

Technical Report 2010-8

**A REVIEW OF ADULT SALMON AND STEELHEAD LIFE HISTORY AND
BEHAVIOR IN THE WILLAMETTE RIVER BASIN: IDENTIFICATION OF
KNOWLEDGE GAPS AND RESEARCH NEEDS**

**Prepared by:
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Department of Fish and Wildlife Resources
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FINAL

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EXECUTIVE SUMMARY

This review compiles and synthesizes information on adult Pacific salmonids (*Oncorhynchus* spp.) relevant to their life history, migration, and survival in the Willamette River basin. Upper Willamette River spring Chinook salmon (*O. tshawytscha*) and winter steelhead (*O. mykiss*) were the focal species for the review because these runs are listed as threatened under the U.S. Endangered Species Act. Information was also summarized for fall Chinook salmon, coho salmon (*O. kisutch*), chum salmon (*O. keta*), and non-native summer steelhead.

A primary review objective was to identify biologically-based information gaps relevant to managing adult salmonids in the Willamette River basin. Forty-one inter-related information gaps were identified for the listed runs in the general categories of: (1) life history; (2) behavior; (3) prespawn mortality; (4) environmental and operational effects; (5) ocean, estuary and climate effects; and (6) population-specific gaps. Many of the identified gaps apply equally to the unlisted Willamette basin salmonid populations. Our recommendations are directly relevant to many of the management actions in the 2008 Biological Opinion for the Willamette project, and are intended to help managers identify and prioritize research needs.

1.0 INTRODUCTION

The primary objective of this review is to compile and synthesize reported information on adult Pacific salmonids (*Oncorhynchus* spp.) relevant to their life history, migration, behavior, and survival in the Willamette River basin (Figure 1). Topics covered in the review were developed in consultation with U.S. Army Corps of Engineers (USACE) biologists as part of a coordinated effort to identify critical knowledge gaps and to provide a context for prioritizing adult salmonid research needs. The presented material is intended to supplement the USACE's implementation strategy for the Willamette Project Biological Opinion (BiOp; NMFS 2008), which includes water management, fish passage, dam operation, water quality, hatchery program, and habitat components. While we conducted this review while considering the management actions presented in the BiOp's reasonable and prudent alternative (RPA), we generally used a biologically-based approach to identifying information gaps.

Upper Willamette River spring Chinook salmon (*O. tshawytscha*) and winter steelhead (*O. mykiss*) are the focal species for the review. Historically, Willamette Falls (near Oregon City, Oregon) blocked or seasonally restricted upstream fish passage, resulting in reproductive isolation and genetic divergence for salmon and steelhead populations upstream from the falls (Reisenbichler et al. 1992; Myers et al. 2006; McElhany et al. 2007). Despite recent hatchery influences and construction of fish passage facilities at Willamette Falls that largely eliminated the seasonal barrier, upper basin spring Chinook salmon and winter steelhead remain genetically distinct from populations downstream from Willamette Falls and from both coastal and Columbia River populations (Myers et al. 1998; NMFS 2008). Both upper Willamette River runs were listed as threatened under the U.S. Endangered Species Act in 1999 (NMFS 1999a, 1999b).

The upper Willamette populations have declined for a variety of reasons, including habitat degradation, habitat loss associated with dams without fish passage facilities, overharvest, pollution, changes in hydrologic and thermal regimes, and direct and indirect effects of artificial propagation (NMFS 2008). The ESA listing decision for spring Chinook salmon noted that significant natural production for this run now occurs primarily in the McKenzie River sub-basin, with limited natural production in other historic habitat (Federal Register 1999a). Natural production of winter steelhead is primarily in tributaries above Willamette Falls that drain the western Cascade Range, and especially in the North and South Santiam, Molalla, and Calapooia sub-basins (Howell et al. 1985; Chilcote 1998). The current distributions of both spring Chinook salmon and winter steelhead are substantially different than historical distributions in the upper Willamette basin. There have been constrictions in many areas as a result of blocked access. However, there has also been expansion into new habitats because fish produced at supplementation hatcheries have spawned in the wild and because fish have been outplanted into areas that did not historically support significant populations.

The Willamette River basin also has adult runs of fall Chinook salmon, coho salmon (*O. kisutch*), and summer steelhead. These populations are not a focus of the review because they are not federally listed and were either introduced or were historically restricted to sites downstream from Willamette Falls. However, we have included some information on all adult salmon and steelhead runs given spatial and temporal overlap with the focal populations, shared management issues, and important interactions among species and seasonal runs across life history stages.

Literature for the review was collected by searching in a peer-reviewed database (Web of Science), by searching for grey literature reports posted on USACE and Oregon Department of Fish and Wildlife (ODFW) websites, and by searching for material specific to adult salmonids in the Willamette BiOp documents. We also searched the citation lists in the most relevant papers and reports and attempted to locate additional electronic files using Google Scholar. Relevant unpublished reports and those unavailable in electronic form were also solicited from personnel at the various agencies conducting adult salmonid research and monitoring in the Willamette River basin. Information from the reviewed papers and reports were organized into several basic categories (i.e., distribution and status, life history, survival, behavior, etc.) and these were used to frame the synthesis. Importantly, we did not focus on hatchery activities or hatchery effects (except as needed to provide context) because hatcheries are part of a separate independent review (Tom Friesen, ODFW, *personal communication*). Similarly, we did not review material related to ocean or in-river harvest as these topics are part of ongoing monitoring and management plans. This report's final section includes a discussion of current information gaps relevant to managing adult salmonids in the Willamette River basin.

1.1 ADULT SALMON AND STEELHEAD RUNS

1.1.1 SPRING CHINOOK SALMON

The Willamette River basin supports a group of Chinook salmon populations with at least two basic life history types (Myers et al. 1998). Typical spring-run or ‘stream-type’ Chinook salmon were historically abundant in most tributaries draining the western Cascades. Core populations spawned in the Middle Fork Willamette (10s of thousands of spawners), Clackamas (10–20,000) McKenzie (10–20,000), North Santiam (1,000s), South Santiam (1,000s), Molalla (1,000s), and Calapooia (100s) rivers (Myers et al. 2002; NMFS 2008). Satellite populations may have spawned in some western tributaries, but it is not known whether these groups were genetically distinct or otherwise differentiated from the core populations.

Dams constructed in the 1940s to 1960s blocked access to headwater spring Chinook salmon spawning areas in most of the Willamette River tributaries upstream from Willamette Falls. Estimated habitat loss was ~70–100% in the Middle Fork Willamette and both Santiam sub-basins and ~25% in the McKenzie sub-basin. Downstream from Willamette Falls, habitat loss in the Clackamas sub-basin was ~20%. Many of the Willamette basin dams also degraded adult spring Chinook salmon habitat downstream from the dams by altering flow and temperature regimes and reducing recruitment of spawning gravel and woody debris.

To mitigate for lost habitat and production, spring Chinook salmon hatcheries were built in most sub-basins. Broodstock at the hatcheries were developed from local populations, but there was widespread mixing of sub-basin groups among the hatcheries. This resulted in extensive introgression and a relatively homogenized upper Willamette River spring Chinook population (Myers et al. 2006). The Clackamas River spring Chinook salmon population is also genetically very similar to the upper Willamette group as a result of extensive hatchery exchange. Five hatchery stocks are associated with Willamette River spring run: North Fork Santiam, South Fork Santiam, Middle Fork Willamette, McKenzie, and Clackamas hatcheries. These core hatcheries have additional satellite facilities, including a variety of traps and weirs. In most basins, hatchery-produced adult fish spawn in-river with naturally-produced adults. Total adult escapement is driven by hatchery production, with more than 90% of returns of hatchery origin in many years (Myers et al. 1998).

Adult spring Chinook salmon passage at Willamette Falls typically begins in mid-March, peaks in late April or May, and then continues through July (Figure 2). Since 1953, an average of ~38,000 adults have been counted at the falls (*range* = 13,000–96,000). In the tributaries, redd building and some spawning starts in August, though most occurs in late September and October (e.g., Schroeder et al. 2007). Current spawn timing is generally believed to be later than historically (i.e., pre-dam) as a result of dam-related temperature changes and hatchery effects.

Spring Chinook salmon emergence timing also differs from likely pre-dam patterns, with large differences among spawning sites (i.e., above and below dams and among tributaries) as a result of prevailing water temperatures. This variability has produced a variety of juvenile emigration strategies. The traditional spring yearling emigration is predominant (Myers et al. 2006), but more complex strategies include subyearling emigration to the lower main stem Willamette River in fall and winter and subyearling emigration from spawning areas followed by rearing in tributary reservoirs and

other locations. The preponderance of hatchery releases and mixing among hatchery and wild fish on the spawning grounds has made it difficult to clearly delineate the life history variability of naturally-produced fish.

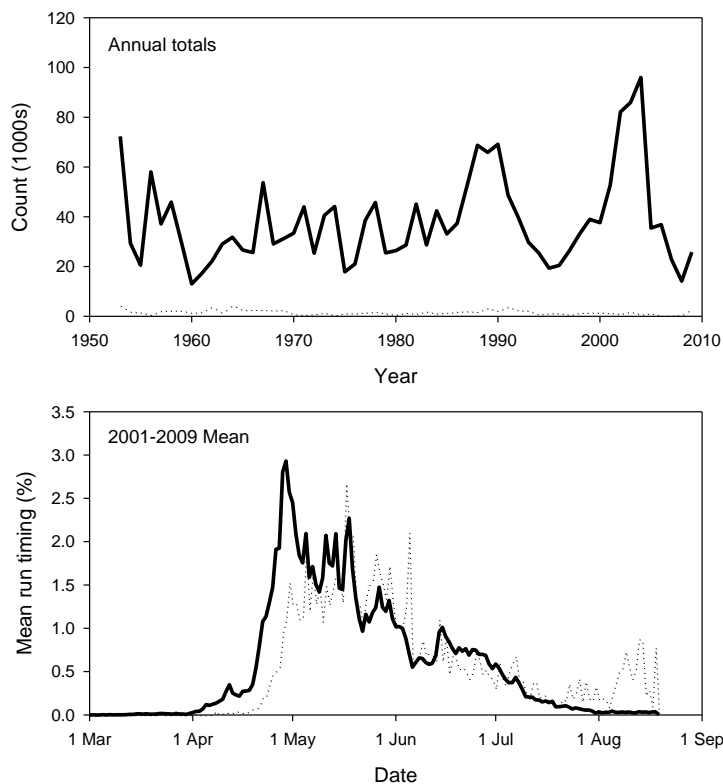


Figure 2. Annual (top) and mean daily (bottom) counts of spring Chinook salmon at Willamette Falls. Solid lines represent adult counts, dashed lines represent jack counts. Source: ODFW count data (http://www.dfw.state.or.us/fish/fish_counts/).

1.1.2 FALL CHINOOK SALMON

Ocean-type, fall-run Chinook salmon were historically abundant in the Willamette River basin, but were restricted to sites downstream from Willamette Falls (Fulton 1968; NMFS 2008). Native fall Chinook salmon were considered a ‘tule’ run that entered the Columbia River estuary relatively early in the fall in mature condition and spawned shortly thereafter. They primarily spawned in the main stem Clackamas River, with many at sites upstream from the current location of North Fork Dam. This run was severely depleted or extirpated by the mid-20th century as a result of habitat loss and water quality problems in the lower Willamette River (Myers et al. 2006). After pollution problems were addressed, fall-run Chinook salmon were reintroduced into the Clackamas River using stocks from lower Columbia River hatcheries. Hatchery outplanting occurred from the 1950s to 1981. The Clackamas population declined when hatchery production ended, but a small tule population persists and spawns in the lower main stem downstream from River Mill Dam (Taylor 1999; McElhany et al. 2007).

Both tule and bright fall Chinook salmon were introduced into several sub-basins upstream from Willamette Falls in the 1950s and 1960s after rebuilt fish ladders at Willamette Falls improved upstream passage (Wevers 1992). The late-migrating ‘bright’ population introduced from Cowlitz Hatchery did not persist after releases stopped. However, the Willamette River tule fall Chinook salmon hatchery program produced millions of fall Chinook salmon (from the ‘Stayton Ponds’ stock) through the mid-1990s. The current population of natural spawners upstream from Willamette Falls derived from this program. Self-sustaining fall Chinook salmon populations currently exist in the North Santiam, South Santiam, McKenzie, Calapooia, and main stem Willamette River. The aggregate fall Chinook population upstream from Willamette Falls has substantially declined since hatchery production ended (Figure 3).

During the last decade, adult fall Chinook passage at Willamette Falls has started in mid-August, when there is some overlap with late-timed spring Chinook salmon, and continued into early November (Figure 3). Since 1966, a mean of ~9,400 adult fall Chinook salmon have been counted at the falls (*range* = <1,000–34,000), with smaller numbers (< 5,000) in recent years. Peak passage at the falls is in mid- to late September. Return timing to spawning sites upstream from the falls is from September–November with a peak in late September. Most spawning occurs in late September and October. During spawning, there is both spatial and temporal overlap between some fall- and spring-run Chinook salmon populations (e.g., Lindsay et al. 2000; Schroeder et al. 2005). In general, however, spring Chinook salmon spawn at higher elevation sites within each sub-basin while fall Chinook salmon spawn in lower main stem reaches.

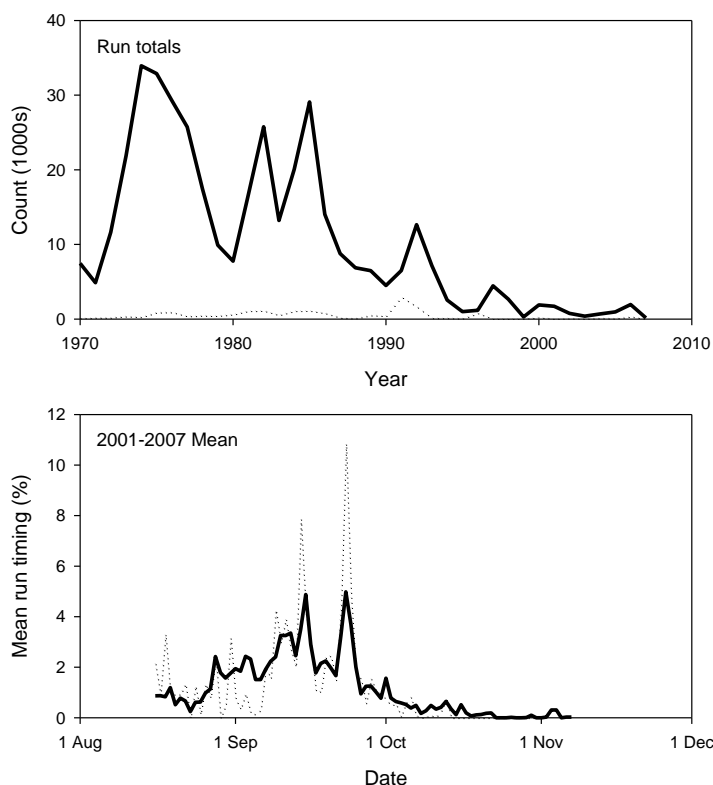


Figure 3. Annual (top) and mean daily (bottom) counts of fall Chinook salmon at Willamette Falls. Solid lines represent adult counts, dashed lines represent jack counts. Source: ODFW count data. (http://www.dfw.state.or.us/fish/fish_counts/)

1.1.3 WINTER STEELHEAD

Willamette River winter steelhead are a genetically and phenotypically distinct group. In genetic analyses, the upper Willamette River fish clustered separately from coastal winter steelhead, lower Columbia River winter steelhead (including those downstream from Willamette Falls), and inland Columbia River winter and summer-run fish (Schreck et al. 1986; Reisenbichler et al. 1992). The upper Willamette River winter steelhead are relatively late-migrating, entering fresh water in March and April (Korn 1961; Howell et al. 1985). This is in contrast with the mid-winter entry by most coastal and lower Columbia River winter-run populations. Downstream from Willamette Falls, Clackamas River winter steelhead are most strongly affiliated with the coastal winter runs (Reisenbichler et al. 1992).

Pre-dam, winter steelhead from the upper Willamette River run primarily spawned in the North and South Santiam rivers, with some in tributaries up to and including the Calapooia River (Fulton 1970). The McKenzie and Middle Fork Willamette rivers had limited or no production (Howell et al. 1985). There may also have been some limited spawning in west-side tributaries like the Tualatin, Yamhill and Coast Fork (Fulton 1970; Howell et al. 1985; Kostow 1995; NMFS 1999). Improved passage at Willamette Falls allowed successful introduction of early-run winter steelhead (Big Creek stock) and summer-run steelhead (Skamania stock) to upper basin tributaries. These non-native stocks occupy habitat that was not used by the native winter steelhead run as well as some habitat in their historic range. In addition, hatchery-reared fish derived from the native winter run have been stocked into previously unoccupied or lightly-occupied habitat. As a result, the upper Willamette winter run is believed to have a broader geographic distribution than historically.

As with spring Chinook salmon, many hatchery winter steelhead spawn in the wild and there is concern about interactions, competition and introgression with native stocks. There are similar or greater concerns about effects of introduced summer steelhead, particularly in the Mollala, North Santiam, and South Santiam rivers where summer-run adult returns have been relatively large. Downstream from Willamette Falls, hatchery summer steelhead have often been predominant in Clackamas River steelhead spawning areas (Kostow et al. 2003). In the Clackamas sub-basin, wild winter steelhead productivity has been significantly reduced by the presence and abundance of hatchery summer steelhead (Kostow and Zhou 2006). Similar negative ecological effects are assumed to occur in tributaries upstream from Willamette Falls.

There is considerable uncertainty about the size of pre-dam winter steelhead populations upstream from Willamette Falls. However, the general consensus is that the run was stable, as evidenced by its genetic distinctiveness, and it may have persisted at relatively low levels in several small closely-related populations (Federal Register 1999b). It is not clear whether there was significant differentiation among sub-basin populations. Over the last several decades, the native winter steelhead run has steadily declined (Chilcote 1998, 2001). Even with the introduction of Big Creek winter steelhead, the adult run size at Willamette Falls has declined (Figure 4). The several years with larger combined early and late winter steelhead returns in the early 2000s have been associated with improved ocean conditions.

Winter steelhead passage at Willamette Falls typically begins in November and continues at low levels until late January; this first group is considered to be primarily Big Creek winter stock (Figure 4). The

native winter run passes the falls from February through May, with peak counts typically in February and March. It is not clear how much temporal overlap exists between Big Creek and native stocks, but there is considerable overlap between the late winter and summer-run steelhead, especially in March, April and May (see Figures 4 and 5). On average, the Willamette Falls count of late winter steelhead (the nominal native run) has averaged ~4,200 fish since 1950. The early winter run (Big Creek stock) has averaged ~7,400 fish since 1971. Winter steelhead spawning occurs from March through June, with peak spawning in April in most east-side tributaries (Howell et al. 1985). There is some spatial and temporal overlap between winter and hatchery summer steelhead on the spawning grounds.

