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4.16 SUMMARY OF INFORMATION AVAILABLE TO ASSESS POTENTIAL AQUATIC SPECIES INTERACTIONS IN THE LEWIS RIVER BASIN (AQU 16)

The Lewis River Aquatic Resources Group (ARG) is evaluating strategies for anadromous fish reintroduction above Merwin Dam. This study contributes to that evaluation by describing the potential adverse species interactions and fish health issues that may arise from reintroduction anadromous salmonids to the upper Lewis River watershed. The study emphasizes potential effects on native fish species that currently inhabit the Lewis River stream reaches above Merwin Dam. In addition, this study considers some of the various benefits of reintroduction, both to aquatic resources of the Lewis River and to the ecosystem as a whole.

4.16.1 Study Objectives

The objective of this study is to identify and describe any potential adverse effects of anadromous fish (coho, Chinook, steelhead, and sea-run cutthroat trout) reintroduction on native resident fish species in the upper Lewis River basin.

4.16.2 Study Area

The study area for AQU 16 is the Lewis River basin upstream of Merwin Dam and any stream in the Lewis River basin that may be accessed by hatchery stocks originating from the Lewis River Hatchery Complex.

4.16.3 Methods

Existing information describing species interactions (i.e. habitat competition, predation, disease and interbreeding effects), including evaluations of species interactions elsewhere in the Columbia River basin, was compiled and reviewed. Information sources included PacifiCorp files, University of Washington library system, aquatic abstract databases, internet searches, and personal communications.

Existing information was evaluated and summarized in comparison to the aquatic resources of the Lewis River watershed, using data specific to the basin whenever possible. In addition, gaps in available information were identified and noted within the text.

4.16.4 Key Questions

The objectives of this study were derived from key questions developed through the Lewis River watershed scoping process. Specifically those questions are:

- What types of interspecific interactions may occur with various options for reintroducing anadromous fish?

- How would reintroduction of anadromous fish affect bull trout and kokanee populations?

- What effects do hatchery operations have on competition for food and space in rearing areas used by wild or native salmonids?

- What effects might hatchery releases have on predation of wild juvenile salmonids (including direct predation by hatchery fish and increases in predation by other fish or avian predators)?

- What are the potential effects on wild populations of supplementing hatchery populations with wild broodstock on a regular basis?

- What would be the benefits of using acclimation sites for release of fish from hatcheries to increase homing to hatcheries of origin and reduce straying to other basins or competition with wild fish on spawning grounds?

All but the last of these questions is at least partly addressed by this study.

4.16.5 Results and Discussion

The results of this study focus on the potential for intra- and inter-specific competition, disease transmission, predation, and interbreeding between resident species in the upper watershed and those anadromous species that are being considered for reintroduction. The species of concern in this study include Chinook, coho, and kokanee salmon and steelhead/rainbow, cutthroat, and bull trout, with northern pikeminnow and tiger muskellunge considered in the predation section. A list of the species discussed in this study, whether they inhabit the upper basin, and if they are being considered for reintroduction is included in Table 4.16-1.

Table 4.16-1. Species evaluated in the species interaction study.

Species	Present in the upper basin	Considered for reintroduction
Chinook Salmon		√
Coho Salmon		√
Steelhead Trout		√
Rainbow Trout	√	
Kokanee	√	
Coastal Cutthroat Trout	√ (resident)	√ (anadromous)
Bull Trout	√	
Northern Pikeminnow	√	
Tiger Muskellunge	√	

4.16.5.1 Habitat Competition

Life History and Spawning and Rearing Habitat Requirements for Salmonids in the
Lewis River Basin

In order to understand how salmonid species may interact and compete in the Lewis
River watershed, it is important to understand each salmonid species' basic ecology as a
first step in identifying potential inter-species overlaps in run timings and habitat
requirements. The following descriptions provide a general life history overview, while
the subsequent section analyzes how overlaps in run timing and habitat requirements may
lead to adverse species interactions. The majority of the life history descriptions were
summarized from the PacifiCorp and Cowlitz PUD (2002a) study (AQU 1) regarding life
history, habitat requirements, and distribution of aquatic species in the Lewis River
watershed.

Chinook Salmon – Chinook salmon (*Oncorhynchus tshawytscha*) are anadromous (adults
migrate from marine waters to spawn in rivers and streams) and semelparous (die after
spawning once) and have a broad range of life history traits, including variation in age at
seaward migration; variation in freshwater, estuarine, and ocean residence; variation in
ocean distribution; and in age and season of spawning migration (Healey 1991, Myers et
al. 1998). Most of this variation is exhibited in 2 distinct behavioral forms (races). These
races are commonly referred to as spring and fall Chinook (stream-type and ocean-type).
Both spring and fall Chinook are native to the Lewis River basin, although existing
spring Chinook stocks have been heavily influenced by hatchery programs. Lewis River
fall Chinook have experienced little hatchery influence (Myers et al. 1998).

Spring Chinook reside in freshwater for a year or more before migrating to sea and return
to their natal river in spring or summer, several months prior to spawning. Fall Chinook
migrate to sea in their first year of life, usually only a few months after emergence, and
return to their natal river in the fall, a few days or weeks before spawning (Healey 1991).
The Lewis River supports populations of both spring and fall Chinook.

Of particular importance to this study are the run timing and freshwater habitat
requirements of both Chinook stocks. Adult fall Chinook enter the Lewis River from late
August through mid-October. Lewis River spring Chinook (a mix of different hatchery
stocks) enter from late March through May (Figure 4.16-1) (pers. comm., E. Lesko,
PacifiCorp, October 2000). The peak spawning period for the naturally spawning spring
Chinook occurs from early September through late October (pers. comm., E. Lesko,
PacifiCorp, October 2000, PacifiCorp and Cowlitz PUD 1999, WDF and WDW 1993).
The peak spawning period for Lewis River fall Chinook occurs from late October
through late November (Figure 4.16-1).

SPECIES	LIFE STAGE	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
Spring Chinook	*Adult Migration*				███	███							
	Spawning									███	███		
	Fry Emergence		███	███									
	Rearing	███	███	███	███	███	███	███	███	███	███	███	███
	Juv. Outmigration					███	███	███					
Fall Chinook	*Adult Migration*									███	███		
	Spawning											███	
	Fry Emergence			███	███								
	Rearing			███	███	███	███	███					
	Juv. Outmigration				███	███	███	███					
Coho Salmon	*Adult Migration*									Type S	Type N		
	Spawning										███	███	███
	Fry Emergence			███	███								
	Rearing	███	███	███	███	███	███	███	███	███	███	███	███
	Juv. Outmigration					███	███						
Summer Steelhead	*Adult Migration*						███	███					
	Spawning			███	███								
	Fry Emergence						███	███					
	Rearing	███	███	███	███	███	███	███	███	███	███	███	███
	Juv. Outmigration				███	███							
Winter Steelhead	*Adult Migration*	███	███										███
	Spawning			███	███	███	███						
	Fry Emergence							███	███				
	Rearing	███	███	███	███	███	███	███	███	███	███	███	███
	Juv. Outmigration					███	███						

Figure 4.16-1. Periodicity chart for various life stages of fish species (with known life history information) in the Lewis River basin.

Note: Periodicity is based on peak times and fishes of wild origin.

SPECIES	LIFE STAGE	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
Coastal Cutthroat	*Adult Migration*												
	Spawning												
	Fry Emergence												
	Rearing												
	Juv. Outmigration												
Kokanee (Cougar Cr.)	*Adult Migration*												
	Spawning												
	Fry Emergence												
	Rearing												
	Juv. Outmigration												
Bull Trout	*Adult Migration*												
	Spawning												
	Fry Emergence												
	Rearing												
	Juv. Outmigration												

Figure 4.16-1. Periodicity chart for various life stages of fish species (with known life history information) in the Lewis River basin (cont).

Note: Periodicity is based on peak times and fishes of wild origin.

In general, Chinook spawning can occur in tributaries as small as 2 meters (m) (6.6 feet (ft)) wide or in the mainstem of large rivers (e.g. Columbia and Lewis rivers). Generally, spring Chinook prefer to spawn in middle and upper reaches of the mainstem areas, while fall Chinook prefer the middle and lower mainstem areas (WDFW 1994). Preferred spawning depths for both spring and fall Chinook are generally greater than 24 centimeters (cm) (9.4 inches (in)) with velocities ranging from 30 to 91 cm/sec (11.8 to 35.9 in/sec) (Bovee 1978, Bell 1986, Bjornn and Reiser 1991). Preferred gravel sizes range from 1.3 to 10.2 cm in diameter (0.51 to 4.0 in). Spawning water temperatures are reported to range from 5.6 to 13.9°C (42 to 57°F) (Bell 1986). In the Lewis River basin, most spring and fall Chinook spawning occurs within the 6.4 km (4.0 mile) stretch of the Lewis River between the Lewis River Hatchery and Merwin Dam (NPPC 1990).

In the Lewis River basin, the emergence of fall Chinook generally occurs from mid-February through mid-April. Spring Chinook emergence extends from early February through mid-March (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000). After emergence, the freshwater residency of fall and spring Chinook differs considerably. The majority of fall Chinook emigrate at 60 to 150 days after emergence, while spring Chinook do not emigrate until their second and sometimes third spring (Myers et al. 1998). Thus, in the Lewis River basin, wild spring Chinook rear in freshwater year round, fall Chinook rear in freshwater from mid-March through the end of June (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000).

Preferred habitat after emergence is in the lower velocity margins of the stream or river, with fall Chinook moving steadily downstream to the estuary. The low velocity marginal areas provide cover in the form of wood, root wads, overhanging vegetation, and/or undercut banks (Healey 1991, NESC 1984). As juvenile Chinook grow, they tend to move into the deeper, higher velocity portions of the channel (Myers et al. 1998). As with other salmonids, water temperature influences the physiology, behavior, and mortality of juvenile Chinook salmon. The upper lethal temperature for Chinook fry is 25.1°C (77.2°F); the preferred temperature is 12 to 14°C (53.6 to 57.2°F) (Scott and Crossman 1973). The optimum temperature for growth depends on food availability, and salmonids will not grow until their metabolic requirements are met (Murphy 1995).

While rearing in freshwater, juvenile Chinook feed opportunistically on insects and small crustaceans. Chinook residing in freshwater do not tend to demonstrate piscivorous (fish eating) tendencies (Scott and Crossman 1973).

Coho Salmon – Like Chinook salmon, coho salmon (*O. kisutch*) are anadromous and semelparous. They spend the first half of their life cycle rearing in streams and small freshwater tributaries. The remainder of the life cycle is spent foraging in estuarine and marine waters of the Pacific Ocean prior to returning to their stream of origin to spawn and die. Most adults are 3-year-old fish, however, some precocious males, known as "jacks," return as 2-year-old spawners. Both early (Type-S) and late (Type-N) coho salmon are endemic to the Lewis River basin. The early coho salmon historically utilized reaches above the Merwin Dam site and possibly Cedar Creek, while the late coho were found in Cedar Creek and the East Fork Lewis River (Smith circa 1943, WDF and USFWS 1951). The existing North Fork Lewis River coho salmon population is maintained through hatchery production. Type-S coho enter the Lewis River from late

August through October, with peak returns occurring in September and October. Type-N coho enter the river approximately 6 weeks later, from mid-September through November (pers. comm., E. Lesko, PacifiCorp, October 2000). Returning adults are either 2-year-old jacks (precocious males) or 3-year-old adults. Coho tend to have less variation in their ages at maturity than other salmonid species. In the Lewis River, both coho stocks spawn from October through late-December (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000).

The majority of returning coho are captured at the Merwin Dam Anadromous Fish Collection Facility, though an estimated 5 to 10 percent spawn naturally within the mainstem Lewis River below Merwin Dam and in several tributaries including Ross, Cedar, Chelatchie, Johnson, and Colvin creeks (WDF and WDW 1993). Wild coho tend to spawn in smaller rivers and tributaries. Optimum spawning habitat is considered to be streams with widths of 1 to 5 m (3.3 to 16.4 ft), relatively low velocities, and gradients less than 3 percent. Coho typically spawn in gravelly transition areas between pool and riffle habitats, with gravel sizes ranging from 0.2 to 10 cm (0.08 to 3.9 in) in diameter. Preferred water depths range from 10 to 53 cm (3.9 to 20.9 in) with velocities from 30 to 91 cm/sec (11.8 to 35.8 in/sec) (Bjornn and Reiser 1991, Ecocline Consulting 2001). Preferred spawning water temperatures range from 4.4 to 9.4°C (Larsen 1998).

The incubation period for coho salmon eggs is predominantly a function of water temperature. In general, the emergence of naturally spawned coho in the Lewis River basin occurs from late February through late April (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000). Following emergence, coho fry spend the remainder of the spring and summer rearing within their natal streams, although larger, more dominant fish may displace smaller fish downstream (Sandercock 1991). Juvenile coho are very territorial and are generally intolerant of the presence of other fish, especially other coho (Stein et al. 1972).

Coho fry are often associated with cover such as overhanging or submerged logs, undercut banks, overhanging vegetation, or large substrate. As coho rear in freshwater, they begin to occupy low velocity areas in the main channel which are adjacent to higher velocity areas (i.e. headwaters of pools). Coho tend to be drift foragers, feeding primarily on insects delivered by stream flow. However, at the yearling stage, coho may become piscivorous, supplementing their insect diet with the fry of their own and other species (Sandercock 1991). Juveniles spend either 1 or 2 years in freshwater before migrating to the ocean. Due to this extended freshwater residency, overwintering habitat in pools, side channels, and backwater channels are crucial to juvenile coho survival. These areas provide protection from peak flows, freezing temperatures, and predation (Sandercock 1991). As water temperatures and flows increase during the spring months, Lewis River coho smolts outmigrate from mid-April through the beginning of June (Figure 4.16-1) ((pers. comm., E. Lesko, PacifiCorp, October 2000).

Steelhead Trout – Steelhead trout (*O. mykiss*) are considered by many to have the greatest diversity of life history patterns of any Pacific salmonid species. Life history distinctions include varying degrees of anadromy, differences in reproductive biology, and plasticity of life history between generations (Busby et al. 1996).

Biologically, the anadromous steelhead can be divided into 2 reproductive races based upon their state of sexual maturity at the time of river entry and duration of their spawning migration. These 2 ecotypes (races) are termed summer (stream maturing) or winter (ocean maturing) steelhead. Summer steelhead enter freshwater as sexually immature individuals during the summer months and require several months of maturation before they spawn. Winter steelhead enter freshwater ready to spawn in late winter or early spring (Busby et al. 1996). The Lewis River supports both native winter and summer steelhead stocks; however, hatchery production has influenced native steelhead populations since 1954. Lewis River steelhead are currently managed for both hatchery and wild production.

In the Lewis River, summer steelhead begin their entry and migration into freshwater during June through mid-August, over-wintering in freshwater until they spawn the following March and April (pers. comm., E. Lesko, PacifiCorp, October 2000). Normally, summer and winter steelhead would spawn at approximately the same time. However, Washington Department of Fish and Wildlife (WDFW) has intentionally developed early spawning winter steelhead hatchery stock to maximize harvest opportunities and minimize hatchery/wild steelhead interactions. Therefore, Lewis River hatchery winter steelhead enter the stream in December and January and spawn in January and February. Yet, wild winter steelhead in the Lewis River enter freshwater from early December through mid-March, with peak migration occurring in March, and spawn from mid-March through late June. Peak spawning for both summer and winter steelhead occurs from mid-March through April.

Today, approximately 5 to 10 percent of returning Lewis River steelhead spawn naturally. Spawning activity has been reported in the mainstem Lewis River below Lake Merwin and in several tributaries including Ross, Cedar, Chelatchie, Johnson, and Colvin creeks (WDF and WDW 1993). Spawning steelhead prefer relatively small, fast flowing streams with cool, clear, and well oxygenated water. They most commonly create their redds at the tail of a pool close to the point where the smooth surface water breaks into the riffle below. Water depths for spawning usually range from 10 to 138 cm (3.9 to 54.3 in) with velocities of 30 to 110 cm/sec (11.8 to 43.3 in/sec) and stream temperatures of 3.9 to 9.4°C (39 to 49°F) (PacifiCorp and Cowlitz PUD 2002a, Bovee 1978, Levy and Slaney 1993). Preferred spawning gravel sizes range from 0.6 to 13 cm (0.2 to 5.1 in) in diameter (Barnhart 1991). After spawning most steelhead die, although approximately 4 to 5 percent survive to potentially spawn again (Busby et al. 1996).

Emergence of wild Lewis River summer steelhead fry begins in mid-June and continues through early August (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000). Emergence of winter steelhead occurs from early July through mid-August. After emergence, juvenile steelhead fry feed primarily on food sources associated with the stream bottom. Food sources include insects and other benthic invertebrates (Graves 1982). Yearling steelhead are also known to prey on other salmonid fry when available (Hawkins and Tipping 1999).

Juvenile steelhead typically spend 2 years in freshwater prior to ocean migration, although some juveniles smolt after only 1 year or after as many as 3 years (Figure 4.16-1) (Hymer et al. 1993). While rearing in freshwater, steelhead fry form small

schools and inhabit the margins of the stream. As they grow larger and more active, they slowly begin to disperse downstream. The individual fish then establish territories (microhabitats that contain feeding lanes and resting areas), which they defend (Barnhart 1991). Juvenile steelhead tend to prefer swift moving areas in riffles, although some of the larger fish inhabit pools or deep, fast runs (Barnhart 1991). Instream cover such as large rocks, logs, root wads, and aquatic vegetation are very important for juvenile steelhead. This cover provides resting areas, visual isolation from competing salmonids, food, and protection from predators (Bjornn and Reiser 1991). Outmigration of steelhead smolts takes place from late April through early June, typically at the time of spring runoff (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000).

Coastal Cutthroat Trout – The life history of coastal cutthroat trout (*O. clarki*) is extremely complex. Both migratory and non-migratory (anadromous, adfluvial, fluvial, and resident) forms may be present within the same population. These variations in life history may be related to environmental conditions, particularly those affecting growth rates (Johnson et al. 1999, Trotter 1991). Anadromous (sea-run) coastal cutthroat trout are generally smaller than the other anadromous salmonids. In addition, they rarely over-winter at sea and do not make extensive ocean migrations (Johnson et al. 1999). Fluvial and adfluvial coastal cutthroat trout migrate entirely within freshwater environments, while resident cutthroat remain as non-migrants in small headwater tributaries. The Lewis River basin supports both resident and anadromous coastal cutthroat trout (WDFW 2000).

All cutthroat trout, regardless of their life history type, are spring spawners. Actual spawning time depends on latitude, altitude, water temperature, and flow conditions (Trotter 1991). Sea-run coastal cutthroat trout enter the Lewis River basin from early October through mid-December (WDFW 2000). Spawning typically starts in early February and continues through late April, with a peak in February (Figure 4.16-1) (PacifiCorp and Cowlitz PUD 2000). Both sea-run and resident coastal cutthroat spawn in the upper reaches of small, low gradient streams and in the upper reaches of small tributaries of moderate-size streams. The volume of water in spawning streams seldom exceeds 10 cfs during the lower flow period, with most streams having average flows less than 5 cfs (Johnston 1982, Trotter 1991). In general, spawning occurs at water temperatures between 6.1 and 17.2°C (43 to 63°F) in low gradient riffles and pool tail-outs at depths between 15 and 45 cm (5.9 and 17.7 in), with velocities ranging from 11 to 72 cm/sec (4.3 to 28.3 in/sec) (PacifiCorp and Cowlitz PUD 2002a, Levy and Slaney 1993). Preferred spawning substrates are gravels ranging from 0.5 to 5 cm (0.2 to 2.0 in) in diameter (Trotter 1997). Often, the preferred spawning sites are located near deep pools, which are presumably used by adults for cover.

Depending upon stream temperatures, coastal cutthroat fry emerge from the gravel from March through June, approximately 8 to 9 weeks after spawning occurred (Figure 4.16-1) (Trotter 1991, PacifiCorp and Cowlitz PUD 2000). While rearing in freshwater, young coastal cutthroat trout are opportunistic feeders. As fry they feed on small invertebrates and as they grow, they utilize aquatic and terrestrial insects, salmon eggs, and small fish (Pauley et al. 1989). In the absence of other salmonids, cutthroat will utilize productive stream areas, such as the upstream ends of pools. However, in the presence of larger

competing salmonids, coastal cutthroat are often displaced to higher velocity riffle areas or remain in the less productive upper watersheds to avoid competition and predation (Trotter 1989).

Coastal cutthroat migrate to sea between the ages of 1 and 6. Most commonly, individuals migrate between the ages of 2 and 4 (Trotter 1997). Outmigration in the Lewis River extends from early April through late June (Figure 4.16-1) (PacifiCorp and Cowlitz PUD 2000). After spending several months in the ocean, all sea-run cutthroat trout return to their natal streams to over-winter, even if they don't engage in spawning activity.

Bull Trout – Bull trout (*Salvelinus confluentus*) exhibit 2 primary life history patterns; migratory (adfluvial, fluvial, anadromous) and resident. Resident bull trout complete their entire life cycle in the tributary streams in which they spawn and rear. Migratory bull trout spawn in tributary streams where the juveniles rear from 1 to 4 years before migrating and maturing in large rivers (fluvial), lakes (adfluvial), or saltwater (anadromous). It is likely that both resident and adfluvial bull trout are present in the Lewis River basin, although there is more existing information regarding those displaying adfluvial life history traits. Adfluvial bull trout are present in all 3 project reservoirs. In addition, a small number of unidentified adult char (bull trout or Dolly Varden) have been captured in the ladder at the Lewis River hatchery downstream of Merwin Dam (PacifiCorp and Cowlitz PUD 2000).

Bull trout are an iteroparous species (capable of spawning multiple times), which leads to a variety of body sizes at spawning. Adfluvial bull trout adults grow to the largest sizes, with resident adults being the smallest at maturity. Adults as large as 82 cm (32.3 in) have been captured in the Lewis River basin (WDFW 1995). Spawning generally occurs in late summer to early fall as water temperatures begin to drop (Goetz 1989). In the Lewis River basin, bull trout residing in Swift Reservoir migrate into tributary streams from late May through early August, and spawn from early August through the middle of September (Figure 4.16-1) (Faler and Bair 1992; Graves 1982; pers. comm., E. Lesko, PacifiCorp, October 2000). The adfluvial population of bull trout in Yale Lake migrates into tributary streams from the middle of August through late-September. Spawning (primarily in Cougar Creek) occurs from late-September through mid-October (Figure 4.16-1) (Graves 1982; pers. comm., E. Lesko, PacifiCorp, October 2000).

Cool stream temperatures are a crucial habitat component for bull trout, with optimum temperatures ranging from 2 to 10°C (35.6 to 50°C) (63 FR 39936). Preferred spawning habitats generally consist of low gradient reaches with loose, clean gravels ranging in diameter from 1.6 to 6.4 cm (0.6 to 2.5 in) (Post and Johnston 2002). Redds are commonly found in glides and the tail-outs of pools at depths from 24 to 61 cm (9.4 to 24 in) with water velocities ranging from 4 to 61 cm/sec (1.6 to 24 in/sec) (Fraley and Shepard 1989). The majority of bull trout spawning in the Lewis River watershed takes place in Cougar, Rush, and Pine creeks (tributaries to Yale Lake and Swift Reservoir) (Faler and Bair 1992, Lesko 2001). Lake Merwin does not appear to contain appreciable bull trout spawning habitat and individuals inhabiting the lake are believed to have moved downstream from Yale Lake.

