforts conducted prior to restocking these lakes with game fish. Pygmy whitefish populations are especially vulnerable to local extinction because recruitment of new fish is usually impossible among isolated lake systems.

Figure 3.5-9. A school of pygmy whitefish in the Cedar River upstream from Chester Morse Lake.

The pygmy whitefish population in the Chester Morse Lake/Masonry Pool system has benefited from the lack of development in the drainage basin, the absence of non-native fish species, and the preservation of high quality water. Pygmy whitefish are the most abundant salmonid in the lake, and they are one of the major prey items for the bull trout population (R2 Resource Consultants, in preparation).

Much of the information in this section regarding the status of pygmy whitefish in Chester Morse Lake was obtained from a recent study on resident fish habitat and populations in the upper Watershed (R2 Resource Consultants, in preparation). This study combined hydroacoustic surveys, gill net sampling, and stream surveys to estimate the pygmy whitefish population, to determine their distribution, and to gain an understanding of their life history. Additional information on the pygmy whitefish population was gathered from other fisheries studies in the Cedar River Watershed (Wyman 1975; EVS 1984), published literature, and field observations by City biologists.

LIFE HISTORY, AGE, AND GROWTH

In the Cedar River Watershed, pygmy whitefish are found in Chester Morse Lake and Masonry Pool. They are also found in some tributaries to Chester Morse Lake, although their use of the rivers and tributaries of the system appears to be limited to spawning. City biologists observed spawning migrations of pygmy whitefish in the Cedar River, Rex River, and Boulder Creek during early December. Pygmy whitefish are a relatively short-lived species. In Chester Morse Lake, the population is comprised mostly of fish in the age class 2+ and 3+ and a few fish in age class 4+ (R2 Resource Consultants, in preparation). In the Flathead Lake system in Montana, researchers observed that male pygmy whitefish generally live only to age 3, whereas some females reached age 5 (Weisel et al. 1973). They also noted that males generally mature at age 2 while females first mature at age 3, and likely spawn in consecutive years. The earlier maturation and shorter life spans of the males may be an evolutionary strategy that reduces the use of limited food resources and helps sustain pygmy whitefish stocks (Weisel et al. 1973).

Pygmy whitefish in Chester Morse Lake are the largest known pygmy whitefish in the world. They are larger than fish from other lakes in Washington State (Hallock, M., WDFW Freshwater Fish Division, January 21, 1998, personal communication), and they are larger than fish from Montana, Alaska, and northern Asia (Weisel et al. 1973; Chereshnev and Skopets 1992; Eschmeyer and Bailey 1954). The total length of fish in Chester Morse Lake ranged from 195 to 220 mm for age class 2+ fish (n=23), 208 to 216 mm for age class 3+ fish (n=10), and 210 to 246 mm for age class 4+ fish (n=2) (R2 Resource Consultants, in preparation). Known sizes of fish from other populations contain only one report of a fish larger than 200 mm. The greater body size of the Chester Morse Lake fish suggests that this is a relatively productive and unique stock.

ABUNDANCE

The population of pygmy whitefish in Chester Morse Lake was estimated to be approximately 51,000 fish, based on the results of hydroacoustic surveys (R2 Resource Consultants, in preparation). However, hydroacoustic techniques underestimate bottom-oriented fish populations, such as the pygmy whitefish. When fish are near the lake bottom the hydroacoustic signal is compromised by bottom noise.

Masonry Pool likely supports a lower density of pygmy whitefish than Chester Morse Lake, because Masonry Pool does not have deep-water habitat comparable to the deep areas of Chester Morse Lake. The population of pygmy whitefish in Masonry Pool is estimated to be less than 1 percent of the population in Chester Morse Lake, or approximately 300 fish (R2 Resource Consultants, in preparation).

REARING DISTRIBUTION AND HABITAT USE

In Chester Morse Lake all of the pygmy whitefish collected in gill nets were within approximately 1-10 ft from the bottom, and most were in water deeper than 100 ft (R2 Resource Consultants, in preparation). The fish appeared to remain near the bottom during the day and were more active at night.

Feeding habits of pygmy whitefish in Chester Morse Lake were determined by stomach content analysis (R2 Resource Consultants, in preparation). Their diet was found to be fairly consistent among spring, summer, and fall sampling events. In each season, the fish predominantly consumed benthic organisms that live in the deeper parts of the lake. The dipteran families Chironomidae and Ceratopogonidae were the most common food organisms and were present in more than 65 percent of all stomachs. Various small clams were the second most commonly consumed prey, and, depending on the season, appeared in 22-48 percent of all stomachs. Other less commonly consumed groups include zooplankton, benthic amphipods, and other insect taxa. Fish and terrestrial organisms were absent from the diet of pygmy whitefish, indicating their food source is primarily from the lake bottom, especially the deeper profundal zone.

Observations that Chester Morse Lake pygmy whitefish feed at or near the bottom are corroborated by research at Flathead Lake in Montana (Weisel et al. 1973). This same study also examined stomach contents in December during the pygmy whitefish spawning period. Results indicated that spawning fish in the creek actively fed on fish eggs and on chironomids and other insects.

SPAWNING DISTRIBUTION AND HABITAT USE

Little is known about pygmy whitefish spawning behavior, incubation, and early life history. In the Chester Morse Lake system, large aggregations of sexually mature fish move into the Cedar River, Rex River, and Boulder Creek during early December. Observations during these spawning migrations are limited, but City biologists observed pygmy whitefish in the Cedar River above Camp 18 as individuals or in groups ranging in size from 2 to approximately 1,000 fish during the first 2 weeks of December in 1996 and 1997. The fish were observed in pools, over pool tailouts, and in shallow riffles. Many of the groups of fish were actively swimming upstream while other groups appeared to be maintaining their position in the channel. In 1997, during the same period, fish were also observed in the lower Rex River and in lower Boulder Creek. No fish were observed in Cabin, Otter, McClellan, Green Point, Shotgun, or Lost (historic) creeks nor along the beach area of Lost (historic), Shotgun, or McClellan creeks.

It is presumed pygmy whitefish spawn by broadcasting their eggs on clean gravel (Wydoski and Whitney 1979). This is supported by observations by City biologists of pygmy whitefish eggs within the gravel of a shallow riffle upstream of Camp 18 in 1996. It is unknown if pygmy whitefish also spawn along the shoreline of the lake, but it is probable that this strategy is used by other populations that occur in lakes without surface water inlets (Hallock, M., WDFW Freshwater Fish Division, January 21, 1998, personal communication).

POTENTIAL EFFECTS OF RESERVOIR OPERATIONS ON PYGMY WHITEFISH

It is unknown if seasonal changes in lake levels from reservoir operations (sections 2.2.4, 3.2.4, and 4.5.6) significantly affect the pygmy whitefish population. Management of Chester Morse Lake can result in a maximum elevation change of 38 ft between maximum full pool and the gravity flow drawdown limit. The lake level of Masonry Pool can fluctuate 70 ft. At the higher lake levels, Masonry Pool and Chester Morse Lake join to form a single water body. At the lowest level, Masonry Pool is essentially a flowing channel. Because Masonry Pool supports such a low density of pygmy whitefish relative to Chester Morse Lake, the effect of such a drastic change in Masonry Pool is not expected to significantly affect the total pygmy whitefish population.

3.5.8 Sockeye Salmon

GENERAL DESCRIPTION

The sockeye salmon (*Oncorhynchus nerka*) (Figure 3.5-10) is a common and relatively well-studied species of the family Salmonidae. The sockeye is the third most abundant of the seven species of Pacific salmon and has been targeted in major commercial fisheries for most of the twentieth century[2]. Spawning populations of sockeye have been reported from the Sacramento River in the south to the rivers of Kotzebue Sound in the north, and east to basins that drain into the Sea of Okhotsk (Burgner 1991). Size at maturity varies considerably between and within populations of sockeye, with larger fish typically spending additional time at sea. The average weight of sockeye returning to the Cedar River is approximately 5.25 pounds (James M. Montgomery Inc. 1990).

The Washington Department of Fisheries et al. (1993) identified four populations of anadromous sockeye salmon in Puget Sound: one population in the Baker River and three populations that occur in the Lake Washington watershed (Cedar River, Issaquah/Bear Creek, and Lake Washington beach spawners). Hendry et al. (1996) used analysis of variation in allelic frequencies to suggest a slightly different population structure in Lake Washington. Their work suggests two subgroups in the Lake Washington watershed: a potentially native stock that spawns in Bear and Cottage Creeks at the north end of the system and a second stock derived from transplants of Baker River sockeye in the 1930s and 1940s that spawns in the Cedar River, Issaquah Creek, and on the beaches of Lake Washington. Within the Baker River sub-group, Issaquah Creek fish could be distinguished from Cedar River and lake spawning fish. However, allelic frequency was not sufficient to distinguish between Cedar River and lake-spawning fish. In a companion study, Hendry and Quinn (1996) were able to use morphological differences in spawning fish to distinguish between Cedar River and lake-spawning fish. The results of these studies could be used to support the hypothesis that the fish introduced from the Baker River are beginning to diverge into distinct sub-populations.

Although considered by the state as a wild stock (naturally reproducing), the Cedar River stock is not presently considered by NMFS to constitute an Evolutionarily Significant Unit (ESU) under the Endangered Species Act. However, the stock present in Bear Creek, a tributary to the Sammamish River, is potentially of native origin (Fed. Reg., Vol. 63, No. 46, pp. 11749-11771; Waples, 1998). The Bear Creek stock is considered by NMFS to be a provisional ESU, although it is not believed to be presently in danger of extinction nor likely to become endangered in the foreseeable future if present conditions continue (Fed. Reg., Vol. 63, No. 46, pp. 11749-11771).

The majority of sockeye returning to Lake Washington spawn in the Cedar River. The north Lake Washington sub-group also exhibits significant returns in most years. Returns to Issaquah Creek are typically lower than returns to the north-end tributaries. Lake spawners typically account for the smallest portion of the run, usually three orders of magnitude less than returns to the Cedar River (Hendry et al. 1996).

IMPORTANCE IN THE LAKE WASHINGTON WATERSHED

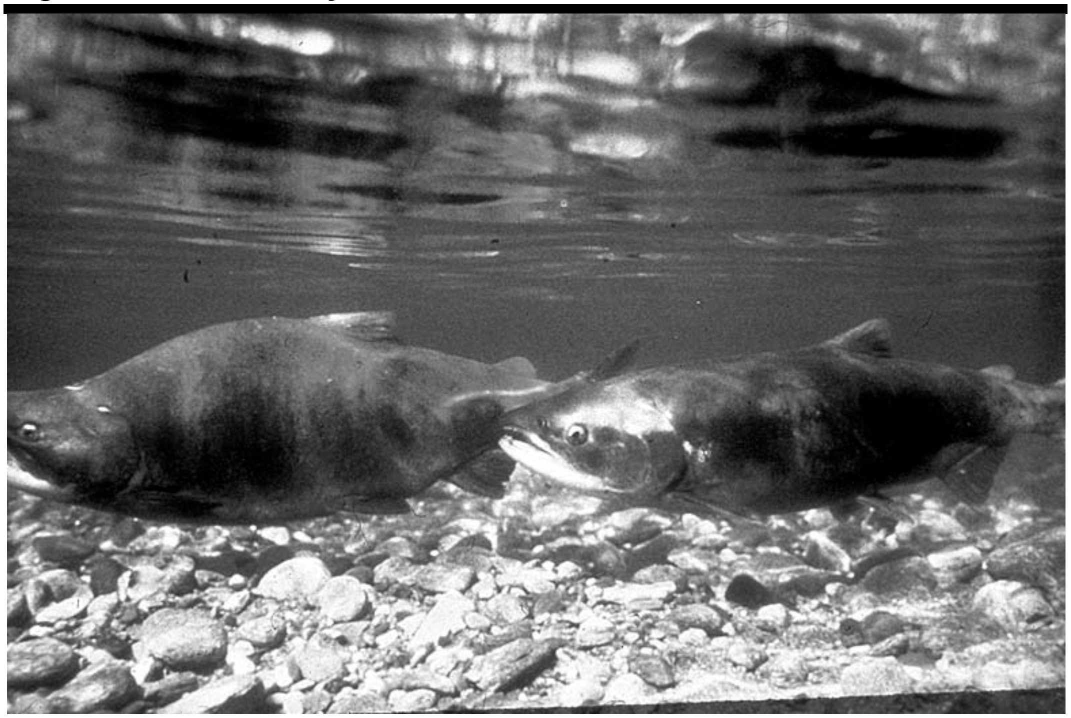
Sockeye are an important component of the Lake Washington ecosystem. Post-spawning salmon carcasses contribute nutrients to the biotic communities in streams and lakes (Kline et al. 1994; Bilby et al. 1996). Returning spawners excavate and, to some degree, redistribute significant amounts of gravel in spawning areas each year (Burgner 1991). A number of fish species in the system feed upon sockeye eggs and juveniles (Foerster

[2] Rainbow and cutthroat trout are now included in the genus *Oncorhynchus*. For convenience, we follow the convention of Groot and Margolis (1991) to maintain the common distinction between salmon and trout and do not include these two species when referring to the seven species of salmon in the Pacific basin.

1968; Stober and Hamalainen 1980; Beauchamp 1993; Tabor and Chan 1996). Birds and mammals scavenge on carcasses (Cederholm et al. 1989), and a number of bird species, such as dippers, kingfishers, and mergansers, feed on eggs and juvenile fish (Burgner 1991). During their extended rearing period in the lake, juvenile sockeye are important predators that consume significant amounts of zooplankton (Foerster 1968; Woodey 1972; Chigbu and Sibley 1994).

Sockeye salmon are the most numerous naturally reproducing salmonids in the basin and, in years of high abundance, the population has supported a significant Tribal treaty harvest and one of the largest sport fisheries in the state (Fresh 1994). The migration of sockeye through the fish ladder at the Ballard Locks attracts thousands of visitors each year. The observation of spawning sockeye in the Cedar River, Bear Creek, and Issaquah Creek has become a popular fall outdoor recreation activity for many people in the region.

Figure 3.5-10. Sockeye salmon.



LIFE HISTORY OVERVIEW

Sockeye salmon exhibit a typical salmon life history pattern that integrates anadromy (juveniles migrate to the ocean where they mature and return as adults to spawn in fresh water), homing (adults generally return to their natal streams to spawn), and semelparity (adults die after spawning once) (Section 3.2.3). Sockeye can also exhibit a resident life history that is similar to the typical pattern, but lacks the feature of anadromy (Burgner 1991). These resident sockeye are called kokanee. Although small numbers of sockeye in the Lake Washington Basin exhibit the resident life history pattern, including a population in Walsh Lake in the Cedar River Watershed (sections 3.6 and 3.2.4), the vast majority of the population is anadromous. Unlike any of the other species of Pacific salmon, juvenile sockeye rear primarily in freshwater lakes.