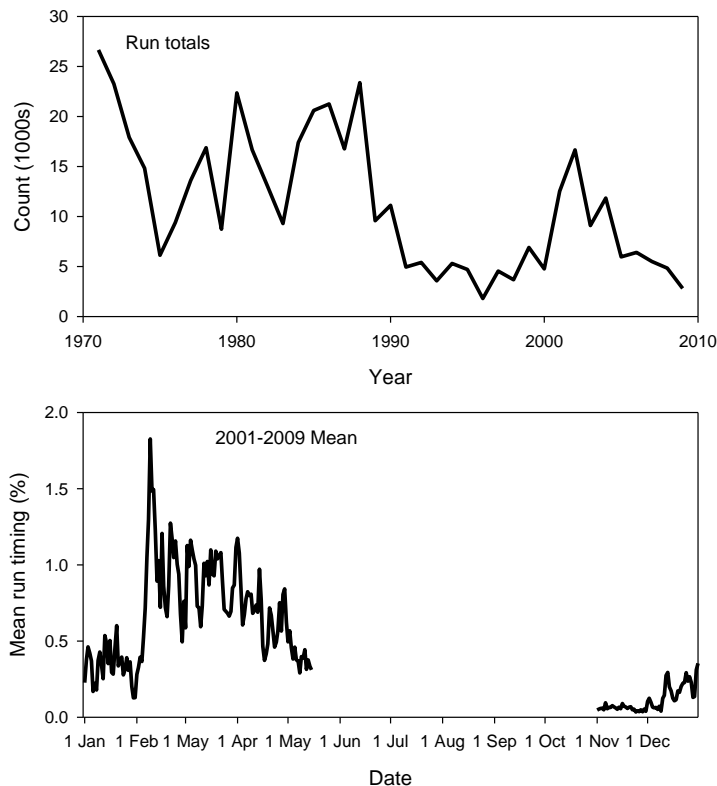


Figure 4. Annual (top) and mean daily (bottom) counts of winter steelhead at Willamette Falls. Source: ODFW count data. (http://www.dfw.state.or.us/fish/fish_counts/)

1.1.4 SUMMER STEEHEAD

Non-native Willamette River summer steelhead are all derived from Skamania stock hatchery fish, a summer-run composite developed using Klickitat River and Washougal River fish in the 1950s (Reisenbichler et al. 1992). Skamania summer steelhead were transferred to the South Santiam Hatchery in the 1970s (Kostow et al. 2003) and were subsequently distributed among many Willamette tributaries to support recreational fisheries. Summer steelhead populations are currently in the Clackamas, Mollala, North Santiam, South Santiam, McKenzie, and Middle Fork Willamette rivers, with some satellite populations at other sites (ODFW 1990; NMFS 2008). In some sub-basins, adult summer steelhead have been collected at dams (i.e., North Fork Dam on the Clackamas River) and recycled downstream to supplement fisheries. The combination of high exploitation rates in fisheries and broodstock collection is believed to limit the number of hatchery fish spawning in the wild. Nonetheless, summer-run spawners are widespread in the Willamette basin (Clack HGMP 2006).

Since 1970, annual adult summer steelhead counts at Willamette Falls have averaged ~14,300 fish and ranged as high as ~40,700 fish (Figure 5). The summer run passes through the lower Willamette from March through late October, with the bulk of the run passing from mid-April to mid-July (Figure 5). Although there is some spatial and temporal overlap in winter and summer-run spawning, overall differences between the two runs are believed to limit opportunities for interbreeding (Chilcote 1998; Firman et al. 2005). However, there is evidence that juveniles produced by summer steelhead have density-dependent and other negative ecological effects on winter steelhead production, and the presence of hatchery summer-run steelhead has been widely associated with native winter-run declines in Willamette basin tributaries (e.g., Chilcote 1998; Kostow et al. 2006; NMFS 2008).

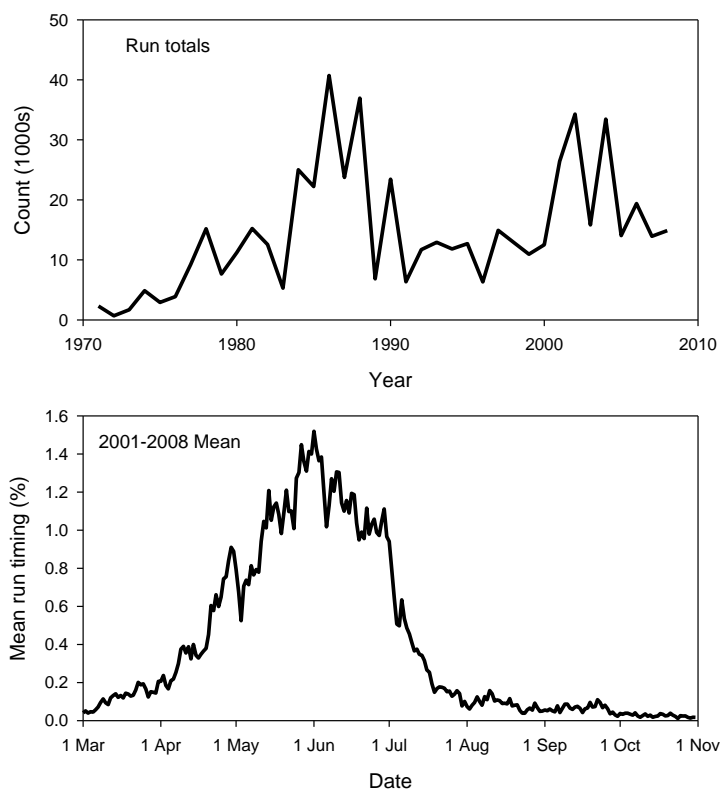


Figure 5. Annual (top) and mean daily (bottom) counts of summer steelhead at Willamette Falls. Source: ODFW count data. (http://www.dfw.state.or.us/fish/fish_counts/)

1.1.5 COHO SALMON

A single native coho salmon stock is present in the Willamette River system. This small population of late-run coho salmon is part of the Lower Columbia River ESU (Weitkamp et al. 1995). Adults return to sites upstream from North Fork Dam on the Clackamas River and are considered the only remaining viable wild population in the Willamette River system (Cramer and Cramer 1994; Weitkamp et al. 1995). A hatchery-derived, early-run coho salmon population is also self-sustaining, with spawning groups in the lower Clackamas River and some upstream from North Fork Dam (Weitkamp et al. 1995). Hatchery rearing occurs at Eagle Creek NFH (Clackamas HGMP 2002).

Willamette Falls was probably a natural barrier to coho salmon prior to modifications (Weitkamp et al. 1995). However, hatchery coho salmon were released for several decades into tributaries upstream from Willamette Falls starting in the mid-20th century, with the last releases in 1998. Self-sustaining populations persist at several sites, including in the Tualatin, Pudding and Yamhill rivers on the west side of the basin and the Molalla and main stem and North Fork Santiam rivers on the east side. On average, annual passage at Willamette Falls has been about 3,200 adult coho salmon, with peak runs greater than 17,000 fish (Figure 6). The vast majority of the run passes Willamette Falls in September and October.

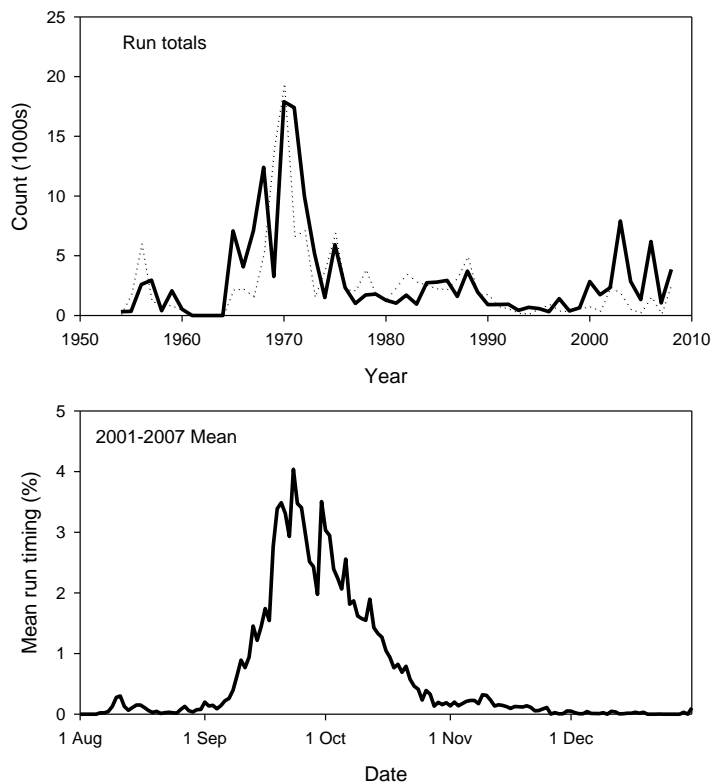


Figure 6. Annual (top) and mean daily (bottom) counts of coho salmon at Willamette Falls. Source: ODFW count data. (http://www.dfw.state.or.us/fish/fish_counts/)

1.1.6 CHUM SALMON

The historical distribution and abundance of chum salmon (*O. keta*) in the Willamette River and lower Columbia River basins have not been fully established (Myers et al. 2006; NMFS 2008). However, a chum salmon run entered and spawned in the Clackamas River main stem in late fall through the early 20th century. This population is believed to have been extirpated by approximately 1944, at least in part due to water quality problems in the lower Willamette River (Dimick and Merryfield 1945; NMFS 2008). The species is considered functionally extinct in the Clackamas (USFWS 2007).

1.2 DAMS AND OTHER BARRIERS

1.2.1 ADULT PASSAGE BARRIERS

In total, there are an estimated 1,491 anthropogenic barriers (i.e., dams, culverts, etc.) that completely or partially block fish access to approximately 15,000 km of habitat in the Willamette River basin (Steel et al. 2004; Sheer and Steel 2006). These authors estimated that barriers block access to an average of 50% of the 1st and 2nd order tributaries in the primary Willamette sub-basins. On a per-basin basis, the greatest percentages of blocked access for anadromous fish are in the North Santiam and Middle Fork Willamette rivers where there are large, impassable dams.

There are more than 350 dams in the Willamette basin, including 18 non-federal hydropower projects and 25 major Federal Columbia River Power System (FCRPS) dams (Fulton 1968; Bennett 1994; Upper Willamette Recovery Plan). Of the 25 FCRPS dams, 11 are single-purpose hydroelectric projects operated by public or private utilities and 13 are multipurpose USACE dams and reservoirs collectively called the “Willamette River Basin Project” (Figure 7). These 13 dams provide flood control and generate hydropower. Construction began in 1941 with Fern Ridge Dam on the Long Tom River and ended in 1969 with completion of Blue Ridge Dam on the McKenzie River (Table 1).

Eight of the 13 USACE Willamette Project dams were constructed without adult fish passage facilities (Table 2). These include Big Cliff and Detroit (North Fork Santiam), Blue River (McKenzie), Hills Creek and Lookout Point (Middle Fork Willamette), Cottage Grove and Dorena (Coast Fork), and Fern Ridge (Long Tom) dams. Adult salmonid trapping facilities are operated at Foster (South Fork Santiam), Cougar (McKenzie), and Dexter and Fall Creek (Middle Fork Willamette) dams. An adult trap at Green Peter Dam (South Fork Santiam) is no longer in operation. There are at least 16 additional sub-basin dams with direct relevance to adult passage. At these sites, fish ladders allow adult passage at nine dams, there is a trap at one dam, and six have no passage facilities (Table 2). (Note: Appendix 1 includes sub-basin maps that show locations of dams and secondary tributaries.)

1.2.2 ENVIRONMENTAL EFFECTS

The environmental effects of high-head storage dams on the Willamette River basin salmonid habitats have been described in several other publications (e.g., Hughes and Gammon 1987; Benner and Sedell 1997; Wentz et al. 1998; Steel and Lange 2007; NFMS 2008; Upper Willamette Recovery Plan) as well as in sub-basin plans (e.g., ODFW 1990a, 1990b; Murtagh et al. 1992). We provide here only a brief synopsis of the environmental effects of Willamette dams on salmonids. Additional discussion related to specific aspects of adult migration and life history in relation to river environment is provided in subsequent sections.

High-head storage dams in the Willamette River system have substantially altered hydrologic and thermal regimes in the main stem and tributaries. The dams have also affected physical characteristics of floodplains, riparian areas and adult spawning habitats by reducing peak flows, limiting recruitment of substrate and woody debris, and reducing channel complexity. In general, cool-water releases from flood-control dams have lowered summer water temperatures at downstream sites in the main stem Willamette River and in the lower reaches of several tributaries (e.g., Rounds 2007; NMFS 2008). The cooler temperatures may result in slower upstream adult passage and delayed onset of spawning.

Table 1. Summary of spring Chinook salmon and winter steelhead extinction risks, hatcheries, and significant dams in each of the major Willamette River sub-basins.

Tributary	Extinction risk ¹			Hatcheries	USACE high-head storage dams (year)	Other dams
	Spring Chinook	Winter steelhead	Other runs			
Clackamas	Low	Not rated	Fall Chinook Coho Chum	Clackamas Eagle Creek		Faraday North Fork River Mill Oak Grove
Molalla NF Santiam	Very high Very high	Moderate Moderate	Summer steelhead ² Summer steelhead	Marion Forks Minto Pond	Big Cliff (1954) Detroit (1953)	Elkhorn Falls Minto Spillway Bennett (upper and lower) Lebanon
SF Santiam	Very high	Moderate	Summer steelhead	South Santiam Foster Dam	Foster (1968) Green Peter (1968)	
Calapooia McKenzie	Very high Moderate	Moderate	Summer steelhead	Leaburg McKenzie	Blue River (1969) Cougar (1963)	Leaburg Smith Trail Bridge Walterville
MF Willamette	Very high			Dexter Pond Willamette	Dexter (1954) Fall Creek (1966) Hills Creek (1961) Lookout Point (1954)	
Coast Fork	Not rated		Summer steelhead ³		Cottage Grove (1942) Dorena (1949)	
Long Tom	Not rated				Fern Ridge (1941)	Ferguson Monroe Stroda

¹ From McElhany et al. (2007)

² Hatchery releases stopped in 1997

³ Strays reported spawning

Table 2. Summary of adult and juvenile passage facilities at dams in Willamette River sub-basins. Source: USACE.

Tributary	USACE high-head storage dams	Adult passage	Juvenile Passage	Other dams	Adult passage
Clackamas				Faraday North Fork River Mill	Fish ladder Fish ladder Fish ladder
Molalla					
NF Santiam	Big Cliff Detroit	None None	None None	Elkhorn Falls Minto Spillway	None Trap Fish ladder
SF Santiam	Foster Green Peter	Trap Trap (closed)	Incomplete Surface collector	Bennett (lower) Bennett (upper) Lebanon	Fish ladder Fish ladder Fish ladder
Calapooia					
McKenzie	Blue River Cougar	None New trap	None 'Horns'	Leaburg Smith Trail Bridge Walterville	Fish ladder None None Unclear
MF Willamette	Dexter Fall Creek Hills Creek Lookout Point	Trap Trap None None	None 'Horns' None None		
Coast Fork	Cottage Grove Dorena	None None	None None		
Long Tom	Fern Ridge	None	None	Ferguson Monroe Stroda	None Fish ladder None

¹ Bennett dams part of 'Stayton hydroelectric complex'

However, reservoir drawdown operations in late summer and fall tend to release relatively warm water (e.g., Sullivan and Rounds 2004), resulting in temperatures that are warmer than pre-dam means during spawning and incubation periods. Elevated fall temperatures have been associated with delayed spawning, high adult prespawn mortality (e.g., Keefer et al. 2010), and premature fry emergence (e.g., ODFW 2004) in spring Chinook salmon. High temperatures have also been associated with increased incidence and lethality of a variety of bacterial and fungal infections in adult migrants (Beidler and Knapp 2005).

Changes in the frequency and magnitude of high- and low-discharge events in the Willamette River system have had dramatic effects on channel structure, habitat complexity, and the exchange and movement of nutrients and various substrate classes (e.g., Seedang et al. 2008). These changes have been associated with redd scouring (ODFW 1990a), loss of spawning habitat downstream from dams, and loss of secondary channels due to main stem channel incision. Homogenization and channelization (i.e., from bulkheads, revetments, and other flow-control structures) of the accessible habitat have reduced adult access to thermal refugia and holding habitats, limited upstream migration routes, and concentrated fish in some constricted areas, especially downstream from dams without passage facilities. The effects of these changes on adult salmonid migration behavior, survival, and fitness have not been well studied and mitigation options are poorly understood.

2.0 SPRING CHINOOK SALMON

2.1 DISTRIBUTION AND STATUS

There is a general consensus that there were seven demographically independent spring Chinook populations in the Willamette River basin: Clackamas, Molalla/Pudding, Calapooia, North Santiam, South Santiam, McKenzie, and Middle Fork Willamette (ODFW 1998; McClure et al. 2003; Myers et al. 2003; HSRG 2009). Based on genetic persistence and/or natural production capacity, four core populations remain (Clackamas, North Santiam, McKenzie, and Middle Fork Willamette) and two have significant self-sustaining natural production (Clackamas and McKenzie). The status of these populations differs along a series of gradients, including their extinction risk, degree of hatchery influence, available habitat, production potential, life history diversity, management-related recovery options, and many others.

Extinction risk for the seven principal Willamette River spring Chinook salmon populations was evaluated using a multi-metric status assessment by McElhany et al. (2004, 2007). The overall conclusion of those evaluations was that the Clackamas population was at low risk, the McKenzie population was at moderate risk, and the remaining populations (Molalla, North and South Santiam, Calapooia, Middle Fork Willamette) were at very high risk of extinction (Table 1). The Clackamas and McKenzie stocks had less risk because abundance, productivity, and life history diversity in these populations were relatively high and negative hatchery impacts were relatively limited (especially in the McKenzie). In addition, the spatial distribution of habitat was considered favorable in the McKenzie and Clackamas sub-basins, as well as in the Mollala sub-basin (McElhany et al. 2007).

A central debate in the recovery process for Willamette River spring Chinook salmon is about the role of hatcheries. Artificial propagation has affected all of the Willamette populations, and hatchery-produced fish predominate in all sub-basins except the McKenzie. However, the degree of genetic introgression and the scope of hatchery-related phenotypic effects have differed among sub-basins. There has been considerable recent research directed towards understanding hatchery effects in the basin (see reviews in HSRG 2009 and Johnson et al. *in prep*). Genetic studies have implicated homogenization and introgression across sub-basins, but there is also evidence that Willamette spring Chinook salmon have remained differentiated from other regional populations in the Columbia River and along the west coast (e.g., Schreck et al. 1986; Myers et al. 2003; NMFS 2008). There is also evidence that natural production in some stronghold areas (i.e., in the McKenzie) and in areas where hatchery fish can be separated from wild fish (i.e., at some hatchery and dam traps) has allowed for continued genetic and phenotypic differences to persist among tributary populations. In addition to hatchery-specific and system-wide hatchery evaluations, a multi-year field monitoring program has focused on identifying the distribution and abundance of hatchery-origin fish in natural spawning aggregations (e.g., Schroeder et al. 2007; Kenaston et al. 2009).

The following brief descriptions of the status of each sub-basin spring Chinook salmon population and their basic hatchery history were gleaned from population summaries developed by the HSRG (2009). We note again, however, that a full consideration of hatchery effects was beyond the scope of this review. (Note: Appendix 1 includes sub-basin maps that show locations of dams and secondary tributaries.)

2.1.1 CLACKAMAS

The Clackamas River sub-basin historically supported a large spring Chinook salmon population. Construction of Faraday and River Mill dams early in the 20th century severely restricted access to the upper basin, and Cazadero and River Mill dams further limited access. A flood destroyed the Cazadero Dam fish ladder in 1917 and eliminated passage to the upper basin for more than 20 years (ODFW 1992). During that period, natural production was limited to the lower section of the main stem Clackamas River and to Eagle Creek. When upstream passage was restored, recolonization progressed slowly, presumably using residual Clackamas River stocks as source fish but also strays from other tributary populations. Later, production by the Clackamas Hatchery helped rebuild the population using a stock derived from a variety of upper Willamette River stocks. Recent genetic analyses have indicated that the Clackamas population is similar to the upper Willamette group, reflecting the extensive mixed-origin derivation.

Currently, the majority of natural spring Chinook salmon production (~85%) in the Clackamas occurs upstream from North Fork Dam (Taylor 1999; King et al. 2000). A trap at the dam is used to remove hatchery fish from this population, a strategy that is effective under most (but not all) environmental and operational conditions. Several thousand fish return to the upper basin in most years and spawn at a variety of tributary and main stem sites. The remaining ~15% of natural production in the basin occurs downstream from North Fork and River Mill dams, with some spawning in tributary (i.e., Eagle Creek) and main stem sites. Production at Clackamas Hatchery supplements fisheries and contributes to spawning aggregates downstream from North Fork Dam; only Clackamas River returns have been used in this hatchery program since 1988. However, rearing of some Clackamas Hatchery fish occurs in other Willamette basin facilities, and this may contribute to adult straying.