Following spawning, bull trout embryos incubate in the gravel for 50 to 250 days depending upon water temperatures (Fraley and Shepard 1989). In the Lewis River basin, emergence is believed to occur from late January through early March (Figure 4.16-1) (pers. comm., E. Lesko, PacifiCorp, October 2000). While rearing in the tributary streams, juvenile bull trout are territorial, opportunistic feeders. Juveniles under approximately 11 cm (4.3 in) in length tend to be benthic foragers, feeding predominantly on insects, leeches, snails, and salmonid eggs. As they grow, bull trout become increasingly piscivorous (Goetz 1989). Bull trout juveniles are strongly associated with cover, including the interstitial spaces in the substrate, which makes them especially vulnerable to effects of sediment deposition, bedload movement, and changes in channel morphology (Pratt 1985, USFWS 1998).

Adfluvial bull trout juveniles remain in tributary streams for up to 6 years before migrating downstream to lakes, although most migrate at 2 to 3 years of age (Goetz 1989, Fraley and Shepard 1989, McPhail and Murray 1979). These individuals then spend an additional 2 to 3 years in the lentic environment before migrating back to their natal stream to spawn (total age of 4 to 7 years) (Goetz 1989). In lakes and reservoirs, bull trout are found throughout the water column during the fall, winter and spring, often near the mouths of migration routes (USFWS 1998). In the summer, as water temperatures begin to warm, they are reported to move into deeper water, often below the thermocline (Goetz 1989).

Kokanee – Kokanee are the resident form of the anadromous sockeye salmon *(Oncorhynchus nerka)*. These semelparous fish complete their entire life cycle in freshwater and are closely associated with lentic environments. Due to the lower productivity in freshwater, in comparison to the ocean, kokanee are usually smaller than sockeye at maturity (Meehan and Bjornn 1991). Kokanee were introduced into the North Fork Lewis River above Merwin Dam in the late 1950s.

In the Lewis River basin, kokanee mature and migrate to their natal streams in mid-September through mid-October, peaking in October (Figure 4.16-1). Peak spawning also occurs in October (pers. comm., E. Lesko, PacifiCorp, October 2000). The majority of the spawning in the system occurs in Cougar Creek, although limited spawning has been observed in the Swift No. 2 bypass reach, Ole Creek, Canyon Creek, and Speelyai Creek. However, it is likely that there is minimal natural production in the tributaries to Lake Merwin and that the primary recruitment of kokanee into the reservoir is from Yale Lake, resulting from fish passing the dam. Spawning in streams usually occurs at temperatures between 5.0 and 12.8°C (41 and 55°F) in slower moving riffle areas (15 to 75 cm/sec velocities) near the stream margins at depths of 6 cm (2.4 in) or greater (Levy and Slaney 1993, Larsen 1998). Preferred spawning gravel sizes range from 1.3 to 1.9 cm (0.5 to 0.7 in) in diameter (Meehan and Bjornn 1991). Some individuals may also spawn in gravel areas along lake shores in areas with groundwater upwelling and gravel sizes from 0.3 to 2.5 cm in diameter. Lake spawning depths range from 0.3 to 9.1 m (1.0 to 29.9 ft).

In the Lewis River basin, kokanee emerge from the gravel from February to early-March and spend only a short period of time in streams before they outmigrate to the lakes from

mid-March through April (Figure 4.16-1). Kokanee outmigration is highly synchronized and occurs during the night, so that thousands of fry swim/drift en masse to the lake in an attempt to minimize predation (Burgner 1991).

While rearing in lakes, juvenile kokanee are primarily plankton feeders. When they first enter the lake they remain in nearshore areas at depths of less than 9 meters (29.5 ft). As they grow, they begin to school together and display a diel pattern that includes inhabiting surface waters from dusk to dawn and descending to the deeper, cooler waters to spend the daylight hours (Burgner 1991). Such a pattern aids in avoiding predators during feeding times (dusk and dawn) and provides for energy conservation during the daylight hours.

The primary purpose for kokanee introduction in the 1950s was to create a recreational reservoir fishery. The species' introduction has been quite successful, as there are now self-sustaining populations in both Yale Lake and Lake Merwin, although the majority of kokanee production is from Yale Lake.

Competitive Interactions

In general, competition occurs when a number of organisms of the same (intra-specific) or different (inter-specific) species exploit common resources, the supply of which is limited (Birch 1957, Larkin 1956). For salmonid species (i.e., Chinook, coho, steelhead, kokanee, bull trout, and cutthroat trout), both intra-specific and inter-specific competition occurs primarily for habitat space rather than directly for food or other resources. In other words, fish are competing for the ability to forage in a given area and are not actually competing for individual food sources. Competition for habitat space is most critical during spawning and fry emergence and when seasonal low flow periods (summer and early fall) limit available instream space (Fresh 1997).

During both spawning and juvenile rearing, intra-specific competition is normally more of a factor in the survival of individuals than inter-specific competition (Fresh 1997, Hearn 1987). This occurs because individuals of the same species share the exact same habitat requirements, while habitat preferences between species differ in important aspects. During spawning, conspecific (individuals of the same species) females compete for available spawning locations, while males compete for access to the females (Wilson 1997). During rearing, conspecifics compete for particular microhabitat locations that afford an energetic advantage (Hearn 1987).

Levels of both intra-specific and inter-specific competition appear to be highly density-dependent, particularly during the first few months of juvenile rearing when mortality is a crucial population regulating process for salmonids. This mortality is generally attributed to competition resulting from peak fry densities that far exceed a stream's carrying capacity for producing smolts. After the initial period of competition for limited habitat, salmonids populations are generally believed to be regulated by density-independent factors, thereby reducing the need for competitive interactions (Hearn 1987).

The concern regarding reintroduction of anadromous salmonids to the upper Lewis River basin relates to the Competitive Exclusion Principle, which states that competition

between 2 species having similar resource requirements may lead to the exclusion of one of the species. Therefore, to successfully co-exist, species must possess distinct habitat niches that provide for a partitioning of habitat and resources between species (Hearn 1987, Larkin 1956).

Researchers have pointed out that in streams throughout the northwest, Pacific salmonid species have lived successfully in sympatry and that some stream systems have supported all salmonid species concurrently. These successes substantiate the general belief that salmonids have species-specific behavioral and genetic preferences for different microhabitats or niches, which aid in successfully partitioning available habitat (Fresh 1997). Such microhabitat parameters include preferences for certain water depths and velocities, temperatures, availability of cover, proximity to other fish, and location of the stream segment within the watershed (Fresh 1997, Griffith 1988, Fausch 1993). Therefore, for competitive interactions to contribute to species decline in a given area, something must be altered that compromises the balance of species' abundance and specific habitat niche availability (Fresh 1997, Nakano et al. 1992). In the Lewis River basin there have been numerous alterations of the natural system (i.e. dam construction, non-native fish introductions, hatchery production, habitat degradation, water temperature and water quality modifications, and harvest) that may disrupt the natural balance between fish species and could potentially lead to competitive interactions and exclusion (Fresh 1997).

Even with species that have lived sympatrically in other systems, inter-species competition may be substantial when species that have evolved in allopatry in a given stream are suddenly placed together. This is attributed to a process termed "ecological release," where a species' niche expands in the absence of other fish with similar life-history timing and habitat requirements (Hearn 1987). Individuals of ecologically released species may occupy habitats typically dominated by competitor species when in sympatry. In such cases, species reintroductions may cause intense inter-specific competition and declines in the ecologically released species, even though these species may have once co-existed in the system (Hearn 1987, Fresh 1997). The intensity of the competition is difficult to forecast, as species may quickly develop niche shifts that provide for successful cohabitation or interactions could escalate to the point that a particular species would be competitively excluded from the system (Fausch 1988). However, re-introduction of a once native species would be expected to result in less inter-specific competition than introductions that bring together species that are not naturally sympatric (Hearn 1987). Even in instances when direct competition for resources does not occur after re-introduction, the original resident species may experience reduction in biomass and growth rates resulting from compression of their habitat niche (Hearn 1987).

Due to the importance of habitat niches in determining species interactions, it is crucial to first examine potential timing overlaps and habitat requirements of the salmonid species inhabiting the Lewis River basin. From examining the periodicity chart for Lewis River salmonid species (Figure 4.16-1) reveals overlap in salmonid use at key life history points (i.e., emergence, spawning). However, simply because there is overlap in timing does

not necessarily mean that competition will result. Many of these species have differing microhabitat needs, which aids in segregating available habitat to avoid competition.

Overlaps in Spawning Timing and Spawning Habitat Requirements – One of the evolutionary mechanisms that allow a variety of salmonid species to successfully coexist within a single watershed is variation in spawning timing. However, modifications of "natural" or locally adapted spawning periods caused by hatchery production (i.e., selective spawning) or the introduction of new stocks/species can lead to competition for available spawning habitat. In the Lewis River basin, there are spawning overlaps in the fall and early winter reproducing species such Chinook, coho, kokanee, and bull trout in addition to spawning overlaps in the spring reproducing species, steelhead and cutthroat trout (Figure 4.16-1). The following paragraphs define these spawning overlaps and discuss how differences in habitat utilization aid in partitioning available spawning areas.

Chinook and Coho Salmon – The wild Lewis River spring and fall Chinook spawning period in October and November overlaps with the wild coho spawning period (Figure 4.16-1). As discussed previously, spring and fall Chinook prefer to spawn in the mainstem areas of a watershed, while coho usually spawn in smaller tributary streams (WDFW 1994). Therefore, although the 2 species' run timings may coincide, it is unlikely that Chinook and coho would be attempting to utilize the same spawning areas, at least under normal circumstances.

Both Chinook salmon and coho do, however, share similar spawning microhabitat preferences (i.e., water depths, velocities, and gravel sizes) (Table 4.16-2). Thus, when the preferred habitat of the 2 species is limited, there may be some overlap, as individuals must attempt spawning in less suitable areas. Because of this potential for spawning overlap, it is possible that Chinook or coho individuals spawning in the same area may excavate redds of fish that have already spawned. However, this is more of an intra-specific than an inter-specific concern, as wild Lewis River adult Chinook would be expected to build their redds and deposit their eggs at greater depths, thereby making them relatively safe from later spawning coho that may occur in the same area (Essington et al. 2000). It is important to consider, that these are only general spawning trends and it is possible that the overlap in timings would result in some disturbances of Chinook and coho redds. Because of the differing preferences for mainstem (Chinook) versus tributary (coho) spawning locations, such effects would not be expected to be substantial.

Table 4.16-2. General spawning habitat characteristics for Chinook and coho.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Chinook	Mid-October through Mid-November	Most commonly, middle and upper reaches of mainstem rivers, but also tributaries greater than 2m wide	>24cm	30 to 91 cm/sec	1.3 to 10.2 cm in diameter	5.6 to 13.9°C
Coho		Tributary streams with widths of 1 to 5m	10 to 53 cm	30 to 91 cm/sec	0.2 to 10 cm in diameter	4.4 to 9.4°C

Chinook Interactions with Kokanee – Kokanee spawn throughout the month of October, which overlaps with the latter part of spring Chinook spawning and the beginning of fall Chinook spawning (Figure 4.16-1). Kokanee only spawn in the tributary streams of Yale Lake and possibly tributaries of Lake Merwin while Chinook prefer to spawn in mainstem and larger tributary habitat. Kokanee also require considerably smaller spawning substrates and shallower water depths than Chinook (Table 4.16-3). Due to these considerable differences in spawning microhabitat requirements, it is unlikely that the overlap in spawning timing between these 2 species would result in interactions that would limit productivity.

However, if Chinook spawning habitat were severely limited, the fish may use areas with the smaller substrates and shallower water depths preferred by kokanee for spawning. In such instances, the larger Chinook salmon would likely displace kokanee spawners to less desirable spawning locations. Furthermore, Chinook spawning activities in areas already used for kokanee redd construction would likely result in excavation of the kokanee eggs, as Chinook construct redds that are generally deeper than 22.5 cm (8.9 in) in the substrate, while kokanee deposit their eggs at very shallow depths (Evenson 2001).

Table 4.16-3. General spawning habitat characteristics for Chinook and kokanee.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Chinook	Throughout October	Most commonly, middle and upper reaches of mainstem rivers, but also tributaries greater than 2m wide	>24cm	30 to 91 cm/sec	1.3 to 10.2 cm in diameter	5.6 to 13.9°C
Kokanee		Margins of lower and middle reaches of tributary streams	>6 cm	15 to 75 cm/sec	1.3 to 1.9 cm	5.0 to 12.8°C

Chinook and Bull Trout – The bull trout and spring Chinook spawning period overlaps from mid-September through the middle of October (Figure 4.16-1). For many of the same reasons discussed above for kokanee, overlaps in bull trout and Chinook salmon spawning timing would not likely result in adverse interactions, as the 2 species have differing microhabitat requirements (Table 4.16-4). Bull trout almost exclusively spawn in the upper reaches of tributary streams, while Chinook are primarily mainstem spawners. Even though water depth, velocity, and temperature requirements are similar for the 2 species, there is a low likelihood that the fish would be spawning in the same stream reaches even if habitat were limiting.

Table 4.16-4. General spawning habitat characteristics for Chinook and bull trout.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Chinook	Mid-September through mid-October	Most commonly, middle and upper reaches of mainstem rivers, but also tributaries greater than 2m wide	>24cm	30 to 91 cm/sec	1.3 to 10.2 cm in diameter	5.6 to 13.9°C
Bull Trout		Middle and upper reaches of tributary streams	24 to 61 cm	4 to 61 cm/sec	1.6-6.4 cm in diameter	2 to 10°C

Bull Trout and Kokanee – Bull trout and kokanee populations in the Lewis River basin have a spawning period overlap in early and mid-October (Figure 4.16-1). Past studies regarding the interactions between these 2 species found that adverse effects due to spawning overlap may be minimized by differing microhabitat requirements, as illustrated in Table 4.16-5. Bull trout tend to spawn in the upper reaches of smaller spring-fed streams like Cougar Creek, while kokanee spawn in the lower reaches. Most kokanee in the Lewis River basin, those found in Yale Lake and Lake Merwin, spawn after bull trout and deposit their eggs in very shallow redds, whereas bull trout eggs are buried at depths averaging 10-15 cm (3.9-5.9 in), which would minimize the potential for disturbance of either species' redds from subsequent spawning by the other species (Shellberg 2002). These differences in microhabitat preferences and spawning behaviors would be expected to effectively minimize potential adverse interactions between bull trout and kokanee during spawning. Furthermore, the fact that these 2 species have co-existed successfully in the Lewis River basin since the introduction of kokanee, suggests that differences of spawning timing and microhabitat preferences are adequate to partition available habitat and support self-sustaining populations of both species.

Table 4.16-5. General spawning habitat characteristics for kokanee and bull trout.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Kokanee	Early through mid-October	Lower and Middle reaches of tributary streams	>6 cm	15 to 75 cm/sec	1.3 to 1.9 cm	5.0 to 12.8°C
Bull Trout		Middle and upper reaches of tributary streams	24 to 61 cm	4 to 61 cm/sec	1.6-6.4 cm in diameter	2 to 10°C

Coho and Kokanee – Coho and kokanee spawning in the Lewis River would overlap from mid- through late October (Figure 4.16-1). Both coho and kokanee would be expected to use lower tributary stream reaches, although the considerably larger size of adult coho in comparison to kokanee results in differing microhabitat preferences, as shown in Table 4.16-6. Coho spawn in glides and the transition area between pools and

riffles, whereas kokanee spawn along the stream margins of glides and slow moving riffles. In addition, coho use areas with considerably larger gravels than those found in kokanee spawning areas. These differing microhabitat preferences should aid in partitioning available spawning habitat, although when spawning densities are high, overlap may occur. Kokanee have been found to be sensitive to density dependent variables (Burgner 1991, Rieman and Myers 1992). Thus, the addition of coho to those areas of the watershed that support substantial kokanee spawning (primarily Cougar Creek) could reduce the reproductive success of kokanee if fish densities were substantially increased. Furthermore, in areas where the spawning activities of the 2 species overlapped, kokanee redds would be highly susceptible to excavation by coho, as kokanee in the Lewis River have almost completed spawning before the deeper coho redds are constructed (Essington et al. 2000). The potential severity of these effects is uncertain since it would depend on the availability of preferred spawning habitat for both species, as overlap would only likely occur when preferred spawning habitat was limited. Monitoring of interactions between spawning kokanee and coho may be warranted if coho were reintroduced to the upper Lewis River basin.

Table 4.16-6. General spawning habitat characteristics for kokanee and coho.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Kokanee	Mid- through late October	Lower and middle reaches of tributary streams	>6 cm	15 to 75 cm/sec	1.3 to 1.9 cm	5.0 to 12.8°C
Coho		Tributary streams with widths of 1 to 5m	10 to 53 cm	30 to 91 cm/sec	0.2 to 10 cm in diameter	4.4 to 9.4°C

Bull Trout and Coho – Coho and bull trout spawning in Lewis River tributaries would likely overlap for a short period in mid-October. Bull trout would generally be expected to spawn further upstream in tributaries than coho, although there may be overlap if available spawning locations are limiting. Both of these species tend to spawn in glides and pool tail-outs and other similar microhabitat characteristics as shown in Table 4.16-7. The egg burial depths for both coho and bull trout are comparable. Coho burry their eggs at depths of 10 to 27 cm (3.9 to 10.6 in), which are similar to the bull trout burial depths mentioned above (van den Berghe and Gross 1984). Furthermore, both coho and bull trout have been documented as being rather aggressive on the spawning grounds, actively defending their redd locations from other would be spawners (Sandercock 1991, Goetz 1989, Scott and Crossman 1973). Such agonistic (aggressive or defensive) behavior may impact the reproductive success of some of the adults of both species.

Table 4.16-7. General spawning habitat characteristics for bull trout and coho.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Bull Trout	Mid-October	Middle and upper reaches of tributary streams	24 to 61 cm	4 to 61 cm/sec	1.6-6.4 cm in diameter	2 to 10°C
Coho		Tributary streams with widths of 1 to 5m	10 to 53 cm	30 to 91 cm/sec	0.2 to 10 cm in diameter	4.4 to 9.4°C

There has been little research of inter-specific adult interactions on the spawning grounds; therefore, it is not possible to accurately predict how the spawning timing overlaps between coho and bull trout would affect the reproductive success of the species (pers. comm., T. Quinn, UW School of Fisheries, June 2002). However, some studies have found that inter-specific competition between salmonids for similar microhabitats can lead to density dependent reductions in reproductive success. In general, either the earlier spawning species or the species with the smaller body size suffers the greatest, due to the susceptibility to disturbance of their redds (Essington et al. 2000). Yet such information is not tremendously helpful in the case of coho and bull trout in the Lewis River, as spawning timing and redd depths are highly comparable. Because of this uncertainty regarding potential effects, monitoring of adult coho and bull trout interactions should be conducted if coho were reintroduced above the Lewis River projects.

Steelhead/Rainbow Trout and Coastal Cutthroat Trout – The March and April spawning period of both summer and winter steelhead in the Lewis River basin overlaps with that of coastal cutthroat trout (Figure 4.16-1). However, these species have a long history of sympatry and appear to have developed mechanisms to partition available spawning habitat. Cutthroat trout tend to spawn in the upper reaches of small tributary streams, whereas steelhead in the Lewis River have been documented as using the mainstem and lower reaches of tributary streams. Yet the spawning depths, water velocities, and preferred gravel sizes for steelhead and coastal cutthroat are comparable (Table 4.16-8). There is also evidence that hybridization between steelhead and coastal cutthroat may occur, as discussed below. Thus, it is difficult to assess how species interactions might occur in the Lewis River basin if both species were released into the upper watershed. Since no studies were found that evaluated adult steelhead and coastal cutthroat interactions, it may be prudent to study the interactions of naturally spawning steelhead and coastal cutthroat in the Lewis River to better understand the potential effects of interactions.

Furthermore, no documentation was found regarding the potential interactions between resident adult rainbow trout and coastal cutthroat trout. The spawning timing of resident rainbow trout and coastal cutthroat trout is thought to generally coincide with the timing of anadromous spawners. For example, the spawning timing of rainbow trout stocked in Swift Reservoir for a recreational fishery would spawn anywhere from October through March (Crawford 1979). Therefore, in addition to potential interactions between

reintroduced steelhead and sea-run cutthroat trout, there is also the potential for spawning interactions with resident rainbow and cutthroat trout. However, the extent to which the stocked rainbow trout survive to reproduce is highly uncertain. Genetic studies of rainbow trout in Canyon and Siouxon creeks suggest that hatchery fish are not regularly interbreeding with wild rainbow trout, as is discussed in greater detail in Section 4.16.5.4.

Table 4.16-8. General spawning habitat characteristics for steelhead and cutthroat.

Species	Spawning Overlap	Spawning Location	Spawning Depths	Spawning Velocities	Substrate Size	Water Temps.
Steelhead /Rainbow	March through mid-April	Mainstem reaches, lower and middle tributary reaches	10 to 138 cm	30 to 110 cm/sec	0.6 to 13 cm	3.9 to 9.4°C
Cutthroat		Low gradient reaches of tributary streams	15 to 45 cm	11 to 72 cm/sec	0.5 to 5 cm	6.1 to 17.2°C

<u>Juvenile Interactions</u> – Due to extended freshwater rearing periods, the majority of the salmonid species inhabiting the Lewis River basin would be expected to occur concurrently in both tributary streams and the mainstem reaches. The notable exceptions would be fall Chinook and kokanee. Fall Chinook migrate to the estuary shortly after emergence, and kokanee migrate to project reservoirs immediately following emergence. Spring Chinook, coho, steelhead, bull trout, and cutthroat trout would be expected to share the habitat of the Lewis River and it tributaries throughout the year. However, as mentioned before, these species frequently occur naturally in sympatry in many Pacific coast streams and all, with the exception of kokanee, occurred in the upper Lewis River basin prior to the construction of the dams. It is generally thought that there are genetic differences in timing and microhabitat preferences that aid in regulating inter-species competition for habitat (Nakano et al. 1992, Hearn 1987, Everest and Chapman 1972).

Perhaps the most obvious potential difference is in the timing of fry emergence. Fry of different species emerging at or near the same time of the year would be expected to be similar in size, and therefore may have similar habitat and feeding requirements.

In the Lewis River basin, bull trout are the first species to emerge from the gravel (from late January through early March) (Figure 4.16-1). If anadromous salmonids were introduced in the upper Lewis River watershed, bull trout emergence would overlap substantially with spring Chinook (emerging from February through mid-March), fall Chinook (emerging from mid-February to mid-April), and coho (emerging from late February through late April). Thus, with such extensive overlap in emergence timing, it is important to evaluate habitat and food requirements in an effort to understand how species interactions may unfold in the Lewis River if anadromous populations were reintroduced above the project dams.

Bull Trout and Coho – In a study that examined the interactions between juvenile coho and juvenile Dolly Varden (a close relative of bull trout), it was found that coho tended to maintain specific focal points, occupying low velocity areas near areas of higher velocities. Coho juvenile also tended to remain near the surface for drift foraging.

