Passage of Adult and Juvenile Salmonids through Federal Columbia River Power System Dams

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Passage of Adult and Juvenile Salmonids through Federal Columbia River Power System Dams

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Executive Summary

Over the past 60 years, a large amount of information has been amassed from studies conducted on the behavior and survival of juvenile and adult Pacific salmon (*Oncorhynchus* spp.) during passage through eight hydroelectric dams operated by the U.S. Army Corps of Engineers on the lower Snake and Columbia rivers. The eight dams are Lower Granite, Little Goose, Lower Monumental, and Ice Harbor Dams on the lower Snake River, and McNary, John Day, The Dalles, and Bonneville Dams on the lower Columbia River. These dams and their reservoirs are the mainstem component of the Federal Columbia River Power System (FCRPS).

This technical memorandum focuses primarily on passage data associated with the dams as they have been configured recently, and not on effects on salmon that might accrue from major changes such as dam removal. The effects of the FCRPS on evolutionarily significant units of salmon listed under the Endangered Species Act and potential benefits to these populations from actions undertaken in the hydropower system are being addressed elsewhere, such as through deliberations of the Technical Recovery Teams, which establish biologically based recovery goals, and Biological Opinions. Here we present a synthesis of the most recent and current information and summary conclusions on the following topics: 1) juvenile salmonid passage through spillways, 2) juvenile passage through mechanical screen bypass systems, 3) juvenile passage through turbines, 4) juvenile passage through sluiceways and surface bypass systems, 5) juvenile diel passage and timing past dams, and 6) adult salmonid passage past dams and through the eight-dam reach.

Spill increases fish passage efficiency (proportion of juvenile fish passing through nonturbine routes) and reduces migration time through forebays and tailraces. In general, when daytime spill is provided, spillway passage of juvenile salmonids occurs at a fairly constant diel rate, and migration timing may be delayed if spill is managed based on passage patterns at the powerhouse. Spill has the potential to reduce the exposure of juvenile salmonids to predation, high water temperatures, and disease. Tailrace residence time for juvenile salmonids passing through spillways is influenced by spill volume and pattern. Generally, spillway passage results in more rapid tailrace egress than bypass system passage. Because piscine predators prefer hydraulic conditions provided in eddies and backwaters, spill patterns that facilitate the egress of juvenile salmonids from the spillway stilling basin and through the tailrace likely increase survival.

Fish survival through spillways can be very high (near 1.00) and is often higher than turbine or bypass system survival when spill passage conditions are optimal. However, survival through spillways with deflectors or shallow stilling basins can be negatively influenced by stilling basin depth and turbulence, hydraulic patterns in the basin, spillbay location within the spillway relative to the spill pattern being used, deflector elevation relative to tailwater elevation and thus total river flow, gate opening, and fish location when passing through the spillbay and under the control gate. Survival through spillways with deflectors or shallow basins can be

considerably less than 0.98, a value typically used in hydropower system modeling exercises. Observations and lessons learned from spillway survival studies at The Dalles, Ice Harbor, and Lower Monumental Dams should be applied to other spillways with deflectors to ensure that safe spillway passage conditions are being provided at these locations.

Estimates of spill efficiency (the proportion of fish approaching a project that pass via the spillway) and spill effectiveness (spill efficiency divided by the proportion of total river flow passing over the spillway) for juvenile salmonids vary by dam, the percentage of river flow spilled, spill pattern, time of day, species/run, and dam configuration. Spill effectiveness is generally between 1.0 and 2.0, but can range considerably higher at certain dams. There is a strong indication that above a certain spill level there is no gain and very likely a reduction in spill effectiveness, although spill efficiency will continue to increase with the percentage of total river flow spilled.

Relatively few studies have been conducted on subyearling Chinook salmon (*Oncorhynchus tshawytscha*) survival through spillways during the summer migration period under contemporary conditions. Estimates of mean spillway passage survival at Ice Harbor and The Dalles Dams since 1997 using traditional spill patterns ranged from 0.75 to 1.00. However, due to problems associated with stilling basin conditions at these dams, this wide range in spillway survival estimates may not be representative of other Snake and Columbia River dams. For example, estimated mean survival through the spillway for John Day Dam in 2002 and 2003 ranged from 0.96 to greater than 1.00 when 30% of river flow was spilled throughout a 24-hour period. In addition, estimated mean survival through the Ice Harbor Dam spillway was 0.96 in 2003 under a "bulk" spill pattern, where higher flow was concentrated in fewer spillbays.