2.1.2 MOLALLA

There is little pre-settlement information about Mollala River spring Chinook salmon. However, a dramatic decline is believed to have occurred by the early 20th century. By the 1960s, the natural component of this sub-basin population was considered extirpated (Cramer et al. 1996).

Currently, there is little evidence of a self-sustaining spring Chinook salmon population in the Mollala River or the Pudding River (the major Mollala tributary) despite annual releases of juvenile hatchery fish from South Santiam Hatchery. Spring Chinook salmon redds found in recent years have been almost exclusively associated with hatchery fish. In most years, more than 90% of carcasses found in spawning ground surveys have been fin-clipped (e.g., Schroeder et al. 2002). Spawner density has been low and there is believed to be limited natural production (McElhany et al. 2007).

2.1.3 NORTH FORK SANTIAM

The North Fork Santiam River was a very productive spring Chinook salmon sub-basin. The 1953 construction of Detroit Dam blocked access to ~70% of the spawning habitat in the sub-basin, and the population rapidly decreased. Returns of naturally-produced adults had declined to several hundred by the 1990s.

Natural spring Chinook salmon production is now restricted to sites downstream from Big Cliff Dam and the Minto barrier weir and trap (Schroeder et al. 2001), except that some juveniles are produced by

adults outplanted upstream. Additional natural spawning occurs in the lower river near Stayton (below upper and lower Bennett dams), and in the Little North Santiam River. In the main stem North Fork Santiam, there is little natural production between the Bennett dams and Minto, and approximately 85% of the spawners in this section are of hatchery origin in most years. Prespawn mortality upstream from the Bennett dams has also been high, at least in part due to upstream dam effects on water temperature. Prespawn mortality has limited natural production in this reach (Schroeder et al. 2006). Hatchery production is from broodstock collected at Minto Pond and reared at Marion Forks Hatchery. This population is locally derived but has had some out-of-basin influence.

2.1.4 SOUTH FORK SANTIAM

The South Santiam sub-basin supported extensive spring Chinook salmon production in many tributaries and in some main stem reaches. Foster and Green Peter Dams blocked access to much of the best spawning habitat in 1968. Prior to dam construction, about 85% of the production was at sites upstream from the Foster Dam site. Run sizes up to about 10,000 adults occurred from the 1960s to 1980s, but runs of natural-origin fish in the last decade have been estimated in the 1,000s (e.g., Firman et al. 2002).

In most recent carcass surveys, most (>80%) of the fish on South Fork Santiam spawning grounds have been of hatchery origin (e.g., Lindsay et al. 2003). Since 1996, adult salmon have been trapped at Foster Dam and outplanted into a variety of spawning tributaries (including into sub-basins other than the South Fork Santiam, including the Calapooia River). Although there has been low redd density and relatively high prespawn mortality, outplanted adults have produced significant numbers of juveniles. The South Fork Santiam hatchery stock is mostly locally-derived, with some out-of-basin transfers. There have been a variety of hatchery strategies in the South Fork Santiam, including rearing at the Willamette and South Santiam hatcheries and some hatchery exchange with the McKenzie system. Out-of-basin rearing may contribute to relatively high straying rates for the South Santiam population (Schroeder et al. 2002). In-basin spawning by hatchery fish is extensive, with carcass surveys indicating ~80% contribution in some recent years (e.g., Schroeder et al. 2002).

2.1.5 CALAPOOIA

The Calapooia River likely supported a relatively small native spring Chinook salmon run. The run had declined to annual returns of several hundred fish by the 1960s (Nicholas 1995; Myers et al. 2003), and the native stock is now considered extirpated.

Hatchery salmon from multiple sources (i.e., South Santiam, Willamette, and McKenzie hatcheries) were released into the Calapooia for several decades, but there have been no recent juvenile releases. However, adults from South Santiam Hatchery have been outplanted into the Calapooia in the last decade. Prespawn mortality has been high for the outplanted salmon, but these fish may contribute to a small naturally-produced population (e.g., Firman et al. 2002).

2.1.6 MCKENZIE

The McKenzie sub-basin is considered the major current production site for naturally-produced spring Chinook salmon in the Willamette River system. Salmon in the McKenzie have had limited influence

from hatchery transfers and presumably maintain many of their locally adapted traits (McElhany et al. 2007). Historic returns were in the 10s of thousands, but declined after Cougar and Blue River dams blocked access to spawning habitat in the 1960s. In recent years, naturally-produced returns have ranged between 3,000 and 10,000 fish.

Most current natural production in the McKenzie sub-basin is upstream from Leaburg Dam, with some additional production in the lower main stem, where most spawners are hatchery fish. The distribution of natural spawners in the basin is approximately 10–20% in the South Fork McKenzie River, 30–40% in the main stem McKenzie downstream from the South Fork, and 45–60% in headwater areas above Cougar Dam as far as Trail Bridge Dam. Natural-origin fish (i.e., without fin clips) have comprised 60–85% of carcass recoveries upstream from Leaburg Dam in recent years (HGMP 2008). Broodstock collection for McKenzie Hatchery occurs near Leaburg Dam, and this program incorporates a relatively high proportion of natural-origin fish. The Leaburg trap is also used to separate hatchery from natural-origin adults, but the trap is inefficient during high-flow conditions. This allows some hatchery fish to reach the upper basin.

2.1.7 MIDDLE FORK WILLAMETTE

As with the McKenzie sub-basin, the Middle Fork Willamette River supported a very large pre-dam spring Chinook salmon population. Several sources suggest that the Middle Fork had the largest runs in the Willamette basin. Construction of Dexter and Lookout Point dams in 1954, both without fish passage facilities, resulted in the loss of about 80% of the spawning habitat upstream from Fall Creek (Connolly et al. 1992). As much as 98% of historic spring Chinook spawning occurred upstream from Lookout Point Dam (Mattson 1948). Fall Creek, the major Middle Fork tributary downstream from Dexter Dam, was also a productive spring Chinook salmon site, but the Fall Creek run was considered extinct by the 1990s. Upstream passage in Fall Creek is blocked by Fall Creek Dam, but returning unclipped adult salmon have been released upstream in the last several years. Overall, natural production in the Middle Fork Willamette has been limited to the lower main stem – an area with relatively limited spawning habitat and unfavorable temperature conditions – and to Fall Creek. This production is largely dependent on the progeny of hatchery releases.

Currently, there is limited evidence to suggest that there is a self-sustaining natural population of spring Chinook salmon in the Middle Fork Willamette River. Recent carcass surveys downstream from Dexter Dam suggested that more than 90% of fish spawning in the wild were of hatchery origin (e.g. Schroeder et al. 2003). Local extinction is considered likely if hatchery releases were to be discontinued. Broodstock collection for the Willamette Hatchery occurs at the Dexter Trap downstream from Dexter Dam. The hatchery population was primarily founded using local stocks, with few historic out-of-basin transfers. Although there have been several decades of artificial selection, this stock is believed to maintain some locally-adapted traits.

For the last several years, Willamette Hatchery adults have been outplanted upstream from Falls Creek Dam and upstream from the Middle Fork Willamette dams. These fish are believed to contribute some natural production, and unclipped adult fish that return to Falls Creek Dam are passed upstream. However, prespawn mortality of outplants in the Middle Fork Willamette has been among the highest recorded among the Willamette basin outplant programs (e.g., Schroeder et al. 2007).

2.1.8 COAST FORK AND LONG TOM

There is little compelling evidence that the Coast Fork supported an independent, self-sustaining spring Chinook salmon population. There may have been intermittent production in this sub-basin, but Dorena and Cottage Grove dams blocked access to most spawning areas. Hatchery-produced juvenile salmon have been released into the Coast Fork, but downstream survival rates were unknown and there have been limited adult returns. In recent years, a small number of surplus adult hatchery salmon have been outplanted into a Coast Fork tributary, Mosby Creek. In combination with strays, these outplants have been associated with some limited spring Chinook salmon production.

There have been anecdotal reports of spring Chinook salmon spawning in other west-side tributaries, but these fish were presumably strays from other sites. No self-sustaining populations are believed to exist.

2.2 LIFE HISTORY

2.2.1 EARLY LIFE HISTORY

There are several early life history strategies among spring Chinook salmon in the Willamette River basin. These range from fall and winter subyearling emigration to more typical stream-type yearling emigration in the spring (Mattson 1963; Myers et al. 2006; NMFS 2008). Juvenile life histories have been inferred from scales and otolith marks, including thermal marking (e.g., Lindsay et al. 1997) as well as from collections of outmigrating juveniles at dams and in traps (e.g., Zakel and Reed 1984; Howell et al. 1988; Keefer et al. 2011). Though juvenile life stages are not a focus of this review, juvenile life history does affect a variety of adult life history traits including ocean residency, age at return, sex ratios, and adult migration timing (Cramer et al. 1996; Myers et al. 2006). Adults with different juvenile life histories also differ in their susceptibility to harvest as a result of age and size differences (Cramer et al. 1996).

Hatchery spring Chinook salmon in the Willamette basin are predominantly yearling emigrants (Myers et al. 1998). Among wild fish, however, the relative contribution of yearling versus subyearling life history types to adult returns varies considerably among sub-basins and among years, suggesting that juvenile survival rates and perhaps emigration strategies vary with in-river conditions and reservoir operations (Keefer et al. 2011). Schroeder et al. (2007) reported that the subyearling portion of wild adults on spawning grounds ranged from less than 5% to more than 80% in sampling over four years in four Willamette sub-basins. On average, the South Santiam had the highest proportion (>50%) of wild adults with subyearling histories and the McKenzie had the lowest proportion (<20%) (Schroeder et al. 2007).

Life history variability results from a complex set of genetic and environmental factors that span across life stages. Adult traits such as migration timing and age, for example, can be directly related to the spawn timing of the previous generation, emergence timing of juveniles, juvenile rearing location, residency times, and emigration behaviors. Understanding relationships among these factors should be considered in any comprehensive plan to manage adult Chinook salmon in the Willamette basin.

2.2.2 OCEAN RESIDENCY

As with most spring Chinook salmon populations, ocean distribution information for Willamette basin fish has primarily been inferred from coded wire tag (CWT) recoveries. A recent 25-year summary of CWT ocean recoveries by Weitkamp (2009) showed that a majority of Willamette River fish were caught between southeast Alaska and Queen Charlotte Island, British Columbia. Smaller numbers were recovered in the major fisheries from the mouth of the Columbia River northward, including in the Washington Coast, Puget Sound, and Vancouver Island fisheries. Almost none were recovered south of the Columbia River mouth. Data in the Weitkamp (2009) summary was for salmon from Marion Forks, McKenzie, and Willamette hatcheries, and distributions for these groups were qualitatively similar. The Willamette River fish also had distributions that were broadly similar to those for upper Columbia and Snake River spring Chinook salmon stocks. Myers et al. (1996), Howell et al. (1988), and Garrison et al. (1994) also reported that Willamette stocks were primarily recovered in fisheries off Alaska and British Columbia, but the Myers et al. (1996) study showed more variability in the distributions among Willamette sub-basin populations, with relatively more North and South Santiam fish in Alaskan fisheries compared to Clackamas and McKenzie stocks.

Almost all of the CWT data used in ocean distribution studies has been for hatchery fish. Given the current genetic similarity between hatchery and wild stocks, distributions for the two groups are also presumably similar. However, this has not been explicitly established, and life history differences (i.e., juvenile emigration age, return age) among hatchery and wild groups may affect behavior and distribution in the ocean.

The age composition of returning adult spring Chinook salmon has been influenced by a variety of hatchery actions (i.e., broodstock selection, juvenile rearing and release protocols) and age selection in both ocean and in-river fisheries (Cramer et al. 1996; Myers et al. 2006; ODFW 2007). For example, combined ocean and in-river harvest rates for Willamette fish tend to be highest for older age classes (i.e., age 5 and 6 fish; Cramer et al. 1996) and this affects age composition on spawning grounds and for hatchery returns. Willamette basin hatcheries also have selected primarily for age 4 and 5 fish and limited collection of jacks (and mini-jacks). As a result, adult returns to the Willamette are dominated by age 4 and 5 fish (Figure 7), and there has been a decrease in older age classes since the middle of the 20th century (Willis et al. 1995). The relative contribution of age 4 and age 5 year classes differs among years (Howell et al. 1985; Wevers et al. 1992; Bennett 1994; McElhany et al. 2007), presumably for a variety of hatchery-related and environmental reasons. On average, less than 5% of recoveries in 10 years of data from the lower Willamette River sport fishery and the Clackamas River sport fishery were age 3 or age 6 (Foster 1997, 1999). These fisheries are almost exclusively for hatchery fish.

In more recent surveys, age composition of wild spring Chinook salmon has been different than for the hatchery fish (e.g., Schroeder et al. 2007). Though still dominated by age 4 and 5 fish, there were substantial differences among wild returns from different Willamette sub-basins (Figure 8). In 2002-2006, for example, there were proportionally twice as many age 5 fish as age 4 fish in the McKenzie wild returns, while this pattern was nearly reversed in the South Santiam basin. Lindsay et al. (1997) also reported predominantly (80%) age 5 unmarked adult salmon at McKenzie Hatchery, compared to a hatchery return that was mostly (76%) age 4. Among-basin and hatchery-wild differences in adult age composition may be related to the relative growth rate, size or survival of each group's juvenile

year-classes (Willis et al. 1995). Alternately, there may be life history-based differences among sub-basin populations that affect juvenile freshwater residency, adult ocean distribution, exploitation rates, or other factors. These possible explanations have not been thoroughly examined for wild fish from any of the sub-basin populations.

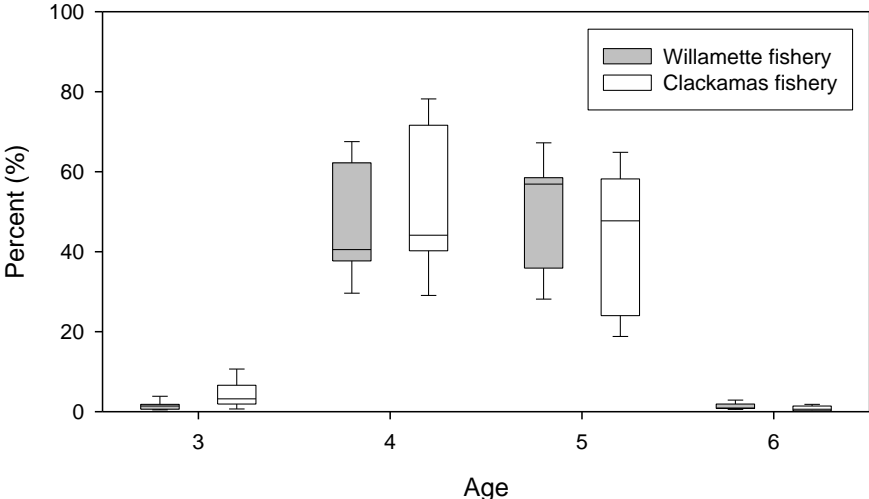


Figure 7. Box plots of adult spring Chinook salmon age composition from 1986–1996 for fish caught in the Willamette and Clackamas River recreational fisheries. Boxes show median, quartile, and 10th and 90th percentiles. Data summarized from Foster (1997).

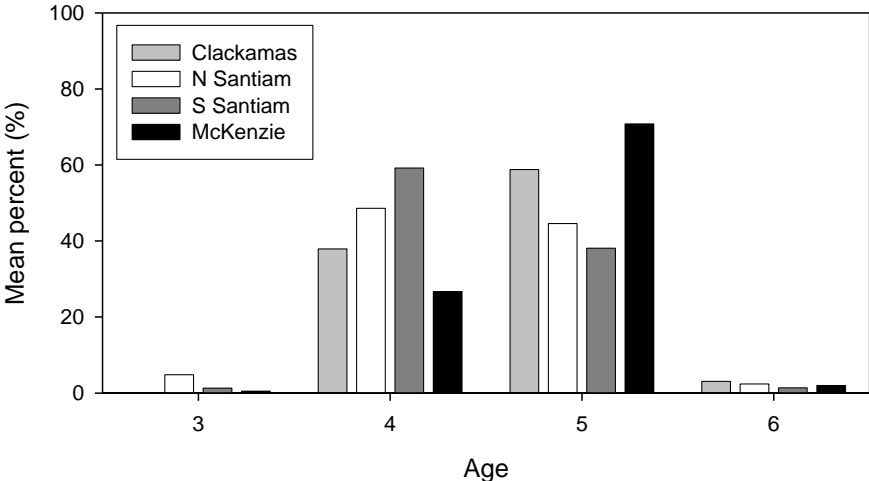


Figure 8. Mean age composition of wild adult spring Chinook salmon that returned to the North Santiam and McKenzie rivers (2001–2006) and Clackamas and South Santiam rivers (2002–2006). Data summarized from Schroeder et al. (2007).

2.2.3 ADULT MIGRATION TIMING

Adult spring Chinook salmon passage at Willamette Falls typically begins in mid-March, peaks in late April or May, and then continues with low daily numbers through July. This migration timing schedule is relatively early when compared with interior Columbia River spring Chinook salmon runs and is believed to be an adaptation to return during late winter/early spring during flows that allowed passage at Willamette Falls (Myers et al. 2006; McElhany et al. 2007). For the overall Willamette run, there is a general consensus that naturally produced salmon and older age classes return earlier than hatchery fish (e.g., Cramer et al. 1996; NMFS 2008).

Over the last decade, median spring Chinook salmon run passage dates at Willamette Falls have varied by more than a month (Figure 9). Although variation in run timing does not appear to have been thoroughly evaluated in the basin, it is likely that water temperature and river discharge affect the upstream progression of the run. In the short time series shown in Figure 9, the late-timed 2008 run was associated with cold April, May and June water temperatures and the early runs in 2001, 2004 and 2005 were associated with relatively warm March temperatures and/or low spring discharge. Run timing at Willamette tributary sites generally lags the distributions at Willamette Falls by several weeks or more. The overall pattern from counts at tributary dams and weirs suggest that spring Chinook salmon either move upstream slowly or spend extended periods holding in the main stem Willamette River or lower reaches of tributaries before moving towards spawning areas.

It should be noted that daily and weekly count data were not readily located in electronic format for many upriver sites (i.e., dams and traps), though such data presumably are available from agencies. A better compilation of count data would be useful for evaluating environmental and operational effects on spring Chinook salmon run timing. Understanding salmon behavior during the period of extended holding may help identify sources of the often high adult mortality that occurs during this period.

Run timing at North Fork Dam on the Clackamas River typically begins in May or June and then peaks between July and September (Clackamas HGMP 2002). In some years, passage at North Fork Dam has been bimodal (e.g., Grimes et al. 1996), with a nadir during peak summer water temperatures. There has also been evidence for age-specific migration timing differences in the Clackamas River with earlier arrival by age 5 versus age 4 fish (Clackamas HGMP 2002). There may also be timing differences between wild and hatchery fish, though this has not been established. North Fork Dam is a relatively short distance (<40 rkm) upstream from the Willamette River, indicating that salmon hold for extended periods in the lower reaches. Holding has also been observed downstream from dams in this system, but the overall spatial distribution of holding behaviors has not been well described.

Passage at upper and lower Bennett dams on the North Santiam River typically starts in mid- to late May or June, but the bulk of the run passes during a relatively short period between late May and July (e.g., Grimes et al. 1996; Lindsay et al. 1998, 2000; Schroeder et al. 1999; Firman et al. 2002, 2004). A much smaller secondary peak occurs in early September in most years, and in some years there have been more than one intermediate peak during the bulk passage in June. It is not clear if large swings in weekly passage are related to dam operations, environmental cues, or if there are finer-scale stock (i.e., hatchery versus wild or specific spawning groups) or age-structured effects. Typically, peak counts at the Bennett dams have lagged peak counts at Willamette Falls by 4–6 weeks (see Figures 10 and 11 for examples).