Conversely, Dolly Varden did not maintain specific focal points and remained in the shallow stream margins near cover, staying primarily near the substrate, presumably feeding on benthic invertebrates (Dolloff and Reeves 1990). In all testing situations (differing sizes and ages) coho tended to be higher in the water column, in shallower water, further from the nearest cover, and further from the nearest fish in comparison to Dolly Varden. These differing microhabitat preferences were demonstrated even when the 2 species were reared in a laboratory stream in allopatry. Furthermore, the peak emergence timing of bull trout in the Lewis River basin tends to be a week or 2 earlier than that of Chinook salmon (Figure 4.16-1). Thus, bull trout may reach a larger size at an earlier time period, which can aid in further partitioning habitat, as individuals of both species grow and tend to utilize areas of deeper water depth and higher velocities (Dolloff and Reeves 1990, Lister and Genoe 1970). Dolloff and Reeves (1990) concluded that differing habitat preferences were a greater determinant of segregation between the species than direct competition and agonistic interactions. Interestingly, in both laboratory and natural environments, coho and Dolly Varden tended to be more tolerant of individuals of the other species than conspecific fish. Another important finding of this study from a habitat management perspective is that both species tended to be more tolerant of other fish in areas of increased habitat complexity (i.e., large woody debris, boulders) (Doloff and Reeves 1990).

Bull Trout and Chinook – The literature review conducted for this study was unable to locate research efforts that specifically addressed interactions between bull trout and Chinook salmon. However, there is literature regarding interactions of juvenile coho and Chinook, which is important to consider for potential reintroductions and also provides limited inferred findings about potential interactions between Chinook and bull trout. It is important to consider that this information is, for the most part, specific to spring Chinook, as they are the stock that demonstrates prolonged freshwater residence, whereas, fall Chinook migrate to the ocean shortly after emergence (PacifiCorp and Cowlitz PUD 2002a).

The emergence timing of wild Lewis River Chinook and bull trout closely overlap; therefore, differences in size at a given time may not play a major role in separating the habitat of the 2 species. Bull trout are even more closely associated with cover than are coho salmon (Dolloff and Reeves 1990). Thus, juvenile bull trout would be expected to be found in the stream margins, whereas Chinook would use areas with deeper water and higher velocities. Furthermore, Chinook are more tolerant of higher water temperatures and can maintain growth rates in warmer, mainstem reaches (Stein et al. 1972). Bull trout, on the other hand, are extremely temperature sensitive and would generally be expected to maintain residence in the cool waters of upper basin tributaries. These differences in habitat preference may preclude adverse interactions between juvenile Chinook and bull trout in the Lewis River watershed. It is important to note that these statements are based upon deductive reasoning, as no documentation was found pertaining to bull trout Chinook salmon interactions studies. Due to the uncertainty regarding potential interactions, it would be important to monitor the status of bull trout if Chinook were reintroduced to the upper Lewis River watershed.

Chinook and Coho – Wild Lewis River spring Chinook and coho salmon emerge from the gravel in early spring, but slight differences in emergence timing and habitat preferences aid in partitioning available juvenile rearing habitat. For example, juvenile spring Chinook have been found to be less closely associated with cover than coho. Juvenile spring Chinook tend to use open water habitat with consistently higher water velocities than the areas inhabited by coho. Due to this preference, Chinook are generally found in the mainstem of rivers utilizing boulders, cobble, and rip-rap as cover rather than large woody debris (Taylor 1988, Stein et al. 1972). Furthermore, Lewis River spring Chinook salmon tend to emerge earlier than Lewis River coho. This can lead to increased growth in Chinook, causing them to be larger than the emerging coho at any given time. This larger body size enables use of deeper water and higher water velocities, while coho have been found to be less adept at maintaining stations at higher velocities. These slight differences in emergence timing may be the cause of these important morphological and behavioral differences that contribute to species segregation (Hearn 1987, Glova 1986).

In stream systems where the emergence timing of the 2 species overlaps, coho generally out-compete Chinook. Chinook have been found to experience somewhat slower growth rates in the presence of coho salmon than when raised in allopatry, while coho growth rates remain the same either way (Stein et al. 1972). This may be due in part to Chinook's tolerance for higher densities of conspecifics in an area, which could limit feeding and growth potential (Stein et al. 1972). Conversely, coho are notably intolerant of the presence of other coho and tend to display more aggression toward conspecifics than other species, which may explain the identical coho growth rates whether Chinook were present or not (Glova 1986). During instances when direct competition occurs, coho and Chinook display agonistic behavior for short periods until a habitat hierarchy is established in which coho tend to successfully defend the cooler, slower moving tributary reaches, while Chinook reside in the warmer, swifter mainstem reaches (Hearn 1987, Stein et al. 1972). This trend also has a biological component, as Chinook have been found to be more able to maintain normal growth rates at higher water temperatures than coho (Stein et al. 1972). These studies support the idea that both biological and behavioral traits exist in coho and Chinook that should allow the 2 species to co-exist without harming the productivity of either of the species.

Cutthroat and Bull Trout – Cutthroat tend to rear in upper reaches of relatively small tributaries in a watershed, although in the absence of other salmonids they may experience ecological release and expand their habitat into lower reaches (Hearn 1987, Glova 1986, Griffith 1988). Cutthroat successfully use riffle habitat in the presence of other salmonids and primarily feed on invertebrates and available fish in the middle to upper areas of the water column (Griffith 1988). The presence or absence of bull trout in a stream has been found to not substantially alter the habitat preferences of either species. As discussed before, bull trout are primarily benthic foragers, while cutthroat are drift foragers. Cutthroat are also less strongly associated with cover than bull trout. At a given age, bull trout tend to be larger than cutthroat, which would also aid in partitioning available habitat (Nakano et al. 1992). These different habitat niche requirements are believed to effectively segregate available space, suggesting that species interactions are

not a major component in successful cohabitation for cutthroat and bull trout (Nakano et al. 1998).

Cutthroat and Chinook – Interactions between juvenile cutthroat trout and juvenile Chinook salmon would not be expected to hinder the health or survival of either species. Although no studies were found that specifically addressed interactions between these species, differences in habitat preferences and emergence timing would be expected to partition habitat. Chinook emerge earlier than cutthroat and would therefore be expected to be larger than cutthroat juveniles at any given time. Chinook use mainstem reaches with higher velocities and warmer temperatures, while cutthroat tend to reside in upper watershed reaches. Thus, adverse interactions between these species would not be expected.

Cutthroat and Steelhead/Rainbow Trout – Competition between cutthroat and steelhead (and resident rainbow trout) trout is one of the better studied of the salmonid interactions. In general, rainbow trout and steelhead are able to out-compete cutthroat trout and can displace individuals to less desirable habitat locations and even cause extirpation of cutthroat when non-native rainbow are introduced to streams (Griffith 1988). Studies in British Columbia and in Washington have found that when rainbow trout and steelhead live in sympatry with cutthroat, the rainbow and steelhead tend to occupy lower reaches of tributary streams, while cutthroat are more abundant in headwater tributaries (Griffith 1988). Rainbow are usually located lower in the water column, utilizing benthic foraging strategies, as opposed to the drift foraging implemented by cutthroat (Bisson et al. 1988, Everest and Chapman 1972). In addition, cutthroat have been found to grow more slowly than rainbow trout and steelhead, which further helps to partition available habitat resources, as the smaller cutthroat occupy different microhabitats than the larger rainbow and steelhead trout.

In the Lewis River watershed, cutthroat fry emerge earlier than wild rainbow trout (mid-March through mid-June compared to mid-June through late-August). This slightly earlier emergence timing has been cited as another genetic mechanism that aids in partitioning the available habitat between cutthroat trout and rainbow and steelhead trout (Griffith 1988). However, due to their similar habitat requirements, the introduction of non-native hatchery rainbow trout to a stream system has been noted as having potentially significant effects on cutthroat populations, as cutthroat are markedly less aggressive than rainbow and steelhead trout and are generally out-competed when interactions occur between the species (Griffith 1988). Furthermore, when the habitat niches of steelhead/rainbow trout and cutthroat trout overlap, steelhead usually displace cutthroat from the riffle areas, which leads to cutthroat occupation of glides or migration further into the headwater tributaries (Griffith 1988, Sabo and Pauley 1997, Hartman and Gill 1968). This is of particular concern in the Lewis River watershed, as rainbow trout have been stocked in project reservoirs to provide a recreational fishery. The extent to which these fish enter streams and may interact with juvenile cutthroat trout is uncertain. In addition, if both sea-run cutthroat and steelhead trout were reintroduced to the upper watershed, the potential effects of interactions between anadromous and resident forms of these species could be highly density-dependent and could require monitoring to ensure there is adequate habitat to support both species.

Cutthroat, Rainbow, and Coho Dominance Hierarchies – Cutthroat trout and other salmonid species, most notably rainbow trout and coho salmon, have been found to form localized dominance hierarchies, which maintain habitat partitioning and confer dominant individuals with greater access to highly productive habitat (Hearn 1987). Coho are dominant over cutthroat trout and tend to quickly drive-off cutthroat from preferred coho habitat. This dominance is manifested through much higher rates of aggressive behavior (approximately 30 percent higher levels of aggressive behavior in comparison to cutthroat) that aids in defining the local social hierarchy (Glova 1986). For example, with coho present in a stream, cutthroat are primarily found in glides and riffles, while coho occupy the heads of pools and glides (Griffith 1988, Sabo and Pauley 1997). In the absence of coho, cutthroat were found to increase their use of pools by approximately 20 to 30 percent (Glova 1986). Therefore, in the absence of coho in the Lewis River, cutthroat may have expanded their habitat and may be increasing their use of pools. Reintroduction of coho may compress the current cutthroat trout habitat niche, thereby, reducing growth rates. However, these 2 species historically existed in sympatry in the Lewis River basin and potential effects on cutthroat would not likely compromise the persistence of the species in the upper watershed, but may result in a shift in habitat utilization. In addition, cutthroat trout are relatively uncommon in the upper Lewis River habitats, in spite of the 50 year absence.

The evidence presented above suggests that cutthroat are the least likely of the 3 species to defend their habitat niche. As mentioned previously, cutthroat are generally less aggressive than either coho or steelhead. It has been hypothesized that this may be due to a lack of morphological specialization for either pool or riffle habitat. In comparison to cutthroat, coho display morphological adaptations that are superior for maneuvering and holding position in pools, while steelhead have similar advantages over cutthroat in riffles. Furthermore, body size has been linked to behavioral dominance, and juvenile cutthroat trout in southwestern Washington streams have been found to weigh approximately 10 percent less than juvenile steelhead of equal length (Griffith 1988). These morphological differences may help to explain why cutthroat are generally the displaced species when interactions with coho and steelhead occur. This suggests the need to consider the health and distribution of cutthroat stocks in the event of anadromous fish re-introductions to the upper Lewis River watershed.

Cutthroat Intra-Specific Competition – Within riffle habitat, cutthroat display isolated instances of intense intra-specific competition. This intra-specific competition appears to be most severe during summer months when available habitat is limited and salmonid densities are at their highest. Furthermore, aggressive behavior is most often displayed during feeding times, as cutthroat compete for the prime feeding areas in the riffle (Glova 1986). Intra-specific competition between cutthroat tends to decline over the summer as social hierarchies are established and densities slowly decline (Glova 1986).

Rainbow/Steelhead and Chinook – Both winter and summer steelhead and rainbow trout emerge later than the majority of the other Lewis River salmonids. However, they rear in freshwater for 1 to 3 years (steelhead) or their entire lives (rainbow). As a result, there is potential for habitat overlap and interactions with the salmonid species that emerge much earlier in the year. Studies regarding interactions between steelhead and the earlier

emerging anadromous salmonid species (i.e. Chinook and coho) have found that sufficient microhabitat preferences exist to preclude substantial agonistic behavior between the species. Everest and Chapman (1972) found no evidence that either steelhead or Chinook changed their habitat preferences in the presence of the other species. In addition, studies that have examined growth rates of Chinook, steelhead, and rainbow trout when raised together or apart found no significant growth rate differences for either species whether raised in sympatry or allopatry (Pearsons et al. 1996, McMichael and Pearsons 1998). These results have primarily been attributed to the earlier emergence of Chinook salmon, which leads to young-of-the-year Chinook being larger than young-of-the-year steelhead and rainbow trout (Hearn 1987). Larger body size leads to different microhabitat and food preferences, which is thought to adequately partition habitat between these 2 species.

Rainbow/Steelhead and Bull Trout – Competition between bull trout and steelhead is controlled by many of the same factors that control steelhead and Chinook competition. Because bull trout emerge earlier than rainbow trout and steelhead, microhabitat requirements and prey preferences successfully segregate the 2 species (Boag 1987). As salmonids grow, they tend to move to deeper areas with higher water velocities, thus the species that emerges first moves to deeper and swifter water which limits overlaps in habitat requirements (Hearn 1987). In the Lewis River, bull trout emerge at least 3 months before steelhead and rainbow trout, which affords substantial time for growth. Thus, even though both species tend to occupy areas low in the water column near the substrate, the actual areas in the stream preferred by bull trout at a given time are thought to be substantially different from those of rainbow and steelhead (Goetz 1989, Everest and Chapman 1972). Studies have found that bull trout and rainbow sampled at the same time of year have substantially different diets. Rainbow tend to feed almost exclusively on insects, while bull trout become increasingly piscivorous as they grow (Boag 1987). In addition, rainbow tend to inhabit areas lower in the watershed than bull trout, which would further contribute to habitat partitioning. In general, competition for habitat between bull trout and rainbow or steelhead is not thought to be a major concern (Boag 1987).

Rainbow/Steelhead and Coho – Even though coho salmon emerge approximately 2 months earlier than rainbow trout and steelhead, competitive interactions have been documented to exist between these 2 species. These interactions are generally attributed to the extended freshwater residence of both species and their similar habitat requirements (Stein et al. 1972, Hartman 1965). During the first year of growth, the larger body size of coho fry appears to adequately partition available habitat, as the larger coho fry would have different microhabitat preferences than the smaller rainbow and steelhead (Hearn 1987, Fausch 1993). This is especially important, as the period after emergence is the time in which juvenile salmonids are the most susceptible to adverse species interactions due to high densities, which can lead to competition for limited habitat resources (Griffith 1988). However, after the first summer of growth, overwintering steelhead and coho of the same age were found to be similar in size, making competition between the species in subsequent years of greater concern (Hartman 1965).

Despite similar body sizes, studies have revealed important behavioral components that allow these 2 species to coexist. In the absence of the other species, both coho and steelhead have been found to prefer pools. When both species are present, a social hierarchy develops in which coho aggressively defend prime feeding locations at the heads of pools and steelhead generally shift to riffles (Hartman 1965). Interestingly, steelhead tend to effectively defend riffle locations from coho encroachment. If this were not the case, coho could potentially drive steelhead from a system by taking over both pool and riffle habitats. Yet the steelhead's behavioral dominance in riffles and coho dominance in pools helps to maintain a balance in the stream, allowing the species to successfully cohabit (Hartman 1965). These findings suggest that if carrying capacity of the habitat is not exceeded, which could lead to disruption of the balance between steelhead and coho, then the upper watershed of the Lewis River should be able to support both species without competitive interactions compromising the viability of either of the stocks.

Coho Intra-Specific Interactions – Up to this point, this study has focused on coho interactions with other species; however, it is also important to consider intra-specific coho interactions. As mentioned above, for all salmonid, species intra-specific competition is generally considered a greater regulator of species populations than inter-species competition (Hearn 1987). This fact may be even more pronounced in the case of coho salmon, as they tend to be extremely aggressive and are generally intolerant of conspecific presence. Coho tend to form rather rigid social hierarchies consisting of 3 primary traits: 1) dominant, 2) subordinate, or 3) nomadic. The dominant fish are those that defend the prime feeding locations at the heads of pools. Subordinate fish maintain specific positions in pools or glides, but they avoid confrontations with dominant individuals by staying in less desirable feeding areas. Nomadic coho actively pursue prey from location to location (Chapman 1962). Of these 3 strategies, the dominant fish are the individuals that attain the largest size and have the highest growth rates. This has been attributed to 3 primary reasons regarding energy expenditure:

1. Holding a specific location has lower search costs.

2. Residing in areas of low velocity near high velocity areas results in lower pursuit costs.

3. Initial aggression to establish dominance in the social hierarchy results in less harassment from conspecifics throughout the growing season, which reduces energy expenditure.

From this information it is apparent that active aggression displayed by coho salmon has developed primarily as a mechanism to gain a competitive advantage over conspecifics. Although coho aggression with other species has been noted, it is generally less intense than with conspecifics and competitive interactions between other salmonids and coho are thought to be regulated through differing habitat preferences and biological adaptations (Glova 1986, Fresh 1997).

Juvenile Overwintering Habitat – Since the majority of the Lewis River salmonid species spend at least one year in streams, with the notable exceptions of fall Chinook and kokanee, over-wintering habitat is a crucial component of juvenile rearing. Studies have consistently found that the levels of both intra-species and inter-species interactions are substantially lowered during the winter (Glova 1986, Hartman 1965, Taylor 1988). Most salmonids display a tendency to increase their use of pools during the winter, which is presumably a behavioral mechanism to reduce vulnerability to flushing during peak flow events (Taylor 1988). Chinook appear to be an exception as they will remain in areas with relatively high velocities even during the winter, although in general they increase their use of pools during winter in comparison to summer (Taylor 1988).

With most salmonids preferring pools during winter, an increase in aggression could be expected. However, this is not the case, as the colder water temperatures reduces metabolism, which in turn leads to decreased foraging activities and levels of aggression toward other fish (Glova 1986). In addition, overall densities of fish are less in winter, as many juveniles perish throughout the summer and fall. Salmonids also demonstrate an ability to fully utilize cover in pools (i.e. substrate, LWD) during the winter to aid in visually separating fish, which tends to reduce levels of aggression (Hartman 1965). Therefore, competition for over-wintering habitat would not be expected to be a major concern if re-introduction activities were conducted.

Competition with Non-Salmonid Species – In general, the habitat requirements of salmonid species are specialized to the point that competitive interactions between salmonids and non-salmonids occurs rather infrequently. However, in systems in which non-salmonid species are unusually abundant or are exotic, competition with these species may increase (Fresh 1997). This investigation was unable to locate studies specifically pertaining to salmonid competition with non-salmonid species, thus the subject is not evaluated in further detail in this report.

Summary of Data Gaps and Uncertainties – To aid in evaluating the potential accuracy and applicability of the analysis presented above, the following bulleted list describes the major data gaps and uncertainties.

- There is a moderate level of confidence in the information regarding life history of the Lewis River species. However, some of the information (i.e. diet, habitat preference) is based upon general information from studies of these species in other watersheds and may not be fully applicable to the Lewis River. Yet, slight variations in species' biology or behavior are not thought to have compromised the validity of the findings presented in this report.

- There is a high level of uncertainty regarding whether the resident species currently residing in the upper watershed have been ecologically released due to the absence of salmon populations. If such release has occurred and resident species are currently using habitats that would normally be utilized by anadromous salmonids, then there may be increased probability of adverse interactions.

- In general, very few studies were found regarding competition for spawning habitat. Thus, the majority of the findings presented in this report are based upon spawning

characteristics (i.e. run timing, habitat preference) for each species. In particular, no documentation was found specifically relating to Chinook and coho spawning interactions. There is a high level of uncertainty regarding whether the species would successfully partition available spawning habitat or whether coho and Chinook spawning habitat would overlap, thereby potentially adversely affecting productivity. Due to uncertainty, this potential interaction may require additional consideration if both species are reintroduced to the upper watershed.

- No documentation was found relating to adult kokanee interactions with spawning coho or Chinook salmon. Such interactions are fairly likely, especially between kokanee and coho, and may have an impact on kokanee spawning success. If this subject is of concern to fishery managers, then this subject may warrant further evaluation prior to reintroduction.

- Adult coho and kokanee interactions are fairly likely, due to their spawning habitat preferences. It is uncertain how these interactions might affect productivity of either species, although it is more likely that the non-native kokanee, rather than coho would be adversely affected, due to the deeper egg burial depths of coho salmon in comparison to kokanee.

- Both coho and bull trout have similar spawning habitat requirements and general egg burial depth characteristics. It is uncertain how the spawning overlap of these 2 species would affect either species. If bull trout have been ecologically released due to the absence of coho and are now spawning in areas historically used by the species, then spawning interactions could adversely affect bull trout. Due to the high level of uncertainty in regards to this interaction and the protection of bull trout under the ESA, this potential competitive interaction may warrant further consideration.

- There is a data gap regarding the adult interactions of steelhead/rainbow trout and coastal cutthroat trout. The spawning habitat characteristics of these 2 species are quite similar, suggesting the potential for species interactions. If either steelhead or anadromous cutthroat trout are chosen as a species to be reintroduced to the upper Lewis River watershed, the potential interactions between both anadromous and resident adult steelhead/rainbow trout and coastal cutthroat trout may warrant further investigation.

- In general, there is a lack of pertinent information regarding species interactions between juvenile salmonid species. Although there has been a great deal of study on this subject, it is not Lewis River specific and the studies reviewed did not deal with the reintroduction of species, but instead focused primarily on the interactions of salmonid population already cohabitating in the study areas. Therefore, it is uncertain how juvenile interactions might unfold once reintroduction efforts were undertaken. Upon initiation of reintroduction activities, monitoring programs geared toward assessing potential adverse interactions between juveniles in the upper Lewis River watershed may be prudent.

- In particular, the information regarding potential interactions between coho and bull trout was based upon a study using coho and Dolly Varden. Although Dolly Varden and bull trout are similar, there may be behavioral differences that could alter the way in which coho and bull trout interact. Furthermore, there were no studies found pertaining to juvenile interactions between bull trout and Chinook salmon, steelhead/rainbow trout, or cutthroat trout. Therefore, there is uncertainty regarding the potential for adverse interactions between bull trout and these species, which may warrant further study.

- Although there were no Lewis River-specific studies regarding juvenile coho and Chinook interactions, there has been a great deal of research conducted on this subject. Therefore, there is moderate confidence in the finding that these 2 species would likely coexist without substantial adverse juvenile interactions.

- Similarly, the interactions between cutthroat trout and steelhead/rainbow trout have been well studied and there is a moderate level of confidence in the findings discussed above, even though no Lewis River-specific studies have been conducted. However, there is no information regarding the behavior of resident rainbow trout released into the Lewis River to provide a recreational fishery. It is not known whether these fish enter tributary streams and their potential for interactions with wild and reintroduced stocks.

- Another data gap exists about the interactions of juvenile coho and Chinook salmon with cutthroat trout. No specific studies were found on this subject, thus the analysis was based upon general behavioral and habitat preference characteristics.

- Although no Lewis River-specific studies have been conducted regarding interactions of juvenile steelhead/rainbow trout with coho and Chinook salmon, there have been numerous studies conducted in other areas. Thus, there is a moderate level of confidence in the conclusions discussed above.

- Finally, no information was found about salmonid competition with non-salmonid species in the Lewis River watershed (i.e. lamprey, sculpin). However, since the behavior and habitat requirements of salmonids differ considerably from these other species, competition for habitat would not be expected to be a major concern.