Adult sockeye salmon begin returning to the Lake Washington watershed through the Ballard Locks in late May with a peak migration in early July. By mid- to late August, essentially all fish have entered the lake (Warner, E., Muckleshoot Indian Tribe, 1998, personal communication). Once in the lake, the fish move into deep, cold areas below the thermocline. Adults will spend from 1 to 4 months in this region of the lake, where they undergo final sexual maturation (Parametrix, Inc. 1991). During this time, gametes mature and the outward appearance of the fish is dramatically transformed by the onset of secondary sexual characteristics. Most fish will move into tributary streams to spawn during the fall, but a relatively small proportion of the population will spawn in selected beach areas along the eastern shores of the lake and along the northern shoreline of Mercer Island (Map 2). The Cedar River supports the largest population of sockeye salmon in the Lake Washington Basin with significant numbers of fish also spawning in the Bear Creek subbasin, and in North Creek, Swamp Creek, and Issaquah Creek. Although there have been exceptions in some years, approximately 90 percent of the returning fish typically spawn in the Cedar River (James M. Montgomery, Inc. 1990).

Cedar River sockeye exhibit relatively protracted periods of spawning and incubation. Mature adults begin to enter the Cedar River in early September. Spawning activity begins to increase in mid-September and continues into January with a peak in mid- to late October (Cascades Environmental Services 1995). Each female selects a site for spawning, digs a redd, and deposits an average of 3,200 eggs. Males compete to court females and fertilize the eggs. After fertilization, the eggs are buried by the female, who guards the site until she dies several days to 2 weeks later. Alevins hatch from the eggs after 2 or 3 months and remain in the gravel for an additional 2-4 months, during which time they are sustained by their yolk sacs as they complete their development into free-swimming fry (Foerster 1968; James M. Montgomery, Inc. 1990).

Fry begin to emerge from the gravel in late January and continue through May, with a peak in late March and early April. Upon emergence, fry immediately begin migrating downstream. Most fry arrive at Lake Washington within 48 hours of emergence (Seiler and Kishimoto 1996). Most juvenile sockeye reside in the lake for approximately 12-14 months, then undergo the process of smoltification as they migrate out of the lake into salt water via the Lake Washington Ship Canal and the Ballard Locks. These migrating smolts move out of the lake and into Puget Sound between April and June (James M. Montgomery, Inc. 1990).

Once in the marine environment, Cedar River sockeye are thought to display distribution and migration patterns similar to those of other northeastern Pacific sockeye stocks as summarized by Burgner (1991). After leaving Puget Sound, subadult sockeye move north along the continental shelf, into the Gulf of Alaska, and then migrate south into the open ocean. Once they reach maturity, the adult fish return to near-shore waters and migrate south along the coastline to Puget Sound and back to Lake Washington. The majority of Lake Washington sockeye return after 2 years at sea, however, a significant proportion from any give year class may return after 3 years at sea. Typically, a very small portion of the population (<1 percent) returns after only 1 year at sea (WDFW 1997e).

HABITAT CHARACTERISTICS AND KEY FACTORS AFFECTING SURVIVAL

A number of factors can potentially affect the survival of Lake Washington sockeye salmon at various stages of their life history, including habitat loss and degradation resulting from a variety of land and water management practices (King County 1993); scour of incubating eggs and alevins during floods (Seiler and Kishimoto 1997); predation by native and non-native fish in the Cedar River and Lake Washington (Beauchamp 1993; Tabor and Chan 1996); food supplies in the lake (Beauchamp 1996); injury to smolts leaving the Lake via the Ballard Locks (Goetz et al. 1997); droughts; and unfavorable ocean conditions. As a result of the population's early run timing, harvest rates for Lake Washington sockeye are typically very low in the marine environment. Occasionally, early season harvests targeting up-river stocks of Fraser River Sockeye are permitted in north Puget Sound. This fishery must be carefully controlled to prevent unintentional over-harvest of Lake Washington sockeye (Warner, E., Muckleshoot Indian Tribe, 1998, personal communication). Although sport and Tribal harvests in Lake Washington are typically well controlled to ensure that adequate numbers of fish return to streams to spawn, Cedar River sockeye can be vulnerable to over-harvest, as demonstrated during the 1996 season when insufficient numbers of fish returned to meet escapement goals after substantial sport and Tribal harvests in the lake.

Clearly, there are a number of ways in which human activities have had impacts on sockeye in the Lake Washington Basin. But perhaps the most profound human impact on the aquatic ecosystem, the alteration of the basin's hydrologic pattern (Chrzastowski 1983), has been beneficial for anadromous sockeye salmon.

Hydrologic Reconfiguration of the Cedar River Basin

In its original configuration, Lake Washington drained south via the Black River near the present day site of the City of Renton (Section 3.2.5). The Black River continued south for a short distance and joined the Duwamish River en route to Puget Sound. The Cedar River flowed into the Black River approximately 0.4 miles downstream of the lake outlet, rather than directly into the lake as it does today. Large numbers of fish reportedly spawned in the Cedar River between its confluence with the Black River and the natural barrier formed by Lower Cedar Falls 34.2 miles upstream. The large numbers of spawning carcasses present in the river were considered a detriment to water quality by early settlers (Bagley 1929). This reference to large numbers of carcasses suggests the presence of mass spawning pink and chum salmon in addition to coho, chinook, and steelhead trout. There have been no reports of significant returns of pink and chum salmon to the Cedar River since that time. Juvenile pink and chum salmon migrate to salt water very shortly after emergence as very small fish. The likelihood that these small fish can successfully navigate through a large natural lake such as Lake Washington and arrive in the near-shore marine environment during the appropriate period in the spring is quite small. If populations of these species were present, they were apparently unable to adapt to the rapid hydrologic changes in the early twentieth century and are now extinct in the Lake Washington Basin. In contrast, sockeye salmon have established themselves in the Cedar River and flourished since the hydrologic changes.

Kokanee (resident sockeye) were thought to have been present in Lake Washington prior to the turn of the twentieth century, and, as mentioned above, are still present in isolated populations. Although it is possible that anadromous sockeye may also have been present in small numbers, their presence was not conclusively determined prior to the introduction of Baker River fish in the 1930s (U.S. Fish Commission 1897; Cobb 1916; Burgner 1991).

This apparent lack of substantial numbers of anadromous sockeye is perhaps not surprising when one considers the hydrology and ecology of Lake Washington and the Cedar River subbasin prior to the early twentieth century. The largest potential spawning area for sockeye in the Lake Washington watershed was located in the Cedar River. However, the Cedar River emptied into the Black River downstream of Lake Washington and therefore did not provide direct access for juvenile sockeye into Lake Washington for rearing. In addition, any adult sockeye that might have returned to the system would likely have been competing with large numbers of pink and chum salmon for spawning habitat.

When stream flows in the Cedar River were very high, the direction of flow in the Black River would apparently reverse and some Cedar River water would flow into the lake. During protracted periods of high run-off, the water level of Lake Washington would rise by as much as 7 ft (Chrzastowski 1983). While it is possible that some sockeye fry might have arrived at the lake when high water conditions occurred during the spring, these same conditions would have made it very difficult for outmigrating smolts to find their way out of the lake and into salt water. Also, conditions in Lake Washington were likely conducive to a kokanee life history pattern. On an annual basis, the water in the lake was replaced at approximately one-half the rate at which it is replaced today. The lake was likely very rich in nutrients and food for juvenile and adult kokanee. The highly favorable conditions for salmonid growth and survival in the lake, and the potential lack of a clear pathway for outmigrating smolts prior to the hydrologic changes completed early in1917, may have resulted in conditions that favored a resident life history strategy over an anadromous pattern.

By 1917, the hydrologic pattern of the Lake Washington Basin had been dramatically altered with the rerouting of the Cedar River directly into Lake Washington, the creation of a new outlet via the Ballard Locks, and the lowering of the mean water level in the lake by nearly 9 ft (Chrzastowski 1983). While this alteration likely had significant negative effects on some salmonid species, it created conditions under which anadromous sockeye salmon could flourish in the Cedar River. In contrast to the other anadromous salmonids in the watershed that rear as juveniles for extended periods in stream habitats (steelhead trout, coho salmon, and chinook salmon), sockeye move downstream immediately after emergence from the gravel and begin to take advantage of the comparatively vast rearing areas and abundant food resources offered by the lake.

In this new hydrologic configuration, sockeye fry produced in the Cedar River were provided with a very direct pathway to the lake and outmigrating smolts were provided with direct access to salt water. In only 30 years, the transplanted Baker River sockeye grew into a robust, naturally reproducing population from relatively small initial plantings. This rapid population growth over a limited number of years and the presence of substantial numbers of potentially native sockeye in the north Lake Washington tributaries are perhaps good indicators of the generally favorable environment for anadromous sockeye salmon provided by Lake Washington. However, many other effects of human settlement and development on this generally benign environment have also reduced the system's resilience and capacity to support anadromous fish, including sockeye.

Landsburg Diversion Dam

Since the early 1900s, the Landsburg Diversion Dam has blocked the migration of anadromous fish into approximately 17 stream miles of formerly accessible habitat between Landsburg and the natural barrier formed by Lower Cedar Falls. As previously mentioned, anadromous sockeye were not reported in the Cedar River prior to the alterations of the basin's hydrology and subsequent introduction of Baker River fish. However, large numbers of other species of salmon were present in this reach prior to the construction of the Landsburg Diversion Dam. Today, naturally reproducing sockeye are established in the lower river and every year significant numbers of adult fish migrate upstream as far as the migration barrier at Landsburg. Exclusion from the habitat upstream of the diversion limits the productive capacity and resiliency of the Cedar River sockeye population.

Stream Flow

Sockeye begin spawning in the early fall when stream flows often recede to their lowest levels of the year. During the last half of September and early October, the amount of spawning habitat available to sockeye can be limited by low stream flow. By mid- to late October, stream flow typically exceeds the levels that provide the maximum amount of spawning habitat for sockeye (Cascades Environmental Services 1991).

In the Cedar River, redd scour during flood events is thought to be a dominant factor controlling the survival of incubating eggs and alevins (Thorne and Ames 1987). Floodplain development, diking, bank armoring, and flood management practices have reduced the width of the functional stream channel and reduced the river's ability to spread into the floodplain and dissipate energy during high water events (Section 3.2.5). High flows are now confined within a relatively narrow corridor that has had the effects of increasing water velocity, transporting sediment, and subsequently increasing the frequency and degree of redd scour. Redd scour starts to occur when streamflow, as measured near the mouth of the river in Renton, exceeds approximately1800-2000 cfs. Scour rates increase quite rapidly as flows increase beyond this level (Cascades Environmental Services 1991; Seiler and Kishimoto 1997b). Redds located near the stream margins appear to be somewhat less vulnerable to scour than redds located in the center of the channel. However, the amount of spawning habitat available along the stream margins is relatively limited and generally available to spawning fish only at relatively high flow levels that exclude fish from much of the spawning habitat in mid-channel areas (Cascades Environmental Services 1991).

Recent information suggests that newly emerged sockeye fry can experience significant mortality during their 1-2-day migration downstream to Lake Washington. The survival of outmigrating fry appears to be higher during periods of elevated flows, and survival at similar flow levels can vary significantly from year to year (Seiler 1994, 1995; Seiler and Kishimoto 1996, 1997a). Other factors that may affect outmigrant survival include water clarity, temperature, and light intensity. One of the key mechanisms causing mortality during outmigration to the lake is hypothesized to be predation by sculpin (Tabor and Chan 1996). Sculpin population size may in turn be partially controlled by peak winter flood events.

Disease

Fish disease is not thought to be a major factor affecting the survival of Cedar River sockeye. However, a virus present in the population could potentially be of concern. Cedar River sockeye, like most sockeye, carry infectious hematopoietic necrosis virus (IHNV), the causative agent of the potentially fatal fish disease, infectious hematopoietic necrosis (IHN). This disease most often afflicts developing alevins and fry near and shortly after the time of emergence. Although disease outbreaks are rarely observed in natural conditions, the prevalence of IHNV in spawning adult Cedar River sockeye can be quite high (Amos et al. 1989) and has been detected in outmigrating wild sockeye fry from the Cedar River, indicating some mortality does occur as a result of IHN (Thomas, J., WDFW, 1998, personal communication). Since 1991, the WDFW Fish Health Division has been monitoring the incidence of IHNV in adult and juvenile fish in the Lake Washington Basin. This information will help improve our understanding of IHNV and its potential impacts on sockeye in Lake Washington and elsewhere in the region.

Juvenile Sockeye Survival in Lake Washington

In recent years, the survival of juvenile sockeye during their residence in Lake Washington has been significantly lower than in the past and is now quite low compared to other lakes that support sockeye. The factors causing this poor survival are unclear, and a number of hypotheses are being tested as part of the Lake Washington Ecological Studies (Section 2.3.8). The hypotheses may be grouped into two categories: (1) those that consider the effects of predators on juvenile salmon, and (2) those that consider trophic relationships and the carrying capacity of Lake Washington. There are a number of native and non-native predators that prey on juvenile salmon. However, attempts to understand the magnitude of predation, especially in offshore areas where young sockeye spend most of their lives, have not yet been successful. Sockeye smolts leaving Lake Washington are consistently among the largest in the world (Burgner 1991), which suggests that food may be quite abundant. However, recent bioenergetic modeling exercises indicate that, in years when planktivorous fish are abundant and zooplankton populations are relatively sparse, newly emerged sockeye fry that enter the lake during the early period of their migration could experience difficulty in securing an adequate food supply (Beauchamp 1996).

All sockeye smolts must migrate through the facilities at the Ballard Locks to reach Puget Sound. There are approximately five pathways for juvenile sockeye through the Ballard Locks. Under certain operating regimes, some of these pathways can injure or kill a portion of the fish migrating through the system. The ACOE, in cooperation with Lake Washington Ecological Studies program, is currently investigating the factors affecting the survival of outmigrating salmonids as they pass through the locks and testing methods to provide better downstream passage conditions (Goetz et al. 1997).

PRESENT STATUS IN THE LAKE WASHINGTON WATERSHED

After building to relatively robust levels in the 1960s and 1970s, the Lake Washington sockeye population has experienced a period of significant decline. The mean spawner return ratio during the last 11 brood years for which full return data is available is 0.79. This means that, on average, for each 100 fish that successfully spawns in the basin, only 79 fish have returned to spawn in the subsequent generation. Since record keeping began in 1967, the escapement goal for the system of 350,000 adult fish has been met or has been exceeded four times. Since the escapement goal was last achieved in 1988, the mean run size has been approximately 135,000 fish (WDFW 1997e). Washington Department of Fisheries et al. (1993) classify the Lake Washington sockeye population as depressed in the Cedar River and elsewhere in the basin.