Migration timing patterns at Leaburg Dam on the McKenzie River are broadly similar to those in the North Santiam. Passage at Leaburg Dam typically begins in late April or early May, peaks in June, has a passage nadir in August, and then a smaller secondary peak in late August or September close to the onset of spawning (Grimes et al. 1996; Firman et al. 2002, 2004; McElhany et al. 2007). As in the North Santiam River, secondary peaks in some years suggest that there may be stock- or environment-related timing differences. There is some evidence that hatchery and wild fish timing differs in some years (Figures 10 and 11). The spatial distribution of fish holding during the prespawn period has not been well described for either South Santiam or McKenzie River populations.

Migration timing data from dams with traps rather than ladders may be biased by trap operation timing, but still provide some relative passage index. At the Foster Dam trap on the South Santiam River, for example, monthly trap data from 1988-2003 showed that trap percentages averaged about 6% in May, 31% in June, and 20% each in July, August and September (Streamnet). Year-to-year variability was relatively high, particularly after 1 July, and it was not possible to infer the duration or spatial distribution of holding downstream from the trap. Extended holding has been observed downstream from Dexter and Fall Creek dams on the Middle Fork Willamette River. Operation of adult traps at these sites and observational data suggest that adults begin to arrive in April or early May, with peak arrival timing in June and July and continued new arrivals through August and a secondary peak in September associated with prespawning movement (ODFW 1990a). Cool water released from the dams during the summer may delay migration timing in the Middle Fork, but the scale of this effect has not been well quantified.

We found little migration timing information for Calapooia or Molalla populations.

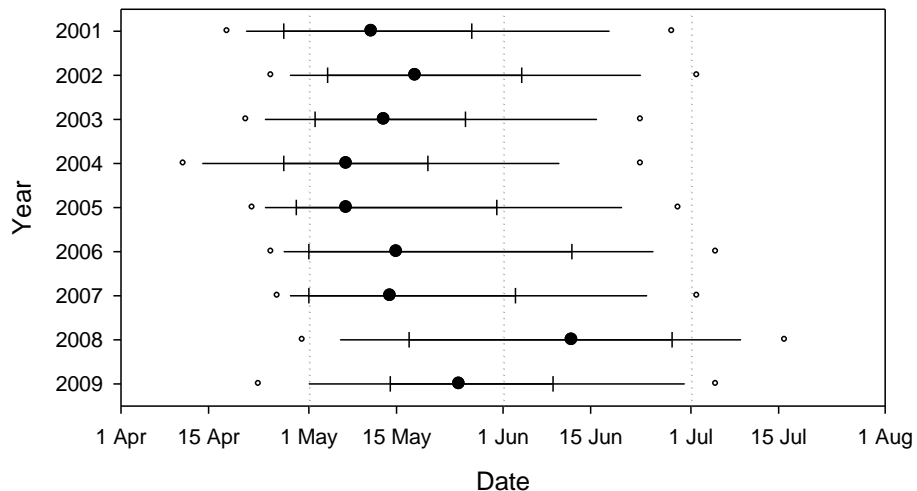


Figure 9. Annual migration timing distributions for spring Chinook salmon counted at Willamette Falls, 2001-2009. Symbols show median (●), quartile (vertical lines), 10th and 90th percentiles (ends of horizontal lines), and 5th and 95th percentiles (○). Data summarized from ODFW daily counts (http://www.dfw.state.or.us/fish/fish_counts/).

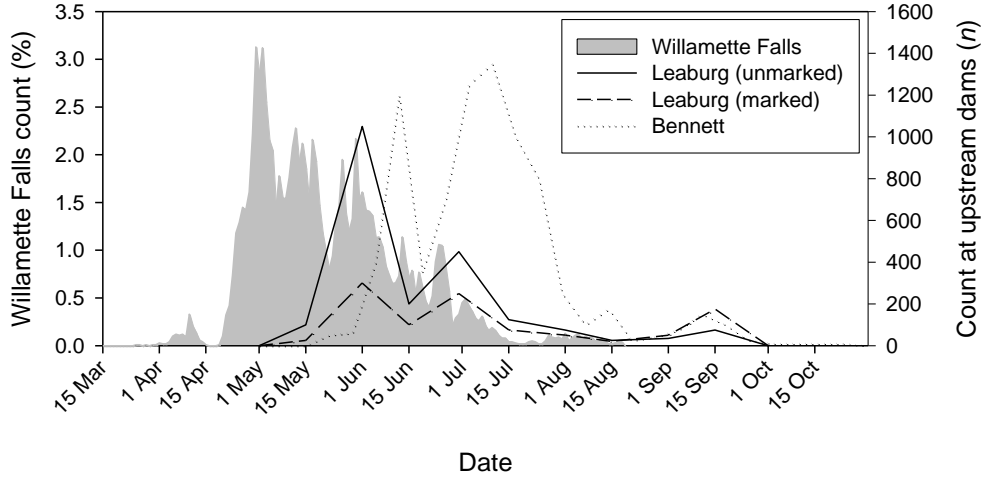


Figure 10. Counts of adult spring Chinook salmon at Willamette Falls, Leaburg Dam (McKenzie River), and upper and lower Bennett dams (South Santiam) in 2002. Willamette Falls data from ODFW daily counts (http://www.dfw.state.or.us/fish/fish_counts/). Upstream counts were extrapolated from figures in Firman et al. (2002).

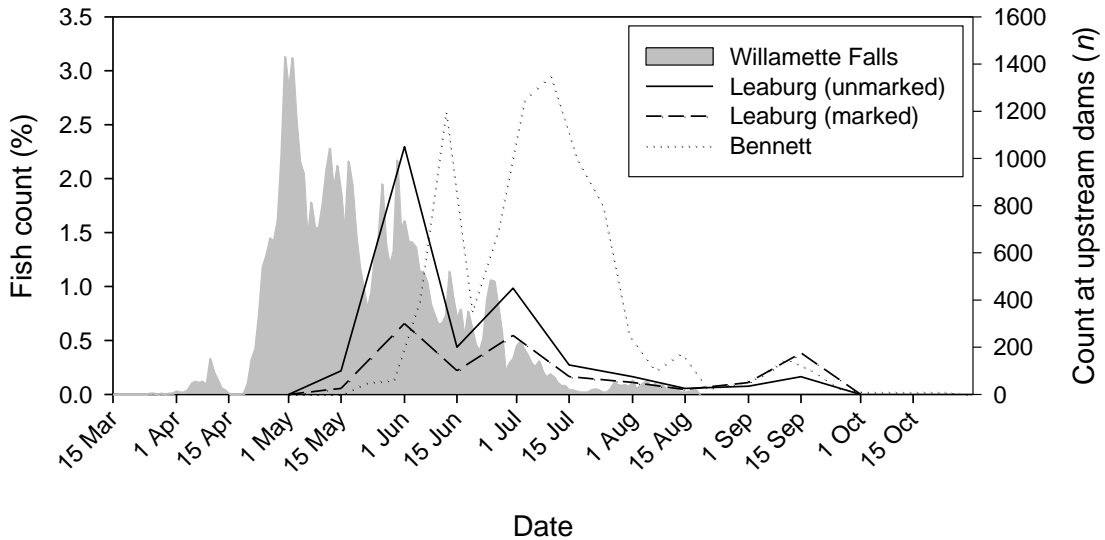


Figure 11. Counts of adult spring Chinook salmon at Willamette Falls, Leaburg Dam (McKenzie River), and upper and lower Bennett dams (South Santiam) in 2002. Willamette Falls data from ODFW daily counts (http://www.dfw.state.or.us/fish/fish_counts/). Upstream counts were extrapolated from figures in Firman et al. (2004).

2.2.4 RUN COMPOSITION

We found little information on in-season spring Chinook salmon run composition (i.e., the relative abundance of sub-basin populations) in the lower Willamette River, at Willamette Falls, or in upstream

reaches. Instead, run composition and relative population size has typically been inferred from adult returns to hatcheries, traps, and spawning grounds (e.g., Cramer et al. 1996). Inference from escapement data does not provide information on relative population abundance early in migration, particularly if some populations are more or less susceptible to mortality prior to reaching collection sites.

A general assumption in the Willamette basin, perhaps justifiably given the dominant hatchery lineage, is that stock-specific passage by spring Chinook run is temporally well-mixed. However, this has not been clearly established from tagging studies or genetic sampling. It is also probable that there is greater among-population variability in run timing for naturally-produced fish (because there has been no equivalent to the unidirectional hatchery selection for early-timed migrants), and consequently run composition presumably is more variable both within and among years for this group. A better understanding of run composition through time would be useful, particularly if there are situations where system operations can be modified to assist vulnerable groups.

2.2.5 SPAWN TIMING

At the basin-wide scale, spring Chinook salmon spawning occurs from early to mid-August through October (McElhany et al. 2007; NMFS 2008). In general, in-river spawning is believed to occur later than historically in most sub-basins and is more temporally concentrated, particularly in the McKenzie and Clackamas River basins, as a result of artificial selection and changes to river discharge and temperature (Howell et al. 1988; Willis et al. 1995; McElhany et al. 2007). Variability among sub-basin populations is related to prevailing water temperatures and the distribution and relative abundance of hatchery fish spawning in the wild versus naturally-produced spawners. Within sub-basin, spawn timing also differs in response to environmental conditions, with different timing in some secondary tributaries and downstream from flow-control dams that affect water temperature. There have been anecdotal reports of spawning initiation following September rain events at several sites.

In the Clackamas River and Eagle Creek, a major spawning tributary, spawning generally occurs from late August through October (Willis et al. 1995; Clackamas HGMP 2002). In spawning ground surveys, dates of 50% spawning have been reported from mid- to late September (Grimes et al. 1996; Lindsay et al. 1997, 1998). There is often temporal overlap between spring and fall Chinook salmon spawning in the main stem Clackamas River. (As an aside, an early-spawning spring Chinook salmon population believed to have spawned in early to mid-July is no longer present in the Clackamas sub-basin [Willis et al. 1995]).

Spawn timing in the North and South Santiam sub-basins generally starts in early August, with peaks in mid- to late September. Dates of 50% spawning have often been in the third or fourth week of September in the upper North Santiam basin, but have been in early October in lower river reaches influenced by relatively warm water releases (Homolka and Downey 1995; Grimes et al. 1996; Lindsay et al. 1997, 1998; Firman et al. 2002). Although dam operations and trapping schedules affect collection of adults in the North Santiam, Willis et al. (1995) concluded that there has not been a significant change in spawn timing in the upper reaches of that basin, at least since the early 20th century.

Spawn timing in the Middle Fork Willamette and McKenzie rivers appears to begin slightly later than in the Santiam basin, with initiation generally in late August to mid-September (Homolka and Downey 1995; Lindsay et al. 1998; Firman et al. 2002). Peak spawning has been reported primarily in late September and early October, and some McKenzie fish spawn through late October (Homolka and Downey 1995; Willis et al. 1995). Dam effects on water temperature are believed to have affected spawn timing in these basins, though the magnitude, directionality and year-to-year variability of the effects have not been well studied.

2.3 ADULT MIGRATION BEHAVIOR

There have been relatively few adult tagging studies in the Willamette basin and consequently there are few reports that describe spring Chinook salmon behaviors during active migration, at or near dams, or on spawning grounds. Much of the behavioral information described below was collected in the radiotelemetry study of Schreck et al. (1994), which was one of the few active tag studies in the basin. In addition, adult spring Chinook salmon were jaw-tagged or floy-tagged at or near Willamette Falls in a multi-year hooking mortality study described in Lindsay et al. (1997, 1998, 2000, 2004; Schroeder et al. 1999). More recently, there have been some small-scale radiotelemetry, floy tag, and passive integrated transponder (PIT) tag studies associated with adult spring Chinook salmon outplanted from hatcheries and traps (e.g., Keefer et al. 2010, and *unpublished data* from G. Taylor and D. Garletts [USACE] and R. Mann and C. Caudill [University of Idaho]). A PIT-tag interrogation site was installed in the Willamette Falls fishway in 2006 and small numbers (i.e., < 50/year total) of adult salmon and steelhead have been detected. Data from PIT-tagged fish is expected to provide important juvenile-to-adult survival and migration timing information, but the single interrogation site (thus far) can be expected to provide only limited behavioral data. The addition of PIT tag detection sites (or manual scanning) at upstream traps would allow estimation of passage times from Willamette Falls to these sites.

2.3.1 MIGRATION RATES

A total of 224 adult spring Chinook salmon were radio-tagged and released into the Willamette Falls fishway in the four years of the Schreck et al. (1994) study. The authors arbitrarily separated the tagged fish into early (April), middle (May), and late (June) components based on the timing of the run at the falls and some basic behavioral differences. Migration rates varied widely among years and within year, and differences were associated with a variety of environmental conditions.

Early-run fish migrated relatively slowly, on average. Migration rates ranged from 6 to 26 km/d over a variety of reaches (primarily starting from the release Willamette Falls release site). The telemetry data showed considerable variability in the early group, with many salmon holding in the main stem upstream from Willamette Falls, others moving downstream, and a few moving rapidly upstream. The middle group, associated with peak passage, showed more consistent directed upstream movement. Typical migration rates were between 25 and 40 km/d for this group, though slower passage was observed once fish exited the lower main stem Willamette River (i.e., passed the Santiam River confluence). Late-run salmon also migrated relatively rapidly initially, with typical rates between 16 and 40 km/d and as high as 59 km/d. As the season progressed, the late-run group had a wider variety of behaviors, including extended holding and downstream movements that were similar to the early-

run fish in some years. These slow passage behaviors were associated with elevated mortality rates across study years (Schreck et al. 1994).

Analyses of environmental effects on salmon migration rates in the Schreck et al. (1994) were qualitative. They suggested that many salmon stopped or significantly slowed migration during periods of high discharge (i.e., flood events) and low water temperature, although fish responses were not entirely consistent across years. The authors also suggested that water clarity may have affected the willingness of fish to migrate throughout diel cycles, with slowed movement during highly turbid conditions and restricted daytime movements during periods of low turbidity. Although in-river conditions appear to have a significant effect on spring Chinook salmon migration timing, behaviors, and upstream passage, we found almost no other reports on these effects.

In the multi-year hooking mortality study summarized by Lindsay et al. (2004), median times from initial capture at or near Willamette Falls to recovery at hatcheries or traps throughout the Willamette basin were about 60 days (*range* = 10–168 days). Median time from capture to recovery on spawning grounds was 146 days (*range* = 118–168 days). These times incorporate all active migration plus holding prior to entering traps or hatcheries. As such, they represent maximum travel times and minimum migration rates. Calculating migration rates was not a principal objective of the mortality study, and neither annual reports (e.g., Lindsay et al. 1997, 2000) nor the peer-reviewed publication (Lindsay et al. 2004) provided times to specific final locations. However, it may be possible to revisit this dataset to calculate more site- and population-specific travel times.

2.3.2 PASSAGE AT AND NEAR DAMS

We found very few data on adult spring Chinook salmon behavior in dam tailraces, in fishways, or in reservoirs. A general exception was that adult trapping facilities have been evaluated as part of the BiOp process (NMFS 2008), with an emphasis on fish handling protocols, efforts to minimize injuries, and fish separation issues (i.e., separating hatchery from wild stocks and broodstock from surplus or outplant fish). There appears to be very little information about upstream migration delay at dams, fishway entrance and/or trap collection efficiencies, passage or collection failure, or the effects of either environmental conditions or dam operations on adult behavior or passage success. Passage failure and delay at dams have a variety of potential impacts on adults, including reduced escapement, increased inter-basin straying, downstream displacement of spawners, and direct mortality. Passage delay can also indirectly affect prespawn mortality by affecting migration timing, altering temperature exposure (i.e., degree day accumulation), reducing energetic reserves, lowering fish condition, or increasing exposure to pathogens or parasites. Such direct and indirect effects of behavior near dams have not been well studied in the Willamette system.

The radiotelemetry research by Schreck et al. (1994) provided some information on spring Chinook salmon passage and behavior at Willamette Falls. They noted that some fish used several days to more than a week to pass through fishways at the falls, while others failed to pass and instead moved downstream. Passage times at this site were longest during flood events when turbidity and discharge were high and water temperature was low or decreasing (Schreck et al. 1994). The radiotelemetry study also provided some anecdotal information, based on behaviors of a few fish, about passage delay downstream from the Bennett dams (North Santiam) during low-flow conditions.

2.3.3 FALLBACK

Downstream movement past dams by adult salmon, a behavior termed ‘fallback’, has been associated with fish injury and migration failure at a variety of locations in the Willamette and Columbia River systems. At sites where escapement estimates are based on adult fish counted in ladders or traps, prespawn fallback can lead to escapement overestimation. Counting errors are exacerbated when fish fall back and then reascend ladders one or more times (i.e., they are counted multiple times). Adult fish have been recorded falling back via a variety of routes at dams, including through turbines, over spillways, down fishways, and through a variety of fish and trash bypass systems. At most sites where fallback has been monitored, fallback rates differ in response to environmental and operational conditions. Fallback rates also tend to be higher when spawning areas, tributaries, or hatcheries are located in close proximity to dams.

There has been little systematic study of adult fallback in the Willamette River system despite probable expression of the behavior at most sites where adult salmon can volitionally pass upstream and where adults are trapped and released upstream from dams. Spring Chinook salmon fallback rates have been estimated intermittently at Willamette Falls on the main stem, at upper and lower Bennett dams (North Fork Santiam), at Leaburg Dam (McKenzie), and at North Fork Dam (Clackamas). In general, reported data in the reviewed literature were limited to tagging study results from a small number of years at each site. In some cases, estimation methods were poorly described.

At Willamette Falls, annual spring Chinook salmon fallback rates in the radiotelemetry study by Schreck et al. (1994) were reported as 10%, 12%, and 4%. Fallback was highest during the early portion of the runs, when discharge was higher. Salmon that fell back had a variety of subsequent behaviors that included return to the Columbia River, entry into downstream tributaries (i.e., the Clackamas River), and reascension at Willamette Falls (Schreck et al. 1994). Other reports of fallback at Willamette Falls have largely been anecdotal (e.g., Lindsay et al. 1997). Understanding variability in fallback patterns at the falls is important because counts there affect management decisions for the aggregated upriver populations.

In the North Santiam sub-basin, salmon have been marked with caudal fin punches to evaluate fallback and reascension, particularly at upper and lower Bennett dams (e.g., Lindsay et al. 1998). It is not clear how often this method has been applied, but fallback-adjusted count corrections have been reported in several publications (e.g., Lindsay et al. 1997; Firman et al. 2002, 2004, 2005; Schroeder et al. 1999, 2002). Annual fallback corrections for spring Chinook fallback at the Bennett dams have ranged from 2–5%. Researchers have also reported seasonal fluctuations in fallback at the Bennett dams. For example, monthly fallback estimates ranged from 0–8.2% in 2003 (Firman et al. 2004). It should be noted that fallback estimates based on external marks tend to underestimate fallback because there is a bias towards identifying only those fish that survive and reascend, unless there is a significant downstream collection effort.

Fallback at Leaburg Dam on the South Santiam River has been indirectly estimated using the ratios of fin-clipped to unclipped fish counted passing the dam and ratios of those counted during spawning ground surveys upstream. This method is used because the majority of the fallback at this dam is believed to be by hatchery fish that have migrated upstream past the McKenzie Hatchery (~3 km downstream from Leaburg Dam) and subsequently fall back downstream to reach the hatchery

(Schroeder et al. 2004, 2007; Firman et al. 2005; McLaughlin et al. 2008). The reviewed reports provided little explanation of the estimation method, of fallback rates for non-hatchery fish, or of potential biases in the approach.

2.3.4 HOLDING

An extended holding period prior to spawning is part of the life history of spring Chinook salmon, because they enter freshwater and then reside and mature for several months before spawning. Typically this holding occurs in or near eventual spawning areas in headwater streams. In several Willamette River tributaries, however, the holding period has been temporally and spatially altered as a result of impassable dams. For example, salmon can spend weeks to more than a month holding below dams on the Middle Fork Willamette, South Santiam, and North Santiam rivers, well downstream from headwater spawning areas. Prespawn mortality while holding below dams can be relatively high (e.g., Beidler and Knapp 2005; McLaughlin et al. 2008). In addition, there are likely indirect and delayed effects on maturation and physiology that manifest closer to spawn time, contributing to the often high mortality among adults outplanted upstream from dams.