Summary of Competition Findings – From a survival viewpoint, intra-specific competition is more of a concern than inter-specific competition, as individuals of the same species share identical lifestage timing and habitat requirements. Coho tend to be the species least tolerant of the presence of conspecifics or individuals of other species and they tend to dominate when competitive interactions occur. Levels of both types of competition are highly density dependent and are most prominent during early juvenile rearing and spawning.

Reintroduction of salmonids into the upper Lewis River basin may compress the habitat niches of resident species that have likely expanded their habitat use due to the absence of anadromous salmonids species. However, the impact of this competition in the Lewis River may be relatively minor, as the introduction of fish into a watershed is less likely to

result in adverse competitive interactions if the fish species was once native to the area. Table 4.16-9 provides an overview of the potential species interactions discussed in this section.

In the Lewis River watershed, the spawning timing of many of the salmonid species overlaps; however, differences in microhabitat preferences and egg burial depths may preclude substantial levels of intra-specific competition related to spawning. Still, if spawning habitat is a limiting factor, competition for available spawning habitat may increase. In general, when competitive interaction related to spawning occurs, the species the either spawns earlier or has a smaller body size suffers the greatest, due to their redds susceptibility to disturbance.

Differences in juvenile emergence timing aids in partitioning available rearing habitat, as fish that emerge at different times would vary in size, thereby creating differing microhabitat preferences. Furthermore, dominance hierarchies are rather quickly established in rearing areas, with coho tending to be the most aggressive and dominant, which relegates other species to different areas in the stream. Chinook generally reside in warmer, mainstem areas, while rainbow and cutthroat generally inhabit riffles, which aids in partitioning rearing habitat. Bull trout are highly cover and temperature dependent and usually retreat further upstream in a watershed when faced with potential competitive interactions.

With reintroduction, competitive interactions between juveniles may be rather intense while dominance hierarchies are established and the existing resident species shift their habitat niches to account for the increased salmonid densities. However, once these initial adjustments have been made, differences in timing, growth, and microhabitat requirements would be expected to reduce the significance of inter-specific competition. It is important to note that coho are the most aggressive of the salmonids considered for reintroduction; therefore, inter-species competition may be the highest in areas where coho are introduced.

Competition and aggression within all salmonid species tends to subside during the cooler winter months; therefore, competition for overwintering habitat would not be expected to be a major concern, assuming that adequate overwintering habitat exists to provide for increased salmonid abundance in the upper Lewis River watershed.

4.16.5.2 Potential for Disease Transmission

Disease is a naturally occurring component of fish ecology. Although aquaculture has brought greater attention to the deleterious effects of disease epidemics, such disease outbreaks cannot be solely attributed to hatchery operations. Hedrick (1998) addressed the common perceptions and misconceptions held by the public and scientific communities regarding the spread of disease-causing agents from hatchery fish to wild fish. He suggested that it is true that human activities have adversely affected fish health through direct changes in habitat and ecosystems; however, these changes do not necessarily mean that fish pathogens have been actively introduced to the wild through such actions (Hedrick 1998, Flagg et al. 2000). In fact, it has been suggested that disease

may be far more prevalent in the wild than is commonly acknowledged but that the primary victims of disease may be salmonid fry, a life stage at which we expect fairly high mortality. Disease-related mortality may serve as an effective mechanism for culling inferior fry, allowing only the fittest individuals to continue development and exploitation of the limited aquatic resources (Coutant 1998). However, this is only a hypothesis and there has been little research conducted regarding the actual impacts of disease in wild salmonid populations.

In a recent study regarding the effects of artificial production on the abundance of wild salmon populations, Flagg et al. (2000) discussed the 8 most important salmonid diseases, as defined by the Pacific Northwest Fish Health Protection Committee. These diseases include:

- *Bacterial*
  Bacterial kidney disease (BKD)
  Bacterial coldwater disease (BCD)

- *Viral*
  Infectious hematopoietic necrosis virus (IHNV)
  Erythrocytic inclusion body syndrome (EIBS)

- *Parasitic*
  Ceratomyxosis
  Whirling disease
  Ichthyophthiriosis ("ich")

- *Fungal*
  Miscellaneous species

The following discussion will focus primarily on these diseases, which are thought to be the most problematic in Pacific Northwest aquatic environments.

Bacterial Kidney Disease

Bacterial Kidney Disease (BKD) is a chronic systemic, highly virulent bacterial infection, with salmonids being the primary hosts (Egusa 1992). The disease, caused by the organism *Renibacterium salmoninarum*, can cause overt infections or be present in a benign carrier state (Plumb 1999). Epidemic outbreaks of the disease have only been documented as occurring in hatchery produced salmonids. The percentage of infected fish in hatcheries can range from 10-100 percent, although in the majority of instances, *R. salmoninarum* is present in low amounts and does not result in the manifestation of clinical symptoms (Rhodes et al. 1998). It is also common to find *R. salmoninarum* in wild salmonids, but displays of infections in these environments are rare (Plumb 1999). The pathogen can infect both juvenile and adult salmonids, although it is most prominent in zero age fish (Egusa 1992).

Table 4.16-9. Potential competitive interactions between Lewis River salmonid species.

Species	Chinook	Coho	Kokanee	Steelhead/Rainbow	Cutthroat	Bull Trout
Chinook *Adult*	Intra-species spawning interactions are density dependent dealing with female access to preferred habitat and male access to preferred females. When habitat is limiting, intra-species competition can be important.	Generally use different areas in the watershed. Similar water depth, velocity and substrate preferences. Limited available spawning habitat could lead to redd superimposition. Adverse interactions are generally unlikely.	Chinook spawning virtually complete prior to kokanee spawning. Deeper egg burial depths for Chinook make adverse interactions unlikely.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.	Chinook spawning virtually complete prior to bull trout spawning. Deeper egg burial depths for Chinook make adverse interactions unlikely.
Fry/Emergence/ Juvenile Rearing	Juvenile intra-species competition is for specific rearing micro-habitat. If such habitat is limiting, intra-species competition can be important.	Chinook juveniles utilize warmer mainstem reaches with higher velocities in comparison to the slow moving tributary reaches used by coho. Coho more closely associated with cover than Chinook. Growth rates of Chinook slightly lower in sympatry with coho than in allopatry, although habitat preferences and partitioning would make adverse interactions unlikely.	Kokanee migrate to project reservoirs shortly after emergence, thereby precluding the potential for adverse interactions.	Chinook fry emerge months before steelhead; therefore, disparities in size lead to differing habitat requirements at a given time. Studies of growth rates found that the presence or absence of the other species had no effect on either species' growth rates. Adverse interactions between these 2 species would be unlikely.	Chinook emerge earlier than cutthroat and would be expected to be larger than cutthroat at any given time. Chinook resided in mainstem reaches, while cutthroat are strongly associated with upper watershed reaches. Adverse interactions between these 2 species would be unlikely.	Emergence times substantially overlap. Bull trout are strongly associated with stream margins and cover, while Chinook use deeper and swifter areas. Bull trout are more sensitive to higher water temperatures than Chinook. These habitat preferences would make adverse interactions unlikely.
Coho *Adult*		Intra-species spawning interactions are density dependent dealing with female access to preferred habitat and male access to preferred females. When habitat is limiting, intra-species competition can be important.	Same lower watershed spawning areas. Kokanee use stream margins and smaller gravel than coho. Kokanee sensitive to density dependent variables. Risk of kokanee redd excavation in areas where spawning overlaps. The level of overlap and significance of effects depends on the availability of preferred spawning habitat for both species.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.	Spawning would be separated by bull trout preference for upper reaches versus coho preference for lower reaches. All other spawning habitat requirements are very similar (i.e., water depths and velocities, substrates, location in stream channel). If preferred habitat is limiting, adverse interactions may be likely.
Fry/Emergence/ Juvenile Rearing		Coho juveniles are the most aggressive of salmon species. Coho have adapted several foraging strategies that help in partitioning habitat between conspecifics. Unless habitat carrying capacity was exceeded, intra-specific interactions would not be expected to adversely affect coho populations.	Kokanee migrate to project reservoirs shortly after emergence, thereby precluding the potential for adverse interactions.	Coho emerge approximately 2 months before coho, which limits fry interactions. After first winter the individuals are of similar size. Agonistic interactions in early spring and summer establish social hierarchy that drives rainbow to riffles and pool bottoms. Rainbow/steelhead successfully defend riffles from coho encroachment, which maintains balance between the species. Agonistic behavior would occur but would not likely jeopardize either species unless densities exceeded habitat carrying capacity.	Extended freshwater residence leads to juvenile rearing overlap. Coho are substantially more aggressive than cutthroat. Agonistic interactions in early spring and summer establish a social hierarchy. Cutthroat move to riffles, while coho remain in the heads of pools. When habitat is limiting, cutthroat may be driven further upstream to habitat not utilized by coho and may experience reduced growth rates over current conditions without coho.	Bull trout tend to emerge before coho, which aids in partitioning habitat. Chinook are drift foragers, while bull trout are benthic foragers. Adverse interactions between these juveniles would be unlikely.
Kokanee *Adult*			Intra-species spawning interactions are density dependent dealing with female access to preferred habitat and male access to preferred females. When habitat is limiting, intra-species competition can be important.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.	Bull trout generally spawn before kokanee. Bull trout use upper watershed areas in comparison to the lower reaches utilized by kokanee. Adverse interactions unlikely.
Fry/Emergence/ Juvenile Rearing			Juvenile intra-species competition is for specific rearing micro-habitat. If such habitat is limiting, intra-species competition can be important.	Kokanee migrate to project reservoirs shortly after emergence, thereby precluding the potential for adverse interactions.	Kokanee migrate to project reservoirs shortly after emergence, thereby precluding the potential for adverse interactions.	Kokanee migrate to project reservoirs shortly after emergence, thereby precluding the potential for adverse interactions.

Table 4.16-9. Potential competitive interactions between Lewis River salmonid species (cont.).

Species	Chinook	Coho	Kokanee	Steelhead/Rainbow	Cutthroat	Bull Trout
Steelhead/Rainbow *Adult*				Intra-species spawning interactions are density dependent dealing with female access to preferred habitat and male access to preferred females. When habitat is limiting, intra-species competition can be important.	Spawning would be separated by cutthroat preference for upper reaches versus rainbow/steelhead preference for lower reaches. All other spawning habitat requirements are very similar (i.e., water depths and velocities, substrates, location in stream channel). If preferred habitat is limiting, adverse interactions may be likely.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.
Fry/Emergence/ Juvenile Rearing				Juvenile intra-species competition is for specific rearing micro-habitat. If such habitat is limiting, intra species competition can be important.	Rainbow/steelhead occupy lower reaches than cutthroat. Rainbow are located lower in the water column than cutthroat. Cutthroat are generally smaller than rainbow/steelhead, which further partitions habitat. When habitat is limiting, cutthroat are usually displaced further upstream. Adverse interactions between these 2 species would depend upon densities of both and habitat carrying capacity.	Bull trout emerge months before rainbow/steelhead, which leads to larger sizes at any given time. Differences in size leads to differing microhabitat requirements, which partitions habitat. Bull trout are also more piscivorous than rainbow, which further limits agonistic behavior. Adverse interactions between these species would not be expected.
Cutthroat *Adult*					Intra-species spawning interactions are density dependent dealing with female access to preferred habitat and male access to preferred females. When habitat is limiting, intra-species competition can be important.	Spawning timing does not overlap; therefore, adverse spawning interactions would not occur.
Fry/Emergence/ Juvenile Rearing					Juvenile intra-species competition is for specific rearing micro-habitat. If such habitat is limiting, intra-species competition can be important.	Presence of the other species has been demonstrated to not affect habitat preference of either. Bull trout are benthic foragers, while cutthroat are drift foragers. Bull trout more strongly associated with cover. At a given age, bull trout are larger than cutthroat further partitioning habitat. Due to habitat partitioning, adverse interactions would be unlikely.
Bull Trout *Adult*						Intra-species spawning interactions are density dependent dealing with female access to preferred habitat and male access to preferred females. When habitat is limiting, intra-species competition can be important.
Fry/Emergence/ Juvenile Rearing						Juvenile intra-species competition is for specific rearing micro-habitat. If such habitat is limiting, intra species competition can be important.

Susceptibility to BKD and the visible signs of infection vary among salmonid species (Plumb 1999, Beacham and Evelyn 1992). The most common symptoms are dark pigmentation, abnormally protruding eyes, bleeding at the base of the fins, enlargement of the abdominal cavity, and lethargic swimming (Plumb 1999, Egusa 1992, Wiens and Kaattari 1999). As BKD progresses, grayish-white lesions appear in the kidney, possibly extending to the liver and/or spleen. In advanced stages of the disease, the number and size of the lesions increase in all internal organs and the kidney becomes swollen (Plumb 1999, Egusa 1992, Wiens and Kaattari 1999). Death from the disease is usually attributed to the complete degradation of normal kidney and liver tissue by bacterial lesions leading to general organ dysfunction, possibly in conjunction with heart failure (Wiens and Kaattari 1999).

The presence of *R. salmoninarum* in a fish does not necessarily lead to overt infections. There have been numerous reports in which the *R. salmoninarum* organism was present in salmonid populations, but infections did not result. It has been suggested that in these instances, the BKD causing organism remains present at low levels unless the host is stressed (Plumb 1999).

Two of the key stressors that have been noted as leading to clinical BKD infections are the smolts' transition from freshwater to seawater and then the adult transition back to freshwater (Plumb 1999, Mitchum and Sherman 1981, Elliott et al. 1997). Studies have also found that water quality may influence the disease severity and associated mortality. Higher rates of BKD infection have been noted at hatcheries with soft water (low mineral content) than at facilities using water with high total hardness, although this relationship has not been fully confirmed (Egusa 1992, Plumb 1999).

Water temperature also has a profound effect on the occurrence and advancement of BKD (Elliott et al. 1997). Most instances of Chinook and sockeye infection occur during fall and winter with water temperatures ranging from 8 to 18°C (46.4 to 64.4°F). Once the BKD has been expressed, the advancement of symptoms is quite temperature dependent. In cooler waters, the disease progresses more slowly (approximately 8°C (46.4°F)), while in warmer waters (11°C (51.8°F)), infected individuals can expire less than a month after initial exposure (Plumb 1999). However, some research conflicts with this finding, suggesting that temperature is not the only determining factor in the rate of BKD progression (Egusa 1992).

Studies have found that *R. salmoninarum* can be transmitted both horizontally (from fish to fish) and vertically (from parent to progeny). For horizontal transmission of the pathogen, individuals do not necessarily need to come into direct contact, as water infected by a BKD-positive fish can transmit *R. salmoninarum*. The primary sites of entry into a fish are through the gills, gut, eye, and external abrasions or lesions (Balfry et al. 1996).

One of the key factors in the possibility of horizontal transmission is the ability of the bacteria to survive outside of the salmonid host. Austin and Rayment (1985) found that low numbers of *R. salmoninarum* could survive up to 21 days in feces excreted from BKD infected fish. It is possible that benthic feeding salmonid species could become positive for *R. salmoninarum* by inadvertently consuming infected feces while foraging

for food. It has been suggested that in aquaculture environments consumption of infected feces could be a significant route for the horizontal transmission of BKD (Austin and Rayment 1985). In addition, *R. salmoninarum* can survive outside of the host for limited periods in both freshwater and saltwater. In filtered stream water, the bacteria has been found to survive independently for 28 days. However, in normal, unfiltered stream water, *R. salmoninarum* was found to be rather short-lived, with all cells perishing within 4 days (Austin and Rayment 1985). This finding suggests that in normal stream water, the *R. salmoninarum* organism is unable to compete with other members of the aquatic microflora and rather quickly perishes. In saltwater, the bacteria have been found to survive for up to one week outside of a host organism (Balfry et al. 1996). The ability to survive outside of the salmonid host, albeit limited, supports the finding that horizontal transmission of BKD is possible. However, the short lifespan of the organism in water under normal circumstances, supports the general conception that horizontal transmission of the pathogen is more likely in a hatchery setting where salmonids are reared in close proximity, thereby, increasing the odds of finding another host once *R. salmoninarum* organisms are expelled from a BKD infected fish.

Numerous research efforts have focused on the potential transmission of BKD from infected hatchery fish to wild populations and vice versa. Mitchum et al. (1979) found that wild rainbow trout were infected by *R. salmoninarum* following the introduction of BKD positive hatchery fish. This finding is consistent with earlier studies that found that in certain situations, the stocking of infected hatchery fish may have a serious impact on wild salmonid populations. However, the extent of this impact cannot be predicted, as *R. salmoninarum* may be present in salmonids in the carrier state and never lead to clinical display of BKD. Furthermore, Mitchum and Sherman (1981) found that stocked hatchery fish could be infected with BKD by being stocked in a stream in which *R. salmoninarum* is enzootic (constantly present in a given geographic area). The findings of these studies suggest that the transmission of *R. salmoninarum* between hatchery and wild fish is possible, although there is no information regarding the likelihood that such pathogen transfers would occur. It is also not possible to determine whether the horizontal transfer of *R. salmoninarum* would result in full blown BKD, or whether the organism would remain dormant in the carrier state. As mentioned previously, different species are more susceptible to *R. salmoninarum* infections, which could increase the risks associated with pathogen introduction. For example, Eliott et al. (1997) stated that spring Chinook is one of the most vulnerable salmonid species to BKD. Chinook smolts collected at dams on the Snake and Columbia rivers had infection rates ranging from 86 to 100 percent, although visible symptoms of BKD were found in only 1 to 11 percent of the Chinook sampled. At the Lewis River hatcheries, BKD occurs fairly frequently and antibiotics have been only minimally effective in controlling the disease (pers. comm., L. Durham, WDFW, August 2002).

Considering the existing literature summarized above, it is not possible to predict how salmonid populations might react to introduction of BKD infected fish into the upper watershed. Due to this uncertainty, WDFW fish pathologist Larry Durham recommended that the release of fish with active BKD into the upper watershed should be avoided (pers. comm., L. Durham, WDFW, August 2002).

Cold Water Disease

Cold water disease, also referred to as low temperature or peduncle disease, is a bacterial infection caused by the organism *Flexibacter psychrophilus* (Moeller 2001). Flexibacter bacteria are found throughout aquatic environments and cold water disease is not exclusively related to salmonid species. Under normal circumstances, salmonid species are thought to be able to effectively defend themselves against *F. psychrophilus*, and it is only through high stress that they succumb to cold water disease. It is called cold water disease because it usually affects fish in water temperatures ranging between 4 and 10°C (39 and 50°F) and normally cannot survive at temperatures above 25°C (77°F) (Shotts and Starliper 1999). Transmission of the bacteria can be both horizontal and vertical. Vertical transmission has been demonstrated in salmonid species with the bacteria being present on the surface of deposited eggs and in the ovarian fluid (Brown et al. 1997). Horizontal transmission is generally thought to occur only if a fish already has lesions or open wounds, or other forms of stress (Fox 2002). There has been very little research regarding incidences of cold water disease in the wild, as *F. psychrophilus* is difficult to isolate and outbreaks of the disease in natural settings are thought to be rare. However, the bacteria are presumed to be present in the upper Lewis River watershed (pers. comm., L. Durham, WDFW, August 2002).

Initial symptoms of the disease include darkening of the skin, lesions on the head and peduncle regions, and hemorrhaging at the base of fins. As the disease progresses and spreads throughout the body, hemorrhaging from internal organs occurs, and fins are sometimes destroyed, especially the caudal fin (Shotts and Starliper 1999, Fox 2002, Egusa 1992). The cause of death from the disease is from general system failure. Mortality tends to be much higher in alevins than older juveniles. Of all the salmonid species, juvenile coho and steelhead are the most susceptible to cold water disease, although the reasons behind this trend are not well understood (Egusa 1992, Shotts and Starliper 1999).

F. psychrophilus has been observed at the Lewis River hatcheries and there is no cure for the disease, although treatments in a hatchery setting have been rather effective at reducing mortality once a disease outbreak has been observed (pers. comm., L. Durham, WDFW, August 2002). Transmission from hatchery fish to wild fish would not likely be a major concern, as horizontal transfer requires open wounds or lesions and wild fish are not under the same type of density-related stress that occurs in hatchery settings. Furthermore, the *F. pshychrophilus* is thought to be endemic to the Lewis River watershed, thus fish inhabiting the upper watershed have most likely been exposed to the pathogen. Therefore, cold water disease would not be a major concern for reintroduction of anadromous salmonid species into the upper watershed.

Erythrocytic Inclusion Body Syndrome

Erythrocytic Inclusion Body Syndrome (EIBS) is a viral infection that has only been discovered within the last 2 decades. The actual virus that leads to the disease has not yet been isolated. Evidence of EIBS has been found in both aquacultural and natural settings, although the source and potential distribution in streams is not currently understood. The pathogen has been found to be successfully transmitted horizontally via

water (Piacentini et al. 1989). Vertical transmission of the disease-causing agent has not been demonstrated, although there has not been adequate research on the subject to dismiss the potential for such transmission pathways (Takahashi et al. 1992). All salmonid species appear to be vulnerable to EIBS, however, the infection tends to be more prominent in coho. Past studies have demonstrated EIBS presence in coho salmon raised at Lewis River hatcheries (Michak et al. 1992). However, EIBS inclusions have not been observed in Lewis River hatchery fish over the past 7 years (pers. comm., L. Durham, WDFW, August 2002).

In salmonids, infected fish may appear lethargic and swim near the surface. In addition, the gills may become pale and yellow in color (Takahashi et al. 1992). Internally, the disease is consistently characterized by the presence of inclusion bodies in the red blood cells (erythrocytes). Inclusion bodies are structures that contain extremely high concentrations of aggregated protein and can disrupt the normal function of the red blood cells (Piacentini et al. 1989, Leek 1987). The inclusion bodies generally result in hemolytic anemia, which is anemia caused by excessive destruction of red blood cells (Maita et al. 1996, Foott et al. 1992). In addition, infected fish have high incidences of external fungi, bacterial kidney disease, and cold water disease, which are often the maladies that actually lead to mortality associated with EIBS (Piacentini 1989, Michak et al. 1992). Mortality resulting specifically from EIBS is difficult to ascertain due to the other ailments that result from clinical expression of the disease, yet associated mortality is thought to often exceed 25 percent (Piacentini et al. 1989). Water temperatures have been found to have a substantial impact on the rate and persistence of EIBS infections. Lower temperatures (i.e. 3-6°C (37 to 43°F)) are associated with delayed clinical displays of infection but longer lasting inclusions, whereas higher temperatures (i.e. 15°C (59°F)) resulted in quicker displays of inclusions, but shorter periods of infection (Piacentini et al. 1989). In laboratory studies, fish that successfully recovered from EIBS showed little signs of the disease when re-exposed to the pathogen.

The ability of the virus to be transmitted horizontally makes it a potential concern in regards to reintroduction of salmonids into the upper watershed. It is not known whether the virus that leads to EIBS is present in the upper watershed. However, the risks associated with EIBS are likely low, even if some of the salmonids that were released into the upper watershed were positive for the pathogen (Durham 2003).