Sockeye harvest opportunities have recently declined in frequency. In 8 of the 22 years between 1967 and 1988, Tribal and sport fishers harvested substantial numbers of sockeye in Lake Washington. Since 1988, Tribal and sport harvests have been conducted in Lake Washington only in 1996 (WDFW 1997e). Although the 1996 return of approximately 450,000 adult fish indicates that the system has retained some potential to produce significant numbers of fish, the general trend in the sockeye population remains one of relatively steep decline.

3.5.9 Coho Salmon

GENERAL DESCRIPTION

The coho salmon (*Oncorhynchus kisutch*) is one of the most popular sport fishes in the family Salmonidae. For most of the twentieth century it has been the mainstay of the average west coast salmon fishing trip. Coho salmon occur along the Pacific coast from Monterey Bay, California, northward to Point Hope, Alaska (Wydoski and Whitney 1979). The typical size of adult coho salmon in the Lake Washington Basin is between 4 and 7 pounds, although fish as large as 10 pounds have been observed. The largest coho in the state weighed 21 pounds, but in recent years large coho have been rare.

The average size of Puget Sound coho weighed at terminal landings has declined from an average of 8.8 pounds to 4.4 pounds from 1972 through 1993 (Bledsoe et al. 1989). It is not clear whether the reductions in size of Puget Sound coho salmon are a result of harvest practices, effects of fish culture, declining ocean productivity, density-dependent effects in the marine environment, or a combination of these factors. Wild spawners on the Cedar River range from 6 to 7 pounds and tend to be somewhat larger than Issaquah Creek Hatchery spawners, which range from 4 to 5 pounds (Antipa, B., WDFW, 1998, personal communication).

The coho population in the Lake Washington watershed is composed of both natural and hatchery subpopulations. Significant releases of hatchery yearlings were made from the early 1950s to the early 1970s, and regular fingerling and fry plants were made from the mid-1970s to the present. These releases have included coho salmon from the Minter, Green, and Skykomish rivers. There are also annual yearling releases from the Issaquah Creek Hatchery and the University of Washington.

Natural spawning populations of coho salmon are common in tributaries to Lake Washington and the Cedar River, including the Lake Walsh subbasin. The extent of historical and current mixing between hatchery coho and wild spawning populations, both spatially and temporally, is unknown. It is also unknown what straying rates occur from the on-station programs. As a result of this uncertainty, the two stocks in the Lake Washington Basin are designated as mixtures of native and non-native stocks (WDF et al. 1993).

The population of coho salmon in the Cedar River is somewhat unique and is defined by the timing of their spawning (late October to late February) as well as by their geographic separation from other significant coho streams in the drainage (WDF et al. 1993). It is unknown how spawner interchange or differences in off-station planting has influenced the Cedar River subpopulation. Until a genetic evaluation is made of the various subpopulations in the basin, designations between Cedar River spawners and other geographical groups are tentative.

IMPORTANCE IN THE LAKE WASHINGTON WATERSHED

Coho salmon are the second most abundant anadromous salmonid in the Lake Washington Basin next to sockeye salmon and contribute significantly to recreational catch in the Puget Sound region. Historically, coho have been harvested in sport, Tribal, and non-Indian commercial fisheries at relatively high rates in mixed stock fisheries in south Puget Sound. Once the fish enter the Shilshole Bay area, they are harvested at considerably lower rates (King County 1993). In years of high abundance, coho are also harvested by sport and Tribal fishers in Lake Washington.

As a result of the recreational popularity of coho, significant efforts have been made to supplement natural spawning populations. Coho fish culture practices are relatively well developed and provide an effective research and teaching tool for fisheries students at the University of Washington. Coho returns to the University of Washington hatchery average about 300 fish per year. Coho salmon are also widely used in elementary school classrooms to help teach students about aquatic organisms and their habitats in the Lake Washington Watershed.

Coho salmon are an important component of the Lake Washington ecosystem. Post-spawning carcasses contribute nutrients to the biotic communities in streams (Bilby et al. 1997). Because adult coho salmon generally spawn in small streams and tributaries, they are particularly important as a nutrient source for areas away from the mainstem of rivers. Carcasses are also important food for some species of mammals and birds (Cederholm et al. 1989). A number of fish species in the system feed upon coho eggs and young fish (Sanderock 1991). While rearing in the river, juvenile coho can consume significant amounts of aquatic insects and affect the distribution of other juvenile salmonids (Allee 1974).

LIFE HISTORY OVERVIEW

General Patterns

Like all eastern Pacific salmon, coho are anadromous (Section 3.2.3) and return to their natal streams to spawn. Coho salmon have one of the more predictable life histories of the Pacific salmon. Juveniles spend approximately 18 months in freshwater and go to sea after their second spring. After growing to maturity in the ocean, they return to their natal streams after 18 months. Coho salmon exhibit two alternative and less common life histories that vary from this pattern. In many populations, a small percentage of coho (typically males) return to spawn after only one summer in salt water. And in some populations, a significant percentage of juveniles spend an extra year rearing in fresh water (Sandercock 1991).

Upstream Migration and Spawning

Adult coho typically begin returning to Lake Washington through the Ballard Locks in late August and continue through early to mid-November (Warner, E., Muckleshoot

Indian Tribe, 1998, personal communication). Historically, a group of late-spawning coho entered the locks in January, but the current status of this late returning subpopulation is not known (Antipa, B., WDFW, 1998, personal communication). After entering Lake Washington, most coho will remain in the lake for several weeks if river flows are low.

When river flows rise with fall rain, coho begin to stage at the mouth of the Cedar River. If flows continue to stay high, coho will move upstream and locate preferred spawning habitat in small tributaries with adequate gravel. Cedar River coho are thought to begin spawning in mid-October and continue into February (Cascades Environmental Services 1991). Females select a site to spawn and dig the redd, typically about three square meters in size (Bell 1991). Fecundity depends on size, but the average Cedar River female coho lays approximately 3,200 eggs (Antipa, B., WDFW, 1998, personal communication). Males will compete with one another to court females and fertilize the eggs. After fertilization, the eggs are buried by the female, who will then guard the site until she dies 3-15 days later (Sandercock 1991).

Incubation and Early Rearing

The specific development rate and emergence timing of Cedar River coho has not been well documented. In most coho populations in this region, eggs hatch in about 2-3 months. Alevins remain in the gravel for an additional 2-3 months sustained by their yolk sac (Sandercock 1991). Coho fry probably begin to emerge from the gravel in early March and continue through late May with peak emergence in mid-April.

Juvenile coho rear in freshwater for at least 1 year. After a short period of schooling behavior immediately after emergence, Coho fry become very territorial and typically maintain distinct feeding territories during daylight hours (Sandercock 1991). Some coho may remain in the same tributary for a full year before they migrate downstream. Others may migrate downstream to larger streams or possibly to the lake to continue rearing prior to smoltification the following spring. However, the role of Lake Washington in juvenile coho rearing and migration is not well understood.

After rearing for approximately 1 year in fresh water, most juvenile coho undergo the process of smoltification and migrate to salt water. Specific size data on Cedar River coho smolts is not presently available. An extensive review by Sandercock (1991) suggests that coho smolt size at the time of migration does not vary greatly across the range of the species in North America and averages between 9 and 12 cm. Coho smolt outmigration has not been extensively documented, but typically occurs from late April through early July, with peak migration occurring in mid- to late May (Goetz et al. 1997).

Distribution in the Marine Environment

Once in the marine environment, coho from the Cedar River are assumed to undergo migrations similar to other coho from the Puget Sound region. This migration takes coho primarily northward into the coastal waters of British Columbia. Coho salmon released from Puget Sound are recovered in Washington (23 - 72 percent of the fish), British Columbia (27 - 74 percent of the fish), and Oregon (0 - 3 percent of the fish), with essentially no recoveries from Alaska or California (NMFS 1995).

HABITAT CHARACTERISTICS AND KEY FACTORS AFFECTING SURVIVAL

Coho salmon are native to the Cedar River and may have been present in Lake Washington tributaries prior to the turn of the twentieth century. However, it is unclear to what extent anadromy existed in Lake Washington and its tributaries as a result of the Lake's outlet connection to the Black River (Section 3.2.5). The response of the original population of coho salmon in the Cedar River to the rather dramatic changes in the hydrology of the Lake Watershed in the early twentieth century is not known. It is not clear to what degree the present Cedar River coho population is derived from the original population that eventually found their way back to the river. Nor is it known if strays from other nearby systems or from past plantings of hatchery fish have contributed significantly to the present day population. Regardless of the source, a naturally reproducing population of coho salmon has evidently persisted in this altered environment.

There are a number of factors that can potentially affect the survival of Lake Washington coho salmon at various stages of their life history. These factors occur in both the fresh water and marine environment. Factors in fresh water include habitat loss and degradation (Scott et al. 1986), predation, droughts, floods (NMFS 1995), and injury or mortality at the Ballard Locks (Goetz et al. 1997). Factors in the marine environment include predation, unfavorable ocean conditions, and harvest (NMFS 1995). Although sport and Tribal harvests in Lake Washington are typically well controlled to ensure an adequate escapement, there is little control over harvest of coho in Puget Sound and Canada.

Since 1916, the Landsburg Diversion Dam has blocked the migration of coho to approximately 17 miles of formerly accessible mainstem and associated tributary habitat within the Cedar River Municipal Watershed. Historical reports indicate that large numbers of salmon spawned in the river between Landsburg Dam and the historical barrier at Cedar Falls (Bagley 1929). In its original configuration, the Cedar River and its tributaries likely formed ideal habitat for coho salmon. It is likely that coho salmon were present in quite significant numbers. Coho salmon currently spawn in the Cedar River downstream of Landsburg Dam every year. If provided with passage over the diversion dam, these fish would likely colonize the habitat above Landsburg Dam quite rapidly, provided that other factors outside the watershed do not adversely affect their survival.

It is believed that redd scour during flood events is a dominant factor controlling the survival of species such as sockeye that spawn in the mainstem Cedar River (Thorne and Ames 1987). Because coho salmon spawn principally in smaller streams and tributaries to the Cedar, mainstem redd scour does not significantly affect coho production. However, urbanization below Landsburg Dam has had significant impacts in smaller tributaries entering the Cedar River. These impacts include sedimentation resulting from urban development in upstream plateau areas and reduction in the complexity of stream channels, riparian areas, and wetlands (King County 1993). Other areas have been modified to pass higher peak flows during storm run-off and have resulted in significant bed and bank scour and channel shifting (WDFW 1993; NMFS 1995). These factors have significantly altered spawning gravel quality and stability and calm water areas used by juveniles for refuge during flood events.

Low summer base-flow conditions can have significant effects on species like coho that rear in the river for an extended period. During low flow periods, juvenile fish can be stressed by factors such as high water temperatures and crowding, which in turn can increase rates of disease, competition, and predation (Zillges 1977; Baranski 1989). According to an extensive, collaborative instream flow study conducted by Cascades Environmental Services (1991), flows in the Cedar River typically exceed levels required to produce maximum coho rearing habitat except from mid-July to mid-September. During this period, flows typically provide approximately 95 percent of the maximum rearing habitat for juvenile coho (Cascades Environmental Services 1991). Although these summer base flows provide substantial levels of habitat, they are typically lower than pre-diversion flows (King County 1993). These reduced flows, in addition to extensive riparian clearing, increased impervious surfaces, reduced amounts of large woody debris, increased sedimentation, and channel confinement, have reduced channel complexity and pool habitat and caused a decline in the quality of coho summer rearing habitat in the Cedar River basin downstream of Landsburg Diversion Dam. Urban development in the Lake Washington Basin has also changed the structure of fish communities. The typical native Puget Sound fish community, with a diverse assemblage of salmonids and non-salmonids, is replaced with a less diverse species assemblage in which cutthroat trout predominate (Scott et al. 1986).

Similar to other salmonids in the Lake Washington watershed, coho must migrate through the facilities at the Ballard Locks to reach Puget Sound. Some of the pathways through the locks can injure or kill a portion of the juvenile fish migrating through the facility. The degree to which migrants are injured has not been well quantified (Goetz, F., ACOE, 1998, personal communication). In an effort to determine the extent of the problem and identify improvements, the ACOE is currently analyzing the factors affecting the survival of outmigrating salmonids and is beginning to develop measures to improve downstream migrant survival (Goetz et al. 1997).

PRESENT STATUS IN THE LAKE WASHINGTON WATERSHED

With some exceptions, coho populations in the Lake Washington Basin have undergone a significant decline. Coho escapement peaked at over 30,000 fish in 1970, but declined to less than 2,000 fish in 1992 (Fresh 1994; King County 1993). The escapement goal for Lake Washington coho is 15,000 fish. Based on available habitat, coho returns to the Cedar River are estimated to be usually 12-15 percent of the total return to the Lake Washington Basin (King County 1993). Therefore, recent returns of approximately 2,000 coho represent a run of only 270 fish to the Cedar River. Although the status of Cedar River coho salmon was determined to be healthy in 1992 (WDFW et al. 1993), this assessment acknowledged that the stock would fall into the depressed classification if future returns similar to those in 1991 were observed. As a result of the continuation of the downward population trend (Fresh 1994; King County 1993), coho salmon are now considered depressed in the Cedar River and elsewhere in the Lake Washington Basin.

With continued low returns of coho salmon over the past 7 years, harvests in the Lake Washington Basin and the Cedar River have declined to almost nil. Recreational fishing on the Cedar River is currently closed and is not expected to fully reopen until significant improvements in returns of all anadromous salmonids are reported. The current outlook for the population is one of continued decline.

In response to a petition to list coho salmon under the Endangered Species Act, NMFS (1995) completed a comprehensive status review of coho salmon along the west coast of the United States. Within this range, the status review identified six Evolutionarily Significant Units (ESUs) of coho salmon, each of which contains numerous spawning populations. Because coho from Puget Sound and the Strait of Georgia formed a coherent genetic cluster, it was determined that this population was unique. This population includes coho from Lake Washington and the Cedar River. In comparison to other populations along the California and Oregon coasts, NMFS determined that coho salmon in Puget Sound and the Strait of Georgia were generally stable and a listing was not warranted. However, because of limited information regarding the health of this population and definitive information on the risks to naturally reproducing fish, NMFS decided to add the Puget Sound/Strait of Georgia population to the federal list of candidates for threatened and endangered species. Upon reevaluation at any time, NMFS may reconsider the present candidate listing and propose to list the Puget Sound/Strait of Georgia population as threatened or endangered.