There has been little direct evaluation of either spatial or temporal aspects of salmon holding behavior in the Willamette basin, or of holding effects on adult physiology or survival to spawning. Many radio-tagged spring Chinook salmon in the Schreck et al. (1994) study held in large, deep pools in the main stem Willamette, Santiam, and McKenzie rivers for days at a time. The authors attributed this behavior to staging prior to spawning and to adverse river conditions that included high discharge and turbidity during flood events. A group of radio-tagged fish also stopped migration in the North Santiam River near Lower Bennett Dam when there was insufficient flow to allow passage to the fish ladder. Schreck et al. (1994) also recorded more frequent, short-duration holding downstream from difficult passage areas and near river confluences and temperature gradients. In some instances, upstream movement slowed or stopped during daylight, with fish holding in pools below riffles. More generally, extended holding by spring Chinook salmon has been observed downstream from several tributary dams. The behavior has been associated with trap operation timing (i.e., below Dexter Dam on the Middle Fork Willamette River) and with water temperature changes related to dam operations. However, these observations have been anecdotal rather than quantitative.

2.3.5 STRAYING

Some inter- and intra-basin straying is a natural feature of most anadromous salmonid populations. There have been a variety of spring Chinook salmon straying evaluations in the Willamette basin, primarily associated with hatchery protocols (i.e., rearing and releases strategies) as well as with straying of hatchery fish into natural spawning populations. Stray rates have been calculated based on tag returns (primarily coded-wire tags [CWT]), from fin clips, and from otoliths marked during juvenile rearing at hatcheries. Recoveries at hatcheries and in fisheries have also been used in some analyses.

In the 1970s, adult straying in the Willamette was linked to trucking of juvenile spring Chinook salmon from upriver hatcheries to releases sites downstream from Willamette Falls (e.g., Smith et al. 1985; Cramer et al. 1996). This result was consistent with results from a suite of juvenile salmon transportation studies in the Columbia River basin and elsewhere. More recently, a multi-year study

evaluated adult straying by hatchery salmon from McKenzie, South Santiam, and Middle Fork stocks that were released as juveniles at a variety of local (i.e., within sub-basin) and downstream locations (Schroeder and Kenaston 2004; Schroeder et al. 2005). Based on CWT data, they found that adult straying was much higher for groups that were released into sites outside their local sub-basins, and especially for fish released into the lower Willamette River. The latter group included direct releases and fish that were held in acclimation pens, with higher straying by the direct-release group (Schroeder and Kenaston 2004). Straying adults entered the Clackamas River as well as many upper Willamette tributaries and stray rates were >50% in some cases (Schroeder et al. 2005).

Relatively high stray rates have also been associated with salmon transfers among sub-basin hatcheries (Cramer et al. 1996). Out-of-basin juvenile rearing has been convoluted at times, and mixed imprinting for these fish presumably contributes to adult straying behaviors. As example results, adult strays into the North Santiam were from South Santiam Hatchery releases into the Molalla sub-basin (Schroeder et al. 2005, 2007) and strays in the McKenzie River have been from Clackamas stock reared in the McKenzie River but released in the lower Willamette (Lindsay et al. 1998). Similarly, juvenile salmon reared in upper Willamette River hatcheries and then released into the Clackamas River tend to stray at relatively high rates.

Spawning ground surveys have been another primary source of spring Chinook salmon straying information in the basin. In addition to identifying strays based on tags (i.e., coded wire tags), inter- and intra-basin strays have been identified by fin clips and thermally-marked otolith analyses. The distribution and abundance of hatchery fish on spawning grounds were evaluated as part of the multi-year ODFW spawning ground studies (e.g., Schroeder et al 2007; McLaughlin et al. 2008). In some cases, these types of spawning ground surveys can identify hatcheries of origin, including from hatcheries in other sub-basins. They have also been used to show the proportion of adults on spawning grounds that were of local hatchery origin (i.e., they did not return to the hatchery trap but rather spawned in the wild) (e.g., Willis et al. 1995). We note that the biological implications of ‘straying’ at within-basin and inter-basin scales can be quite different.

In general, inter-basin straying by spring Chinook salmon is believed to be relatively low (i.e., <10%) among naturally-produced populations. This appears to be true for wild Willamette sub-basin populations, based on genetic analyses and recoveries of tagged fish (Cramer et al. 1996; Myers et al. 2006; McElhany et al. 2007). However, as described above, stray rates can be relatively high for hatchery fish, and within-basin straying by hatchery fish into wild spawning aggregates has been very high (i.e., >60%) in some Willamette River cases (e.g., Willis et al. 1995; Cramer et al. 1996). There is a need to better quantify the degree of straying at these different scales and a need to identify factors – such as hatchery collection efficiencies for adults, rearing or release strategies for juveniles, or migration timing effects such as river environment – that affect the behaviors.

2.3.6 THERMAL REFUGIA USE

Main stem Willamette River water temperatures are routinely greater than 21° C during the summer and exceed 24° C in some reaches in some years (USGS gage data; NMFS 2008). Although most spring Chinook salmon exit the main stem prior to peak summer water temperatures, some fish presumably encounter temperatures near thermal limits. Such conditions have been associated with behavioral thermoregulation in adult spring Chinook salmon and a variety of other anadromous

salmonids in many river systems. Nonetheless, we found few data related to the thermal experiences of adult migrants in the Willamette River or its tributaries. We expect that the proportion of the spring run that encounters high temperatures varies considerably among years and differs among tributary populations given relatively large inter-annual differences in snow pack, winter rain, river discharge, elevation, and other hydrogeomorphic features.

Some of the holding behaviors reported for late-run spring Chinook salmon, such as extended use of deep pools, are presumably a response to unfavorable temperatures. Two small-scale radio-tag studies, including one that used ten archival temperature data storage transmitters, found limited evidence for refugia use by migrating spring Chinook salmon but also did not report the availability of refugia in the study area (EA Engineering, Science, and Technology 1991; Schreck et al. 1994). More recent temperature monitoring in the Middle Fork Willamette River has indicated that there is considerable thermal heterogeneity related to secondary tributaries and springs (USACE, *unpublished data*), and use of these variable habitats by adult spring Chinook salmon is being evaluated. Despite their potential importance for juvenile and adult salmonids in the Willamette basin, the spatial distribution of thermal refugia and their use by salmonids have been inadequately described. Given the loss of channel complexity and riparian canopy cover in the basin, thermal refugia are believed to be more spatially and temporally restricted than historically, with unknown consequences for adult migrants (Benner and Sedell 1997; PNW-ERC 2002; Rounds 2007; Seedang et al. 2008).

2.4 PRESPAWN MORTALITY

Adult prespawn mortality can significantly affect production, particularly in small natural-spawning populations. In the Willamette basin, adult mortality in the migration corridor, during prespawn holding downstream from dams, and on the spawning grounds can be substantial in some years. Prespawn mortality has been reported from almost all sub-basin populations, and there has been some recent research effort to describe inter-annual and inter-basin variability, to identify causal effects, and to relate the mortality to environmental and operational conditions. To date, the focus of prespawn mortality research has predominantly been on spawning grounds.

2.4.1 MIGRATION AND HOLDING MORTALITY

There have been few direct estimates of *en route* migration mortality for adult spring Chinook salmon in the Willamette basin. However, in some years far fewer fish are counted returning to hatcheries, traps, and on spawning grounds than would be expected from the adult counts at Willamette Falls, even after correcting for harvest rates in the main stem and in tributaries. In the radiotelemetry study of Schreck et al. (1994), it was estimated that non-harvest mortality during migration and holding downstream from spawning areas ranged from 20–40%. Estimates were based on recoveries of radio-tagged carcasses and from observations of untagged carcasses in non-spawning areas. Spatial and temporal patterns differed among years, with substantial migration mortality recorded as early as early May in some years, and greater mortality in June and July in all years. Carcasses were recovered at a variety of main stem locations, from sites up and downstream from Willamette Falls (also reported by Foster [1997]) and upstream to near Salem and Eugene (Schreck et al. 1994).

In the radiotelemetry studies, migration mortality was associated with both specific salmon behaviors and river environment. For example, adult Chinook salmon that moved downstream and those that held position for extended periods rather than moving upstream were more likely to die before reaching tributaries. High mortality was also associated with low discharge and warm water temperatures, with greater mortality in warmer years (Schreck et al. 1994). These authors concluded that fungal growth and other disease-related processes may have been accelerated at higher temperatures and likely contributed to the migration mortality.

2.4.2 SPAWNING GROUND MORTALITY

Prespawn mortality on spawning grounds has been evaluated for several years during carcass surveys upstream and downstream from tributary dams. Point estimates have been well summarized by Schroeder et al. (2007) and Kenaston et al. (2009) and show that mortality rates vary widely among years and among sub-basins (Figures 12 and 13). On average, recent prespawn mortality rates in Willamette basin spawning areas have been at the high end of the range for the species and for spring runs. During the last decade, prespawn mortality greater than 50% has been reported in the Clackamas, North Santiam, South Santiam, McKenzie, and Middle Fork Willamette rivers (Schroeder et al. 2007; McLaughlin et al. 2008; Kenaston et al. 2009). Prior to 2000, carcass survey data were somewhat less systematically collected, but high mortality rates were recorded for a variety of sites (e.g., Cramer et al. 1996; Grimes et al. 1996; Foster 1997; Lindsay et al. 1998). The combined data suggest that relatively high prespawn mortality is common in the basin, perhaps as a result of systemic changes (i.e., dams and other water management activities) or because the basin is relatively low elevation and in the southern portion of the spring Chinook range.

There has been little systematic evaluation of the factors that affect inter-annual variability in prespawn spawning ground mortality. A variety of factors may be important, ranging from temperature effects on disease prevalence and severity, maturation timing, physiology (i.e., condition), handling effects at traps, or transportation effects (for outplanted adults). In a four-year USACE radiotelemetry study, prespawn mortality rates in the Middle Fork Willamette River were highest in years with warm water temperatures (Keefer et al. 2010), consistent with temperature effects that have been implicated in other spring Chinook salmon studies in the Willamette basin (e.g., Schreck et al. 1994; Beidler and Knapp 2005; Caudill and Mann *unpublished data*). The relatively low prespawn mortality rates recorded at most carcass survey sites in 2006 and 2008 (Figure 12) also suggest an overall river environment effect; these years had relatively high discharge and low water temperature.

Relatively low mean prespawn mortality rates in the Clackamas and McKenzie River basins, where fish have fewer passage barriers to spawning areas, may suggest that holding below dams or trapping activities in other sub-basins increase adult mortality. The generally higher prespawn mortality downstream versus upstream from dams (e.g., McLaughlin et al. 2008) also suggests that the holding effect may be related to prevailing environmental conditions or to differences in overall habitat quality between these sites. Concentrations of fish below dams may also increase disease transmission rates, affect stress levels or maturation rates, or have other direct or indirect effects on physiology and mortality.

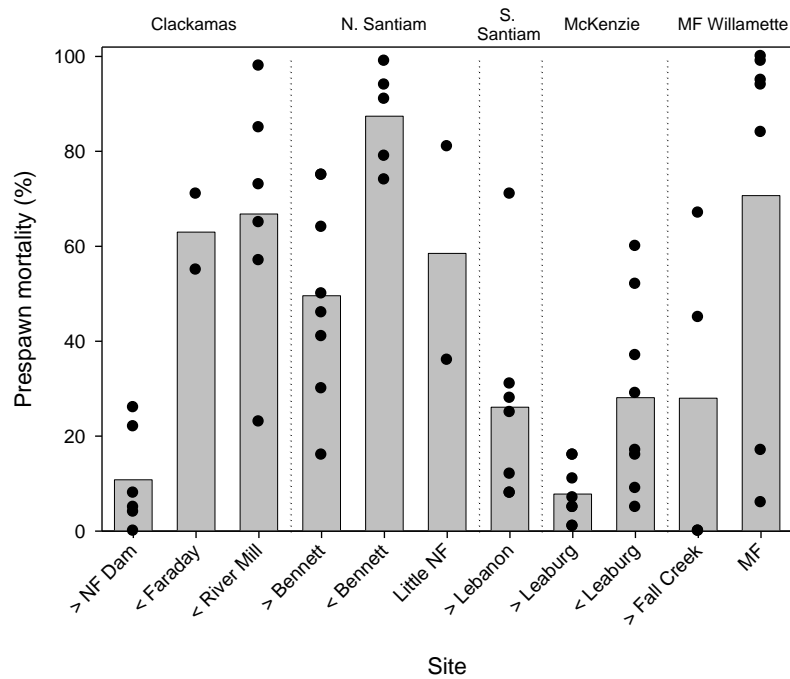


Figure 12. Mean (bars) and annual (●) pre-spawn mortality estimates for female spring Chinook salmon examined in carcass surveys from 2001-2008, by survey site. Data are from Schroeder et al. (2007) and Kenaston et al. (2009). Some individual estimates are from surveys that began late or ended prior to the end of peak spawning time; see original documents.

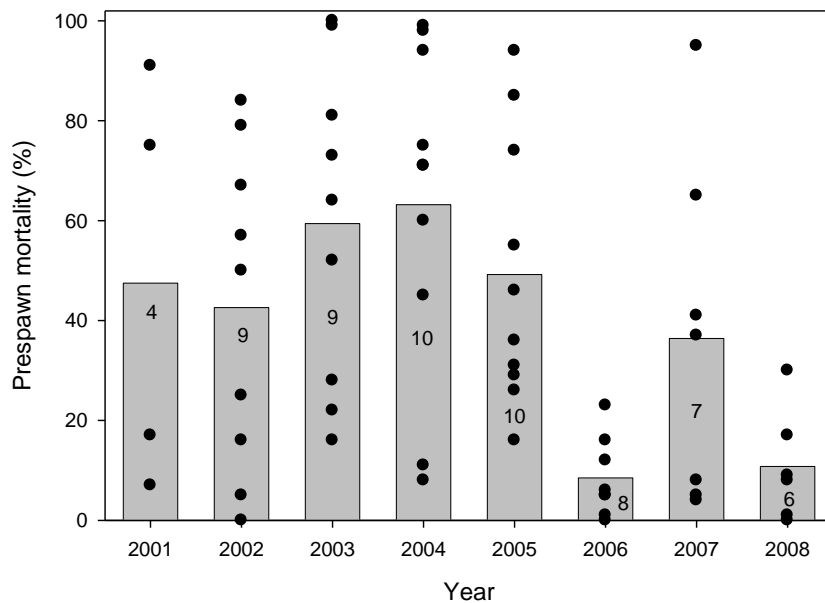


Figure 13. Mean (bars) and annual (●) pre-spawn mortality estimates for female spring Chinook salmon examined in carcass surveys from 2001-2008, by year. Numbers inside bars are the number of survey sites within year. Data are from Schroeder et al. (2007) and Kenaston et al. (2009). Some individual estimates are from surveys that began late or ended prior to the end of peak spawning time; see original documents.

2.4.3 MORTALITY SOURCES

Small numbers of prespawn spring Chinook salmon mortalities were necropsied in Willamette basin tributaries in 2004 and 2008 (Oregon State University Pathology Laboratory, reported in Beidler and Knapp 2005; Schreck et al. *unpublished data*). Pathology results indicated severe fungal infections in most fish, plus a variety of bacterial infections (e.g., *Aeromonas* and *Pseudomonas* sp. [furunculosis] and *Renibacterium* sp. [BKD]), lesions, and parasites (e.g., copepods, *Nanophyetus*, *Ceratomyxa shasta*). In 2008, the prespawn mortalities showed more severe visceral and gill lesions, and had higher intensity of infections by several parasites than fish that survived to spawn. Most notable infections in the prespawn fish were very heavy infections by metacercariae of *Apophallus* sp. (Digenea) in the gills, heart infections by *Nanophyetus*, and severe *Ceratomyxa shasta* infection, which were likely the cause of death. While fungal or bacterial infections have been implicated as causal agents in several prespawn mortality evaluations in the basin (e.g., Foster 1997; Beidler and Knapp 2005; Schreck et al. *unpublished data*), we note that infection timing, individual susceptibility, transmission rates, disease progression, and the roles of environmental and operational effects remain poorly understood at almost all sites.

Outside the Willamette River basin, *en route* migration mortality and prespawn mortality on spawning grounds have been associated with a variety of additional physiological conditions. These include low energy reserves, elevated stress indicators, advanced reproductive development, and low osmoregulatory preparedness. We found no published reports indicating that these types of physiological indicators have been examined on Willamette basin spring Chinook salmon, although some data have been archived for hatchery and natural production broodstock fish (Tom Friesen, ODFW, *personal communication*). There is also a pilot study of energetic status currently underway in the Middle Fork Willamette River (Caudill et al. *unpublished data*).

2.5 OUTPLANTING

In the late 1980s and early 1990s, ODFW began outplanting surplus hatchery adult spring Chinook salmon and some naturally-produced fish above barrier dams in the Willamette system. The program was developed to supplement natural production, to re-establish self-sustaining, locally-adapted populations, and to restore some of the ecological functions of adult salmon (e.g., marine-derived nutrients) to tributaries (Lindsay et al. 1998; Mamoyac and Ziller 2001; Beidler and Knapp 2005; Taylor and Garletts 2007). Primary release locations were initially above Minto Dam on the North Santiam River (starting in 1986), above Foster Dam on the South Santiam River (1996), above Cougar Dam on the McKenzie River (1996), and above dams on the Middle Fork Willamette River (1993). The outplant program expanded to include releases into habitat downstream from some dams and into areas where satellite populations had been extirpated. There has also been an increased focus on outplanting natural-origin fish at sites where separation is possible.

Outplant data summarized by Firman et al. (2002, 2004, 2005) and McLaughlin et al. (2008) for several recent years show outplanting programs have operated from Marion Forks, South Santiam, Dexter, McKenzie, and Leaburg facilities (Table 3). Releases during these years occurred in at least 20 locations, including into secondary tributaries and across sub-basins (i.e., into the Calapooia and Coast Fork basins). Over six years, counts of outplanted adults ranged from approximately 15,500 to 23,700 per sub-basin, the equivalent of several thousand fish per basin per year (Table 3).

Table 3. Summary of the numbers of adult spring Chinook salmon captured at hatchery and trap facilities and then outplanted into potential spawning areas, 2002-2006. Data from Firman et al. (2002, 2004, 2005) and McLaughlin et al. (2008).

Hatchery	Release location	Year						Total
		2002	2003	2004	2005	2006	2007	
Marion Forks	North Santiam above Detroit	2,677	2,914	2,511	528	1,123	574	10,327
	North Santiam above Minto	729	203	144	30	758		1,864
	Little North Fork Santiam River	399	2668	377	329	130	195	1,695
	Breitenbush River				86	720	420	1,226
	Recycled Downstream	69	102	216				387
South Santiam	South Santiam downstream	2,959	3,258	5,551				11,768
	South Santiam above Foster	765	556	2,580	936	932	403	6,172
	Wiley Creek	546	101	247	166			1,060
	Thomas Creek	461	155	237	193	256		1,302
	Crabtree Creek	359	175	246	143	180		1,103
	Calapooia River	351	140					491
Dexter	Middle Fork Willamette	3,330	1,498	2,011	1,052	694	176	8,761
	North Fork of Middle Fork	3,737	1,697	2,703	798	827	555	10,347
	Lost Creek	20		405				425
	Salt Creek	1,367	631	1,192	405	381	72	4,048
	Mosby Creek (Coast Fork)					119	43	162
McKenzie	McKenzie River	73	33	6			17	129
	Mohawk River	197		137	33	265	191	823
	South Fork McKenzie above Cougar	4,242	2,981	3,430	863	1,018	743	13,277
	Trail Bridge reservoir (or upstream)	99	141	122	116	116	128	722
Leaburg Trap	McKenzie River above trap		609					609
	Mohawk River	20	125					145
	South Fork McKenzie River	619	904	9				1,532
	Trail Bridge reservoir	34						34

Despite the scale of the outplant program, analyses of program effects have been relatively limited. Beidler and Knapp (2005) provided a fairly comprehensive review of program activities through 2004, including a summary of information needs. They noted that there was limited data on prespawn mortality, fish condition, disease issues, delayed mortality, redd construction, spawning success, or juvenile production. They also suggested that more information was needed to evaluate the effects of salmon handling and transportation, the effects of river conditions (i.e., temperature and flow), and effects of release locations and release timing (Beidler and Knapp 2005). At sites where outplant mortality rates have been evaluated, mortality has varied considerably among years and among release sites. Some of the variability has been attributed to holding times downstream from impassable dams, the efficiency of collection facilities, transportation distance, and the quality of habitat at release sites (Beidler and Knapp 2005; ODFW 2007). Qualitatively, the outplant program in the McKenzie sub-basin has been relatively successful, while those in the Middle Fork Willamette and South Santiam sub-basins have been somewhat less successful (as measured by fish:red ratios and prespawn mortality surveys).