Infectious Hematopoietic Necrosis Virus

IHNV is a naturally occurring virus that causes a highly contagious disease in certain salmonid species, but no other hosts have demonstrated susceptibility. The disease caused by IHNV has been called the most serious viral disease for Pacific salmonid species (Plumb 1999, LaPatra 1998, Anderson et al. 2000). The virus is thought to be endemic to the Pacific Northwest and can be present in a carrier state that does not necessarily lead to clinical displays of disease (Meyers 1998, Anderson et al. 2000). Rainbow and steelhead trout, and Chinook, sockeye, and kokanee salmon tend to be the most susceptible to IHNV infection, while coho and pink salmon have demonstrated very low susceptibility to the virus (Plumb 1999). No published information was found regarding the susceptibility of bull trout to IHNV infection. Unpublished data from the Oregon Department of Fish and Wildlife suggests that bull trout are resistant to the virus.

Laboratory challenges conducted in 1999 and 2002 using high levels of IHNV failed to produce significant mortality in juvenile bull trout. The bull trout used in the study were different stocks (Oregon and Idaho), and the virus was a different strain than is typically found in the Lewis River drainage, but the results are encouraging (pers. comm., L. Durham, WDFW, February 2003).

High rearing densities, such as those demonstrated in hatchery settings, are closely correlated with outbreaks of IHNV. This has been attributed to the ability of the pathogen to be transmitted horizontally between juveniles (LaPatra 1998, Bootland and Leong 1999, Beacham and Evelyn 1992). IHNV is one of the more problematic pathogens at Lewis River hatcheries, as there are no effective treatments (pers. comm., L. Durham, WDFW, August 2002). IHNV may also be present throughout the Lewis River watershed, although reports of IHNV outbreaks in natural systems are rare (LaPatra 1998). The epizootiology of IHNV is not fully understood and the source of the virus in aquatic systems has not been isolated. However, studies have found that IHNV can live up to 30 days in freshwater outside of a salmonid host, which aids in horizontal transmission of the virus. In addition, since mortality in very young salmon in hatchery environments is rather high, vertical transmission is also thought to play a role in the perpetuation of IHNV, although such transmission pathways have not been fully proven (Bootland and Leong 1999).

Water temperature is the most important environmental factor affecting the severity of IHNV infections. Outbreaks of the disease most commonly occur during the spring and fall, when water temperatures are at about 10 to 12°C (50 to 54°F). Normally, infections of the pathogen do not occur at water temperatures above 15°C (59°F), although some studies have noted infections at temperatures up to 18°C (64°F). Once clinical expression of the disease has occurred, the time period until death is generally inversely related to temperature in that the higher the water temperature, the shorter the time until the disease induces mortality (Bootland and Leong 1999).

When symptoms of disease are present, IHNV leads to death (necrosis) of body tissues responsible for blood cell creation (hematopoietic tissues), primarily in the kidney and spleen (Bootland and Leong 1999). Symptoms of the disease include darkening of the skin, swollen abdomens, pale gills, hemorrhaging at the gills and fins, and an opaque fecal cast oftentimes trails from the vent of the fish (Plumb 1999). Internally, the organs appear anemic and the digestive tract is absent of food and instead is filled with fluid (Plumb 1999). Behaviorally, fish demonstrating clinical expression of IHNV tend to be lethargic and move to stream margins or bottoms. In advanced disease stages, fish are often frenzied, swimming in circles and quickly flashing (Plumb 1999).

Salmonids tend to become more resistant to IHNV with increased age and weight. Alevins and fry up to 2 months of age are highly susceptible and can experience up to 100 percent mortality in hatchery environments (LaPatra 1998, Plumb 1999). Fish up to 6 months of age typically experience less than 50 percent mortality. The virus has been noted to kill juvenile (up to 2 years of age) sockeye, kokanee, and rainbow trout, but with fish of this advanced age and weight, mortality is generally low (Bootland and Leong 1999).

After returning to their natal streams to spawn, IHNV infections have also been documented in kokanee, Chinook, chum and coho salmon in addition to rainbow and steelhead trout (Bootland and Leong 1999). It is not known whether these fish are actually infected as adults or whether the disease is an expression of exposure to IHNV as juveniles, perhaps brought on by the stress of adult biological and morphological changes. Female adults are more susceptible to IHNV infection than are males, which may be attributable to the differences in hormone levels or possibly due to the difficulty of isolating the virus in milt (Bootland and Leong 1999, Meyers 1998). It is also possible that infected adults could horizontally transmit IHNV to nearby juveniles, which is a consideration for the reintroduction of anadromous adult salmonid species to the upper Lewis River watershed (Hastein and Lindstad 1991, Bootland and Leong 1999). However, the densities of salmonids in the riverine habitat of the upper Lewis River would likely be insufficient to provide for horizontal transmission of the virus from infected adults to juveniles rearing in the streams (Durham 2003).

It is important to note that there are numerous strains of IHNV. Some salmonid species are more susceptible to certain strains and some strains may cause more virulent reactions in certain salmonid species than others (LaPatra et al. 1990, Bootland and Leong 1999). In general, the strains of IHNV found in the Columbia River are rather virulent for steelhead trout and Chinook salmon, but reactions are less severe in kokanee and coho (Bootland and Leong 1999). However, these are only general trends and the severity of reactions of Lewis River salmonid populations to different IHNV strains may be different.

Therefore, due to the ability of the virus to be transmitted horizontally and the inability to locate the source of naturally occurring IHNV in aquatic systems, it would be advantageous to screen adults for IHNV to ensure protection of wild salmonids in the upper watershed and downstream hatchery production (Anderson et al. 2000, Durham 2003). However, it may not be practical or prudent to screen every adult prior to release into the upper Lewis River basin (Durham 2003).

Ceratomyxosis

Ceratomyxosis in salmonid species is caused by the myxosporean parasite *Ceratomyxa shasta*. Infections of the disease are known to occur in rainbow, steelhead, brown, and brook trout and pink, Atlantic, chum, coho, sockeye, kokanee, and Chinook salmon. The range of the infective stage of the parasite is restricted to the Pacific Northwest. The Columbia River and many of its tributaries are known locations of *C. shasta*. In the Lewis River watershed, only the East Fork Lewis River has been documented as an area inhabited by *C. shasta*, although it may be present at low levels in other areas of the drainage (Bartholomew 1998; pers. comm., L. Durham, WDFW, August 2002).

Distribution of the disease can be isolated to specific areas, such as the East Fork Lewis River, due to its rather unique life history. The myxosporean parasite on its own is not infectious for salmonid species. It is only after the parasite has been processed by a specific intermediate freshwater polychaete (*Manayunkia speciosa)* that it takes on the infective actinosporea form (Bartholomew 2001). Therefore, distribution of ceratomyxosis is limited to the locales in which the polychaete intermediate host is

present. In the Lewis River watershed, the intermediate host is only known to exist in the East Fork, although no studies were found that confirmed the absence or presence of *C. shasta* in the North Fork Lewis River.

Clinical symptoms of the disease vary somewhat among salmonid species. Yet, in the majority of the cases, at least some of the following symptoms would be displayed: anorexia, lethargy, darkening of the skin, swollen abdomen, a swollen and hemorrhagic vent, emaciation, and protruding eyes (Bartholomew 2001). Internally, the disease tends to primarily attack the digestive tract, especially the posterior intestine and pyloric caeca. Ultimately, this destruction and subsequent failure of the digestive system and other internal organs, is the ultimate cause of *C. shasta* related mortality (Lom and Dykova 1995). Rates of mortality after exposure to the parasite are rather variable depending upon the species and their susceptibility to the pathogen, although mortality of 75 percent or higher is not uncommon (Bartholomew 1998). Yet fish from streams in which *C. shasta* and the intermediate polychaete host are endemic are generally less susceptible to ceratomyxosis infections.

Most importantly from a salmonid reintroduction perspective is the fact that the disease cannot be transmitted between fish. As mentioned previously, *C. shasta* can only reach its infective state through processing by the polychaete host. The waste products, body fluids, and other components of diseased fish are not infective to other fish (Bartholomew et al. 1989). Therefore, the introduction of fish into the upper North Fork Lewis River watershed would have no impact on the presence or absence of ceratomyxosis outbreaks. As of now, the intermediate host has not been found in the North Fork Lewis River, thereby precluding ceratomyxosis outbreaks (Hoffmaster et al. 1988, Bartholomew et al. 1989). However, any transplants of bedload or organic materials from the East Fork to the North Fork should be avoided to ensure that the intermediate polychaete host is not inadvertently introduced into the North Fork.

Whirling Disease

Whirling disease is caused by the metazoan parasite *Myxobolus cerebralis*. The parasite penetrates the head and spinal cartilage of susceptible trout. It then multiplies rapidly, putting pressure on the organs that govern equilibrium. This pressure causes the characteristic, erratic swimming (whirling) that is associated with this disease. Continued growth of the parasite in the cartilage can lead to head, spine, and tail deformities, and neural damage. *M. cerebralis* can cause changes in feeding behavior that lead to malnutrition and death (Markiw 1992).

As with *C. shasta,* development of whirling disease requires an intermediate host, which is the aquatic worm, *Tubifex*. The worm creates the infective stage of the disease, therefore, distribution of whirling disease depends upon the presence of *Tubifex*. Unlike *C. shasta*, there is evidence that direct horizontal transmission of whirling disease may be possible, although vertical transmission does not occur (Markiw 1992).

All of the Lewis River salmonid species, including bull trout, have been documented as being susceptible to whirling disease (Egusa 1992, Lorz 2002). However, the presence of *M. cerebralis* in the Lewis River drainage is uncertain, although its existence in the

watershed is unlikely (pers. comm., L. Durham, WDFW, August 2002). Furthermore, studies have found that the parasite is rarely found outside of hatcheries in areas in which it is not endemic (Lom and Dykova 1995).

Therefore, using existing Lewis River stocks to reintroduce anadromous salmonids to the upper watershed should not result in potential whirling disease infections, as the intermediate host does not appear to be present. Any transfers of fish or materials from other watersheds should be done in a manner which would ensure that the *Tubifex* worm was not introduced into the Lewis River (Hulbert 1996).

Ichthyophthiriosis ("ich")

I. multifiliis is the most pathogenic of the protozoan parasites of fishes and results in a condition referred to as ichthyophthiriosis or "ich". It is found world-wide and is responsible for substantial economic loss related to aquaculture. The pathogen is thought to be ubiquitous to the Lewis River drainage and outbreaks have been documented at project hatcheries (pers. comm., L. Durham, WDFW, August 2002). All fish appear to be susceptible to the parasite, as there are no records of any species with natural resistance (Dickerson and Dawe 1995).

Growing aquaculture production of salmonids has been attributed to increased infections in both wild and hatchery raised fish (Dickerson and Dawe 1995). The parasite is most easily established in areas of high fish densities, as are generally found in hatchery settings. Ichthyophthiriosis is most likely to occur when fishes are stressed and water temperatures are quite warm (25-28°C (77-82°F)). However, outbreaks at cooler water temperatures are common, although the parasites are less active and their life cycle takes longer to complete (Dickerson and Dawe 1995).

Ichthyophthiriosis is also known as white spot disease, due to the raised white nodules (trophonts) that attach themselves to the host fish. The parasite tends to congregate on the dorsal surface of fish, particularly the head and fins. In the early stages of parasite infection, fish tend to flash or rub their bodies against objects in reaction to the skin irritation caused by the attached trophonts. Infected fish also tend to swim more quickly and frequently leap from the water. The skin irritation from parasite attachment causes substantial increases in surface mucus production and can lead to skin necrosis and sloughing (Durborow et al. 1998). Internally, fish infected by the parasite have enlarged spleens and kidneys and pale livers. These internal changes may not be a direct result of the parasite, but may instead be caused by opportunistic bacterial or fungal infections. Trophont attachment to fish gills is an extremely important component in the virulence of infection. In severe infections, gill tissues around the areas of trophont attachment can become necrotic, which can lead to a loss of gill function and death to the host (Durborow et al. 1998, Traxler et al. 1998). Presence of trophonts on the gills can be rather common, as this area is constantly exposed to surrounding water and the gills lack the mucus covering, which helps protect the skin from parasitic infections (Dickerson and Dawe 1995).

Previously infected fish develop a protective immunity against future afflictions and *I. multifiliis* can only live for a day or 2 outside of a fish host (Durborow et al. 1998).

Therefore, it must maintain at least low levels of infection in fish populations to persist in a given locale. Again, this is another reason why hatchery situations are especially suitable for Ichthyophthiriosis, as there is a constant supply of potential hosts for the parasite. Furthermore, in natural settings, there does not tend to be enough host species to allow for severe infections of *I. multifiliis*, which helps to maintain a balance between host and parasite. It is generally only in the artificial densities of hatcheries that adequate numbers of host fish are present to lead to lethal levels of infection (Traxler et al. 1998, Dickerson and Dawe 1995).

Since *I. multifiliis* is already thought to be present in the upper watershed, reintroduction of anadromous salmonid species would not likely have a substantial effect on potential Ichthyophthiriosis infections.

Fungal Disease

Of the many fungal diseases, those of the genus Saprolegnia are the most significant to the Lewis River and particularly to hatchery operations (pers. comm., L. Durham, WDFW, August 2002). These fungi are considered ubiquitous to the entire Lewis River drainage but usually only cause clinical displays in fish that are physically injured, stressed, or infected with other diseases (Bruno and Wood 1999). Saprolegnia may occur anywhere on the body of fish, but normally appears as a circular or crescent-shaped, white, cotton-like mycelium, particularly around the head and the caudal and anal fins (Bruno and Wood 1999). All salmonid species appear to be susceptible to infection by this disease.

Saprolegnia fungi are transmitted horizontally and no intermediate host is required. In hatcheries, factors such as overcrowding and handling stress can lead to clinical displays of Saprolegnia. In wild fish, redd digging and spawning have been associated with increased displays of the disease (Bruno and Wood 1999).

In general, these types of fungal diseases are not a major concern for reintroduction activities, as the fungi are thought to exist throughout the Lewis River basin and only occasionally result in clinical displays of infection in wild fish (Bruno and Wood 1999).

Summary of Data Gaps and Uncertainties – To aid in evaluating the potential accuracy and applicability of the analysis presented above in Section 4.16.5.2, the following bulleted list describes the major data gaps and uncertainties.

- In general, there was is very little documentation of the prevalence of fish pathogens in the wild. Therefore, it is difficult to assess disease potential for wild resident and anadromous fish in the upper Lewis River watershed. However, disease outbreaks are generally considered to be rare in natural stream settings. Furthermore, the presence of a pathogen does not necessarily mean that there is disease in that area. Pathogens express themselves as disease when environmental conditions allow for transmission and reproduction of the pathogen and in the wild, fish densities are generally too low to support substantial disease outbreaks (Durham 2003). Thus, there is a moderate level of uncertainty pertaining to the discussions regarding disease potential in the upper Lewis River basin.

- Studies found that it is common to find *R. salmoninarum* (the bacteria that causes BKD) in wild salmonids, although clinical infections in the wild may be rare. It is not known how prevalent *R. salmoninarum* is in the upper Lewis River watershed. Resident rainbow, cutthroat, kokanee, etc. may currently carry the *R. salmoninarum* bacteria, but may never develop clinical signs of the disease. It is highly uncertain whether handling stress at juvenile or adult collection or fish passage facilities (which may be constructed as part of reintroduction efforts) would result in clinical expression of BKD is some fish, which could lead to horizontal transfer to other fish held at collection facilities. Furthermore, since the current distribution of the bacteria in the watershed is not known, passage of fish from reservoir to reservoir or the upper mainstem Lewis River may introduce *R. salmoninarum* to areas where it does not currently exist. However, the potential for BKD infections in the wild is highly uncertain. Such potential BKD related concerns may warrant further consideration prior to reintroduction and efforts should be made to protect against the transport of BKD infected fish to the upper watershed.

- The *F. psychrophilus* bacteria that causes coldwater disease is thought to be present in the Lewis River, but this assumption has not been studied. Although expression of this disease in the wild is thought to be rare, it is not known if handling stress at juvenile or adult collection or fish passage facilities would be sufficient to lead to clinical manifestation of cold water disease. However, in general there is a fairly high level of certainty that cold water disease would not be a major concern in reintroduction efforts, as fish in natural stream settings are generally not subject to the high levels of stress that are associated with clinical displays of cold water disease.

- The understanding of EIBS is still limited and it is not known whether the virus that causes EIBS is present in the upper watershed. There have been documented cases of EIBS in coho at the Lewis River hatcheries, although the pathogen has not been detected in the past 7 years. The risks associated with EIBS in relation to reintroduction efforts are likely low.

- It is uncertain whether IHNV is present in the upper watershed, as outbreaks of the virus are rare in natural stream settings. Expression of the disease is brought on by stress, and it is uncertain how handling stress at collection or fish passage facilities may affect the expression of IHNV symptoms. Furthermore, observations of adult salmonids showing symptoms of IHNV upon their return to freshwater have been documented, as has the passing of IHNV from infected adults to juveniles. However, the likelihood of such transmission pathways in the wild is unlikely, as fish densities are quite low. Screening for IHNV at passage facilities would be advantageous, but testing of each adult fish prior to reintroduction may not be feasible.

- Studies pertaining to *C. shasta* distribution in the Columbia River basin have not identified the mainstem Lewis River as being an area where the pathogen is present. However, there are no known studies that have explicitly researched the presence or absence of *C. shasta* in the North Fork Lewis River. Yet, since the pathogen cannot be transmitted without the presence of the intermediate host, reintroduction efforts should not pose a risk for increasing *C. shasta* distribution.

- It is uncertain whether the *Tubifex* worm required for whirling disease transmission is present in the Lewis River watershed; however, due to the need for the intermediate host, reintroduction efforts would not be expected to lead to a spread of the pathogen.

- The distribution of *I. multifiliis* and fungal diseases in the Lewis River basin is uncertain, although these organisms are assumed to be present throughout the watershed. Expression of symptoms is brought about by stress, thus it is also uncertain how handling at juvenile or adult collection or fish passage facilities might affect expression of these diseases. However, since *I. multifiliis* and disease producing fungi are already thought to be present throughout the Lewis River watershed, reintroduction efforts should have no impact on the distribution of this disease.

Summary of Disease Findings

BKD and IHNV are the 2 salmonid diseases that are of the greatest concern for reintroduction activities, as both diseases are already prevalent in the Lewis River hatcheries and treatments are of limited success. Therefore, health screenings of adults, juveniles, and/or eggs prior to transplants into the upper watershed may be beneficial, when feasible.

Ceratomyxosis may not be a major concern for reintroduction efforts, as the intermediate host required for infection of salmonids is not thought to inhabit the upper Lewis River watershed. However, the presence or absence of C. Shasta has not been confirmed and may warrant further consideration. Also, whirling disease has not been isolated in the Lewis River watershed, thus there is little concern regarding this disease from a reintroduction standpoint.

The organisms that cause bacterial coldwater disease, ichthyophthiriosis, and various fungal infections are thought to be present throughout the Lewis River watershed. Therefore, reintroduction efforts would not result in the introduction of these pathogens to previously uninhabited areas. The likelihood of infection by these pathogens is related to salmonid densities and stress, thus disease outbreaks in the wild are unlikely. Yet, it is uncertain how handling at fish collection and passage facilities may impact the clinical expression of these diseases.

4.16.5.3 Predation Concerns Regarding Reintroduction

The potential effect of predation on the success of establishing self sustaining anadromous salmonid stocks in the upper Lewis River basin is a key concern. Since the time anadromous salmonids inhabited the upper Lewis River, the watershed has undergone substantial changes, most notably hydroelectric development, which have altered the dynamics of predator-prey relationships. Current habitat conditions and species composition could contribute to substantial predation on anadromous salmonid juveniles, which could compromise the success of reintroduction efforts. The following discussion examines the 2 primary habitat areas where predation on reintroduced salmonids would occur (project reservoirs and instream) and briefly discusses management options that may reduce predation risks.

Reservoir Predation

The species of primary concern from a reservoir predation standpoint is northern pikeminnow (*Ptychocheilus oregonensis*), which is a well documented predator of salmonid species. In addition, tiger muskellunge (*Esox masquinongy X Esox lucius*) were introduced to Merwin Reservoir in 1995 and the species' potential for predation on salmonids is of interest for reintroduction activities.

Northern Pikeminnow - In the Lewis River project reservoirs, northern pikeminnow are considered to be the primary piscivorous predator of salmonids. These fish are opportunistic feeders that generally forage near shore both in lakes or reservoirs and near the substrate of slow moving streams (less than 2.3 feet per second) (Scott and Crossman 1973, Faler 1988 as cited in Mesa and Olson 1992). They tend to prey on the most abundant resident insect and fish species, focusing on salmonids during seasonally high abundance (Poe et al. 1991). These fish are long-lived (15 to 20 years), which makes them a persistent threat to salmonid prey.

Although northern pikeminnow are considered indigenous to the Lewis River, their populations were likely rather small prior to dam construction due to their preference for stillwater habitat. Following construction of the hydroelectric projects and the creation of substantial reservoir habitat, northern pikeminnow populations appeared to have increased substantially (PacifiCorp and Cowlitz PUD 2002a). Large numbers of the fish have been observed in Lake Merwin, with a smaller population occurring in Yale and possibly Swift reservoirs. There has not been extensive research regarding the abundance of northern pikeminnow in the Lewis River watershed, although in 1961 the Lake Merwin population was estimated to be about 350,000 fish (WDF 1970). Furthermore, creel surveys conducted in 1995 at Yale Lake estimated that 19,337 angler hours were expended to catch 3,068 kokanee, 511, coho, 20 rainbow trout, and 20,764 northern pikeminnow (Hillson and Tipping 1999). These and other sources suggest that the species is abundant throughout Merwin and Yale reservoirs.

From a predation standpoint, the presence and abundance of northern pikeminnow in the Lewis River reservoirs is an important consideration. Preliminary studies conducted during the 1960s (Hamilton et al. 1970), found that approximately 15 percent of the northern pikeminnow in Lake Merwin were consuming coho. Furthermore, the ratio of coho to northern pikeminnow at the time was estimated at about 7 to 1. Therefore, if each northern pikeminnow would have eaten just 7 coho juveniles, then the population would have been eliminated (Hamilton et al. 1970). Such evidence suggests that northern pikeminnow predation in Lake Merwin may be a major concern. In addition, there have been numbers studies conducted in the mainstem Columbia River, which also suggest the importance of northern pikeminnow predation on salmonids. Since the late 1980s, studies in the John Day Reservoir of the Columbia River have assessed the nature of northern pikeminnow predation on outmigrating salmonid smolts. The results of these studies are not directly applicable to the Lewis River reservoirs, but they do reveal general characteristics and possible impacts of northern pikeminnow predation.

Fish, particularly salmonids, become the most important food source as northern pikeminnow approach and exceed fork lengths of approximately 250 mm (10 in) (Friesen

and Ward 1999). Studies have found that after reaching 250 mm in length, an individual northern pikeminnow in the Columbia River will consume between 100 and 400 juvenile salmonids throughout the remainder of its life, based upon the probability of survival to a given size and size-specific consumptions rates in John Day Reservoir (Beamesderfer 1996). Furthermore, as northern pikeminnow continue to grow beyond 250 mm, consumption rates of juvenile salmonids tend to increase exponentially as size of the predator increases (Beamesderfer et al. 1996). Salmonid consumption by individual pikeminnow has been found to range from approximately 0.5 to 2.0 fish/day, when salmonid juveniles are available (Vigg et al. 1991). Such predation rates, when considered for the entire pikeminnow population, can have a drastic impact on the number of outmigrating salmon smolts. It has been estimated that throughout the Columbia River system around 16.4 million salmon juveniles are consumed annually by pikeminnow, which accounts for approximately 8 percent of the 200 million juveniles produced in the system (Beamesderfer et al. 1996).