3.5.10 Chinook Salmon

GENERAL DESCRIPTION

The chinook salmon (*Oncorhynchus tshawytscha*) (Figure 3.5-11) is the largest of the seven species of Pacific salmon. Mature adults can reach weights in excess of 40 kg. Chinook are the least numerous of the five Pacific salmon species that occur in North America. In the eastern Pacific, spawning populations range from the central coast of California, north to the drainages of Kotzebue Sound. In the western Pacific, the species is somewhat less numerous and ranges from the Anadyr River, which drains into the northern Bering Sea, south to the island of Hokkaido.

Chinook salmon have been commercially harvested since the mid-nineteenth century. They have been highly valued by indigenous peoples for thousands of years. Today, they are also a highly prized sport fish throughout their range in North America. Chinook is the only species of salmon that has been successfully introduced in the southern hemisphere. Naturally reproducing populations have become established in New Zealand from introductions of North American chinook in the early part of the twentieth century (Healey 1991).

Individual spawning populations of chinook salmon tend to be relatively small, typically not more than a few tens of thousands. Healey (1982) reports that 80 percent of the chinook populations in British Columbia average fewer than 1,000 spawners. Larger river systems tend to support the largest populations (Healey 1991).

According to WDF et al. (1993), there are 26 stocks of chinook salmon in Puget Sound. At the time of their report, the authors classified the population status of approximately half of the stocks as depressed. However, since that time, there has been a sharp decline in the abundance of Puget Sound chinook, and nearly all naturally reproducing populations in the area are now considered depressed (Johnson et al. 1997; Smith, C., WDFW, 1998, personal communication).

Three stocks of chinook are present in the Lake Washington Watershed: (1) the Issaquah Creek stock, a composite population that is at least partially sustained by production from the Issaquah Hatchery; (2) the Cedar River stock, classified as native/wild; and (3) the north Lake Washington tributary stock also classified as native/wild. Annual counts of spawners for the period from 1989 to 1996 averaged approximately 1,600 fish in Issaquah Creek, 420 fish in the Cedar River, and 285 fish in the north Lake Washington tributaries (Smith, C., WDFW, 1998, personal communication). Recent genetic analyses indicate that Cedar River chinook are clearly members of the South Puget Sound, Hood Canal & Snohomish Summer/Fall chinook Genetic Diversity Unit described by Marshall et al. (1995). They are closely associated with the Green River Hatchery population but are distinct from this population and all other populations within the Genetic Diversity Unit. The degree to which the present Cedar River population has been affected by past interbreeding with hatchery fish is not known (WDF et al. 1993; Marshall, A., WDFW, 1998, personal communication).

Figure 3.5-11. Chinook salmon.

The Lake Washington watershed has a long history of being stocked with hatchery-reared salmonids (Ajwani 1956). Today, the majority of chinook salmon returning to the basin originate from the Issaquah and University of Washington hatcheries. Hatchery-reared chinook were planted in the Lake Washington Basin as early as 1914 (Darwin 1916). Ajwani (1956) reported extensive plantings of Issaquah and Green River hatchery chinook into Cedar River during the period from 1943 to 1954. According to a 1948 WDF report, salmon returns to the Cedar River were at one time negligible, but were significantly enhanced by plantings from the Issaquah and Green River hatcheries (WDF 1948). Like many early artificial production programs, the effectiveness of this planting program was not rigorously monitored. Therefore, it is difficult to confirm the former status of salmon in the Cedar River. Currently, there are no releases of hatchery chinook into the Cedar River.

Puget Sound chinook salmon, including the populations in the Lake Washington Basin, were recommended for listing as threatened under the federal Endangered Species Act on February 26, 1998 (Fed. Reg. Vol. 63, No. 45, March 9, 1998).

IMPORTANCE IN THE LAKE WASHINGTON WATERSHED

Of the three species of salmon returning to Lake Washington, chinook salmon are the least numerous. However, because of their large size and high quality flesh, they are highly prized by tribal, sport, and commercial fishers. Anecdotal information suggests indigenous people may have harvested spring-run chinook, which are not present in the river today. Precise chinook harvest information is not available for Cedar River chinook. However, harvest data is available for the nearby Green River Hatchery chinook and is likely representative of harvest patterns for Cedar River chinook. From 1985 to 1994, approximately 44.9 percent of the harvest of Green River Hatchery chinook occurred off the coast of Canada, 42.5 percent occurred in Puget Sound sport and net fisheries, 11.7 percent in commercial troll fisheries off the coast of Washington and Oregon, and 0.3 percent of the harvest was taken in Alaskan waters (Pacific Salmon Commission 1996). Wild chinook are not typically targeted for harvest in Lake Washington.

Although the population of Lake Washington chinook salmon is typically two orders of magnitude smaller than the sockeye salmon population, it is an important component of the ecosystem. Post-spawning carcasses contribute nutrients to the biotic communities in streams (Bilby et al. 1997). A number of fish species in the system feed upon chinook eggs and juveniles (Healey 1991). Birds and mammals scavenge on carcasses and some species also feed on chinook eggs and juveniles (Cederholm 1989). Additionally, while rearing in the river, juvenile chinook can consume significant amounts of aquatic insects and affect the distribution of other juvenile salmonids (Chapman and Bjornn 1969).

Chinook salmon are viewed by visitors to the Ballard Locks fish ladder. Chinook salmon returning to Issaquah Creek downstream of the Issaquah hatchery offer an unusual and exceptional opportunity to view large fish spawning in a very small stream. The annual return of these fish is the centerpiece of the Issaquah Salmon Days Festival, which attracts thousands of visitors each year.

LIFE HISTORY

General Patterns

Like all eastern Pacific salmon, chinook are anadromous (Section 3.2.3), they return to their natal streams to spawn, and they are semelparous (die after spawning). In an extensive review of the literature, Healey (1991) used differences in life history patterns to divide eastern Pacific chinook salmon into two broad races: stream-type populations and ocean-type populations. While there is significant variation in specific life history patterns between and within stocks in each race, it is possible to discern broad, general patterns unique to each race. In North America, spawning populations of stream-type chinook are predominant north of latitude 56°N and in headwater areas of large river systems throughout the species' range. Ocean-type populations predominate south of latitude 56°N, except in headwater areas of large river systems. Table 3.5-4 summarizes the key life history attributes of each race. Note that stocks in the extreme south and north of the chinook's range may depart somewhat from this general model (Kjelson et al. 1982; Hallock and Fry 1967; Yancey and Thorsteinson 1963).

Table 3.5-4. Comparison of the life history characteristics of stream-type and ocean-type races of eastern Pacific chinook salmon (summarized from Healey 1991).

Life History Stage	Stream-type	Ocean-type
Spawning migration	Enter rivers in spring and early summer and may hold in fresh water for up to several months before spawning.	Enter fresh water in summer and fall and spawn shortly after entry into fresh water.
Spawning	Spawn in summer and fall.	Spawn in fall and early winter.
Juvenile rearing	Rear in fresh water for at least one full year. Move through the estuary fairly quickly as yearling smolts and into near-shore areas of the marine environment.	May move directly downstream to estuary immediately after emergence in the spring; or may rear in streams for up to three months. Rear in estuary for up to several months before dispersing into near-shore areas of marine environment.
Adults at sea	Move rather quickly through the near-shore areas and into the open ocean where they tend to exhibit extensive migrations in the North Pacific Ocean.	Tend to remain in continental shelf waters and typically range less than 1,000 km from natal stream.

Cedar River chinook appear to be relatively well-matched with the description for ocean-type chinook. Their natal stream is located well south of 56° N, but is still within the central portion of the range of eastern Pacific chinook populations. Adult chinook enter Lake Washington through the Ballard Locks from late June through September with a peak in late August (Warner 1998). They spawn from early to mid-September through mid- to late November with a peak in early to mid-October (Cascades Environmental Services 1995; WDF et al. 1993). Although extensive surveys have not been conducted, juvenile chinook have not typically been found in the Cedar River after mid-summer.

Spawning populations of ocean-type chinook are not commonly found above large natural lakes. Although there are a few examples of chinook spawning upstream of natural lakes in this region (e.g. Nanaimo River, Vancouver Island), most of these populations are thought to exhibit the stream-type life history and use the lakes primarily as over-wintering habitat (Carl and Healey 1984; Healey 1980, 1982). The position of Lake Washington between the Cedar River and the marine environment raises some interesting questions regarding the ocean-type life history pattern and is discussed later.

Spawning

Chinook spawning behavior is similar to that of other salmonids (Section 3.2.3). The female selects an appropriate spawning location over gravel and small cobble substrate where she excavates the redd. After spawning, females have been reported to remain on the redd from 4 to 26 days until they die or become too weak to hold in the current (Neilson and Green 1981; Neilson and Banford 1983). During this period, females will vigorously defend the redd against the spawning activity of newly arriving fish.

Fecundity is quite variable within and between populations of chinook salmon. Fecundity increases with fish size and generally increases from south to north across the range of the species. Reported fecundity values range from approximately 4,000 eggs per female for small fish from southern populations, to over 14,000 eggs per female for large fish in northwest Alaska (Healey 1991). Fecundity data is not available for the Cedar River stock, however, fecundity for chinook returning to the nearby Issaquah Hatchery in 1996 and 1997 averaged approximately 4,400 eggs per female (Antipa, B., WDFW, 1998, personal communication).

Incubation and Early Rearing

Chinook eggs in this region typically hatch 2 or 3 months after fertilization. The larval fish, or alevins, remain in the gravel for an additional 2 or 3 months, then emerge into the stream as free-swimming fry. There is little data on the precise development rate and emergence timing of Cedar River chinook. In the Cedar River, fry probably begin to emerge in February and continue through March and perhaps April.

Chinook fry typically emerge at night and tend to exhibit an immediate downstream dispersal (Reimers 1971; Healey 1980; Kjelson et al. 1982). Within the ocean-type race, Healey (1991) distinguishes two life history variations: (1) fry that emerge from the gravel, disperse downstream to the estuary in a matter of hours or days where they then rear for an extended period; and (2) fry that emerge, disperse a shorter distance downstream, then stop and rear in the river for up to 3 months before migrating downstream to the estuary for another period of extended rearing. In several well-studied rivers in southern British Columbia, the movement of newly emerged fry to the estuary typically occurs from early March through early May. A second migration of fry that have reared in the river and are approximately twice the size of the early migrants occurs from mid-May to mid-June (Healey 1991). The degree to which Cedar River chinook exhibit these two alternative behaviors at emergence is not known. In addition, the distribution and behavior of chinook fry in Lake Washington and the role that the lake plays as a rearing area and migration corridor are not well understood.

Distribution in the Marine Environment

Healey (1991) cites a large number of studies that have reported the importance of estuaries as rearing habitat for ocean-type chinook. The behavior and distribution of juvenile Cedar River chinook, after they have migrated through the Ballard Locks and into salt water, has not been studied.

No data are available on the specific distribution of Cedar River chinook in Puget Sound or the North Pacific. However, harvest data for the Green Hatchery stock indicate that nearly all fish that are taken in sport and commercial fisheries are harvested off British Columbia, the coast of Washington, and in Puget Sound. Less than 1 percent of the fish are harvested off the coast of Alaska (Pacific Salmon Commission 1996). This information suggests that the ocean distribution of Cedar River chinook is likely similar to that described by Healey (1991) for ocean-type populations in this region.

There is little information on the specific age structure of returning adult Cedar River chinook. However, age at return is thought to be similar to the Green River population. While age at return can vary considerably between brood years, the average age at return for the 1990, 1991, and 1992 brood years at the Green River Hatchery counting from fertilization to subsequent spawning is: 4 percent 2-year old fish, 18 percent 3-year old fish, 65 percent 4-year old fish, and 13 percent 5-year old fish (Kimble, M., WDFW, 1998, personal communication).

HABITAT CHARACTERISTICS AND KEY FACTORS AFFECTING SURVIVAL

Stouder et al. (1997) discuss the complexity of identifying the factors contributing to the decline of Pacific salmon in the Northwest. They point out the need to consider the cumulative effects of small incremental changes in habitat and population structure, the potential synergistic effects of seemingly unrelated factors, the far-ranging nature of individuals in the populations, and potentially profound and uncontrollable cyclic changes in the marine environment. Further, all of these issues must be considered in the context of very complicated governance and regulatory structures. Faced with such a complex system, it is helpful to view the factors that affect salmon survival in a relatively broad context.

There are a number of factors that have affected the survival of Cedar River chinook salmon, including loss and degradation of stream habitat resulting from a variety of land and water management practices; predation by native and introduced species in the river and lake; injury to juvenile fish exiting the lake via the Ballard Locks; droughts; floods; over-harvest; and unfavorable ocean conditions. All of these effects should be viewed in the context of what may have been the most significant single anthropogenic effect on the ecosystem, the alteration of the basin's hydrologic configuration.

Hydrologic Reconfiguration of the Cedar River Basin

At about the turn of the twentieth century, early settlers reported the presence of large numbers of spawning salmon and carcasses throughout the Cedar River from near its confluence with the Black River, upstream to Cedar Falls (Bagley 1929). At this time, the Cedar joined the Black River just downstream from the outlet of Lake Washington. The Black flowed a short distance into the Duwamish River and the Duwamish then flowed another short distance into an extensive estuary (Map 2; Section 3.2.5). In this configuration, the Cedar River likely formed ideal habitat for ocean-type chinook salmon. Although reports of early observations of salmon in the river did not differentiate between species, significant numbers of chinook salmon were undoubtedly present in the Cedar River in its original configuration.

Between 1912 and 1917, the hydrology of the Cedar River and Lake Washington was dramatically altered when the Cedar was rerouted into the lake and the outlet of the lake was rerouted through the Ballard Locks to Salmon Bay. The response of the original population of chinook salmon in the Cedar River to these changes is not known. It is not clear to what degree the present chinook population is derived from members of the original population that eventually found their way back to the river. Nor is it known if strays from other nearby systems, or past plantings of hatchery fish, have contributed significantly to the present day population. Regardless of the source, a naturally reproducing population of chinook salmon has persisted in this highly altered environment. Between 1967 and 1991 annual counts of adult chinook salmon in the Cedar River ranged from a low of 488 to a high of 1,745 fish (WDF et al. 1993).