Overall, data collection and record keeping for the spring Chinook salmon outplant program have been irregular, with few data collected on fish characteristics or in-river conditions that may affect survival to spawning or reproductive success. An exception is the Cougar genetics pedigree study, which has been collecting morphometric data and genetic samples for outplanted adults in the South Fork McKenzie River. In any case, there should be opportunities to retrospectively evaluate some of the outplant data, and there is also considerable room for improved evaluation going forward.

3.0 WINTER STEELHEAD

3.1 DISTRIBUTION AND STATUS

Long-term trends in adult returns have been negative for the aggregated native winter steelhead run upstream from Willamette Falls and for most individual sub-basin populations (Kostow 1997). The recent exception is that there were a few strong returns in the 2000s associated with good ocean conditions (NMFS 2008; McElhany et al. 2004). Among the four core sub-basin populations upstream from Willamette Falls, the Molalla, upper portion of the South Santiam, and Calapooia stocks are considered to be at relatively high risk of extinction, while the North Santiam and lower portion of the South Santiam stocks are somewhat more stable (Nehlsen et al. 1991; Chilcote 1998; NMFS 2006). Downstream from the falls, the Clackamas River population is considered stable. Declines in the native winter steelhead populations both upstream from Willamette falls and in the Clackamas sub-basin have been associated with a mix of poor outmigration conditions for juveniles and post-spawn kelts, low productivity, and the negative effects of introduced summer steelhead (i.e., Skamania stock) and early-run winter steelhead (i.e., Big Creek stock). In addition, elevated temperatures and high *Ceratomyxa shasta* infection rates have occasionally been implicated in low adult survival.

Winter steelhead populations above and below Willamette Falls have had varying degrees of hatchery influence (Howell et al. 1985). In addition to native winter stocks, there is an upper Willamette hatchery stock derived from local winter-run fish (mostly from the North Santiam), a similar, mostly locally-derived Clackamas River winter stock, and the Big Creek hatchery stock developed from lower Columbia River winter-run populations. The presence of hatchery steelhead, of both summer- and winter-run origin, can potentially negatively impact native winter-run populations in the Willamette basin (e.g., Chilcote 2003, 2007; Kostow and Zhou 2006).

The following brief descriptions of the status of each sub-basin winter steelhead population and their basic hatchery history were gleaned from population summaries developed by the HSRG (2009) and from the status review material in Chilcote (1998), McElhany et al. (2004, 2007), and NMFS (2008). We note again, however, that a full consideration of hatchery effects was beyond the scope of this review. (Note: Appendix 1 includes sub-basin maps that show locations of dams and tributaries.)

3.1.1 CLACKAMAS

Returns of adult winter steelhead to the Clackamas River have ranged from about 500 to several thousand fish in recent years. It is considered a stable population, and the component above North Fork Dam is managed for native production by separating returning adult hatchery from wild fish at the dam. Clackamas River winter steelhead have been significantly influenced by introduced Big Creek hatchery fish (early run) and hatchery production of a local, late-winter population developed at Eagle Creek Hatchery (Howell et al. 1985). Although the hatchery contribution is important downstream from North Fork Dam, the majority of the production in the Clackamas basin has been natural for the last decade or more, with wild fish spawning in secondary tributaries downstream from North Fork Dam and in the wild fish sanctuary above the dam (Murtagh et al. 1992; Chilcote 1998; Hutchison et al. 2007).

3.1.2 MOLLALA

Estimated adult returns to the Mollalla River basin have been in the hundreds to low thousands of fish historically and in the last decade, but dropped below 100 fish in some years in the 1990s (Goodson 2005 [cited in McElhany et al. 2007]). The native population is considered relatively stable, but may be vulnerable as evidenced by the low native fish returns in the 1990s. Although Big Creek winter steelhead were released into the basin for several decades (discontinued in 1997), recent production is believed to be almost exclusively by late-run native stocks. Summer steelhead hatchery releases continue.

3.1.3 NORTH FORK SANTIAM

On average, the North Santiam winter steelhead population has returned 2,000-3,000 adults in recent years. The population was supported by releases of the locally-derived Willamette River winter stock, but this supplementation ended in 1998. Though not rigorously studied, the introduced and wild components of the North Santiam winter steelhead population appear to have similar life history traits. Releases of hatchery summer steelhead and natural production by progeny of summer-run fish continue in the basin and are a winter steelhead management concern.

3.1.4 SOUTH FORK SANTIAM

Winter steelhead returns to the South Santiam have been approximately 2,500 fish in recent years, an increase from the 1990s when returns were about 1,000 fish, on average. As in the North Santiam, releases of hatchery winter steelhead ended in the late 20th century (McElhany et al. 2004, 2007; Goodson 2005). The winter hatchery fish appear to have been from several Willamette basin sources, but it is not clear how much interbreeding there was between hatchery and native stocks. McElhany et al. (2004) noted that there are continuing concerns about introgression between hatchery summer steelhead and native winter steelhead in the South Santiam. Adult separation at Foster Dam allows populations above and below the dam to be managed separately. Extinction risks are considered relatively high for the small native population upstream from Foster Dam, while the population downstream from the dam is relatively stable.

3.1.5 CALAPOOIA

Adult steelhead returns to the Calapooia River have been in the hundreds of fish in recent years, and dropped below 100 fish per year during the 1990s. There have been no (or very limited) direct hatchery releases of winter steelhead in this basin.

3.1.6 MCKENZIE AND MIDDLE FORK WILLAMETTE

The McKenzie and Middle Fork Willamette sub-basins probably did not historically support independent winter steelhead populations, but now have small, apparently self-sustaining populations that were founded by outplants from the Willamette hatchery stock, strays from other populations, and perhaps winter-summer hybrids (Howell et al. 1985; ODFW 1990a). Summer steelhead juveniles reared at several hatcheries continue to be released into both basins and there is some natural

production by summer-run fish (Firman et al. 2005); interactions between summer and winter fish, including interbreeding, have not been well documented.

3.1.7 WEST-SIDE TRIBUTARIES

The small winter steelhead populations in west-side tributaries (i.e., Tualatin, Yamhill, Rickreall, and Luckiamute rivers) have had relatively little direct hatchery influence, but there have been periodic introductions of both Big Creek and Willamette hatchery stocks. It is not clear whether winter steelhead that return to these drainages derived from native or hatchery stocks, and there may be some inter-breeding with non-anadromous populations (Myers et al. 2003; McElhany et al. 2007).

3.2 LIFE HISTORY

3.2.1 EARLY LIFE HISTORY

Almost all winter steelhead in the Willamette basin rear in spawning tributaries or reservoirs for 1–3 years before smolt outmigration. The predominant freshwater residency time is two years followed by three years (Buchanan and Wade 1982; Howell et al. 1985; ODFW 1992; Wevers et al. 1992; Busby et al. 1996; Clackamas HGMP 2002). We found few specific details about reservoir habitat use by juvenile fish, except that extended use has been reported for Clackamas River steelhead in North Fork reservoir (Beamesderfer et al. 2001). Some individual steelhead in the Beamesderfer et al. (2001) report resided in the reservoir for as long as two years prior to outmigration. Rearing may also occur in lower reaches of the primary tributaries and in the main stem Willamette, though there is some evidence that juvenile steelhead primarily use the main stem below Willamette Falls as a migration corridor (e.g., Friesen 2005). The types and variability in steelhead juvenile life history strategies have not been well described in the basin.

In general, winter steelhead smolts outmigrate from late spring through early summer, but there are some fall and winter outmigrants as well (Howell et al. 1985; Friesen 2005). Outmigration timing appears to be related to rearing environment (i.e., longer residency at sites with lower productivity and short residency at sites with stressful summer or fall temperatures) and steelhead size (i.e., larger steelhead tend to outmigrate earlier). Hatchery winter steelhead have been recorded passing Willamette Falls primarily from late April through May, and this period is also an active migration window in several tributaries (Howell et al. 1985; Buchanan and Wade 1982).

3.2.2 OCEAN RESIDENCY

Little is known about the ocean distribution of specific winter steelhead populations, but recoveries of fish with coded wire tags from along the west coast indicate that marine migrations are extensive. Several studies have shown that North American winter steelhead use a broad swath of the central North Pacific Ocean, the Gulf of Alaska, and an area along and south of the Aleutian Island chain (Light et al. 1988; Burgner et al. 1992; Myers et al. 1996). Distributions of winter-run fish have generally been similar to those for summer steelhead, though winter fish may have a more western and northern distribution (McKinnell et al. 1997). We found few data specific to Willamette River populations.

The majority of Willamette basin winter steelhead spends two years at sea. Sea age estimates for adults that returned to the Clackamas, Molalla, and North and South Santiam rivers suggest that approximately 70–90% of adults were two-ocean fish, 10–30% were three-ocean fish, and a small number (generally <1%) were four-ocean fish (Howell et al. 1985; Chilcote 1998). Because most spent 2-3 years in freshwater as juveniles, returning adults are predominantly age 4 or 5. McElhany et al. (2007) reported adult winter steelhead age composition at Willamette Falls as 48% age 4, 41% age 5, 10% age 6, and 6% age 7. It should be noted that there are a wide range of life history combinations in winter steelhead (Myers et al. 2006), with varying juvenile freshwater residency, ocean residency, and repeat spawner age groups. For example, Leider et al. (1986) reported more than 20 juvenile age–adult age–repeat spawner combinations in wild winter steelhead from the nearby Kalama River (lower Columbia River tributary).

3.2.3 ADULT MIGRATION TIMING

An adult winter steelhead tagging project in the lower Columbia River in the 1950s (Korn 1961) provided some of the first migration timing data for Willamette River stocks. This study showed that Willamette fish were predominantly in the lower Columbia in March, with smaller percentages in January, February, and April. Compared to other lower Columbia populations (the source for the Big Creek stock), the Willamette steelhead were relatively later-timed. This timing difference presumably reflects the selective pressures of passage at Willamette Falls.

At Willamette Falls, the native winter run passes primarily from February through May, with peak counts typically in February and March (Howell et al. 1985; Firman et al. 2004, 2005; also see Figure 4). Steelhead from the introduced Big Creek stock pass Willamette Falls earlier, typically starting in November and continue at low levels until late January. As mentioned previously, it is not clear how much temporal overlap exists between Big Creek and native stocks or if interbreeding between these populations have produced phenotypes with intermediate migration timing traits. We found no analyses of the factors that affect year-to-year variability in steelhead migration timing at Willamette Falls, though there have been some qualitative observations that suggest the run slows or stops in response to high-discharge events and low water temperatures.

Migration timing patterns in the Clackamas River parallel those at Willamette Falls for both native lineage and Big Creek populations. Passage at North Fork Dam begins in November and continues at low abundance levels through February. This portion of the run is presumed to be primarily fish of Big Creek origin. Greater numbers are counted at the dam starting in March, when more native fish are present. Annual peak passage is typically in April or May and some fish continue to pass into June in some years (Everest and Sedell 1983; Howell et al. 1985; Olsen et al. 1992). Peak returns to Eagle Creek Hatchery are also in April or May in most years (Clackamas HGMP 2002).

In the North and South Santiam rivers, most steelhead return from February through June, with typical peak passage in March or April (Howell et al. 1985; ODFW 1990a; Wevers et al. 1992). Counts at Foster Dam on the South Santiam and passage at the Bennett dams near Stayton Island on the North Santiam follow the same general pattern (e.g., Firman et al. 2004, 2005). Trapping at the Minto collection facility (North Santiam) occurs during April and May in most years, though fish have been reported to arrive as late as July (DeCew 1969). Winter-run fish passage at these sites appears to

precede hatchery summer-run passage by 6-12 weeks, with the summer steelhead passing mostly in June and July (Firman et al. 2002).

We found few data describing adult run timing in the Molalla or Calapooia rivers. The relatively limited available information is due in part to an absence of counting facilities (i.e., no major dams or traps) in these sub-basins. However, adult arrival timing in these rivers is likely similar to timing in the Santiam. Further upstream, return timing of the introduced winter steelhead populations in the McKenzie and Middle Fork Willamette rivers are mostly similar to or slightly later than the timing distributions of the native populations in the Santiam basin. In 18 years of count data at Fall Creek Dam on the Middle Fork, for example, Howell et al. (1985) reported peak returns in April, with some winter fish recorded from February through June. Much of the data available in electronic format provide monthly or weekly summaries, a scale that is relatively coarse for evaluating differences among tributary populations or among years. There is some archived daily count data (largely unpublished) that could be used to examine potential phenotypic or environmental effects.

3.2.4 RUN COMPOSITION

We found little information on native winter steelhead run composition (i.e., the relative abundance of sub-basin populations) in the lower Willamette River, at Willamette Falls, or in upstream reaches. However, there are clear timing differences between the aggregate native population and the introduced Big Creek stock. The degree of overlap between these two groups has not been clearly established – the two are arbitrarily separated in early February at Willamette Falls – and composition during the transition period remains uncertain. At Willamette Falls and other sites, there are also composition questions during the significant overlap period for native winter and hatchery summer steelhead.

Overall winter run composition in the basin has been inferred from adult returns to hatcheries and traps and from counts at dams (e.g., Wevers et al. 1992; Chilcote 1998). However, inference from escapement data does not provide information on relative population abundance during migration, particularly if some populations are more or less susceptible to mortality prior to reaching collection sites, are less likely to be counted at upstream sites, or have different behaviors (i.e., holding) during migration. It is probable that there is greater among-population variability in run timing for naturally-produced fish compared to either Big Creek or Willamette hatchery winter steelhead given greater life history variability in wild fish. Consequently, run composition presumably is more variable both within and among years for wild fish. A better understanding of run composition through time would be useful, particularly if there are situations where system operations can be modified to assist vulnerable groups.

3.2.5 SPAWN TIMING

Native winter steelhead begin spawning as early as February and early March in the Willamette basin, with the earliest initiation generally in the west-side tributaries (Hutchison et al. 1966). Peak spawning in the native range is typically in April or May and can continue into June and even July (Murtagh et al. 1992; Chilcote 1998; Firman et al. 2005; Myers et al. 2006; Hutchison et al. 2007). Spawning in the wild by fish from the early-timed Big Creek stock typically is in January and February, and occasionally as early as December (Howell et al. 1985; Chilcote 1998). Where Big Creek and native

populations overlap, as in the Clackamas River, spawning by the early-timed Big Creek stock typically precedes spawning by the native stock (Clackamas HGMP 2002).

There have been relatively few systematic, multi-pass spawning ground surveys for winter steelhead in the basin, although single-pass redd counts have been routinely conducted at some sites (i.e., in the North and South Santiam and Calapooia rivers, Wevers et al. 1992; Chilcote 2001; ODFW 2004). More intensive surveys were conducted in 2006-2007 in the Clackamas River downstream from North Fork Dam (Hutchison et al. 2007), but these appear to have been the exception in terms of spatial and temporal coverage. In part, winter steelhead spawning ground surveys are limited because redds can be difficult to observe during the high and turbid flows that can occur from late winter through spring at many locations. In addition, steelhead redds can be difficult to differentiate from Pacific lamprey (*Lampetra tridentata*) redds and the two species have spatial and temporal overlap at some sites (e.g., Hutchison et al. 2007).

3.2.6 ITEROPARITY

Iteroparity (i.e., repeat spawning) in winter steelhead has not been well studied in Willamette River populations. In other lower Columbia River tributaries, 5–12% of returning adults have been reported as repeat spawners (e.g., Long and Griffin 1937; Leider et al. 1986; Busby et al. 1996). Chilcote (2001) similarly reported a 10–11% repeat spawning rate for Clackamas, Molalla, Santiam, and Calapooia populations. It was not clear, however, whether the rates in Chilcote (2001) were inferred from other studies or were based on quantitative data for these tributaries. Howell et al. (1985) reported that 2–4% of the winter steelhead at Foster Dam (South Santiam) were repeat spawners in 1979-1981.

Kelt (post-spawn adult steelhead) outmigration occurs over several months, with most individuals moving downstream immediately following spawning. Numerically large kelt outmigrations have been reported in the Clackamas River (Gunsolus and Either 1970; Howell et al. 1985), with peak movement in May and June, but these outmigrations have not been associated with significant repeat spawner returns. Kelts have also been reported in most upper Willamette tributaries, but enumeration is difficult and we found few data. It is possible that mortality is high as these fish move downstream. For example, turbine passage mortality rates for kelts at Foster Dam (South Santiam) were reported as 22–51% in the 1960s and 1970s (Wagner and Ingram 1973; Buchanan and Moring 1986). Given the timing of outmigration (i.e., spring and early summer), many kelts also potentially encounter warm water temperatures in the lower river at a time when the fish are emaciated and have limited somatic energy reserves, which may increase mortality. Behavior and survival during this life stage have not been well studied in the Willamette system, and it is not clear if most mortality occurs along the migration corridor or after fish exit the Willamette River and enter the estuary or ocean.

3.3 ADULT MIGRATION BEHAVIOR

There have been very few adult winter steelhead tagging studies in the Willamette River basin and consequently there is little information about steelhead behaviors during active migration, at or near dams, or on spawning grounds. We found no winter steelhead analog to the radiotelemetry study of Schreck et al. (1994), which provided much of the available migration behavior data for spring

Chinook salmon. However, there have been some recent summer steelhead tagging programs (e.g., floy tagging at Bennett Dams; Firman et al. 2005) that may provide some relevant information. The PIT-tag interrogation site installed in the Willamette Falls fishway in 2006 may also eventually provide important juvenile-to-adult survival and migration timing information, but the single interrogation site (thus far) can be expected to provide only limited behavioral data. In an early 2010 query of the PTAGIS database, no winter steelhead were detected in 2008 or 2009 at the Willamette Falls site.

Winter steelhead migration rates can be estimated using fish counts at Willamette Falls and at upstream dams and traps, but there have been few summary analyses. We did not find any migration rate data based on individual fish. Similarly, winter steelhead fallback rates at dams or at Willamette Falls have rarely been directly estimated. An apparent exception was a 6.6% fallback correction used for winter steelhead at Bennett Dams on the North Santiam River (Firman et al. 2005). From this report, however, it was not clear if the correction was based on tagged winter steelhead or inferred from mark-recapture studies that used summer steelhead.

As with spring Chinook salmon, we found few data on adult winter steelhead behavior in dam tailraces, in fishways, or in reservoirs. Steelhead passage challenges in the main stem and in tributaries presumably parallel those for Chinook salmon, with information gaps associated with upstream migration delay at dams, fishway entrance and/or trap collection efficiencies, passage and collection failure, and the spatial and temporal patterns of migration and prespawn holding. As with salmon, more data are needed on the effects of environmental conditions and dam operations on adult steelhead behavior and passage success. Specific migration problem areas have been described in some of the sub-basin and hatchery management plans, particularly in regards to collection sites and passage and separation activities at dams (e.g., ODFW 1990a, 2004; Wevers et al. 1992).