One reason for this dramatic impact by a single predatory species is due to the human induced changes to the system, which have benefited the northern pikeminnow and potentially increased their consumption rates and predatory success. For example, reservoirs increase the availability of slow moving water, which is preferred by the pikeminnow. Impoundments also increase water temperatures, which escalates digestion and consumption rates by pikeminnow. For example, one study found that northern pikeminnow salmonid consumption rates were 4 times higher in 19°C (66°F) (2.03 salmonids/pikeminnow/day) than in 11.5°C (52.7°F) water (0.5 salmonids/pikeminnow/day) with similar juvenile salmonid densities (Vigg et al. 1991). Furthermore, lentic environments and increased water temperatures may delay juvenile outmigration causing stress and increased disease manifestation in outmigrating juvenile salmon. Studies have found that increased stress in salmonids may change migratory behavior in a manner that makes them less adroit at avoiding predators such as northern pikeminnow (Beamesderfer et al. 1996, Ward et al. 1995).

Furthermore, sediment-related turbidity in reservoirs tends to be less than in more swiftly moving streams. Northern pikeminnow have been found to use vision to detect prey, therefore, turbidity-induced visual obstruction may be less in reservoirs than would be expected in free-flowing streams. Gregory and Levings (1998) conducted studies that supported the general belief that decreased sediment-related turbidity in reservoirs may increase the success of predatory fish species feeding on salmonid juveniles. They found that fish foraging in clear-water sites consumed approximately 2 prey/predator, while similar sized fish foraging in more turbid waters consumed only about 1 prey/predator. The lack of suspended solids is one of the factors attributed to the seemingly lower predation rates in the Columbia River below Bonneville Dam in comparison to the predation rates demonstrated in reservoirs further upstream (Friesen and Ward 1999). However, it is important to note that increased sediment-related turbidity is only beneficial at relatively low levels, as high concentrations may cause stress in salmonids leading to increased susceptibility to predation. Furthermore, in reservoir environments, plankton and algae growth in addition to other suspended organic debris could reduce visibility and decrease northern pikeminnow predation success, although no studies were found relating to this subject.

Numerous studies have also found that northern pikeminnow predation on salmonids is higher near areas of salmonid concentration such as above and below dams, fish passage facilities, or hatchery release points (Ward et al. 1995, Shively et al. 1996, Petersen 1994). For example, studies at McNary Dam on the Columbia River found pikeminnow predation rates were 5 times higher in the tailrace than in the rest of the reservoir (Vigg et al. 1991). This increase in predation rates below dams is thought to be due to the stress on the fish caused by passage over spillways or through the project turbines. Furthermore, stress from crowding, handling, measurement, marking, and/or transportation have been documented as having a substantial impact on the predator avoidance capabilities of salmonids. Mesa (1994) found that Chinook salmon that had recently been released or passed through a dam display lethargic and disoriented behavior for up to an hour after the initial stress inducing event occurred. These findings were generally consistent with other studies using coho that found that the fish generally recover from handling and dam passage stress in approximately 90 minutes (Mesa 1994). However, predation may be reduced by the tendency of pikeminnow to select moribund or dead fish below dams. Therefore, fish that successfully navigated the dam may be spared, as pikeminnow may focus on fish that had already or would suffer dam related mortality (Gadomski and Hall-Griswold 1992). Yet, it is important to note that information regarding this phenomenon is quite limited.

Another important concern from the standpoint of hatchery releases and potential trap-and-haul fish passage strategies (one of the options for fish passage in the upper Lewis River) is the increased predation rates by northern pikeminnow near concentrated smolt release points. Although predation by pikeminnow in free-flowing rivers is generally less than in reservoirs and other slow moving waters, high rates of predation are oftentimes experienced in relatively swift moving stream reaches nearby hatchery or trap-and-haul release areas (Buchanan et al. 1981). Studies have found that pikeminnow tend to concentrate and feed at release points and may even increase their total consumption rates when salmonid smolts are abundant (Collis et al. 1995). One of the most convincing studies regarding this subject was conducted by Shively et al. (1996) on the Clearwater River in Idaho. This study sampled the diets of northern pikeminnow before and after the release of 1.1 million spring Chinook salmon smolts. Prior to the Chinook release, the diet of the northern pikeminnow contained no salmonid remains. However, within 24 hours of the salmonid release, pikeminnow sampled approximately 60 km (37 mi) downstream had already made Chinook fry 54 percent of their diet by weight. At 5 and 7 days after release, Chinook fry constituted 78 and 86 percent of the diet, respectively (Shively et al. 1996). This study displays how quickly northern pikeminnow can respond to increases in salmonid densities and also demonstrates the species preference for salmonid smolts when they are available. Findings such as these suggest the need to consider alternate strategies for hatchery and/or trap-and-haul fish releases in order to minimize the potential of northern pikeminnow predation.

However, it is also possible that increased localized predation near hatchery release sites could reduce predation potential on wild smolts by decreasing the numbers of northern pikeminnow elsewhere in the migratory path of salmonids. Conversely, concentration of pikeminnow near release sites may also increase predation on wild smolts as they pass through the areas of high predator concentrations. How hatchery releases would actually

affect wild smolt survival is uncertain and there is no evidence that increased predator abundance in localized areas either increases or decreases predation on wild salmonids (Flagg et al. 2000). However, studies have found that northern pikeminnow tend to prefer the smaller salmonid smolts, which may equate to more of the smaller wild smolts being preyed upon in comparison to the generally larger hatchery smolts (Poe et al. 1991, Ward et al. 1995). The reasoning behind the pikeminnow preference for smaller smolts has been attributed to size-related differences in swimming abilities that may make smaller juvenile salmonids easier to capture (Collis et al. 1995). One potential management strategy to deal with this higher potential of wild smolt consumption would be to raise hatchery smolts that were more similar in size to wild juveniles to ensure that wild smolts were not disproportionately preyed upon.

Tiger Muskellunge (tiger muskie) – In 1995, Lake Merwin became only the third lake in the State of Washington to be planted with tiger muskies. The muskie program was initiated for 2 primary reasons: to provide a trophy fishery in Lake Merwin and to aid in the reduction of predator-size northern pikeminnow in the reservoir (Tipping 1996).

Tiger muskies are a hybrid species of northern pike (*Esox lucius*) and muskellunge (*Esox masquinongy*). The species has been stocked throughout the United States as a trophy fish and to serve as a predator in niches where other predator fish are inadequate (Hesser 1978, Wahl and Stein 1988). Tiger muskellunge is thought to be a particularly well suited species for predator control programs as it is rather fast growing, attains a large adult size (over 100 cm (39 in) and 10 kg (22 lb)), tolerates warm water temperatures, is exclusively carnivorous and predominantly piscivorous, and sterile (Hesser 1978, Weithman and Anderson 1977, Scott 1964, Newman and Storck 1986, Buss et al. 1978). The following paragraphs provide a brief description of each of these characteristics.

Like many hybrid species, tiger muskellunge are generally faster growing than either of its parent species, especially during its first year (Eddy and Underhill 1974). At age one, tiger muskellunge can reach lengths of 51 cm (20 in), and up to 76 cm (30 in) by their second year (WDFW 2000b). Studies in Mayfield Reservoir on the Cowlitz River found that tiger muskellunge were reaching lengths of 53 cm (21 in) or more by the October of the year after planting (approximately 2 to 3 years total age) (Tipping 1992).

Growth of tiger muskellunge has been found to be closely related to water temperatures, with the highest growth rates occurring during summer and early autumn (July – September) and lower rates during the winter months (November – December) (Chipps et al. 2000). This seasonal variation in growth rates is attributed to increased feeding activity and metabolism at higher water temperatures. For example, one study of muskellunge found that food consumption increased with temperature from less than 0.04 grams of food per gram mean weight of muskellunge per day ($g \cdot g^{-1} \cdot d^{-1}$) at 5°C (41°F) up to a peak of 0.14 ($g \cdot g^{-1} \cdot d^{-1}$) at 25°C (77°F). This increase in food consumption was found to result in increases in growth rate. At 5°C (41°F) the growth of muskellunge was less than 0.01 change in grams wet weight of muskellunge per gram mean weight of muskellunge per day ($\Delta g \cdot g^{-1} \cdot d^{-1}$) and peaked at 25°C (77°F) where growth rate was found to be 0.05 ($\Delta g \cdot g^{-1} \cdot d^{-1}$) (Clapp and Wahl 1996).

Optimal growing temperatures for tiger muskellunge are fairly warm at about 20°C (68°F), although the fish is generally considered to be a cool water species (Meade and Lemm 1986). This optimal temperature compares favorably with water temperatures in Lake Merwin, which have been found to have medians ranging from 17.8°C (64.0°F) to 21.2°C (70.2°F) during the summer and fall (July – September) prime growing months for tiger muskellunge (PacifiCorp and Cowlitz PUD 2002c).

Tiger muskellunge and the other pike species are exclusively carnivorous and are generally considered to have voracious appetites. They are visual feeders and have been known to strike at a variety of animals such as waterfowl, small mammals, and amphibians, although the species is primarily piscivorous (Eddy and Underhill 1974, Miller and Menzel 1986). Tiger muskellunge tend to be bottom oriented feeders that hide and strike-out at their prey rather engaging in active foraging. Yet, muskies will often pursue their prey after a failed strike (Engstrom et al. 1986).

Originally, it was thought that tiger muskellunge would feed upon any prey available. However, numerous laboratory and field studies have found that given a choice, these fish and their parent species display consistent prey selectivity (Weithman and Anderson 1977). Unfortunately, the vast majority of studies conducted regarding tiger muskellunge and its parents have not included analysis of the species preference for salmonids.

In general, tiger muskellunge tend to select soft-rayed fish over spiny-rayed individuals. For example, one study found that muskies tend to select northern pikeminnow and suckers over bass or sunfish (Engstrom et al. 1986). A similar study determined that tiger muskellunge and northern pike selected white suckers and golden shiners (both soft-rayed fish) over yellow perch (a spiny-rayed fish) (Wahl and Stein 1988). This preference for soft-rayed fish over spiny-rayed individuals is generally attributed to the fact that digestion or removal of prey spines increases the metabolic costs of consumption, while such costs are not incurred when feeding upon soft-rayed prey (Wahl and Stein 1988).

Furthermore, the size of prey consumed by tiger muskellunge and its parent species is very closely related to the size of the individual (Wahl and Stein 1993). Muskies tend to consume prey that are from about about 10 to 50 percent of their body length (Wahl and Stein 1988, Bozek et al. 1999). Tiger muskellunge consume larger prey as they grow and large and small muskellunge tend to eat approximately the same overall number of fish, only the size of the prey differs. Muskies also tend to consume only one fish at a time, targeting the largest fish the individual is able to successfully take (Bozek et al. 1999). These dietary characteristics contribute to the tiger muskellunge's attractiveness as a predator control fish, as a 100 cm or larger tiger muskellunge would be able to consume the 30 cm or greater northern pikeminnow that are considered the greatest predators of salmonid juveniles in reservoirs.

However, the research conducted for this study was unable to locate studies in which tiger muskellunge predation potential on salmonids had been specifically tested. Cursory monitoring of tiger muskellunge planted in Mayfield Lake on the Cowlitz River has found that of the 46 stomachs examined, 28 were empty, 12 contained northern pikeminnows, 4 contained suckers, 1 had eaten a redsided shiner, and 1 contained a

brown bullhead (Tipping 2000). The 46 stomachs examined as of 2000, were of tiger muskies that had either died as a result of annual boat shocking efforts, gillnetting, or had been caught in the recreational fishery (Tipping 1991, Tipping 1992, Tipping 1993, Tipping 1994, Tipping 1995, Tipping 1996b, Tipping 1999, Tipping 2000). Thus, with such a small sample and potentially biased sampling techniques, this information does not conclusively suggest that tiger muskies in Mayfield Lake are not consuming salmonids. A more detailed analysis of approximately 100 stomach samples in Mayfield Lake was initiated in 2001. The majority of the full stomachs sampled contained northern pikeminnow and largescale suckers, although the remains of 2 salmonids were observed in tiger muskellunge stomachs. It is important to note that the results of this study are preliminary (pers. comm., S. Caromile, WDFW, December 2002). Yet, the research conducted in Mayfield reservoir suggests that tiger muskellunge predation on salmonids may be occurring.

In addition, a study conducted in Milltown Reservoir, Montana found that during seasonal periods of high abundance, northern pike fed primarily upon salmonids, most notably bull trout (Schmetterling 2001). This is important to potential reintroduction efforts in the upper Lewis River system, as the food preferences of northern pike and tiger muskellunge are very similar (Wahl and Stein 1988).

Schmetterling (2001) studied the diet of northern pike by season, to see if the species responded to seasonal differences in prey abundance. The contents of 57 northern pike were examined from fish collected between March 8 and March 24, 2000. The predominant prey species consumed included largescale suckers (12% of stomachs sampled), mountain whitefish (5%), and northern pikeminnow (4%). The majority of northern pike examined (79%) had empty stomachs. This type of diet is consistent with the other food consumption studies that have been conducted for northern pike, tiger muskellunge, and muskellunge, described above.

However, northern pike collected from May 3 to May 17, 2000 showed a significant change in the prey species consumed. In the Milltown Reservoir, this period overlaps with bull trout and other salmonid migration timing. Of the 57 northern pike stomachs examined in May, 9 contained bull trout (16%) and 7 (13%) contained westslope cutthroat or rainbow trout. In addition to salmonids, the northern pike had consumed 4 (7%) slimy sculpins, 1 (2%) pumpkinseed, 2 (4%) largescale suckers, 1 (2%) yellow perch, some had eaten dragonfly nymphs, and 32 (56%) of the stomachs were empty (Schmetterling 2001). Therefore, bull trout were the single most abundant species collected from northern pike stomachs during this time period.

Northern pike stomachs examined (n = 84) from October 17 through November 6 were also devoid of bull trout, with whitefish (13%), largescale suckers (11%), and redsided shiner (4%) being the species consumed (Schmetterling 2001). For all sample periods, the vast majority of northern pike stomachs that contained food had only 1 fish.

It is important to note, that no bull trout were found in the northern pike stomachs examined on May 3 or May 17, suggesting that pike may only feed on bull trout during migration peaks, when abundance is at its highest. This time period coincided with a spring freshet that likely led to increased salmonid outmigration (Schmetterling 2001).

The results of this study suggest that if anadromous salmonids species were introduced to Lake Merwin, tiger muskellunge may feed upon outmigrating juveniles during periods of high salmonid abundance.

Another important consideration in the use of tiger muskellunge as a trophy fish and predator control strategy is the fact that they are sterile, like many hybrids (Weithman and Anderson 1977, Buss et al. 1978). Therefore, there is no chance that the tiger muskie population will grow beyond the numbers planted in Lake Merwin. If it were decided that tiger muskellunge pose an unacceptable threat to salmonid reintroduction efforts in the upper Lewis River watershed, then the program could be discontinued and muskies would be eliminated from the reservoir within approximately 10 years.

From 1995 through 2000, WDFW introduced 9,945 tiger muskellunge into Lake Merwin, with an average of 1,243 fish planted per year (Table 4.16-10) (Tipping 2001b, Hillson and Tipping 2001).

Table 4.16-10. Tiger muskellunge released into Merwin Reservoir.

Date	Number	Size(#/lb)
September 1995	1,208	5.8
May 1996	375	2.0
May 1997	1,331	4.0
May 1998	1,945	3.5
October 1998	1,717	10.0
May 1999	1,273	2.9
May 2000	968	2.8
June 2000	1,128	3.1

Tiger muskellunge were scheduled to be planted in 2001, although these fish were lost to mortality at the Cowlitz River hatchery facilities during early rearing (Hillson 2002). Furthermore, the fish planted in the first 2 years of the program (1,583 total) were thought to have had poor survival and very few of these fish are likely to still be present in the reservoir (Hillson and Tipping 1999). In the early years of the program, tiger muskellunge were released in Speelyai and Cresap bays, although recent plants have scattered the fish throughout the upper portion of the reservoir (Tipping 1996; pers. comm., T. Hillson, WDFW, December 2002). Tiger muskellunge demonstrate site fidelity; therefore, it can be inferred that the upper portions of Lake Merwin are likely the locations of highest tiger muskellunge density, particularly Speelyai and Cresap bays (Tipping 2001).

Although muskellunge demonstrate site fidelity, the species tends to demonstrate seasonal movements (Hanson and Margenau 1992). Radio telemetry studies conducted in Mayfield Lake on the Cowlitz River, found that the areas occupied by tiger muskellunge differed substantially by season. However, the areas occupied at a given period were rather similar from year to year, supporting the assertion that the species maintains a consistent home range (Tipping 2001). Muskies in Mayfield Lake generally occupied smaller areas and moved less during summer months (July – October) than in winter and spring (November – June). During the summer the average area maintained

by tiger muskies was 0.48 km² compared to an average of 1.39 km² during the winter. The types of habitat used also differed by season. During the summer muskies stayed in shallower water (1.5-2.5 m (5–8 ft)) near aquatic macrophytes, while in the winter and spring, they were found in 5-10 m (16–33 ft) of water (Tipping 2001).

In 2000, WDFW fitted 4 Lake Merwin tiger muskellunge with sonic tags. Only one of the fish was successfully tracked for over a year, although the data provides a general idea of the movements of tiger muskellunge in Lake Merwin. The results of this research were consistent with the findings from the Mayfield Lake study. Tiger muskies demonstrated a preference for shallow water (1.5-3 m (5-10 ft)) during the summer (May – August) and moved to deeper water (4.6-9 m (15-30 ft)) during the fall and winter (September through December) (Hillson 2002). One important distinction between Lake Merwin and other tiger muskellunge impoundments is that the reservoir is devoid of the aquatic vegetation that the species tends to prefer. It is not known how the lack of vegetation in Lake Merwin affects their behavior, although a study of muskellunge diets found that the absence or presence of aquatic macrophytes did not alter their feeding habits (Wahl and Stein 1988).

Since the tiger muskellunge program in Lake Merwin has only been underway for about 7 years, and the first 2 years of plants had poor survival, it is difficult to definitively discuss the success of the program from a predator control perspective. Annual boat shocking conducted in Speelyai Bay found that northern pikeminnow density in 2000 was only about 20 percent of what it had been at the beginning of the program (Table 4.16-11) (Hillson and Tipping 2001, Hillson 2002). However, variables such as flooding in 1997 could have affected northern pikeminnow abundance, thus the decrease in abundance cannot be directly linked to tiger muskellunge predation based upon the available data.

Furthermore, as discussed above, no formal diet analysis research on tiger muskellunge in Lake Merwin has been performed, which would be necessary to define the species preference for northern pikeminnow and to determine whether or not salmonid predation by muskies is occurring.

Table 4.16-11. Number of northern pikeminnow (>12 cm) captured while boatshocking Speelyai Bay.

Year	May	October	Total
1995	471	303	774
1996	519	208	727
1997	491	116	607
1998	210	96	306
1999	133	--	133
2000	83	52	135
2001	125	--	125

Other Potential Reservoir Predators – In the Lewis River system, other fish species such as rainbow, cutthroat, and bull trout likely feed on juvenile salmonids in the project reservoirs; however, the extent to which predation by these species occurs in lentic systems is not well understood. Studies conducted in Alaska and British Columbia found that coho salmon can have a significant impact on sockeye populations in lakes. For example, in Chignik Lake, Alaska coho ate as much as 59 percent of the average population of sockeye fry (Pearsons and Fritts 1999). Based upon these results, it is possible that reintroduced coho would benefit from the food source provided by kokanee juveniles. However, predation by coho may ultimately affect the health and sustainability of the introduced kokanee population in the watershed. Cutthroat trout in lakes have also been found to be quite piscivorous and are capable of maintaining fairly constant rates of predation. In Margaret Lake Alaska, cutthroat trout consumed approximately 50 percent of stocked sockeye salmon (Cartwright et al. 1998). Although these studies provide some insight into salmonid predation on other salmonids, the majority of the research on this subject has been conducted in stream habitats and is discussed in the following section.

However, it is known that in the presence of fish predators of any species in lakes and reservoirs, juvenile salmonids become increasingly dependent on complex habitats where predators cannot forage as effectively. Areas that provide cover in the form of aquatic macrophytes, inundated vegetation, stumps, or large boulders are important nursery areas for salmonids so that they can avoid both salmonid and non-salmonid predators. Tabor and Wurtsbaugh (1991) found that in laboratory studies the probability of mortality was 34 percent lower in areas with cover than in sections without cover. In areas without cover and in the presence of predators, juvenile salmonids experienced reduced growth rates presumably due to intimidation by predator species (Tabor and Wurtsbaugh 1991, Reinhardt et al. 2001). To reduce the potential intimidation factor, juvenile salmonids, particularly coho, tend to school when cover is not available. Schooling behavior may have several safety benefits including earlier detection of approaching predators, confusion of attacking predators, and simple dilution of risk to individuals by increasing the total number of available prey in an area (Grand and Dill 1999). No studies could be found regarding the benefits of schooling versus the use of cover; therefore, it is not possible to discuss which predator avoidance strategy may be more effective.

Although predation by northern pikeminnow and other piscivorous species in the project reservoirs is a potentially significant threat to reintroduced anadromous salmonid species, it is also important to consider predation by other piscivorous species in the project tributaries and lower Lewis River. These types of predatory interactions are discussed in the following section.

Instream Predation

In the mainstem and tributary streams in the Lewis River watershed, reintroduced salmonid species would face different predation risks than in the reservoirs. Since the juveniles of all salmonid species spend at least several months in streams, this type of predation could be very important to the success of reintroduction efforts. Studies have found that due to their early spring emergence and use of mainstem habitats, spring and fall Chinook juveniles may be the species most susceptible to predation by other stream dwelling piscivorous fish (Pearsons and Fritts 1999). Although Chinook may be the most

vulnerable species, predation on all juvenile salmonid species by piscivorous fish has been documented, as is discussed in the following paragraphs.

Northern pikeminnow are presumed to be present throughout the mainstem Lewis River. Studies have found that even in free-flowing stream reaches, the presence of northern pikeminnow can alter the microhabitat preferences of salmonid species. Brown and Moyle (1991) found that before pikeminnow establishment in a test stream, resident salmonid species were found throughout the stream. After pikeminnow were introduced, salmonids typically inhabited shallower or more structurally complex portions of the stream. Rainbow trout for example were found almost exclusively in riffles after pikeminnow introduction, which was the only microhabitat not used extensively by northern pikeminnow (Brown and Moyle 1991). These types of habitat shifts and changes in salmonid behavior may result in reduced growth rates, as discussed above. However, it is important to note that the majority of northern pikeminnow found in stream habitats would likely be found in the lower Lewis River and would therefore only have an impact on reintroduced salmonids during migrations.

In addition, tiger muskellunge have been observed on approximately 5 instances below Merwin Dam. It is not known whether these fish were passed over the spillway or through the turbines (pers. comm. T. Hillson, WDFW, December 2002). Furthermore, no documentation was found regarding the feeding habits of tiger muskellunge in riverine environments. Therefore, it is not possible to evaluate the number of tiger muskellunge in the lower Lewis River, nor the potential for predation on salmonids. Tiger muskellunge that have migrated to the lower Lewis River would only have an opportunity to prey upon reintroduced salmonids during outmigration and with the low numbers of muskies likely present in the lower river, the actual numbers of fish consumed would likely be low.