The effects on Cedar River chinook of rerouting the river into Lake Washington are difficult to ascertain but potentially quite profound. The lake provides a much different migration environment for recently emerged fry than the original river environment. Although the lake could potentially provide rearing habitat for newly emerged chinook fry, it is not clear to what degree ocean-type chinook possess the adaptive capacity to make use of a lake-rearing environment. It is also difficult to quantify the quality of this environment, which has been subjected to extensive shoreline development and is home to a host of introduced species that can prey on young chinook. The requirement for young ocean-type chinook to migrate through Lake Washington could limit the productive capacity of the population.

The highly modified environment at the marine-freshwater interface downstream of the Ballard Locks creates an additional puzzle. This environment is much different than the natural estuary that was present at the mouth of the Duwamish River. Numerous sources as cited by Healey (1991) have reported on the importance of estuarine rearing for juvenile ocean-type chinook salmon. The behavior, growth, and survival of juvenile ocean-type juvenile chinook in the ship canal downstream of the Ballard Locks has not been well studied. However, it seems clear that this environment provides much less favorable conditions than the original estuary at the mouth of the Duwamish River.

The effects of the alterations in the lake and estuary on adult chinook are also unclear. For example, the relatively warm surface waters of the lake might, in some years, have an effect on upstream migration at the locks and perhaps in the lake itself. However, because adult chinook primarily use these environments for migration and are generally more resilient and much less vulnerable to predation than juvenile fish, the effects of these habitat alterations are perhaps less significant for adult fish than for juvenile fish.

Landsburg Diversion Dam

The Landsburg Diversion Dam is a run-of-the-river dam that was built near the turn of the century to serve as the intake point for the City's municipal water supply system. The dam is located on the Cedar River 21.8 miles upstream from Lake Washington and has excluded chinook salmon and other anadromous fish from 17 stream miles of formerly accessible habitat between Landsburg and the natural migration barrier formed by Lower Cedar Falls. This loss of spawning, incubation, and rearing habitat has limited the productive capacity of the chinook salmon. However, ascertaining the actual magnitude of the lost productive capacity for the population is complicated by effects of the hydrological reconfigurations discussed above.

Although the aquatic habitat in the area between Lower Cedar Falls and Landsburg Dam was degraded by extensive timber harvest and other land use practices early in the twentieth century, much of the area has recovered to a substantial degree. Consequently, this portion of the watershed offers some of the best fish habitat in the Lake Washington Basin. With provisions for sufficient releases of water from upstream storage facilities to meet instream flows requirements, this part of the watershed has the potential to provide excellent habitat for salmonids. A robust population of rainbow trout, thought to be derived from the original stock of steelhead present before the construction of the diversion dam, currently occupies this habitat (Section 3.5.11). There are relatively large inputs of high quality ground water throughout this reach. Erosion and sedimentation are largely in balance with other natural processes. Riparian zones are largely intact, and much of the stream channel is shaded by mature conifers. Chinook salmon currently spawn and, to an unknown degree, rear throughout the 21.8 miles of river below the Landsburg Diversion Dam. Provided that downstream factors do not seriously impair the productive capacity of the population, these fish would likely colonize the habitat upstream of Landsburg quite rapidly if allowed to pass above the dam.

Stream Flow

Streamflow represents a very important factor in the quality of habitat for aquatic life in the Cedar River, and particularly for the four anadromous fish species found there. The City's water supply and hydroelectric power generating operations on the river (Section 2.2.3) can affect the total flow volume and the rate of change in those volumes. Although juvenile chinook do not rear for extended periods in the river like steelhead or coho, flows in the mainstem are likely an important consideration for newly emerged fry and are certainly important for chinook spawning and incubation.

Most chinook salmon typically spawn in medium to large size rivers. They have been reported to spawn in habitats from small tributaries 3 meters wide, to the mainstem of the largest rivers emptying into the eastern Pacific Ocean. There does not appear to be a distinction between the physical spawning habitat preferences of stream- and ocean-type races of fish. Data from a number of different studies suggest that chinook salmon will spawn at depths of 10-700 cm and in water velocities of 10-189 centimeters per second (cm/s). However, these same studies report much narrower ranges of 30-56 cm for mean spawning depth and 40-61 cm/s mean water velocity (Healey 1991). As part of the IFIM study discussed in Section 3.3.2, Cascades Environmental Services (1991) established a preferred depth range of 30-104 cm and a preferred water velocity range of 30-107 cm/s for spawning chinook salmon in the Cedar River. Egg deposition has been reported at depths from 10 to 80 cm under the surface of the gravel, with deeper redds typically associated with slower water velocities (Briggs, 1953; Neilson and Banford 1983; Chapman et al. 1986).

Chinook begin spawning in the early fall, when stream flows in the Cedar are often at their lowest levels of the year. During the last half of September and early October, the amount of spawning habitat available to chinook can be limited by low stream flow. By mid- to late October, stream flows typically exceed the levels that provide the maximum amount of spawning habitat for chinook (Cascades Environmental Services 1991). In the Cedar, like many systems that support both sockeye and chinook salmon, spawning sockeye are present in large numbers during the entire time that chinook spawn. Chinook tend to spawn in deeper, swifter water, in larger substrate, and typically bury their eggs deeper than sockeye. While it is not presently considered a major controlling factor, the effects of the overlap between these two species on chinook spawning and incubation success in the Cedar River is not known.

Because chinook tend to spawn in deeper areas of the river, their redds are perhaps somewhat less vulnerable to dewatering than those of other salmonids spawning in the Cedar. However, chinook redds can become vulnerable to dewatering if periods of very low flow occur during incubation. Alevins are much more vulnerable to damage during dewatering than eggs (Becker et al. 1982, 1983).

There is little quantitative data on the effects of floods on chinook incubation survival in the Cedar River. Because chinook redds tend to be constructed in larger substrate and with deeper egg pockets than the redds of other species of salmon, they are perhaps somewhat less sensitive to scour during high flow events. However, major flood events on the Cedar likely cause significant mortality of incubating chinook. In the lower river, human development in the floodplain, diking, bank armoring, and flood management practices have reduced the width of the functional stream channel and reduced the river's ability to spread into the floodplain and dissipate energy during high water events (King County 1993; Section 3.2.5). High flows are now confined within a relatively narrow corridor, which increases water velocity, sediment transport, and subsequently increases the frequency and degree of redd scour. This situation has been further aggravated by the removal of forest cover and by increases in impervious surfaces in the lower watershed, which can increase the amplitude of high run-off events.

Thoughtful water management practices can help to reduce flood peaks and frequency. However, in the Cedar River, water storage facilities only capture water from the upper 43 percent of the basin, leaving flows in the lower 57 percent unregulated. In addition, storage facilities in the upper basin have a relatively limited storage capacity. Although water management activities can help to reduce the magnitude of flood events and, to a limited degree, decrease the frequency of such events, the facilities are not adequate to eliminate the occurrence of major channel forming events.

During the period in which ocean-type chinook fry would be rearing in the Cedar River, stream flows are typically well above the levels that provide the maximum amount of rearing habitat (Cascades Environmental Services 1991). Newly emerged chinook fry tend to occupy the areas near the margins of the stream and are quite sensitive to stranding during rapid reductions in stream flow, especially at night (R.W. Beck and Associates 1989; Hunter 1992). However, because little is known about the precise juvenile life history of chinook in the Cedar River, the magnitude of this problem is uncertain. Fish that stay in the river prior to migrating downstream will be more vulnerable to stranding than fish that move directly downstream to the estuary.

The gap in our understanding of juvenile Cedar River chinook makes it difficult to predict the effects of flow on juvenile rearing and migration. If most juvenile chinook migrate to the lake immediately after emergence, successfully rear, and migrate to salt water, then higher stream flows in the spring would be beneficial. However, if the dominant life history pattern is one in which the fish rear in the stream for longer periods prior to migrating to the lake and estuary, then high flows in the spring may force fry out of their preferred habitat too early.

Disease

Fish disease is not thought to be a major factor affecting the survival of Cedar River chinook. However, a virus carried by sockeye salmon could potentially be of concern. Cedar River sockeye, like most sockeye, carry infectious IHNV (see Section 3.5.8), the causative agent of the potentially fatal fish disease, infectious hematopoietic necrosis (IHN) virus. Chinook salmon are susceptible to IHN. However, the degree to which Cedar River chinook might be affected by the particular strain of IHNV present in Cedar River sockeye is uncertain (Wolf 1988; Hsu et al. 1986).

PRESENT STATUS IN THE LAKE WASHINGTON WATERSHED

Washington Department of Fisheries et al. (1993) classified the status of Lake Washington chinook salmon as unresolved because of differing viewpoints of state and

Muckleshoot Indian Tribe and Suquamish Indian Tribe resource managers. Johnson et al. (1997) describe wild Puget Sound chinook as relatively stable from 1968 to 1990 with a sharp drop in abundance beginning in 1991 because of poor ocean survivals, habitat alterations, and harvest pressures. Recent trend analyses confirm the continuation of this decline and the State of Washington now classifies the demographic status of Lake Washington chinook as depressed (Smith, C., WDFW, 1998, personal communication).

Puget Sound chinook salmon, including the populations in the Lake Washington Basin, were listed as threatened under the federal Endangered Species Act on March 24, 1999 (Fed. Reg. Vol. 64, No. 56, March 24, 1999).

3.5.11 Steelhead Trout

GENERAL DESCRIPTION

Steelhead trout (*Oncorhynchus mykiss*) (Figure 3.5-12) are rainbow trout that display an anadromous life history pattern. Originally, this species was included in the genus *Salmo* but further scientific study suggested that the morphometric, genetic, and physiologic traits of this species more accurately reflect those of the genus *Oncorhynchus*. Officially, the species is now considered to be a Pacific salmon but, considering the present vernacular and the purposes of this document, anadromous and resident forms will be referred to as steelhead trout (steelhead) and rainbow trout, respectively. The primary focus of this section is to relate information on steelhead trout, but references to resident rainbow populations will be included in discussions related to local stocks in the Cedar River Basin.

Steelhead life history characteristics are quite diverse, exemplifying their extensive ability to adapt to a wide variety of environmental conditions. Cumulatively, these characteristics make the steelhead life history the most complex and variable of all the species in the genus *Oncorhynchus*. In Washington State, wild steelhead trout are the least numerous anadromous member of their genus with the possible exception of anadromous cutthroat trout, which have not been studied extensively enough to provide an accurate statewide population estimate.

Steelhead trout inhabit Pacific coast streams of North America and northern Asia. The original native range of North American steelhead extends southward from the northern side of the Alaska Peninsula to northern Mexico. The present range is somewhat smaller because human activities have virtually eliminated steelhead populations south of San Francisco. In western Washington, steelhead are present in most Puget Sound drainages, coastal streams, and tributaries of the lower Columbia River. East of the Cascade Mountains they are found in tributaries of the Columbia drainage such as the Entiat, Okanogan, and Yakima rivers, and tributaries of the Snake River such as the Grand Ronde, Clearwater, and Willawa rivers. Asian stocks are most abundant in streams along the west coast of the Kamchatka Peninsula although they also occur, to a lesser extent, in eastern Kamchatka streams and in scattered areas along the northern coast of the Okhotsk Sea.

The total annual abundance of all North American steelhead stocks (including hatchery fish) was estimated to be 1.6 million fish in 1987 (Light 1987). The center of abundance occurs in the Columbia River Basin, which produces 28 percent of the total coast-wide population, followed by Oregon (21 percent), California (17 percent), British Columbia (16 percent), coastal Washington and Puget Sound (13 percent), and Alaska (5 percent). Fifty percent of total coast-wide estimated abundance was attributed to hatchery production, which ranged from 3 percent of Alaska production to 73 percent of production in the Columbia River Basin.

There are 60 wild steelhead stocks inhabiting the Puget Sound drainage (WDF et al. 1993). Of these stocks, 16 are considered healthy, 14 are classified as depressed and 1 stock is considered to be in critical condition. The remaining 29 stocks in the Puget Sound drainage are designated "status unknown." The Lake Washington Basin is considered to have only 1 stock of native/wild steelhead trout. Historically, natural production has occurred in the Cedar River, Issaquah Creek, and north Lake Washington tributaries such as Bear Creek and the Sammamish River (WDF et al. 1993). The Lake Washington steelhead stock is considered to be depressed, and there is no longer significant natural production from any stream in the basin other than the Cedar River (Foley, S., WDFW, 1997, personal communication).

Hatchery steelhead have been planted extensively throughout the Lake Washington and Lake Sammamish basins with the first recorded plant occurring in 1915 (Ajwani 1956). Between 1915 and 1954, over 1,073,000 steelhead fry were planted in the Lake Washington watershed (Ajwani 1956). Additional hatchery plantings were made in the Cedar River and other Lake Washington and Lake Sammamish tributaries between 1954 and 1993, and the last steelhead planting to occur in the Cedar River was in 1993 (WDF et al. 1993). Like many early artificial production programs, the effectiveness of the early steelhead plantings was not rigorously monitored. Available data indicate that estimated levels of hatchery introgression among wild Cedar River steelhead is low as compared to other wild steelhead stocks in the region (Phelps et al. 1994). In 1997, WDFW, in cooperation with Trout Unlimited, started a wild broodstock program designed to incubate and rear Cedar River steelhead for out-planting in Issaquah and Bear Creeks, with the intent of re-establishing the species in these streams.

IMPORTANCE IN THE LAKE WASHINGTON WATERSHED

Steelhead stocks are not typically exposed to the heavy commercial fishing pressure that is associated with Pacific salmon fisheries. This is, in part, a result of relatively low production levels, a highly dispersed ocean life history pattern, and protracted timing for river return and spawning migrations. Historically, returning Lake Washington steelhead have been harvested by the Muckleshoot and Suquamish tribes in the lower reaches of the Lake Washington ship canal (both tribes) and in Lake Washington (Muckleshoot Indian Tribe only). These fisheries were intended to target hatchery fish between December and January, prior to the arrival of significant numbers of wild fish. The last Tribal fishery for Lake Washington steelhead occurred in December 1989 and January 1990 (WDF et al. 1993). Native American tribes have also historically harvested Lake Washington and Cedar River steelhead in hook and line subsistence fisheries in Puget Sound.

Figure 3.5-12. Steelhead trout.

Steelhead are considered to be one of the most sought-after salmonid species by recreational fishers throughout the Pacific Northwest, Canada, and Alaska. The relative importance of the steelhead is exemplified by the Washington State Legislature's designation of steelhead as the State Fish. Cedar River steelhead were once an important component of the Puget Sound steelhead fishery, but recent declines in escapement levels have required that the river be closed to all recreational fishing until the population recovers. Steelhead are not typically targeted for harvest in Lake Washington and any captured rainbow trout measuring over 20 inches (standard length) must be released.