3.3.1 STRAYING

Inter-basin straying rates for native winter steelhead have not been described in the Willamette River basin, though low-level straying is believed to help sustain some of the satellite populations (i.e., in west-side tributaries and tributaries upstream from the Calapooia River). There have been some summaries of hatchery winter steelhead spawning in the wild (i.e., within-basin straying) in the Clackamas (Clackamas HGMP 2002; Hutchison et al. 2007). Hatchery winter- and/or summer-run fish may also contribute to native spawning groups in the Calapooia River, which is a management concern for that small native population (Chilcote 1997, 1998). There have been few estimates of straying into the Molalla River.

In the Santiam and Clackamas rivers, hatchery summer steelhead strays and summer strays resulting from the fishery recycling program (i.e., when captured hatchery fish are re-released downstream for fisheries) are an issue because of potential interactions with spawning winter steelhead (e.g., Kostow et al. 2003; ODFW 2004). The summer steelhead spawning survey program described in Firman et al. (2004, 2005) suggest that several thousand summer-run fish spawn in the upper Willamette basin each year. These are a mix of strays from hatcheries and naturally-produced summer-run fish. Spatial overlap between summer and winter steelhead spawning sites has been observed at several sites in the Santiam and Clackamas rivers (Kostow et al. 2003; Firman et al. 2004, 2005; Kostow and Zhou 2006).

3.4 PRESPAWN MORTALITY

Unlike for spring Chinook salmon, we found essentially no estimates of prespawn mortality rates for winter steelhead either on spawning grounds, during prespawn holding, or during migration. Compared to spring Chinook salmon, winter steelhead migrate and spawn during relatively favorable environmental conditions (i.e., lower temperatures) and so may be less susceptible to high adult mortality. However, this has not been quantified in the basin despite some evidence that extinction risk may rapidly increase as prespawn mortality increases in winter steelhead. In an assessment by Chilcote (2001), North Santiam and Calapooia winter steelhead populations were considered vulnerable to elevated adult mortality, while South Santiam and Mollala populations were more resilient. Adult mortality ‘trigger points’ of approximately 30% or higher were associated with rapidly increasing population-level risk (Chilcote 2001). The analyses were based on hypothetical fishery-related mortality, but conclusions would be similar for mortality due to other sources and demonstrate the need for total adult mortality estimates. Several potential fishery-related mortality effects could warrant further examination for winter steelhead, including mortality following catch and release and incidental mortality associated with summer steelhead fisheries.

There are several disease- and parasite-related mortality risks for winter steelhead that have been noted in the Willamette populations. These include furunculosis, bacterial kidney disease (bkd), and *Ceratomyxa shasta* infections (ODFW 2004). However, we found little information about the incidence or lethality of these or other bacterial, fungal, and parasitic infections in native stocks. Similarly, there have been no assessments of energetic reserves, stress indicators, or reproductive status of returning adult fish. Each of these factors potentially affect survival, vary among years and sub-basin populations, and may be important to the productivity and fitness of winter steelhead in the Willamette.

4.0 INFORMATION GAPS

4.1 SPRING CHINOOK SALMON AND WINTER STEELHEAD

Information gaps related to adult spring Chinook salmon and winter steelhead are generally similar, although there are greater gaps overall for steelhead. Winter steelhead tend to be more difficult to observe and monitor because they migrate during winter and when discharge is higher. In addition, springtime spawning is more difficult to monitor and therefore traditional salmon survey techniques (i.e., carcass surveys, redd counts) can be less effective for steelhead. Post-spawn survival and kelt emigration by steelhead also present enumeration and monitoring challenges that do not apply to spring Chinook salmon.

The information gaps described below are loosely organized by topic. However, we emphasize that many of the concepts and questions raised are inter-related. For example, effects of facility operations and river environment can be difficult to separate, particularly in the highly-managed main stem and in tributaries with multiple dams. Similarly, life history, behavior, and survival questions are strongly linked together and to the conditions adult salmon and steelhead encounter during migration, at passage barrier, and on spawning grounds. The ordering of information gaps below does not reflect their relative importance or relevance to RPA management objectives.

4.1.1 LIFE HISTORY

- The relationships among juvenile and adult life history characteristics have not been well described for either run. These include clarifying how juvenile rearing locations (i.e., in spawning tributaries, versus in reservoirs or at main stem sites) and emigration strategies (i.e., yearling versus subyearling Chinook salmon) relate to survival to adulthood, adult age at return, adult migration timing, or adult spawn timing.
- The relationship between spring Chinook salmon jack (and mini-jack) returns and subsequent adult returns have not been clearly described. Such information can be useful for forecasting run size, timing, and composition.
- Little is known about the abundance, distribution, emigration timing, or emigration survival of winter steelhead kelts.
- Little is known about the contribution of repeat winter steelhead spawners to population growth rates, either within individual populations or at a basin-wide scale.
- The factors that affect migration and spawn timing have not been well described for either run. The timing of life history events are typically driven by a combination of genetic and environmental factors, and it is possible that age-related, population-specific, and hatchery-wild difference exist in both runs. Identifying these patterns and distinguishing genetic from environmental effects may be used to improve management strategies.
- The effects of extensive spawning by hatchery-produced adults, especially within spring Chinook salmon populations, is poorly understood. The influence of hatchery-origin conspecifics may have

either positive or negative effects on reproductive success and fitness and likely differs among sub-basins.

- In general, there is limited information about how specific life history traits (during any life stage) are associated with population-level fitness.
- Little is known about the relative reproductive success of individuals of either species, including potential genetic and phenotypic effects. Understanding the relative contribution of individual types is important for estimating effective population size.
- A variety of life history traits have been lost or diluted in both runs through artificial selection and extirpation. Managers should consider whether there are opportunities to target some of these traits (i.e., variability in adult age, re-establishment of historical spawn timing) to increase overall life history diversity in the basin.
- There is very little information about potential intra- or inter-specific density-dependent effects for adults of either run. Density dependence may be especially important on spawning grounds (i.e., competition for mates or redd sites), between wild and hatchery fish, and between native and non-native stocks (i.e., summer versus native winter steelhead).

4.1.2 BEHAVIOR

- There are relatively large information gaps related to basic adult salmon and steelhead behaviors, ranging from migration rates, migration routes, and use of specific habitats to behaviors at and near passage barriers and collection facilities.
- Information is needed on adult passage bottlenecks related to both infrastructure (i.e., dams, fishways, culverts, revetments) and river environment (i.e., thermal barriers, high and low flow barriers).
- There is very limited information on adult passage failure at Willamette Project facilities. It would be helpful to develop standardized metrics to evaluate adult collection efficiency at fishways, trapping efficiency at sites without volitional passage, and overall dam passage efficiency at project dams. Establishing baseline metrics will be important for evaluating operational and structural changes at the dams as BiOp plans are implemented.
- A relatively large portion of adult freshwater migration is spent holding in main stem and tributary sites, especially by spring Chinook salmon. However, relatively little is known about the spatial or temporal distribution of holding behavior, the factors that affect the initiation and cessation of holding, or the effects of holding on maturation, fish physiology, or prespawn mortality.
- Large aggregations of spring Chinook salmon hold downstream from project dams, but behavior and survival in these areas have not been well studied, nor have potential delayed or indirect effects such as disease transmission or temperature exposure.

- Spatial and temporal use of thermal refugia by adult spring Chinook salmon has not been investigated during active migration or during prespawning holding, despite the potential importance of refugia to maturation and survival.
- Adult fallback at dams can have important effects on escapement estimation and can lead to adult injury and mortality for fish from both runs, but there has been little research on the behavior. There are information gaps related to the timing and magnitude of fallback, the effects of project operations and river environment, potential age-related or demographic differences in fallback rates, or the relationship between fallback and injury or mortality rates. It may also be important to evaluate route-specific (i.e., spillways, turbines, etc.) fallback effects if fallback-related mortality is high.
- There is little information related to volitional downstream passage by winter steelhead kelts, including passage routes at dams, behavior in reservoirs, or factors that affect emigration survival.
- Although there have been inter-basin straying evaluations related to hatchery protocols, information gaps remain, including permanent straying by wild fish, effects of project operations and impassable dams, and temporary straying related to thermal refugia use or other environmental cues.
- There is limited information on the behavior of outplanted spring Chinook salmon. Given the scale of the outplant program, it is important to have information on behavioral interactions between outplants and natural spawners (where the two groups overlap), movement data in relation to outplant release location and timing, and movement data related to spawn site selection.
- There is a need for studies investigating effects of competition and other potential negative interactions among juvenile hatchery and wild salmonids.

4.1.3 PRESPAWN MORTALITY

- There are significant information gaps related to the incidence, spatial distribution, and timing of non-harvest adult mortality for both species.
- Information is needed on prespawning mortality mechanisms. These include the effects and incidence of injury and disease, fish condition, energetic depletion, and interactions with conspecifics and non-natives. Linkages between behavior (i.e., fallback, passage failure, straying) and mortality need to be examined.
- Prespawning mortality among spring Chinook salmon varies widely among years and populations, but there has been little quantitative evaluation of the factors affecting this variability. A mix of environmental, operational, and behavioral hypotheses should be examined.
- There is very little information on prespawning mortality for winter steelhead, either during migration or on spawning grounds. It also may be useful to measure the incidence of catch-and-release mortality and mortality associated with summer steelhead fisheries.

- The effects of project operations on prespawn mortality need to be addressed. These include project operations that affect water temperature and discharge, project attraction and collection efficiencies, and fish holding and passage behaviors.
- More information is needed on the survival effects of collection, trapping, handling, holding, and transporting adult salmon and steelhead.

4.1.4 ENVIRONMENTAL AND OPERATIONAL EFFECTS

- As described above, there are information gaps related to environmental and operational effects on the behavior and survival of both spring Chinook salmon and steelhead. These include direct environmental effects (i.e., of ‘natural’ patterns in water temperature, discharge, and turbidity) as well as effects related to project operations (i.e., the temperature, timing, and volume of water releases). Information needs exist for both main stem and tributary sites.
- Environmental and operational effects on adult behavior and mortality should be evaluated at a variety of spatial (i.e., migration corridor, at passage barriers, in tributaries) and temporal (i.e., annual, seasonal, daily) scales. Proximate conditions may be very important for behaviors like fishway collection efficiency, holding, fallback, or thermal refugia use, whereas large-scale differences in river environment may broadly affect survival rates.
- Information is needed on how project operations affect collection and passage efficiency for both upstream migrants and emigrating winter steelhead kelts.
- Information is needed on the effects of project ramping flow and alternative water management strategies.
- There has been some evaluation of adult trapping facilities, but a more systematic approach is recommended, particularly as new designs are developed and deployed. Of particular interest are the effects of trap operation timing and fish holding strategies, plus trap effects on injury rates and disease transmission.
- There are several information gaps related to spring Chinook salmon outplanting operations. These include the effects of holding and transportation, the effects of release timing and location, and effects of in-river environmental conditions at release sites. It would also be useful to better understand the relationships between salmon outplant success and a suite of fish characteristics such as initial condition, maturation and stress levels, and disease and parasite infection rates. Lastly, potential outplant effects on conspecific natural spawners and other species should be considered.
- Both operational and environmental factors may prevent adult salmon and steelhead from accessing beneficial holding habitats during migration (i.e., thermal refugia, deep pools) as well as spawning areas. Management activities related to adult habitat creation and access should be evaluated.

4.1.5 OCEAN, ESTUARY AND CLIMATE EFFECTS

- Ocean distributions for wild spring Chinook salmon and winter steelhead are not well described, particularly for individual Willamette River populations.
- Relationships among ocean productivity, ocean environment, estuary conditions, adult return rates and adult condition (i.e., energy reserves) have not been well described for Willamette River populations.
- Although there are a variety of climate models that predict reduced snowpack, an increase in winter rain events, reduced summer discharge, and warmer summer temperatures in the Willamette and Columbia River basins, the effects of these potential changes have not been well integrated in adult Chinook salmon and steelhead management plans.

4.1.6 POPULATION-SPECIFIC INFORMATION

- In general, many of the information gaps described in the previous sub-sections also apply to individual populations of both spring Chinook salmon and winter steelhead. Understanding differences in life history traits or behavioral differences among populations may be important for developing appropriate management strategies and for protecting vulnerable groups.
- There are information gaps related to within and among-year variability in run composition for both species. Runs may vary demographically (i.e., age-structure, proportion of wild fish, relative population abundance) through time and certainly at different sites along the migration corridor. Understanding run composition may also be useful for making operational or other management decisions.
- An important population-related gap is whether different groups experience different mortality rates — likely given the widely variable prespawm mortality rates among sites — either as a result of life history traits (i.e., migration timing, adult age), phenotypic differences, exposure to pathogens (i.e., at collection facilities, downstream from projects, or in specific main stem or tributary sites), or potential differences in environment and operations among tributaries.
- There have been some evaluations of spatial and temporal overlap between hatchery and wild spring Chinook salmon and between winter and summer steelhead in the Willamette basin. However, there is relatively limited information about intra- and inter-specific competition (i.e., for mates or spawning sites) or behavioral interactions.

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6.0 Appendix 1. Willamette River sub-basin maps (from NMFS 2008).

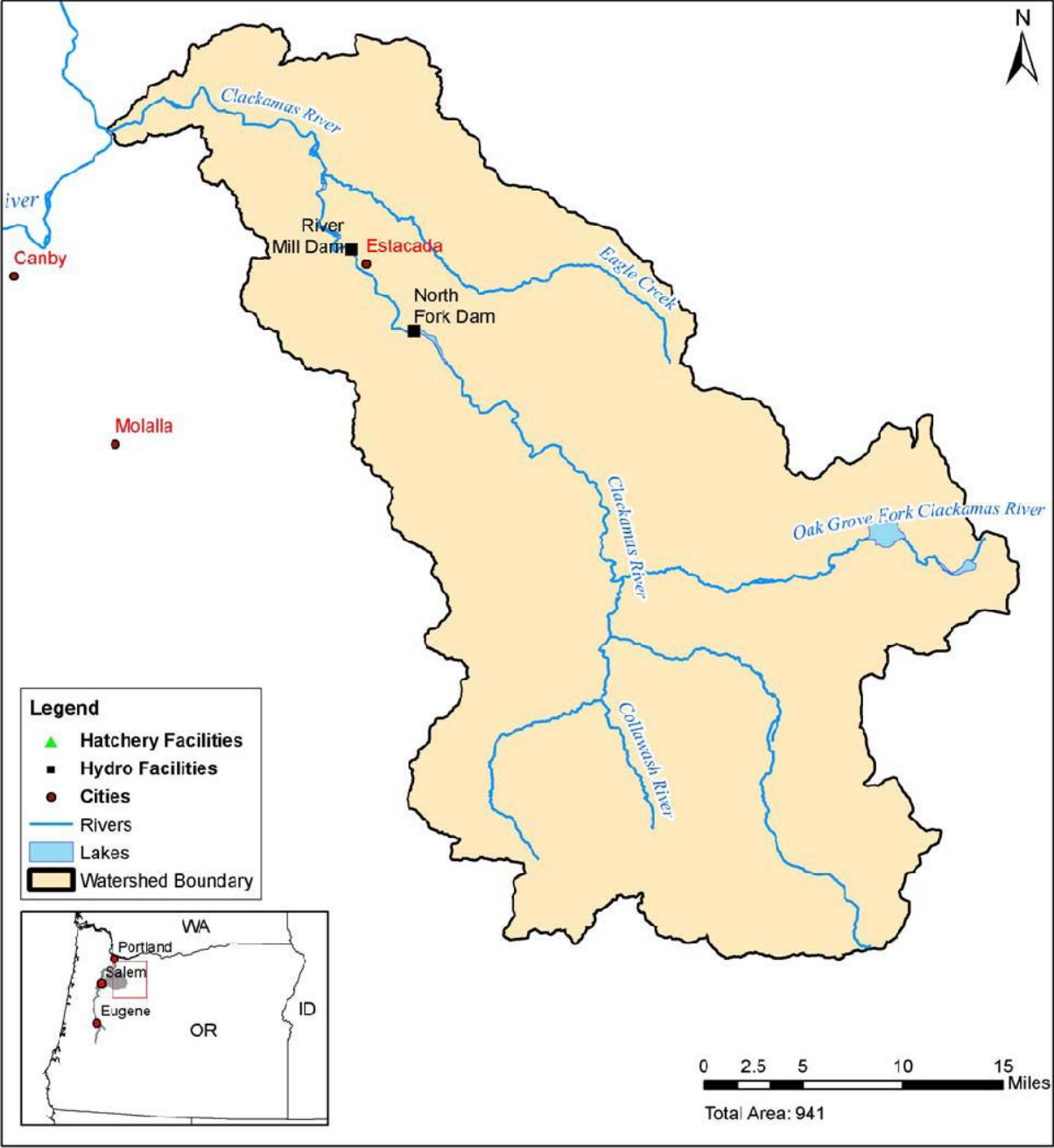


Figure A-1. Clackamas River sub-basin.

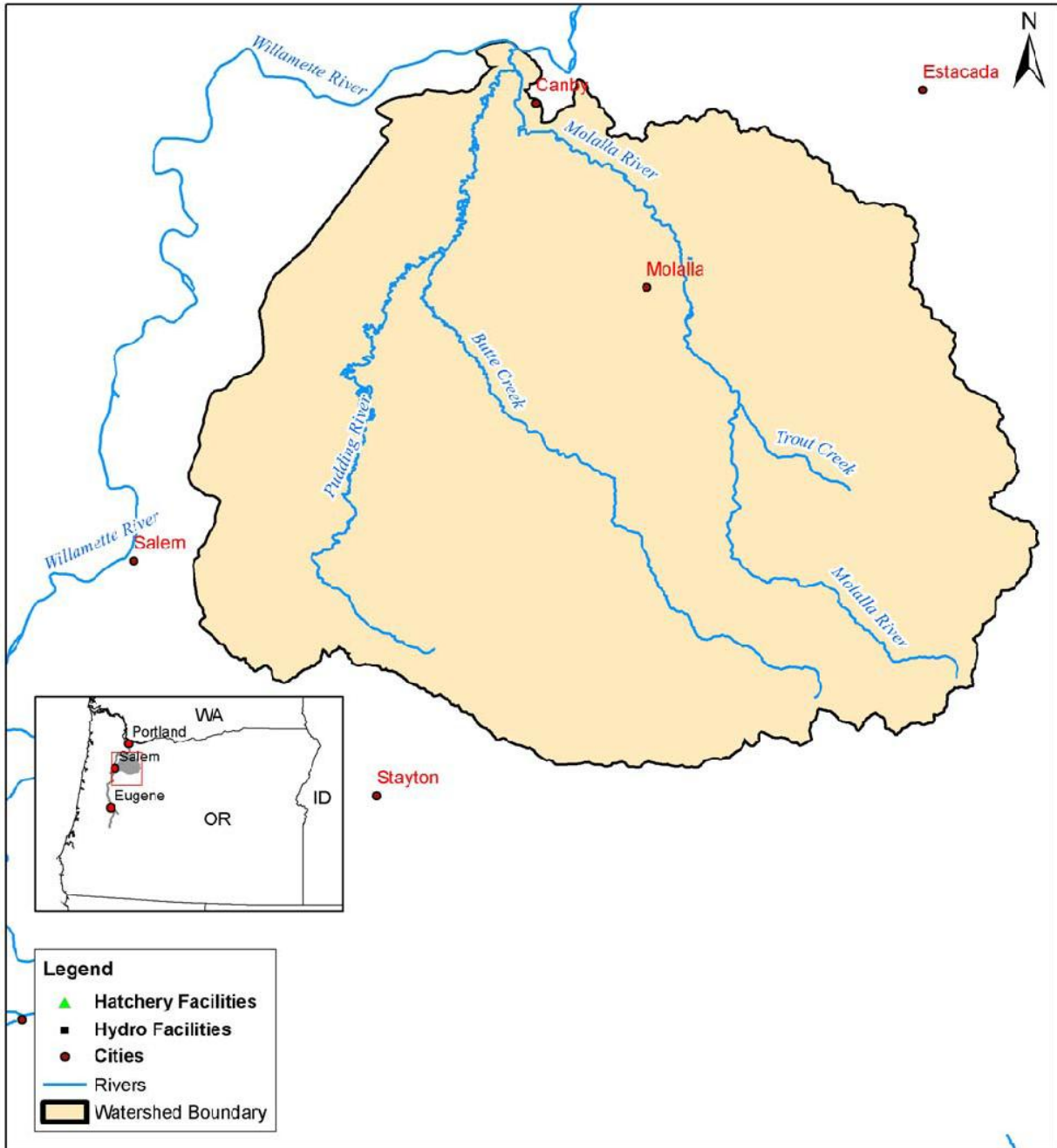


Figure A-2. Molalla River sub-basin.