In the upper Lewis River mainstem and tributary streams the primary piscivorous fish predators on salmonid species would most likely be other salmonids. Numerous research efforts have documented incidences of salmonids consuming other salmonids or their eggs. The majority of these studies have found that salmonid predators tend to feed on fish as only a supplement to their diet. However, when piscivorous salmonid smolts are present in large numbers, even rather low levels of predation on other salmonids can have dramatic effects on population dynamics.

For example, a study in the Salmon River, Idaho found that hatchery steelhead smolts were only consuming approximately 0.00148 Chinook salmon fry per smolt, which is a rather low level of predation. However when calculated for the entire 744,000 steelhead smolts released, the predation accounted for an estimated 4.8 to 21.7 percent of the naturally spawned Chinook fry in the watershed (Cannamela 1993). Predation by steelhead on Chinook salmon may be even more of a concern than the Cannamela (1993) study would suggest. Similar studies of hatchery steelhead smolts and residualized steelhead conducted in northeast Oregon found that predation rates can be as high as 0.49 to 0.84 prey per predator, although the authors of the study acknowledge that these findings are inconsistent with other research on the subject (Whitesel et al. 1994, Jonasson et al. 1994, Jonasson et al. 1995, Jonasson et al. 1996). In addition, studies of steelhead predation on sockeye salmon fry in the Cedar River, Washington found that

hatchery steelhead were not actively feeding on fish. However, the same study revealed that wild steelhead smolts consumed approximately 3.1 to 47.7 percent of all sockeye fry, with the sampled smolts containing between 1.8 and 16.5 fry (Beauchamp 1995).

Coho salmon smolts have also been reported to consume Chinook salmon fry at relatively low levels, with approximately 0.16 to 0.17 percent of steelhead smolts in the Yakima River containing Chinook fry (McConnaughey as reported in Hawkins and Tipping 1999). As mentioned above, even predation at these seemingly low levels can have substantial effects on Chinook fry populations when the abundance of coho smolts basin-wide is considered. It is important to note that coho salmon smolts only consume fry that are between about 40 and 46 percent of their total body length (Pearsons and Fritts 1999). For example, in laboratory studies, coho smolts ranging in size from 129 mm to 174 mm (5 to 7 in) consumed only Chinook salmon ranging in size from 40 mm to 74 mm (1.5 to 3 in). However, even though coho are not likely to consume fish larger than about 46 percent of their total body length, they have been found to attempt to consume larger fish (51 – 58 percent of body length) and may be successful in killing them even though they are unable to actually ingest the larger fry (Pearsons and Fritts 1999).

Although not as well studied as the other salmonid species, bull trout are known to be a highly piscivorous species. In the Muskeg River system in British Columbia, salmonids and rainbow trout eggs were found to comprise 23.3 percent of the bull trout's diet by volume, while the diet of rainbow trout in the area was composed of only 0.04 percent fish and eggs. Since bull trout are present in many of the project's larger tributary streams and in Yale and Swift reservoirs, predation by the species on the juveniles of reintroduced anadromous species could contribute to reductions in the number of smolts produced in the upper watershed. However, information on this subject is limited. Yet it is also important to consider that the increased abundance of salmonid juveniles in the upper Lewis River basin would substantially increase available food sources for bull trout, which are a protected species under the Endangered Species Act (ESA). This increase in food sources for bull trout could increase production of the species in the Lewis River, contributing to recovery. However, it is not known whether available food sources are limiting bull trout production, therefore, the potential beneficial effects are uncertain.

One of the most important studies regarding predation in the Lewis River system was conducted in recent years by the Washington Department of Fish and Wildlife. This study used beach seining and lavage techniques to sample predation rates by salmonid smolts in the Lewis River. The study found that all 4 of the major salmonids species whose smolts inhabit the lower Lewis River were preying on salmonid fry, presumably fall Chinook, although the contents of the stomachs were not keyed to specific salmonid species. The 4 hatchery smolt species examined included spring Chinook, coho, steelhead, and cutthroat trout (Hawkins 1998). The period of most intense sampling occurred in 1997 and 1998, although some investigations were conducted in 1995 and 1997. It was during the earlier sampling that 11 spring Chinook smolts were captured and from which 10 fall Chinook fry remains were found for a total consumption rate of 0.91 smolts per day (Hawkins 1998). The rate is considered daily consumption for the smolts, as the study found that all of the salmonid species examined were able to digest

their stomach contents in about 24 to 25 hours. Therefore, the contents of the stomach at any given time can be considered the daily consumption of an individual smolt (Hawkins unpub.). This rate of evacuation of stomach contents has been supported by previous studies regarding salmonid predation (Cannamella 1993).

WDFW's Lewis River predation study was able to obtain more detailed data for coho, steelhead, and cutthroat trout during the sampling conducted in 1997 and 1998. The results from the 1997 and 1998 sampling are summarized in Table 4.16-12. Again, it was presumed that the salmonid fry consumed were fall Chinook, however, the stomach contents were not definitively assigned to a particular salmonid species. Cutthroat trout were by far the most piscivorous of the salmonid species with daily consumption rates of 2.148, although it is important to note that the total sample size was relatively small (n=54). Steelhead had the second highest salmonid fry consumption rates at .272 fry/day (n=298) followed by coho at .088 fry/day (n=1,884). These predation rates are substantially higher than those reported for the same species in upper Columbia River tributaries. These differences in predation rates are thought to be a reflection of the substantially greater abundance of Chinook fry in the Lewis River compared to most other areas in the Columbia River system (Hawkins and Tipping 1999).

One important aspect of the WDFW's findings was that for cutthroat and coho, wild smolts had higher predation rates than hatchery fish, which is consistent with other salmonid predation studies (Beauchamp 1995). Hawkins (1998) suggested that these differences in predation rates between hatchery and wild smolts may be due to a learning process for hatchery smolts, as they are not accustomed to feeding on live fish. This is supported by Hawkins finding that many of the stomachs of the hatchery fish contained sticks, leaves, rocks, and other debris (Hawkins 1998). Once the hatchery smolts acclimate to their new surroundings, predation rates may increase.

Table 4.16-12. Lower Lewis River salmonid predation results for 1997 and 1998.

COHO													
Year	Stock/ Origin	Number Lavaged	Observed occurrence of predation on salmonids							Coho Predators		Prey Consumed	
			0	1	2	3	4	5	6	No.	Percent	No.	Rate
1997	Hatchery	1,209	1,169	24	11	4	1	0	0	40	3.3%	62	0.051
1998	Hatchery	447	416	19	8	2	1	0	1	31	6.9%	51	0.114
Combined		**1,656**								**71**	**4.3%**	**113**	**.068**
1997	Wild	84	81	2	0	0	1	0	0	3	3.6%	6	.071
1998	Wild	144	120	9	10	3	2	0	0	24	16.7%	46	.319
Combined		**228**								**27**	**11.8%**	**52**	**.228**
Hatchery and Wild Coho Combined Predation Rate													**.088**

Table 4.16-12. Lower Lewis River salmonid predation results for 1997 and 1998 (cont.).

STEELHEAD													
Year	Stock/ Origin	Number Lavaged	Observed occurrence of predation on fall Chinook							Steelhead Predators		Prey Consumed	
			0	1	2	3	4	5	6	No.	Percent	No.	Rate
1997	Hatchery	110	107	3	0	0	0	0	0	3	2.7%	3	0.027
1998	Hatchery	48	25	5	10	4	3	1	0	23	47.9%	54	1.125
Combined		**158**								**26**	**4.3%**	**57**	**.361**
1997	Wild	52	43	6	3	0	0	0	0	9	17.3%	12	.231
1998	Wild	88	78	7	1	1	0	0	0	9	10.2%	12	.136
Combined		**140**								**18**	**12.9%**	**24**	**.171**
Hatchery and Wild Steelhead Combined Predation Rate													**.272**
CUTTHROAT TROUT													
Year	Stock/ Origin	Number Lavaged	Observed occurrence of predation on fall Chinook							Cutthroat Predators		Prey Consumed	
			0	1	2	3	4	5	6	No.	Percent	No.	Rate
1997	Hatchery	1	0	1	0	0	0	0	0	1	100%	1	1.000
1998	Hatchery	15	2	3	3	6	0	1	0	13	86.7%	32	2.133
Combined		**16**								**14**	**87.5%**	**33**	**2.063**
1997	Wild	6	0	1	5	0	0	0	0	6	100%	11	1.833
1998	Wild	32	8	3	9	3	5	2	2	24	75.0%	72	2.250
Combined		**38**								**30**	**78.9%**	**83**	**2.184**
Hatchery and Wild Cutthroat Combined Predation Rate													**2.148**

*   Rate is the number of fry consumed per predator per day, based upon a 24 hour expulsion period demonstrated during the study.

Source: Hawkins unpub.

During the 1997 sampling, wild steelhead also had higher predation rates than their hatchery counterparts. However, in 1998, hatchery steelhead sampled displayed high rates of predation that far exceeded the rates exhibited by the wild fish (Hawkins unpub.). Possible explanations for this anomaly include:

- 92 percent of the hatchery steelhead caught in 1998 were captured after May 27, while most of the 1997 hatchery steelhead catch (92 percent) occurred before May 27. Therefore, the 1998 had been in the stream longer and had more time to discover Chinook as a food source;

- Most of the steelhead caught in 1998 were residuals that had moved upstream;

- Abundance of fall Chinook prey was 3 times higher in 1998;

- Hatchery steelhead lavaged in 1998 were 7 percent longer in length;

- Nearly half (48 percent) of the hatchery steelhead caught in 1998 were predatory versus 3 percent in 1997 (Hawkins unpub.).

Even with the generally lower predation rates displayed by hatchery released fish, the sheer numbers of wild and hatchery smolts in the lower Lewis River makes predation on Chinook salmon fry a significant concern. Furthermore, even though steelhead demonstrated substantially higher rates of predation, coho salmon would be expected to consume nearly 3 times as many fry as steelhead due to the fact that approximately 10 times more hatchery coho are released in the Lewis River than steelhead (Hawkins and Tipping 1999). For all species released into the Lewis River it has been estimated that hatchery smolts alone account for a loss of approximately 10 million juvenile fall Chinook per year, which comprises about 50 to 72 percent of the total lower Lewis River Chinook fry production (Hawkins 1998, Hawkins unpub.). Even in 1998, when hatchery release sites for cutthroat trout and steelhead were moved further downstream to reduce predation, the loss of fall Chinook due to hatchery smolt predation was estimated at 7.5 million (Hawkins unpub.).

Even though the Lewis River study focused on predation on Chinook fry, it is believed that predation occurs on all salmonid fry present in the Lewis River and is relative to fry abundance for each species at a given time (Hawkins 1998).

Reservoir Predation Control Strategies

If after reintroduction, predation by northern pikeminnow becomes a limiting factor on the number of smolts produced in the upper watershed, there are control programs that could be implemented to reduce predation pressure. One of the purposes of the tiger muskellunge program has been to control northern pikeminnow and preliminary data suggests that the program may be proving effective in that capacity. If necessary in the future for the Lewis River, there are other northern pikeminnow control measures that have been in place for over 10 years in the mainstem Columbia and Snake rivers. Three primary control programs (sport-reward fishery, dam angling, and site-specific removal) have been implemented to reduce the number of pikeminnow in the lower Columbia River. The programs are based on 6 key assumptions, which include:

1. the number of retuning adults is a function of survival of juvenile salmon and steelhead during migration in freshwater;

2. northern pikeminnow in mainstem Columbia and Snake river reservoirs consume significant numbers of juvenile salmon and steelhead that would otherwise have survived migration;

3. large, old northern pikeminnow are the most important predators of salmonids;

4. the cumulative effect of a 10-20 percent annual northern pikeminnow removal rate reduces the predation rate from the fish by 50 percent through reducing the population of older predaceous individuals;

5. target northern pikeminnow removal rates can be achieved and sustained with a combination of removal methods;

6. northern pikeminnow population dynamics or the composition of the resident fish community do not compensate for removals (i.e. the remaining fish do not

increase their consumption rates of salmonids as the density of northern
pikeminnow decreases) (Beamesderfer et al. 1996).

The following provides a brief description of each of the 3 control methods and discusses
the program's overall success at reducing predation on migrating salmonid smolts.

Sport Reward Fishery – The premise of the sport-reward fishery system is rather simple.
Bonneville Power Administration (BPA) offers anglers $4 for every pikeminnow, 250
mm (10 in) or longer, caught in the Columbia River from the mouth up to Priest Rapids
Dam, Washington. After an angler turns in 100 fish, the reward is $5 per fish, and $6
after 400 fish are deposited. The season generally runs from the beginning of May to the
end of September (BPA 2002).

Since the program began in 1990, anglers have caught over one million northern
pikeminnow in the Snake and Columbia rivers. The sport-reward program has been the
most successful of the 3 control strategies both in numbers of fish removed and cost
effectiveness. The program costs between $1.5-2 million annually, which in 1993
worked out to $13.62 per pikeminnow removed (pers. comm., R. Porter, Pacific States
Marine Fisheries Commission, September 1998; Hanna and Pampush 1995).

Controlled Angling and Site-Specific Removal – Both of these programs are
administered by local Indian tribes with management and oversight provided by the
Columbia River Inter-Tribal Fish Commission. The controlled angling program consists
of multiple crews using hook-and-line removal methods at specific dams within zones
restricted to other anglers. From 1991 to 1996, the controlled angling program had
removed over 110,000 northern pikeminnow from areas around dams (Friesen and Ward
1999). The total annual cost has ranged from $250,000 to nearly $1 million, which
equates to approximately $38 per pikeminnow removed in 1993 (pers. comm., R. Porter,
Pacific States Marine Fisheries Commission, September 1998; Hanna and Pampush
1995).

The site-specific removal program has been the least successful of the 3. Using gill nets
and trap nets at selected locations, usually at tributary mouths prior to hatchery releases,
3,000 to 10,000 pikeminnow have been removed annually since 1993. The cost ranges
from $250,000 to $400,000, with cost per pikeminnow removed as high as $110 in 1997
(pers. comm., R. Porter, Pacific States Marine Fisheries Commission, September 1998).

Although the Columbia River pikeminnow control programs have been quite expensive,
they have substantially reduced consumption of salmonids in Columbia River reservoirs.
Through these 3 control programs, sustained exploitation rates of over 10 percent of the
northern pikeminnow population have been achieved. At these rates it has been
estimated that losses of juvenile salmonids due to pikeminnow predation in the Columbia
River could be reduced from an estimated 15.2 million individuals annually to 9.4 – 13.1
individuals annually. This would represent a gain of 1.1 to 2.9 percent of the total 200
million downstream migrants (Friesen and Ward 1999). If the assumptions for the
program, as discussed above, are correct, then the program should result in substantially
higher smolt production for the entire system, which in turn should result in more adult
returns. Whether or not such a program is appropriate for the Lewis River system would

depend upon estimates of juvenile salmonid loss to northern pikeminnow once reintroduction efforts were conducted.

Instream Predation Control Strategies

Controlling predation on salmonid fry by other salmonid species is a difficult task; however, changes in hatchery practices may help to reduce the predation risk imposed by salmonid smolts. Hawkins (1998) suggests several possible strategies that may aid in reducing the predation risk associated with hatchery releases. These recommendations are listed below.

- Hatcheries should strive to release fish that emigrate quickly (i.e. fully smolted fish). This might be accomplished through removal of smaller pre-smolts out of the ponds prior to releases, as these fish would likely outmigrate more slowly. Quality, not quantity, should be the goal of hatchery releases into streams with natural salmonid production.

- The practice of volitionally releasing coho smolts at the Lewis River Hatchery should be examined. Field observations show that many parr and pre-smolts are the first to exit the ponds. These smaller fish have been shown to be predatory and reside in the major fall Chinook rearing areas for extended periods.

- Hatchery releases should be as early as possible when in-river flows are high and juvenile salmonid densities are low. Flushing by increasing flows at later dates might be examined, but fall Chinook stranding is a major issue with this strategy.

Management changes such as these would not completely eliminate predation risks to salmonid fry, but they may be effective in reducing the current predation rates that may account for the loss of over 50 percent of emerging Chinook fry, as discussed above.

Summary of Data Gaps and Uncertainties – To aid in evaluating the potential accuracy and applicability of the analysis presented above in Section 4.16.5.3, the following bulleted list describes the major data gaps and uncertainties.

- The abundance and distribution of northern pikeminnow in Merwin, Yale, or Swift reservoirs has not been quantitatively studied in recent decades. Therefore, thorough analysis of the predation potential of the species in these reservoirs is not possible. The discussion presented above is qualitative and assumes that northern pikeminnow are abundant at least in Merwin Reservoir.

- Studies of northern pikeminnow predation conducted in Lake Merwin in the early 1960s suggest that pikeminnow predation on salmonids may be a major concern, although no recent studies have been conducted. The current level of predation on resident species (i.e. rainbow, kokanee) is unknown. However, the extensive data available for the Columbia River is also thought to be generally applicable to the Lewis River reservoirs and there is moderate certainty that northern pikeminnow predation could pose a threat to the success of anadromous salmonid reintroduction efforts.

- The extent to which northern pikeminnow select dead or moribund salmonids over healthy fish is highly uncertain. Data regarding this phenomenon is limited and it cannot be assumed that this phenomenon would adequately protect healthy salmonid smolts from northern pikeminnow predation.

- There is a high level of uncertainty regarding the distribution and abundance of northern pikeminnow in the lower Lewis River. Thus, knowledge regarding the potential for concentrated predation at downstream hatchery release sites is also limited. Furthermore, the extent to which predation on hatchery smolts by northern pikeminnow may increase the survival of wild salmonid smolts is highly uncertain.

- Due to assumed poor survival during the first 2 years of tiger muskellunge release, the actual numbers of the fish inhabiting Lake Merwin is fairly uncertain. The seasonal distribution and habitat preferences of tiger muskellunge in the reservoir are also uncertain, as the telemetry study conducted in the reservoir produced limited data. Furthermore, there have been reported instances of tiger muskies below Merwin Dam, although there is no data regarding the numbers thought to inhabit the lower river, nor is there data pertaining to the manner in which these fish are migrating downstream (i.e. spill or turbines). Thus, the characteristics of lower Lewis River tiger muskellunge inhabitance and characteristics is highly uncertain.

- There have been no Lewis River-specific studies regarding tiger muskellunge diet. There is very limited data from other areas suggesting that muskie predation on salmonids is a possibility, but the extent to which this may occur is uncertain. Available data suggests that tiger muskellunge may be reducing northern pikeminnow abundance in Lake Merwin; however, the levels of reduction are uncertain due to limited data and potentially confounding variables.

- There is a high level of uncertainty about the predation potential of resident cutthroat and bull trout, in addition to reintroduced salmonid juveniles. Cutthroat and bull trout already present in the upper watershed may actively feed on reintroduced salmonid juveniles. The extent to which such predation would hinder reintroduction success and benefit the health of resident populations is highly uncertain. Furthermore, reintroduced salmonids, namely coho, may benefit from predation on other resident and reintroduced juveniles. However, the potential levels of such predation are uncertain.

- The extent to which reintroduced salmonid smolts would prey upon other salmonid juveniles is highly uncertain. However, research from the Lewis River and other watersheds suggests that predation on wild fall Chinook could be extensive, which is a major concern that would require further consideration and monitoring upon initiation of reintroduction efforts.

- The potential adverse effects of bull trout predation on reintroduction efforts is highly uncertain, as are the possible benefits of increased food sources to bull trout in the Lewis River. Thus, the effects of reintroduction on the health of bull trout is an issue that may warrant further consideration.

- The results of the WDFW study (Hawkins unpub.) are preliminary in nature and were not subject to peer review. These results would need to be evaluated in greater detail upon completion of a final report regarding Lewis River predation on fall Chinook juveniles.

- The reservoir predation control programs presented in this section were designed for use in the mainstem Columbia River, therefore, the applicability and potential for success of these programs in the Lewis River watershed is highly uncertain. Furthermore, the potential success of the hatchery release strategy modifications is highly uncertain and monitoring would need to be conducted to evaluate whether such efforts reduced predation on juvenile salmonids.

Summary of Predation Findings

Northern pikeminnow are abundant in Lewis River reservoirs, especially in Lake Merwin. Based upon studies conducted in the Columbia River, predation on salmonids by northern pikeminnow can substantially reduce the number of outmigrating smolts. Therefore, reintroduction efforts should consider the potential effects of predation by this species on the production potential of the upper watershed. If northern pikeminnow predation in the Lewis River is found to be significantly impacting upper watershed salmonid production, predator control programs such as a sport-reward fishery, controlled angling, or gill netting could be implemented.

In mainstem and tributary streams, northern pikeminnow predation is frequently concentrated at hatchery or trap-and-haul release sites. This concentrated predation may impact the rates of predation on wild fish as they migrate downstream.

If reintroduced, juvenile salmonids, specifically coho and steelhead may consume a large percentage of kokanee salmon in Yale Lake and Lake Merwin. The potential impacts and importance of this type of predation should be considered prior to reintroduction, as the abundance of existing kokanee populations could be affected.

Studies of predation on salmonid fry by salmonid juveniles have found that juveniles only use fish as a supplement to their diet. However, even fairly low predation rates can substantially impact fry production, as there are large numbers of hatchery and wild salmonid juveniles in the lower Lewis River. Current predation rates by Lewis River hatchery salmonids are estimated to reduce the number of fall Chinook fry by approximately 10 million. Therefore, the effects of predation by salmonid juveniles on upper watershed productivity should be considered prior to reintroduction. Modifications of hatchery practices could aid in reducing the predation impact of hatchery released smolts. Such management modifications may include moving release sites away from fry rearing areas, releasing fish that emigrate quickly, and releasing fish during flows that facilitate rapid outmigration.

4.16.5.4 Interbreeding and Reintroduction Efforts

With the exception of fall Chinook, anadromous salmonids in the Lewis River watershed have in large part been maintained by hatchery production (PacifiCorp and Cowlitz PUD

2002b). As in most hatchery driven systems, these hatchery fish have the potential to remain in the river and spawn with wild fish. Numerous studies have attempted to analyze the potential effects of interbreeding between hatchery-produced and wild fish, although consensus regarding these potential effects is far from being reached (Kincaid 1995, MacKinlay 2002).

In general, the concern regarding the effect of interbreeding between hatchery-produced and wild fish is that genetic differences between the 2 stocks will adversely affect the long-term fitness and behavioral adaptations of the wild stocks (Hindar et al. 1991, Chilcote et al. 1986, MacKinlay 2002). A number of behavioral and physical differences between wild and hatchery-produced fish have been noted and are shown in Table 4.16-13, below.

Table 4.16-13. Relative differences between wild and hatchery reared salmonids.