Although the Lake Washington steelhead population is small compared to the other anadromous fish populations found in the basin, it remains an important component of the ecosystem. Steelhead fry are subject to predation by adult cutthroat trout and coho salmon smolts. Diving birds such as mergansers and kingfishers also feed upon steelhead juveniles during the stream rearing period. In general, juvenile steelhead feed on various invertebrates including zooplankton, larger crustaceans, insects, snails and earthworms. As they reach smolt size they also feed on the fry of other fishes, including salmonids. In addition to the competition for food between juvenile steelhead and other juvenile salmonids, the territoriality of juvenile steelhead can affect the behavior and distribution of other salmonids such as cutthroat trout (Meehan and Bjornn 1991).

LIFE HISTORY OVERVIEW

General Patterns

Steelhead are anadromous fish that home to their natal rivers to spawn. They exhibit an iteroparous life history, unlike the semelparous Pacific salmons. Steelhead populations are typically divided into two seasonal races of fish that are primarily defined by the timing of adult returns to spawning streams and by the state of sexual maturity upon entry into fresh water (Neave 1944; Shapovalov and Taft 1954; Bali 1959; Withler 1966; Smith 1968). *Summer steelhead* is the term given to fish that return to fresh water between May and October, and *winter steelhead* is the term given to fish that return to fresh water between November and April (Withler 1966; Smith 1968).

The major differences between the seasonal races are the sexual maturity of the fish upon freshwater entry and the time between freshwater entry and actual spawning. When summer steelhead enter fresh water their gonads are only slightly developed. The gonads of winter steelhead are well developed upon freshwater entry. Summer steelhead usually reside in the freshwater environment for several months before spawning while their winter counterparts spend much less time between freshwater entry and spawning (Shapovalov and Taft 1954; Withler 1966). Summer steelhead also tend to have a higher percentage of body fat than winter steelhead when returning to fresh water (Smith 1968). Despite their significant behavioral and physiological differences, both summer and winter steelhead typically spawn between January and May.

Summer and winter steelhead can exist in the same stream but many steelhead streams are only inhabited by one of the races. Summer steelhead tend to be more prevalent in larger drainages whereas winter steelhead typically inhabit smaller streams (Murphy and Shapovalov 1950; Bali 1959; Withler 1966). It is important to note that this is only a general trend that includes many exceptions.

Two major genetic groups of North American steelhead have been established through extensive genetic studies (Allendorf 1975). These genetic groups are termed *inland* and *coastal* and they tend to be separated by a geographic line that coincides with the Cascade Mountains. The inland group exists in drainages east of the Cascade Mountains and is exclusively comprised of summer steelhead from the Fraser River and Columbia River drainages. The coastal group inhabits rivers west of the Cascades and includes both the summer and winter races (Allendorf 1975; Parkinson 1984; Okazaki 1984).

The Cedar River steelhead population is a coastal population of winter-race fish. Historically, adult steelhead enter Lake Washington through the Ballard Locks between December and early May (WDF et al. 1993). They spawn primarily in the mainstem from March through early June (Burton and Little 1997), although there are historic records of steelhead spawning in Cedar River tributaries such as Rock Creek (below Landsburg Diversion Dam) (Pfeifer, R., WDFW, 1998, personal communication).

Although ocean residence can range from one to several years of age, steelhead typically reside and mature in the ocean for 2-3 years. After maturation, steelhead leave their open ocean feeding areas and migrate to their natal streams to spawn. Most steelhead populations have a period of freshwater entry that lasts several months and is comprised of many minor peaks in abundance of immigrants. Entry into fresh water can be influenced by tides and stream discharge. However, steelhead do not typically linger in the estuary if stream conditions are favorable for their spawning migration (Shapovalov and Taft 1954; Withler 1966; Everest 1973; Oguss and Evans 1978). After freshwater entry, steelhead appear to have the ability to delay spawning for short periods to avoid high instream flow events (Burton and Little 1997). The average size of adult steelhead returning to fresh water to spawn is between 625 and 750 mm, and rare individuals can reach lengths that exceed 1 meter (Ball and Petit 1974; Whately 1977). The majority of the fish returning to Washington streams weigh between 5 and 10 pounds, but fish in excess of 30 pounds have been caught in Washington's recreational steelhead fisheries.

Spawning

Steelhead spawning behavior is similar to that of other salmonids (Section 3.2.3). Cedar River steelhead spawn between March and early June, with peak spawning activity occurring in mid-May (Burton and Little 1997). After spawning, the female will leave the redd site to migrate back to the ocean or die. In small streams, up to 30 percent of the adult steelhead may survive to spawn a second or third time but fish that spawn in larger streams that require long freshwater migrations to reach the spawning grounds are prone to much higher rates of mortality after their initial spawning period. In the Columbia River Basin, summer steelhead rarely ever survive spawning and are essentially semelparous (Long and Griffin 1937).

Before steelhead undertake their first spawning migration they are termed *maiden fish*. If these maiden fish survive their first spawning and manage to return to the sea they are referred to as *kelts*. Kelts that return to spawn in the season immediately following their prior spawning period are termed *consecutive spawners*, whereas kelts that remain in the saltwater environment for an additional year before a subsequent spawning migration are referred to as *alternate spawners*. Studies in Alaska and Canada suggest that approximately 80 percent of repeat spawners are females (Hooton et al. 1987; Didier 1990).

Steelhead trout take advantage of a wide range of spawning habitats including large mainstem habitats such as the Skagit River, and small perennial streams such as Rock Creek below Landsburg Diversion Dam. Steelhead usually spawn in medium- to high-gradient sections of streams at the tails of pools or at the heads of riffles, where hydrologic conditions maintain adequate inter-gravel flows that provide an oxygenated environment for egg incubation (Greeley 1932; Orcutt et al. 1968). Depths during spawning are typically in excess of 24 cm and water velocities for spawning range between 40 and 91 cm/s (Smith 1973).

Steelhead fecundity varies with the size of the female and the strain or stock of fish (Bulkley 1967). Small fish can have as few as 1,000 eggs but large fish can produce in excess of 10,000 eggs. In 1996, 22 Cedar River steelhead were captured and spawned as part of a hatchery broodstock program. The average fecundity of the females captured was 5,172 eggs, with a range of 1,378 to 9,597 eggs per female (Antipa, B., WDFW, 1998, personal communication).

Incubation and Rearing

Steelhead egg development typically occurs in the spring and early summer when water temperatures are increasing. Steelhead require a significantly lower number of degree days for embryonic development and emergence than Pacific salmon.

Steelhead typically hatch between 4 and 8 weeks after fertilization and the larval fish (alevins) remain in the redd for an additional 3 - 5 weeks, absorbing nutrients from a yolk sac connected to their abdomen. Emergence studies occurring in the Cedar River during 1996 and 1997 indicate that fry emergence for an individual redd begins approximately 54 days after fertilization and is complete approximately 63 days after fertilization (Burton and Little 1997). The emergence period for Cedar River steelhead lasts from late May to early August with peak emergence occurring in mid- to late July (Burton and Little 1997). The survival rate of steelhead embryos depends on the amount of fine sediments in the redd, predation and disease rates, the frequency and intensity of scour events during spring freshets, and the maintenance of adequate flows (Bley and Moring 1988).

Steelhead alevins emerge at night and begin feeding within days of becoming free-swimming fry. Less than 20 percent of these fry will survive their first year in the stream environment, because they are highly vulnerable to predation and extreme winter and spring flow conditions that can cause significant scour and premature outmigration (Seelbach 1987). Steelhead typically reside in the stream for 2-3 years, although a small number of fish may outmigrate after 1 year. In some northern rivers, juvenile steelhead can rear 4 or 5 years before migrating to the ocean. Freshwater residence time generally increases from south to north for steelhead populations along the coast of North America (Burgner et al. 1992).

Cedar River steelhead rear in the mainstem and tributaries below Landsburg Diversion Dam. The majority of Cedar River fish are believed to outmigrate as smolts after 2 years of freshwater residence. Size, not age, is the main determinant in smolt outmigration. Fish from less productive systems take longer to reach smolt size and, therefore, are older when they begin to migrate to the ocean. Cedar River steelhead smolts tend to attain large sizes compared to other local and regional stocks (Foley, S., WDFW, 1997, personal communication).

Distribution in the Marine Environment

Generally, steelhead outmigration from fresh water occurs in the spring between mid-March and early June. The peak of the smolt migration usually coincides with peak spring runoff in mid-April to mid-May. The majority of steelhead smolts appear to migrate directly to the open ocean and do not spend significant amounts of time in the estuarine or coastal environments around their birth stream (Burgner et al. 1992). Timing of Cedar River steelhead smolt outmigration is not well understood, although there are ongoing studies being conducted at the Ballard Locks.

After spending 2-3 years in the ocean, the majority of steelhead become mature and leave their feeding grounds to migrate back to their birth stream. Very few fish return after only 1 year in the marine environment, and some fish remain in the ocean for up to 6 years. Steelhead in specific rivers of southern Oregon and northern California are unusual in that a large proportion of the population returns to fresh water only a few months after entry into the marine environment (Kesner and Barnhart 1972; Everest 1973). During their initial ocean residence, these fish grow to an average weight of one-half pound, hence the common name half-pounders. Half-pounders are typically sexually immature and tend to return to the ocean without spawning. Fish displaying this life history pattern usually return to spawn in their natal streams after their second summer in the ocean. Most of the fish returning to Washington streams have been at sea for 2 years and weigh between 5 and 10 pounds. In the Green River (the system directly south of the Cedar; see Map 2), 73 percent of returning steelhead migrate to sea as age-2 smolts, and most of those fish spend 2-3 years in the ocean (Pautzke and Meigs 1941).

HABITAT CHARACTERISTICS AND KEY FACTORS AFFECTING SURVIVAL

Hydrologic Reconfiguration of the Cedar River Basin

In 1917, the Lake Washington ship canal was completed and the outlet of Lake Washington was rerouted through Lake Union, down to the Ballard Locks and into Salmon Bay (see Section 3.2.3 for full discussion). As a result of this project, the elevation of Lake Washington dropped approximately 8.8 ft and the Black River was dewatered. After the change in lake elevation, the Cedar River was re-routed into the south end of Lake Washington, cutting off the normal migration corridor for Cedar River anadromous fish populations. The response of the original population of steelhead trout to these alterations is not known, and information concerning the role of the lake in juvenile and adult life history phases is lacking.

In the early 1900s, construction of Landsburg Diversion Dam was completed without fish passage facilities, blocking access to approximately 17 miles of previously productive anadromous fish habitat. By the beginning of the twentieth century the stream habitat between Cedar Falls and Landsburg Dam had been impacted by extensive timber harvesting. Today this habitat has largely recovered from the effects of logging, and its potential to provide excellent habitat for salmonids is indicated by the presence of a robust population of resident rainbow trout. There are relatively large inputs of high quality ground water throughout the Landsburg Dam-Cedar Falls reach. Erosion and sedimentation is largely in balance with the other natural processes, and riparian zones are largely intact, with much of the stream channel shaded by mature stands of coniferous trees.

Steelhead trout currently spawn and rear in the 21.8 miles of mainstem river habitat downstream of the Landsburg Diversion Dam and can be expected to colonize the habitat above Landsburg Dam if fish passage facilities are provided. Access to the upstream habitat would contribute significant benefits to the population if other factors outside the watershed do not adversely affect their survival. Although the habitat in the Cedar River below Landsburg Dam has been modified by channel confinement structures, increased impervious surfaces, commercial and agricultural development, and a general lack of riparian forest cover and large woody debris, it is still considered to provide the best steelhead habitat in the basin (Foley, S., WDFW, 1997, personal communication).

Stream Flow

Streamflow represents a very important factor in the quality of habitat for aquatic life in the Cedar River, particularly for the four anadromous fish species found there. The City's water supply and hydroelectric operations on the river can affect the total flow volume and the rate of change in those volumes. Flows in the mainstem of the Cedar River are an important consideration for protecting steelhead during spawning, incubation, and rearing.

Steelhead typically spawn at a time when the hydrograph is on a decreasing trend (river water levels are decreasing), which can potentially make their redds vulnerable to dewatering. Particularly during years with high spring stream flows, steelhead are able to access spawning habitat that may later become dewatered as instream flows decrease with the declining snow melt and rainfall in June and July. To address this potential problem, a cooperative effort between WDFW and the City was established in 1995 to monitor steelhead redds to determine the relationship between instream flows and impacts to incubating and emerging steelhead. The initial results of the ongoing monitoring program indicate that significant redd dewatering can occur in years with unusually high spring freshet flows if measures are not taken to adaptively manage instream flows to protect shallow, vulnerable redds. The probability of redd dewatering increases significantly in July when the majority of steelhead remaining in their redds have hatched to become alevins. Alevins are much more vulnerable to damage by dewatering than eggs (Becker et al. 1982).

Instream flow levels in the Cedar River can also impact incubating and emerging steelhead by scouring redds during spring freshet events in March and April. There is very little quantitative data on the effects of floods on steelhead incubation survival in the Cedar River. Because steelhead spawn on a descending hydrograph, they are generally less vulnerable to redd scour than their Pacific salmon relatives that spawn and incubate during fall and winter when the hydrograph is increasing and the probability of major flood events is much higher. Nevertheless, significant Cedar River flood events have occurred in March and April, causing potential mortality to incubating and emerging steelhead. In addition, floodplain development, levees, bank armoring, and flood management practices have reduced the width of the functional stream channel and reduced the river's ability to interact with the natural floodplain to dissipate energy during flood events (King County 1993). High flows are now confined within a relatively narrow corridor, which increases water velocity and sediment transport, and subsequently increases the frequency and intensity of flood scour. This situation has been further aggravated by the removal of forest cover and large woody debris, and increases in the impervious surface area in the lower watershed.

Thoughtful water management practices can help to reduce flood peaks and frequency. However, water storage facilities only capture water from the upper 43 percent of the Cedar River Basin, leaving inputs from the lower 57 percent of the watershed unregulated. In addition, storage facilities in the upper basin have a relatively limited storage capacity. Although water management activities can help to reduce the magnitude of flood events and, to a limited degree, decrease the frequency of such events, the facilities are not adequate to eliminate the occurrence of major channel forming events. Studies have shown that significant impacts to sockeye redds from redd scour occur at approximately 2,000 cfs in the Cedar River (Cascades Environmental Services 1991). Water management practices protect incubating sockeye salmon and steelhead trout from scour events by attempting to reduce the frequency of events that exceed this scour threshold.

Disease

Fish disease is not thought to be a major factor affecting the survival of Cedar River steelhead. However, a virus (IHN) carried by sockeye could potentially be of concern. Steelhead trout are susceptible to IHN (McDaniel et al. 1994; Section 3.5.8), but the degree to which Cedar River steelhead might be affected by the particular strain of IHNV present in Cedar River sockeye is uncertain (Hsu et al. 1986; Wolf 1988).