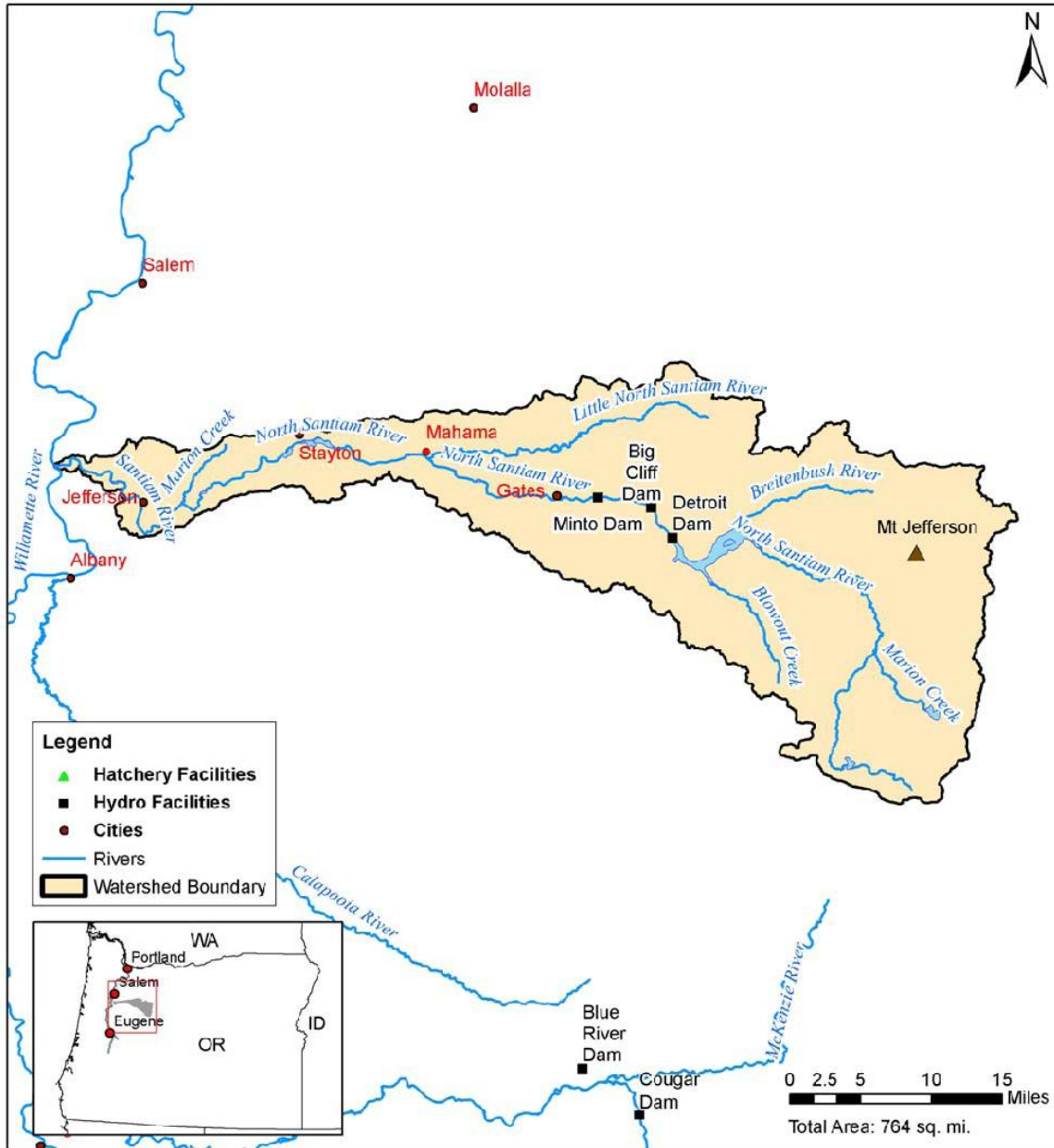


Figure A-3. North Fork Santiam River sub-basin.

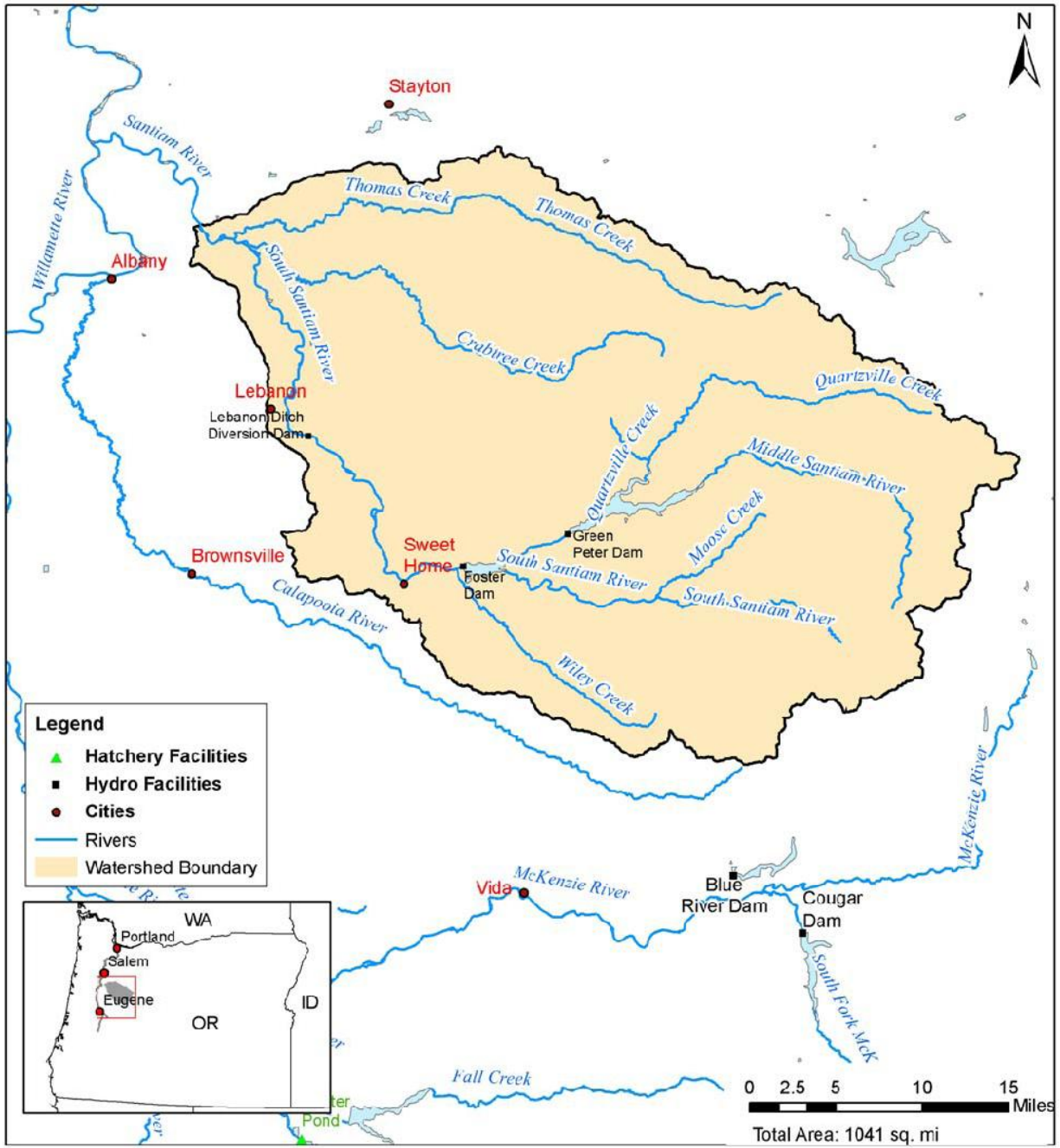


Figure A-4. South Fork Santiam River sub-basin.

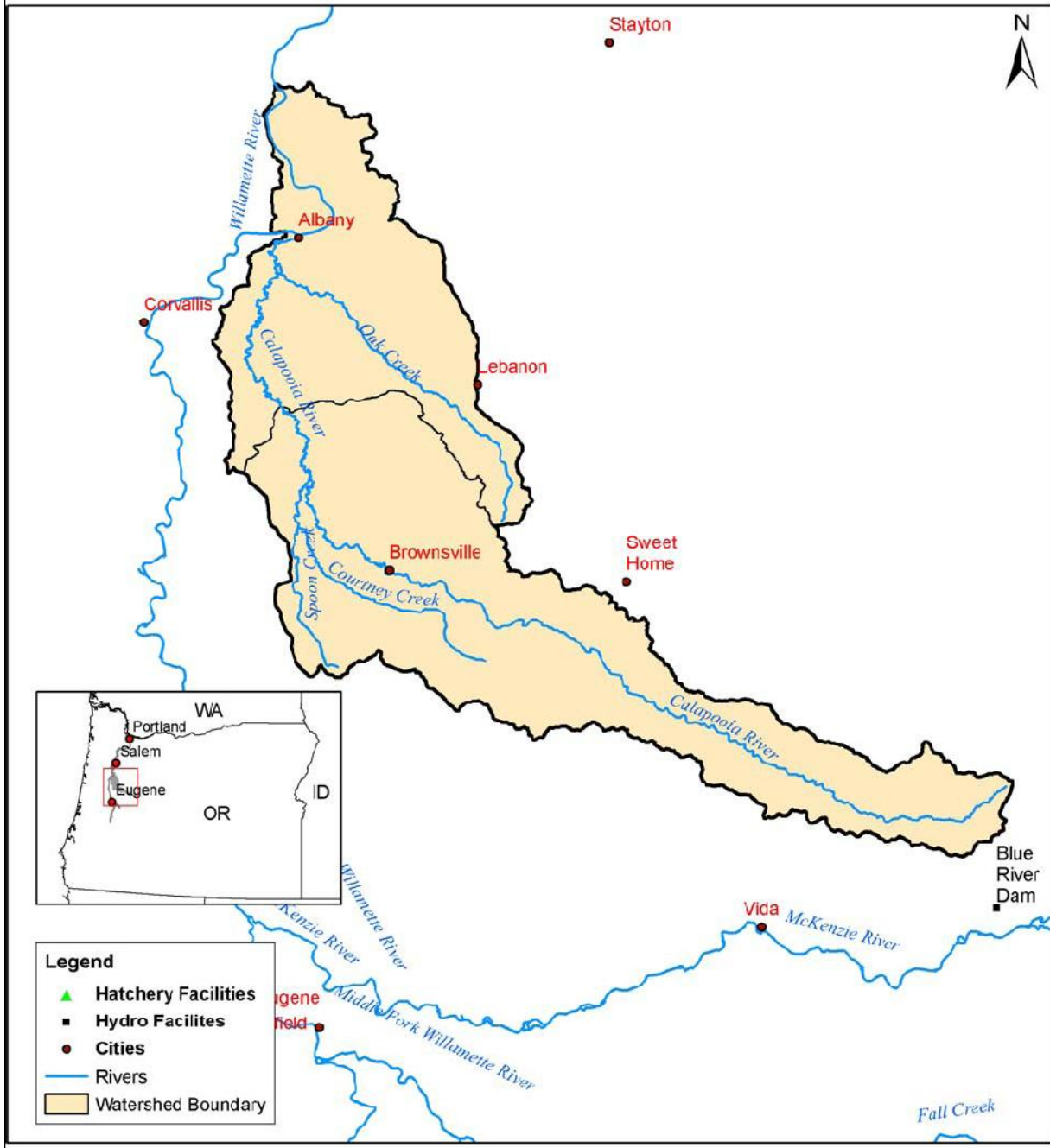


Figure A-4. Calapooia River sub-basin.

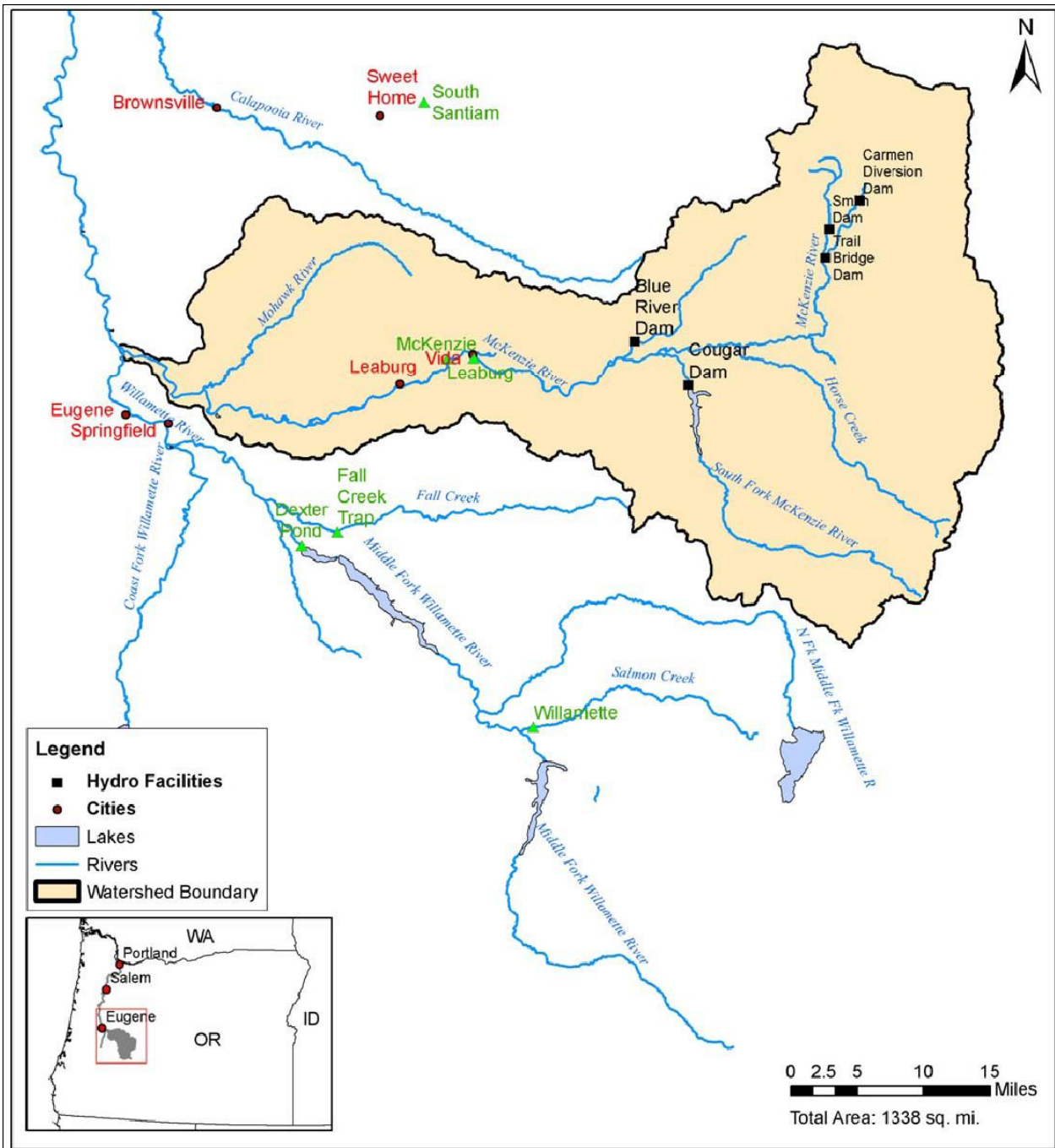


Figure A-5. McKenzie River sub-basin.

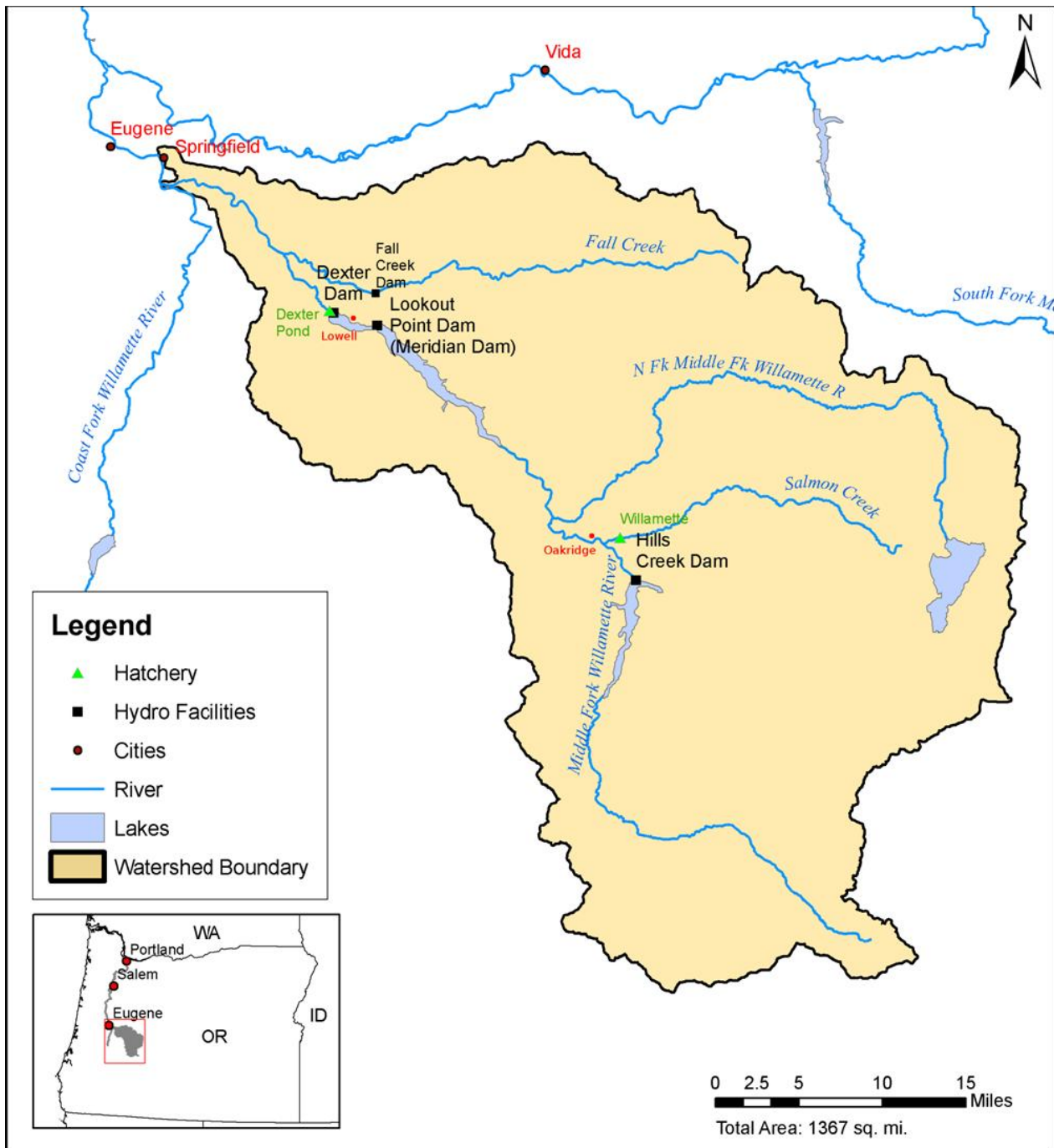


Figure A-6. Middle Fork Willamette River sub-basin.