Category	Wild	Hatchery
Survival		
egg-smolt survival	Lower	Higher
smolt-adult survival	Higher	Lower
Behavior		
foraging ability	Efficient	Inefficient
aggression	Lower	Higher
social density	Lower	Higher
territorial fidelity	Higher	Lower
migratory behavior	Disperse	Congregate
habitat preference	Bottom	Surface
predator response	Flee	Approach
Morphology		
juvenile shape	More variable	Less variable
nuptial coloration	Brighter	Duller
kype size	Larger	Smaller
Reproductive Potential		
egg size	Smaller	Larger
egg number	Lower	Higher
breeding success	Higher	Lower

Source: Flagg et al. 2000

The causes of these differences are highly uncertain, although hypotheses generally focus on potential genetic variations that may be caused by numerous factors including:

- *Inbreeding of parent hatchery fish* – Inbreeding can change genotype frequencies in a population, which can lead to changes in performance traits. Inbreeding has been reported to decrease growth, survival, and digestive efficiency, while increasing the rate of physical abnormalities in salmonid fry (Kincaid 1995).

- *Hatchery fish adaptation to the artificial hatchery environment* – The traditional manner in which fish are raised in hatcheries (i.e. very high densities in tanks or

raceways) bears little resemblance to the natural environment. Thus, fish that are genetically well suited for hatchery environments may differ genetically from wild salmonids (Einum and Flemming 2001, Waples 1999, Reisenbichler and Rubin 1999, Hatchery Scientific Review Group 2000).

- *Use of non-native or out of basin fish for stocking* – Use of fish from other watersheds may increase genetic diversity of the hatchery stock, but it may also disrupt gene complexes that are locally adapted (Waples 1991). Such practices may increase the genetic differences between hatchery and wild fish, which could lead to outbreeding depression when the hatchery and wild fish interbreed (Lynch 1997). Outbreeding depression refers to the potential negative impacts (i.e. loss of locally adapted traits) that may result when individuals mate that have highly different genetic compositions (Lynch 1997).

- *Artificial selection in hatcheries for a desired trait* (i.e. fast growth, earlier run-timing, larger adult body size) – Hatchery management strategies may select fish with particular traits in an attempt to maximize hatchery production. Such traits may be the function of genetics that could differ substantially from those of the wild population in a watershed (Campton 1995).

In the Lewis River watershed, these potential differences between hatchery and wild fish are of particular concern for steelhead. Prior to development of hydroelectric facilities, the Lewis River supported self-sustaining populations of both summer and winter steelhead. Historical estimates of abundance ranged from 1,000 to 11,000 summer and winter steelhead (Smoker et al. 1951, WDFW 1994). Current wild escapement numbers in the watershed are somewhat uncertain and data is limited. Returns of summer steelhead in the East Fork Lewis River were reported to average approximately 80 adults between 1996 and 2000, while returns of wild winter steelhead to Cedar Creek, a Lewis River tributary, averaged approximately 60 adults (WDFW 2001a, WDFW 2001b). Adult captures of wild steelhead at the Lewis River Hatchery traps have been rather low, with only 6 wild summer steelhead captured in 1999 and 1 collected in 2000 (PacifiCorp and Cowlitz PUD 2002b). Naturally spawning winter steelhead in the Lewis River basin are thought to be of native stock, although interbreeding may be occurring with hatchery origin fish. Hatchery adult returns have averaged approximately 420 winter steelhead and 1,340 summer steelhead from 1995 through 1999 (WDFW 2001a, WDFW 2001b).

These limited data sources regarding hatchery and wild adult returns suggest that hatchery produced steelhead adults far outnumber wild, or naturally spawning adults. The extent to which Lewis River hatchery adults spawn naturally in the Lewis River remains highly uncertain and may require further investigation. Furthermore, there have been limited studies conducted to determine the genetic similarities and differences between Lewis River wild and hatchery-produced salmonids, an important component in evaluating potential adverse effects of interbreeding. However, it is important to note that Phelps (1992) examined rainbow trout in Siouxon and Canyon creeks to determine if rainbow trout stocked in Swift Reservoir as a recreational fishery, were regularly interbreeding with wild rainbow trout. This study found that the wild fish exhibit genetic diversity that is atypical of other coastal rainbow trout populations. No evidence of hatchery rainbow trout gene flow was found in the rainbow trout collected in upper

Siouxon Creek and only minor gene flow was found in the lower creek rainbow trout. The Canyon Creek populaton also showed no signs of gene introgression. Introgression in this case refers to the introduction of hatchery gene complexes into the wild stock genes (Hindar 1991). The results of the upper Lewis River rainbow trout genetics study are not necessarily applicable to potential interbreeding in the lower river, but they do suggest that further research may be warranted.

There are several important concerns regarding interbreeding of hatchery-produced and wild steelhead in the Lewis River watershed. One of the key concerns is the potential reduction of reproductive success in the natural environment. Leider et al. (1990) found that the reproductive success of hatchery origin fish in the Kalama River was substantially lower than for wild fish in natural environments and that the progeny of hatchery origin fish spawned in streams was considerably lower than the survival of the offspring of wild fish. Additional studies in the Kalama River watershed found similar results (Chilcote et al. 1986, Hulett et al. 1996). It is also important to note that although the survival of the progeny of hatchery fish was lower, the overall numbers of smolts outmigrating from the Kalama River was dominated by the offspring of naturally spawning hatchery fish. This was because the number of hatchery fish spawning naturally far exceeded the number of wild spawners (Chilcote et al. 1986). Over extended periods of time, it is a concern that the overall numbers of offspring from naturally spawning hatchery fish will lead to a shift in the genetic make-up of the wild steelhead population (Hindar et al. 1991, Einum and Fleming 2001). This provides another reason why it might be prudent to further evaluate the number of Lewis River hatchery origin adults that spawn naturally in the lower river.

The potential shift in wild steelhead genetics, related to interbreeding, may increase overall diversity in the gene pool, although it may displace favorable gene complexes in the wild population that are specifically adapted to the local environment (Waples 1991). For example, anadromous salmonid stocks adapt over time to the characteristics of the watershed (i.e. hydraulics, presence of other salmonid species) and adult run-timings can shift to ensure the highest survival rates. Introduction of hatchery fish can disrupt the gene complexes that govern run-timing and alter such favorable locally adapted traits. However, the likelihood and severity of such effects is uncertain and a topic that requires further investigation (Waples 1991, Waples 1999). Yet, as noted previously, the extent to which Lewis River hatchery steelhead spawn naturally is highly uncertain and would need to be evaluated before these types of risks could be fully evaluated. Furthermore, it is important to note that the potential shift in genetic make-up through the natural spawning of hatchery fish is thought to pose a greater threat when the genetics of the hatchery and wild fish are dissimilar, which also suggests the importance of defining the similarities and differences between Lewis River hatchery and wild stocks (Einum and Fleming 1997, Utter 1999).

Other studies on the topic of interbreeding between hatchery and wild fish have found that hatchery and wild strains adapt fairly quickly to their surroundings and hybrids of the 2 species are not as well equipped for either hatchery or stream rearing. For example, Reisenbichler and McIntyre (1977) found that the survival of hybrid juveniles was considerably lower than for wild x wild offspring. However, hybrid fish living in a

stream environment may grow faster than their wild counterparts, which could increase their egg to smolt survival rates over time (Flagg et al. 2000). Other demonstrated trait differences between wild and hatchery salmonids such as reduced foraging success, increased aggression, and decreased predator responses of hatchery-produced fish have been shown to have genetic links that may also be affected by interbreeding (Flagg et al. 2000, Einum and Fleming 1997, MacKinlay 2002). However, it is crucial to note that many of the studies relating to this subject have tended to focus on the differences between hatchery fish and wild fish and not specifically on evaluating the long-term effects of interbreeding between the 2 (Waples 1999, Sharpe et al. 2000). Some studies have found, including research on the Kalama River, that even in areas with substantial overlap between wild and naturally spawning hatchery fish, substantial genetic introgression has not occurred (Hindar 1991, Sharpe et al. 2000). The reasons that introgression has not occurred in these cases is not well understood (Sharpe et al. 2000). Also, as discussed above, genetic sampling of wild rainbow trout in Swift Reservoir tributaries has shown that substantial introgression is not occurring.

WDFW has attempted to minimize the potential adverse effects of hatchery fish interbreeding with wild Lewis River steelhead by purposefully selecting hatchery fish for earlier spawning times (WDFW 2001a, WDFW 2001b). As stated previously, hatchery steelhead in the Lewis River spawn in January and February, while wild steelhead spawn from mid-March through late June (PacifiCorp and Cowlitz PUD 2002a). However, the extent to which this earlier spawning period for hatchery steelhead reduces interbreeding with wild fish is still uncertain and would require further evaluation (Waples 1999). This question is important, as the engineered earlier spawning period for hatchery steelhead could adversely affect wild steelhead health if this trait were incorporated into the wild genetics. In the Lewis River watershed, peak flow events occurring in December and January could scour the substrate, reducing egg to fry survival rates, thereby compromising the health of wild Lewis River steelhead (Chilcote et al. 1986).

One potential solution that has been proposed to help reduce adverse effects of hatchery and wild fish interbreeding is the use of wild adult fish to supplement hatchery broodstock. The idea behind this approach is to capture some of the benefits of the locally adapted wild fish genetics and increase genetic variability in the hatchery broodstock (Kincaid 1995). The lack of genetic variability in hatchery fish has been attributed to such adverse effects as decreased growth, survival, and digestive efficiency, while increasing the rate of physical abnormalities in salmonid fry (Kincaid 1995, Larkin 1981).

The potential benefits to hatchery fish health and survival through the use of wild broodstock supplementation may also be important if hatchery fish are used in reintroduction efforts. Studies regarding survival of wild x hatchery versus hatchery x hatchery progeny in natural stream settings have found that the hybrid fish tend to be better equipped for life in streams and have higher survival rates than fish of complete hatchery origin (Einum and Fleming 1997, Reisenbichler and McIntyre 1977, Sharpe et al. 2000). However, in areas where wild stocks are fairly strong, the use of hatchery x wild hybrids may be detrimental, as they may out-compete wild fish for habitat due to potentially higher growth rates, which could affect the survival of wild juveniles. In

general, the actual benefits of using wild broodstock in the hatchery setting are still uncertain.

Furthermore, the earlier run timing of Lewis River hatchery stocks may hinder the stock's usefulness in reintroduction efforts, as the redds produced by adults spawning in January and February may have an increased likelihood of being scoured, as mentioned above. However, the use of wild broodstock to supplement hatchery steelhead may help to align the run timing of hatchery and wild fish, thereby increasing the survival of reintroduced hatchery fish (Nickelson 1986 as reported in Flagg et al. 2000). If wild broodstock are used, eggs should be taken from adults throughout the run and in a manner that ensures that adults of varying shapes and sizes have an opportunity to reproduce (Hatchery Scientific Review Group 2002).

Although the discussion above has primarily focused on research relating to steelhead, as was dictated by the final study plan, the same types of potential effects of interbreeding have been documented for the other anadromous salmonids species (Flagg et al. 2000, Hindar 1991). However, the understanding of how interbreeding of hatchery-produced and wild fish actually affects wild populations is still rather limited (Waples 1999, Leider 1997).

<u>Summary of Data Gaps and Uncertainties</u> – To aid in evaluating the potential accuracy and applicability of the analysis presented above in the Interbreeding and Reintroduction section, the following bulleted list describes the major data gaps and uncertainties.

- The extent to which Lewis River hatchery fish stray and spawn naturally is highly uncertain. The numbers of naturally spawning hatchery fish is crucial to the evaluation of potential effects of interbreeding.

- Similarly, there is a data gap pertaining to the actual escapement of wild anadromous salmonids, particularly steelhead in the Lewis River watershed. Such information is needed to assess potential interbreeding effects.

- The genetic differences and similarities between Lewis River wild and hatchery-produced fish is another data gap. The greater the differences between the genetics of wild and hatchery fish, the greater the potential for adverse effects associated with interbreeding. Furthermore, this information would lend insight into the extent to which introgression has occurred between hatchery and wild stocks.

- The causes and significance of the physical and behavioral differences between hatchery-produced and wild fish are still uncertain. There are numerous hypotheses regarding the causes of these differences, although there is not substantial agreement among experts on the subject.

- Although a number of studies have found that hatchery x wild and hatchery x hatchery progeny of naturally spawning adults have lower survival rates than wild x wild offspring, the reasons behind this phenomenon are not well understood. Furthermore, there is a data gap regarding the survival rates of second and third

generation offspring of hatchery origin fish that spawn naturally. Ongoing studies continue to examine this topic, but current understanding is limited.

- The extent to which interbreeding leads to the disruption of locally adapted traits is still highly uncertain. This issue has predominantly been raised on the basis of general genetic theory, rather than actual observations of genetic shifts in wild populations. Empirical evidence of such effects in wild salmonid populations was not found during the research conducted for this report.

- The extent to which WDFW's efforts to create earlier spawning hatchery steelhead has actually reduced interbreeding is uncertain. It is expected that some spawning overlap likely occurs and this is a subject that may require further consideration.

- The potential benefits of supplementing hatchery broodstock with wild fish are highly uncertain. Research conducted for this report was not able to locate empirical evidence documenting the potential benefits or costs of using wild broodstock in hatcheries. Existing discussions of benefits are primarily based upon theoretical observations.

4.16.5.5 Benefits of Reintroducing Anadromous Fish

The benefits of reintroducing anadromous salmonids into the upper Lewis River watershed would exceed the obvious results of increasing salmonid abundance in the basin and providing access to an additional 96 miles of potential stream habitat. Recent studies have found that the ocean-derived nutrients transported to a watershed by spawning anadromous salmonids can have a substantial positive impact on the productivity of the aquatic and riparian environments (Naiman et al. 2002).

Most anadromous salmonid growth occurs in the marine environment, where they accumulate up to 95 percent of their body mass (Groot and Margolis 1991, Chaloner and Wipfli 2002). Thus, anadromous salmonids are a contradiction to the general perception that materials and nutrients flow from land to freshwater and then to the sea. Each year, salmon runs provide a massive transport of organic material and nutrients from marine water to the freshwater and terrestrial ecosystems (Wipfli et al. 1999, Cederholm et al. 1999). These nutrients transported by salmon are referred to as marine derived nutrients (MDN) (Chaloner and Wipfli 2002). Since, most anadromous salmonids are semelparous, their carcasses left in streams after spawning are a significant source of nutrients (Naiman et al. 2002, Cederholm et al. 1999).

MDN may be especially important in Pacific Northwest streams, which tend to be limited in nutrients, especially nitrogen (N) and phosphorous (P), of which salmon carcasses are an excellent source (Wipfli et al. 1998, Cederholm et al. 1989, Chaloner and Wipfli 2002). In larger streams, such as the Lewis River, which support multiple anadromous salmonid species, adult runs span much of the year and salmon carcasses decompose slowly over weeks or months, thus providing a fairly constant source of nutrients to the system (Wipfli et al. 1999). Historically, salmon runs returned approximately 160 to 226 million kilograms (kg) (353 to 495 million lbs) of biomass to Pacific Northwest streams, annually. That number is now closer to 11.8 to 13.7 million kg (26 to 30 million lbs),

annually. Gresh et al. (2000) estimated that the reduction in salmon returns to the Northwest has produced an annual nutrient deficit of 5 to 7 million kg (11 to 15 million lbs) of marine derived N and P. Thus the question becomes, what is the importance of MDN and how would the upper Lewis River watershed benefit from the increased availability of nutrients provided by reintroduced anadromous salmonids?

Recent studies have found that salmon contain higher proportions of heavier N and carbon (C) isotopes than those found in most temperate terrestrial and aquatic plants and wildlife, which provides a natural marker for the study of the transfer of nutrients from salmon carcasses to aquatic and terrestrial plants and wildlife (Ben-David et al. 1998, Kline et al. 1990, Helfield and Naiman 2001).

Using analysis of N isotopes, studies have found that the concentration of the heavier N isotope was significantly higher in some terrestrial plant species in areas where salmon spawn in comparison to reaches absent of salmon runs (Helfield and Naiman 2001, Ben-David et al. 1998, Bilby et al. 1996). As would be expected, the concentration of the heavier N isotope in terrestrial plants tends to diminish with increasing distances from the stream (Ben-David et al. 1998). Furthermore, terrestrial animal species such as brown bears, river otters, mink, and marten inhabiting areas with salmon streams have demonstrated seasonal and annual changes in the heavier N isotope in response to salmon availability, which demonstrates the contribution of MDN to the chemistry of these species (Hilderbrand et al. 1996, Ben-David et al. 1998). Presence of these terrestrial salmon predators has also been attributed to increased concentrations of the N isotope in terrestrial plants, as the feces and urine of these animals provides an important mode of transport for MDN to the terrestrial environment (Ben-David et al. 1998, Helfield and Naiman 2001).

Aquatic species have also demonstrated increased concentration of the heavier N isotope in streams with salmon runs. Aquatic invertebrates demonstrated higher concentrations of the heavier N isotope in salmon bearing streams in comparison to reaches without salmon. In addition, juvenile coho, steelhead, and cutthroat trout exhibited elevated concentrations of the N isotope in areas where salmon were present (Bilby et al. 1996).

From the information presented above it is clear that terrestrial and aquatic plants and wildlife are utilizing the MDN provided by salmon runs. However, simply demonstrating the use of MDN does not definitively suggest that terrestrial and aquatic organisms are benefiting from the nutrient availability. Thus, another important question regarding MDN is how do these nutrients actually promote improved ecosystem health?

One example of the benefits of MDN is illustrated by the increased growth rates of riparian vegetation where salmon are present. Helfield and Naiman (2001) found that among Sitka spruce trees within 25 m (82 ft) of the stream, where MDN inputs tend to be greatest, the mean annual basal area growth was more than tripled at salmon spawning areas, compared to areas without salmon. With this increased growth rate, it was estimated that it would take a Sitka spruce near a spawning area approximately 86 years to attain a diameter at breast height (dbh) of 50 cm, as compared to 307 years for trees in areas without salmon runs (Helfield and Naiman 2001).

This increased growth of riparian trees may increase the quality of large woody debris (LWD) in the system. The influence of LWD on stream habitat is largely controlled by the size of the LWD pieces (Helfield and Naiman 2001). Larger pieces tend to persist for a longer period of time in a stream, as they take longer to decompose and are more difficult to flush out of the system. Furthermore, the longer the tree, the more likely it is that all or part of it will fall into the stream. LWD in streams is commonly thought to contribute to pool formulation, sediment trapping, velocity breaks, increasing structural complexity, and providing cover to salmonids from predators and flushing flows. Therefore, to the extent that wider and taller trees are more likely to enter and remain in streams, the presence of MDN actually enhances the potential benefits derived from the presence of LWD. The presence of high quality LWD may then in turn contribute to increased salmonid populations, thereby creating a positive feedback loop that could ultimately increase long-term salmonid abundance (Helfield and Naiman 2001).

This same type of positive feedback loop may occur in relation to salmon presence and the health of primary producer aquatic invertebrates. Studies have found that the presence of MDN increases the abundance of biofilm in comparison to stream reaches absent of the nutrients provided by anadromous salmonids (Wipfli et al. 1998, Wipfli et al. 1999, Kline et al. 1990). Biofilm is a mixture of microbes contained in the filmy layer attached to stream substrates and is one of the primary components of the aquatic food web (Wipfli et al. 1998). Increased abundance of biofilm means more food available for higher order aquatic invertebrates. For example, benthic macroinvertebrate densities have been found to range up to 5 times higher in streams with salmon carcasses in comparison to those without (Wipfli et al. 1999). Increases in macroinvertebrate abundance enhance food availability for higher trophic level animals such as fish.

Studies have found that the increased macroinvertebrate abundance related to MDN and direct consumption of salmonid carcasses and eggs can substantially benefit juvenile salmonids. Juvenile salmonids in streams where MDN is present have been found to experience statistically significant increases in forklengths and condition factors in comparison to fish in areas without spawning salmon (Gresh et al. 2000, Bilby et al. 1996, Bilby et al. 1998). Increased length and condition factor is closely correlated with improved overwintering, outmigration, and adult to smolt survival (Budy et al. 1998, Wipfli et al. 1999, Bilby et al. 1996). Thus, the benefits to primary production would likely filter through the food web and ultimately benefit the long-term production potential of salmonids in a watershed, thereby creating another positive feedback loop (Bilby et al. 1996).

Another important consideration regarding MDN is the manner in which the nutrients from anadromous salmonid carcasses are transported to the ecosystem. MDN are transported to the terrestrial environment through 3 primary processes, including: 1) deposition of carcasses on land during high flow events; 2) breakdown of the urine, feces, and partially eaten carcasses consumed by terrestrial predators of salmonids, as discussed previously; and 3) terrestrial plants can absorb MDN contained within water flowing through the porous substrate (hyporheic zone) beneath the riparian vegetation (Helfield and Naiman 2001, Edwards 2000). Of these 3 pathways of MDN transport, little is

known regarding the importance of riparian plant uptake of MDN contained in hyporheic flows.

Absorption of MDN in the aquatic environment occurs through 3 primary processes as well, including: 1) direct consumption of salmon carcasses and eggs by fish, macroinvertebrates, and other aquatic species; 2) absorption by aquatic species of MDN suspended in water; and 3) unintentional consumption by benthic micro and macro invertebrates of MDN contained in stream substrates (Wipfli et al. 1998, Chaloner and Wipfli 2002, Bilby et al. 1996).

However, for either terrestrial or aquatic uptake of MDN to occur, the carcasses of spawning salmon must be retained within the system, rather than being flushed out by stream flows. Cederholm et al. (1989) found that instream structures, namely LWD, were crucial to retain salmon carcasses within a stream. In their study of streams on the Olympic Peninsula, they found that approximately 96 percent of the 605 carcasses were retained in the system. Of those retained, 56 percent of the carcasses were held in the system by LWD and an additional 5 percent were trapped in the streams by live roots and branches. Furthermore, pools were an important habitat structure for retaining MDN, with 45 percent of the carcasses caught in pools (Cederholm et al. 1989). Even during high flow events, pools and LWD were found to be effective in holding the majority of salmon carcasses in the stream system, although carcasses were transported further distances during floods (Cederholm et al. 1989). Thus, stream channel complexity is an important factor in ensuring that the MDN provided by salmon carcasses remains in the system. In heavily modified stream channels, the ability to retain salmon carcasses may be substantially degraded, thereby depriving the area of the benefits of MDN.

In addition to restoring MDN inputs to the upper Lewis River watershed, reintroduction may also reduce the need for hatchery production in the basin. If healthy, self-sustaining salmon runs can be restored in the upper watershed there may be less of a need to supplement populations with hatchery production, thereby returning the Lewis River to a more natural condition and function.

Summary of Data Gaps and Uncertainties – To aid in evaluating the potential accuracy and applicability of the analysis presented above in the Benefits of Reintroducing Anadromous Fish section, the following bulleted list describes the major data gaps and uncertainties.

- There is no data regarding whether the aquatic or terrestrial environments of the upper Lewis River system have degraded due to the lack of MDN since hydroelectric development. Therefore, it is not possible to definitively estimate how restoring MDN to the upper basin might improve ecosystem function.

- It is uncertain whether productivity in the upper Lewis River basin is limited by nutrients such as P and N. It can be assumed based upon the general characteristics of Pacific Northwest watersheds; however, no studies of nutrients as a productivity limiting factor in the watershed have been conducted. Thus, the actual benefits of MDN in the Lewis River cannot be fully evaluated.

4.16.6 <u>Schedule</u>

This study is complete.

4.16.7 <u>References</u>

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