Other Factors Affecting Survival

In addition to the hydrological alterations associated with rerouting the Cedar River into Lake Washington, there are a number of other factors that potentially influence the survival of Cedar River steelhead trout. These factors include predation by sea lions at the Ballard Locks; degradation of stream habitat from land and water management practices; predation by native and non-native species in the basin; injury to juvenile fish exiting the lake via the Ballard Locks; excessive recreational harvest; illegal fishing practices (poaching); droughts; floods; and unfavorable ocean conditions.

One of the major factors contributing to the decline of steelhead in the Cedar River is predation from sea lions at the Ballard Locks. The precipitous decline experienced during the 1990s coincides with the arrival of feeding sea lions at the locks in the 1980s. Recent studies have shown that sea lions once consumed an annual average of 60 percent of the adult steelhead migrating through the locks (Fraker 1993). As a result of this impact, there has been an exemption from the Marine Mammals Act that allows problem sea lions at the Ballard Locks to be removed or euthanized. In 1996, three problem sea lions were captured and moved to Sea World in an attempt to reduce the associated predation mortality at the locks.

PRESENT STATUS IN THE LAKE WASHINGTON WATERSHED

On February 16, 1994, a comprehensive petition to list west coast steelhead was submitted by Oregon Natural Resources Council and 15 co-petitioners. In response to this petition, NMFS assessed the best available scientific and commercial data, including technical information from Pacific Salmon Biological Technical Committees and interested parties in Washington, Oregon, Idaho, and California. NMFS also established a Biological Review Team, composed of staff from NMFS's Northwest and Southwest Fisheries Science Centers and Southwest Regional Office, as well as a representative of the National Biological Service, which conducted a coast-wide status review for west coast steelhead (Busby et al.1996).

Based on the results of the Biological Review Team's report, and after considering other information and existing conservation measures, NMFS published a proposed listing determination that identified 15 Ecologically Significant Units (ESUs) of steelhead in the states of Washington, Oregon, Idaho, and California. Ten of these ESUs were proposed for listing as threatened or endangered, 4 were found not warranted for listing, and 1 was identified as a candidate for listing. The Lake Washington steelhead population is included in the Puget Sound ESU, which did not warrant listing.

As previously mentioned, the status of Lake Washington Basin steelhead, of which the Cedar River run is the largest component, was deemed depressed in the WDF Salmon and Steelhead Stock Inventory (1993), a report developed prior to the lowest recorded return (70 fish) in 1994. Between 1983 and 1997, escapement estimates for the Lake Washington Basin ranged from 2,575 fish in 1983 to 70 fish in 1994 (all of which were in the Cedar River). The average escapement for this time period was 800 fish. Very low returns in the early 1990s resulted in the closing of all recreational fisheries in the Cedar River until steelhead numbers return to healthy levels. Since the record low return in 1994, steelhead escapement estimates have increased each year from 126 fish in 1995 to 616 fish in 1997.

Genetic analyses has shown that Cedar River steelhead belong to the Puget Sound Genetic Conservation Management Unit (GCMU) (Leider et al. 1994). Stocks comprising the Puget Sound GCMU are presumed to have differentiated from Columbia River stocks of the coastal lineage because of the substantial geographical isolation of their respective migration corridors. Stocks within the Puget Sound GCMU that might merit their own GCMU status in the future were identified in four areas, including Deer Creek, Hood Canal, South Puget Sound, and North Puget Sound subunits. There are some distinguishing characteristics among some stocks of the South Puget Sound subunit, most notably the presence of alleles that do not appear to have been reported in any other genetic assessments of steelhead or rainbow trout (Phelps et al. 1994). Recent investigations seem to indicate that there is sufficient evidence to create a Central Puget Sound GCMU that is comprised of the Pilchuck, White, Puyallup, Green, and Cedar rivers (Phelps, S., WDFW, 1998, personal communication).

Rainbow Trout in the Municipal Watershed

In addition to the wild population of winter steelhead found below Landsburg Diversion Dam, there are also two populations of resident rainbow trout above the diversion. The first population occurs between Landsburg and Cedar Falls, the historic natural barrier to anadromous fishes. The second population occurs in Chester Morse Lake and its tributaries. Genetic analysis of these populations suggests that rainbow trout in Chester Morse Lake were derived from a hatchery planting, however not necessarily from one of the strains currently maintained at the WDFW hatcheries. In contrast, the rainbow trout population between Landsburg and Cedar Falls are more similar to Cedar River and Puget Sound steelhead than to Chester Morse rainbow trout. However, the rainbow trout population above Landsburg Dam also contains alleles from hatchery rainbow trout. Because these alleles are spread throughout the population, the hypothesis that there has been interbreeding between hatchery-originated and wild fish in this reach is supported.

Because of the introgression with non-native, hatchery-originated rainbow trout, neither of the resident rainbow populations in the municipal watershed are considered suitable for artificial supplementation of steelhead in the Lake Washington Basin (Phelps, S., WDFW, 1998, personal communication).

3.5.12 Bald Eagle

STATUS

Legal Status. The bald eagle (*Haliaeetus leucocephalus*) is a federally listed threatened species and a threatened species at the state level in Washington

The bald eagle was listed as endangered throughout the lower 48 states in 1978, except for Michigan, Minnesota, Wisconsin, Washington, and Oregon, where it was listed as threatened. In 1995, bald eagle populations in other states were down-listed from endangered to threatened by the U.S. Fish and Wildlife Service. On July 4, 1999, the USFWS proposed to remove the bald eagle from the List of Threatened and Endangered Wildlife in the lower 48 United States, citing population recovery due in part to habitat protection, and to reduced levels of organochlorine pesticide residue in the environment (Fed. Reg. Vol. 64, No. 128, July 6, 1999).

Population Status. Biologists estimate that there may have been 25,000 to 75,000 nesting bald eagles in the lower 48 states prior to extensive settlement by Europeans (USDI 1995c). Since then, bald eagle populations have declined precipitously as a result of habitat disturbance and loss, direct killing, lead poisoning, power line electrocutions, and reproductive failures caused by the pesticide DDT (USDI 1995c). By the early 1960s, there were fewer than 450 bald eagle nesting pairs in the lower 48 states (USDI 1995c). Today, through aggressive recovery programs, there are more than 4,500 breeding pairs of bald eagles in the lower 48 states (USDI 1995c).

In Washington State, the number of active bald eagle nests has increased steadily since 1980, and now numbers over 550 (WDFW 1997f). However, for unknown reasons, reproductive rates in the Hood Canal and Lower Columbia River areas remain below the target level of one young per nest per year.

RANGE

The bald eagle historically ranged throughout North America, except extreme northern Alaska and Canada, and central and southern Mexico (USDI 1995c).

Bald eagles are present year-round throughout Washington. Most nesting in the state occurs on the San Juan Islands and along the Olympic Peninsula coastline. Bald eagles have been documented to nest in the vicinity of the Cedar River Municipal Watershed along Lake Washington and Lake Sammamish (Smith et al. 1997). Several nests of bald eagles occur within the city limits of Seattle (Smith et al. 1997). Nesting territories are also found along Hood Canal, on Kitsap Peninsula, along the Columbia River in southwestern Washington, in the Cascade Mountains, and in eastern Washington (USFWS 1986). Primary wintering areas include the Olympic Peninsula, the San Juan Islands, Puget Sound and its tributaries, Hood Canal, the Cowlitz and Columbia Rivers (Taylor 1989), and rivers of the western Cascade slopes (e.g., Skagit River).

LIFE HISTORY

Bald eagles usually reach sexual maturity when they attain adult plumage, in the fifth year after hatching; this species is thought to have a reproductive life of 20 - 30 years (Stalmaster 1987). Bald eagles are monogamous, with pair bonds assumed to last for life (Stalmaster 1987; Ehrlich et al. 1988). Both birds in a pair share the duty of caring for the clutch of 2 or 3 eggs during the incubation period of 34 - 36 days; in addition, both sexes care for the young during the 10- to 12-week period before fledging (Ehrlich et al. 1988). During the winter, bald eagle populations in the Pacific states are primarily associated with open water or major river systems (USDI 1986). Radio tracking studies indicate that wintering populations in Washington state come from nesting territories in coastal and interior British Columbia, Alaska, and the San Juan Islands (USDI 1986).

Diet is highly variable, both seasonally and geographically; when a choice is available, however, bald eagles invariably select fish over other prey (Stalmaster 1987). Carcasses of spawned-out salmon provide a key winter food source in the Pacific Northwest (USDI 1986; Stalmaster 1987); eagles will also catch live fish, or steal prey from ospreys (Terres 1980). Other major prey items include birds (most commonly seabirds and waterfowl) and mammals (usually carrion, but rabbits and smaller mammals are also taken occasionally) (USDI 1986; Stalmaster 1987).

HABITAT

Suitable habitat for bald eagles includes the presence of accessible prey and trees for nesting and roosting (Stalmaster 1987). The availability of adequate, non-contaminated food resources is an important determinant of bald eagle nest and territory distribution, and wintering habitat (Stalmaster 1987; Keister et al. 1987). Important food items during the breeding season include fish, small mammals, waterfowl, seabirds, and carrion (Anderson et al. 1986; USFWS 1986). Carrion, such as "spawned out" salmon, also comprises an important part of the fall and winter diet for bald eagles (Stalmaster et al. 1985). Foraging habitat is usually within a short distance of nesting and perching sites during the breeding season, but may be a longer and more-variable distance from winter roosting sites (Stalmaster 1987). The most common foraging habitats for bald eagles are lakes, rivers, and ocean shorelines (Stalmaster 1987).

Nesting habitat for the bald eagle typically includes mature and old-growth forest within 1 mile of water bodies that support an adequate food supply (USFWS 1986). Nests are most commonly found in large Douglas-fir and Sitka spruce trees, averaging 125 - 186 ft in height (Anthony et al. 1982; Anthony and Isaacs 1989). Mature, tall trees tend to provide an unobstructed view and are sturdy enough to support the eagles' large stick nest (which can be 5-6 ft in diameter) (Stalmaster 1987). Perch trees, used by adults and fledged young for resting and searching for prey, are important components of bald eagle habitat (Stalmaster 1987). Perch trees are usually large and located close to open water or the nest tree (Stalmaster 1987). Snags are often used for perching.

Wintering habitat typically includes daytime perches in close proximity to an abundant food source (e.g., anadromous fish runs, waterfowl concentration areas) and communal night roosting areas (USFWS 1986). Communal roosting habitat provides thermal and wind protection for wintering birds. Communal roosts typically occur in uneven-aged forest stands with some old-growth characteristics, and are frequently in areas sheltered by landforms and close to a rich food source (Anthony et al. 1982). Roost trees are typically the most dominant trees of the site (Anthony et al. 1982).

Destruction or degradation of habitat and human disturbance are the main threats to bald eagle populations. Habitat alteration can limit suitable nesting and roosting habitat, and human disturbance can cause birds to leave their nests and can affect prey availability (Roderick and Milner 1991). Bald eagles are particularly intolerant of human disturbance during the breeding season (USFWS 1986). Human activity has been documented to cause nest abandonment and reproductive failure (Bogener 1980; Lehman 1983).

OCCURRENCE IN THE CEDAR RIVER WATERSHED

Bald eagles are present in the Cedar River Watershed regularly as transients and migrants. They are most often associated with habitats adjacent to major streams and larger lakes, especially Chester Morse Lake. No comprehensive surveys have been conducted and no nests or breeding activity have been documented within the Cedar River Municipal Watershed to date.

Nesting habitat for the bald eagle in the municipal watershed may be found in late-successional and old-growth forest within approximately 1 mile of larger water bodies (such as the Cedar River and Chester Morse Lake). Foraging habitat includes rivers, lakes, and other aquatic habitats. Communal winter roost sites may exist wherever the favorable juxtaposition of protective landforms and mature coniferous forest occurs near anadromous fish spawning areas.

3.5.13 Peregrine Falcon

STATUS

Legal Status. The peregrine falcon (*Falco peregrinus*) was recently removed from the federal endangered species list, but remains a state endangered species in Washington.

Population Status. Peregrine falcon populations in the United States were suffering a gradual decline prior to World War II, the result of habitat loss and disturbance from a growing human population (Peregrine Falcon Recovery Team 1982). Their decline accelerated rapidly following World War II as a result of the widespread use of organochlorine pesticides, primarily DDT (Peregrine Falcon Recovery Team 1982). Pesticide use resulted in further reproductive decline and failure in many populations because of eggshell thinning and disruption of normal reproductive behaviors (Peregrine Falcon Recovery Team 1982).

The historical population level of peregrine falcons in Washington State is not well known, because available information is sketchy and non-systematic (Allen 1991). In 1980, one researcher documented at least 12 historical nest sites on the outer coast, San Juan Islands, Columbia River Gorge, and Snake River canyons of Washington (Allen 1991). Washington's peregrine falcon population increased from one known pair in 1978 to 15 pairs in 1990 (Allen 1991), and is considered to be undergoing a slow, steady recovery (Allen 1991).

RANGE

The peregrine falcon is one of the most widely distributed birds in the world, occurring on all continents except Antarctica. Peregrine falcons occur year-round in Washington, as either nesting or migratory individuals. Breeding evidence was confirmed on Mt. Si, approximately 4.5 linear miles from the municipal watershed in 1996 and 1997(Spencer, R., Wildlife Biologist, WDFW, North Bend, Washington, Sept. 21, 1998, personal communication). In 1990, breeding pairs were found on the outer coast, in the San Juan Islands, and in the Columbia River Gorge (Allen 1991). Peregrines have exhibited an increasing use of urban areas, such as Seattle and Tacoma (Smith et al. 1997).

LIFE HISTORY

Peregrine falcons normally reach reproductive maturity in their third year after hatching, and have been known to live up to 20 years (Terres 1980). Pairs are usually monogamous, with the female assuming primary responsibility for incubation of the clutch of 2 to 6 (usually 3 or 4) eggs (Ehrlich et al. 1988). Young hatch in about 29 to 32 days, and remain as nestlings for 5 to 6 weeks before fledging (Ehrlich et al. 1988). A second clutch may be initiated if the first fails early in the incubation period (Terres 1980). Northern populations usually retreat from their nesting areas in fall, and spend the winter south of their breeding range; populations along the Pacific coast may winter within their breeding range, however (Terres 1980).

Peregrine falcons are noted for their hunting prowess, and admired as the swiftest birds of prey (Bent 1938). Diet consists almost entirely of birds, usually obtained on the wing by attacking from above or chasing from behind (Peregrine Falcon Recovery Team 1982; Ehrlich et al. 1988). Doves and pigeons are the most preferred prey; shorebirds, waterfowl, and passerines are commonly eaten in areas where preferred prey are not available (Peregrine Falcon Recovery Team 1982).

HABITAT

Potential nesting and roosting habitat for the peregrine falcon usually includes cliffs or high escarpments that dominate the nearby landscape, although office buildings, bridges, and river cutbanks have been used for nesting as well (Peregrine Falcon Recovery Team 1982; Craig 1986). Most preferred nesting cliffs are at least 150 ft high, can occur from sea level to 11,100 ft elevation, and are usually near water (Peregrine Falcon Recovery Team 1982). The preferred cliff usually has a small cave or overhung ledge large enough to accommodate three or four large nestlings (Peregrine Falcon Recovery Team 1982). Availability of nest sites may be a limiting factor in some areas (Peregrine Falcon Recovery Team 1982).

Peregrines do not build nests (Terres 1980). Instead, they scrape out a shallow depression in which to lay their eggs. Peregrines are sensitive to disturbance during all phases of the nesting season (Hoover and Wills 1987), particularly disturbances that occur directly above the nest (WDW 1993). Disturbance near the nest site may cause the peregrines to desert their eggs or young, and may cause older young to fledge prematurely (WDW 1993).

Foraging habitat for peregrines includes open areas such as wetlands, lakes, river bottoms, estuaries, intertidal mudflats, coastal marshes, croplands, and meadows (Porter and White 1973). They feed on a variety of songbirds, shorebirds, seabirds, and waterfowl (Johnsgard 1990). Their hunting territory may extend up to 12 to 15 miles from nest sites (Hoover and Wills 1987; WDW 1993).

OCCURRENCE IN THE CEDAR RIVER WATERSHED

No comprehensive surveys to determine the presence or absence of peregrine falcons have been conducted in the Cedar River Municipal Watershed and no incidental observations of this species have been documented to date. Peregrine falcons have been sighted in the vicinity of Mt. Si since 1993. They were observed copulating in 1996, but no nest site was found. A nest was located in 1997 and 3 young were fledged that year; 2 were fledged in 1998 (Spencer, R., Wildlife Biologist, WDFW, North Bend, Washington, Sept. 21, 1998, personal communication). Falcons have been spotted flying from the Mt. Si site toward the general direction of Rattlesnake Lake, but foraging hasn't been confirmed in the Rattlesnake Viewshed (Spencer, R., Wildlife Biologist, WDFW, North Bend, Washington, Sept. 21, 1998, personal communication). Because an active nest site is located within approximately 4.5 linear miles of the watershed boundary on Mt. Si, and because presumably suitable nesting and foraging habitat are both present within the watershed, it is possible that peregrine falcons are currently nesting or will eventually nest within the Cedar River Municipal Watershed.

Potential nesting habitat for the peregrine falcon in the watershed includes cliffs and rock outcrops. Foraging habitat may be provided by naturally open habitats (grass-forb meadows and persistent shrub communities) and open wetlands (palustrine emergent wetlands and palustrine scrub-shrub wetlands).

3.5.14 Grizzly Bear

STATUS

Legal Status. The grizzly bear (*Ursus arctos*) is federally listed as a threatened species in the lower 48 States, and it is listed by Washington State as endangered. The Grizzly Bear Recovery Plan (USDI 1993) identified the North Cascades ecosystem as one of six potential recovery areas for the grizzly bear. The North Cascades Ecosystem Grizzly Bear Recovery Zone extends from the Canadian border south to Interstate 90 to include both east and west slopes of the Cascade Mountains (USDI 1993a).

Population Status. Historical records (Sullivan 1983; Almack et al. 1993) indicate that grizzly bears once occurred throughout much of Washington State. Early explorers in Washington mentioned observations and killings of several grizzly bears from the Okanogan and Columbia rivers (Thompson 1970; Sullivan 1983). Grizzly bear observations occurred more frequently along the crest and east slopes of the North Cascades (Thompson 1970).

The decline of Washington's grizzly bear population, dating from the mid-1800s, was likely a result of intensive killing (first for the fur trade, then by indiscriminate killing of grizzlies as "vermin" and government predator control programs) followed by rapid human encroachment into their habitat (Sullivan 1983; Almack et al. 1993). Today, the grizzly bear has been extirpated from much of its range in Washington. Only the North Cascades ecosystem is considered to have a small resident population, while occasional individuals are sighted further south in the Cascades. In 1997, approximately 5-10 grizzly bears were believed to be resident in the North Cascades, with most of these sightings occurring north of the Skykomish Ranger District of Mt. Baker-Snoqualmie National Forest (Almack, J., WDFW, Sedro Woolley, Washington, November 18, 1997, personal communication).

RANGE

Grizzly bears occurred historically throughout most of central and western North America (USDI 1982) including most of Washington State. Their current distribution within Washington is not well known, but a resident population appears to be confined to remote areas of the North Cascades. The North Cascades Grizzly Bear Evaluation Project (1986-1991) verified 15 observations of grizzly bears in the Cascades of Washington between 1964 and 1991, including 3 on the Cle Elum Ranger District of Wenatchee NF (Almack et al. 1993). An additional 75 observations were considered highly reliable (Almack et al. 1993), including several on the Cle Elum Ranger District, one on the St. Helens Ranger District of Gifford Pinchot National Forest, and one on the Packwood Ranger District of Gifford Pinchot National Forest (Almack et al. 1993). Sightings south of the Skykomish Ranger District appear to be primarily transient individuals.

LIFE HISTORY

Female grizzly bears attain sexual maturity at 3.5 years of age, but successful production of litters doesn't appear to occur until at least 5.5 years (Craighead and Mitchell 1982). Young females may breed in alternate years or every third year; individual bears often show greater intervals between breeding as they grow older (Craighead and Mitchell 1982). Cub production has been observed in females up to 25.5 years of age; litter size ranges from 1 to 4, with 2 cubs most common (Craighead and Mitchell 1982). Cubs usually remain with their mother for one or two years before establishing a separate home range as a subadult (Interagency Grizzly Bear Council 1987). Young males disperse greater distances, while subadult females often establish a home range encompassing some portion of the maternal range (Interagency Grizzly Bear Council 1987). Adult male grizzly bear home ranges are generally much larger than those of adult females; there is generally little overlap in individual home ranges, except in the vicinity of rich food sources (Craighead and Mitchell 1982; Interagency Grizzly Bear Council 1987).

HABITAT

Food, cover, denning habitat, extensive space, and solitude are all important constituents of effective habitat (see glossary) for grizzly bears (Craighead et al. 1982; USDI 1993). Vegetative and topographic diversity and extensive areas with low human activity characterize prime grizzly bear habitat (USDI 1993; Interagency Grizzly Bear Committee 1994). The relative importance of forest cover to grizzly bears, especially for bed sites, has been documented in several studies (e.g., Blanchard 1978; Servheen and Lee 1979; Schallenberger and Jonkel 1980). It is not known whether grizzly bears use forest cover because of an innate preference or to avoid humans (Blanchard 1978). Interspersion of open habitats (e.g., shrub lands, riparian areas, meadows, emergent wetlands, avalanches chutes, talus slopes) with forest cover is also important, as documented by Blanchard (1978). These open habitats are extremely important foraging areas for bears.

Diverse vegetation types provide the abundance and diversity of plant and animal foods used by grizzly bears on a seasonal basis. When they emerge from dens in the spring, grizzlies seek lower elevation areas, valley bottoms, avalanche chutes, and ungulate winter ranges where their food requirements can be met (USDI 1993). Through spring and early summer they tend to move to higher elevations, tracking the emergence of herbaceous plants that are higher in crude protein levels (USDI 1993). In late summer and fall, there is typically a transition to roots, tubers, nuts and berries (USDI 1993). Animals (e.g., ground squirrels, pocket gophers, deer and elk) are preyed upon or scavenged opportunistically throughout the active period of the year.

Grizzly bears sleep during the winter (approximately 5 months, from late November to late April) in dens they excavate. Dens are usually dug on steep slopes where wind and topography cause deep snows to accumulate and where that snow is unlikely to melt during warm periods (USDI 1993). Elevations of dens vary geographically, but they generally occur at higher elevations well away from development or human activity (USDI 1993). Several instances of den abandonment resulting from human disturbance have been reported (USDI 1993).

Individual grizzly bears can have extensive home ranges, sometimes encompassing 1,000 - 1,500 square miles. Thus, large amounts of space are essential to the maintenance of viable grizzly bear populations (USDI 1993). Because grizzly bears can conflict with humans and their land uses, grizzly populations require some protection from human depredation and competitive uses of habitat (including road construction, traffic, logging, mining, grazing, recreation, and human settlement) (USDI 1993). Effective grizzly bear habitat must include some areas isolated from human development and high levels of human activity.

Roads probably pose the most imminent threat to grizzly bear habitat today with mortality being the most serious consequence (USDI 1993). Several studies have documented that grizzlies are more vulnerable to legal harvest and poaching as a consequence of greater road access by humans (e.g., Aune and Kasworm 1989). Bears are also killed by collisions with vehicles (Palmisciano 1986). In addition, numerous studies have documented that grizzly bears, especially females with cubs, avoid using areas near roads, even roads officially closed to public use (e.g., Smith 1978; Zager 1980; Archibald et al. 1987; Mattson et al. 1987; McClellan and Shackleton 1988; Aune and Kasworm 1989; Kasworm and Manley 1990). The Interagency Grizzly Bear Committee (1994) concluded that "core areas, areas free of motorized access during the non-denning period, are an important component of the habitat of adult females grizzlies that have successfully reared and weaned offspring." The Committee defined core areas as those areas greater than 0.3 miles from any open road or motorized trail (Interagency Grizzly Bear Committee 1994). Management consideration should be given to ensure that core areas meet seasonal grizzly bear habitat needs (Interagency Grizzly Bear Committee 1994).

OCCURRENCE IN THE CEDAR RIVER WATERSHED

No comprehensive surveys to determine the presence or absence of grizzly bears have been conducted in the Cedar River Municipal Watershed and no incidental observations of this species have been documented to date. Additionally, recent sighting information suggests that the watershed is at the southern periphery of the current range of grizzly bears in Washington State (Interstate 90 forms the southern boundary of the North Cascades Ecosystem Recovery Zone; this is approximately 3 linear miles north of the Cedar River Watershed administrative boundary).

The occurrence of highly reliable grizzly bear sightings south of the watershed within the past 10 years suggests that an occasional bear may travel through the watershed (e.g., while dispersing).

3.5.15 Gray Wolf

STATUS

Legal Status. The gray wolf (*Canis lupus*) is federally listed as an endangered species and is listed as endangered by Washington State.

Population Status. Once considered common throughout forested areas of Washington (Johnson and Cassidy 1997), the gray wolf was essentially extirpated from the state by the 1930s as a result of trapping for pelts and predator control (Roderick and Milner 1991). However, gray wolves appear to be in the early stages of becoming re-established in Washington State. Since 1984, wolves have been observed in the vicinity of Ross Lake on both sides of the U.S.-Canada border (North Cascades National Park no date). In the past 10 years, wolf family groups have been confirmed in two areas in Washington – North Cascades National Park and the Okanogan area; three other sightings appear to be reliable, but are unconfirmed (Almack, J., WDFW, Sedro Woolley, Washington, November 18, 1997).

RANGE

Gray wolves occurred historically throughout most of the United States, including most of the forested areas in Washington. The species was virtually extirpated from the lower 48 states in the 1800s through large-scale predator control programs and severe reduction of natural prey populations. The species' current distribution in Washington is not well known, but a resident population appears to be confined to remote areas of the North Cascades. Other sightings have been made as far south as the Taneum Creek, Big Creek, and Bald Mountain areas of Wenatchee National Forest, and the Randle area of Gifford Pinchot National Forest (WDFW database 1996). There is one Class 2 sighting (reliable but unconfirmed) of a gray wolf within the Green River Watershed analysis area (USDA 1996). These sightings appear to be primarily transient individuals.

LIFE HISTORY

Wolves gain sexual maturity in their second year, but often do not breed until their third year. It is commonly thought that wolves mate for life, with breeding occurring annually (Mech 1970; Paradiso and Nowak 1982). Breeding occurs from late January through April, depending on latitude; gestation lasts about 9 weeks (Mech 1970). The average litter contains 4 to 7 pups, although as many as 14 pups have been reported (Mech 1970). Pups are born blind and helpless, and remain with their mother for several months after birth (Paradiso and Nowak 1982). The life expectancy of wolves is 16 years, although a 10-year-old wolf can be regarded as a very old animal (Paradiso and Nowak 1982).

HABITAT

The gray wolf is a very wide-ranging species that uses almost any natural habitat (Laufer and Jenkins 1989), including forestlands and natural openings (e.g., alpine meadows, shrublands, marshes), as long as the level of human activity is low and an adequate ungulate prey base is available (Laufer and Jenkins 1989). Suitable denning and rendezvous habitat for the gray wolf is defined as broad valley bottoms away from human disturbance, usually at high elevations (Mech et al. 1988). Wolves avoid areas with greater than approximately one mile of open roads per square mile of land area (Mladenoff et al. 1995). Primary threats to transient and re-introduced gray wolf populations include human disturbance, habitat loss, and lack of ungulate prey (USDI 1987).

Because large ungulates (black-tailed deer, mule deer, elk, caribou, moose) are the principal prey of gray wolves, habitat conditions that favor ungulate species (i.e., interspersion of forest cover with open areas for feeding) would also favor wolves. Ungulate winter ranges and calving areas are especially important habitats for wolves.

Wolves are highly social, living in family groups (packs) with 2 - 25 individuals (Mech 1970). Wolf packs consist of one dominant pair (alpha pair), their offspring from one or more generations, and other non-breeding adults. Den sites are usually underground although abandoned beaver lodges or hollow logs are also used (Mech 1970). Typically, dens are located on south or southwest aspects of moderately steep slopes in well-drained soils, and are usually within 600 ft of surface water (Mech 1970). Wolf packs have territories ranging in size from 50 square miles to over 1,000 square miles, depending on pack size and prey density (Mech 1970). Thus, a viable wolf population requires an extensive geographic area.

OCCURRENCE IN THE CEDAR RIVER WATERSHED

No comprehensive surveys to determine the presence or absence of gray wolves have been conducted in the Cedar River Municipal Watershed. Recent sighting information suggests that the watershed is at the southern periphery of the current range of gray wolves in Washington State. The occurrence of reliable gray wolf sightings east and south of the watershed within the past 10 years suggests that an occasional wolf may travel through the watershed (e.g., while dispersing).