Results of gas bubble disease (GBD) monitoring and research on juvenile salmonids indicate that incidents of high prevalence and severity of GBD signs and probable related mortality are associated with exceptionally high river flows or unusual dam operation problems where powerhouse flows are limited. Under total dissolved gas cap conditions of 115% (forebay) and 120% (tailrace), signs of GBD are minimal or nonexistent, with no apparent GBD-related mortality.

Estimates of how many fish are guided into mechanical screen bypass systems at dams vary by location (dam, turbine), time (season, day, time of day), species/run, and method used (physical netting, hydroacoustics, radiotelemetry, and estimates based on detection of fish tagged with passive integrated transponder [PIT] tags). These variables make it difficult to confidently develop accurate and precise point estimates of fish guidance efficiency. Even so, we present our best estimates by dam. Extended-length screens appear to guide higher percentages of smolts without concomitant increases in descaling and injury rates, but only when appropriately designed vertical barrier screens are installed that can manage increased flows into turbine gatewells. Once fish have been guided into gatewells, estimates of passage efficiency into the bypass system have generally exceeded 70%, the regionally acceptable value, and estimates are higher for extended-length than for standard-length fish guidance screens. Contemporary mechanical screen bypass systems are vastly improved compared to the original systems that operated during the 1970s and early 1980s, based on recent low rates of descaling, injury, and system mortality. However, outfall location and configuration are important considerations for maximizing the survival of juvenile salmonids that are bypassed back to the river below dams.

Mechanical screen systems have the potential to be selective for species/run, life history types, and other attributes such as size. Operations that provide multiple routes to pass fish around dams presumably minimize the risk of reducing biodiversity associated with mechanical screen bypass systems.

Smolt-to-adult return rates for PIT-tagged in-river migrants are related to bypass history. Smolts detected at dams other than Lower Granite Dam often return as adults at lower rates than fish not detected at any dams. For hatchery (but not wild) smolts, detection frequency is inversely related to return rates. A confounding factor is that smaller smolts with lower return rates tend to be more easily guided into bypass systems than larger smolts with higher return rates.

Passage through components of mechanical screen bypass systems induces physiological changes related to the general adaptation syndrome, or stress, in juvenile anadromous salmonids. However, the physiological effects of passage through fish bypass facilities are nominal for juvenile Chinook salmon and steelhead (*O. mykiss*), as measured by blood plasma stress indices, and follow a typical sequence for fish subjected to an acute stressor followed by acclimation to or removal of the stressor. Unless exhaustion or physical injury are involved, the majority of bypassed fish are probably minimally affected by a single, and possibly several, stressful experiences. There is some evidence that multiple stress experiences can cumulatively diminish survivability, and post-stress survival likely depends on a combination of the type of stressor involved and each fish's overall condition and health, and thus, sensitivity to that stressor at the time.

At bypass facilities all fish sizes and species are mixed and small yearling Chinook salmon smolts can become stressed when in the presence of larger steelhead smolts. Separation will be necessary to reduce extended holding of small fish with larger fish, especially in the context of transportation. Prototype high-velocity separators separate in excess of 80% of small from large fish and could potentially be used in future mechanical screen bypass systems.

During summer, juvenile anadromous salmonids migrating through the Columbia and Snake rivers encounter water temperatures that often exceed the 20°C (68°F) threshold mandated in the Washington State Clean Water Act. Temperatures exceeding 25°C (77°F), the upper incipient lethal limit for salmonids, have also been documented during the summer juvenile migration period. High summer water temperatures caused a catastrophic mortality within the juvenile collection facility at McNary Dam in summer 1994, which was thought to have resulted from operating turbine units at the south end of the powerhouse that draw in warmer water. Modifications to powerhouse operations and improved temperature monitoring have prevented a recurrence, because average mortality rates during the most recent five summers ranged from 0.6 to 1.0%.

Passage timing for each route of dam passage varies with dam configuration, flow volume through the route, species, run, and life history type. Most studies were not designed to evaluate 24-hour passage behaviors for each species and life history type, making it difficult to precisely describe passage timing for each dam under varying conditions. In general, juvenile migrants that arrive during daytime tend to not pass through powerhouse routes (turbines and bypass) on arrival but will readily pass through surface-oriented outlets and daytime spill if

provided. For juvenile migrants that arrive at night, passage is more equally distributed amongst available routes.

Results from studies of the Lower Granite Dam removable spillway weir and Wells Dam deep-slot surface bypass systems demonstrate that surface bypass is a viable concept that can produce high rates of nonturbine fish passage with a relatively small percentage of project discharge. Surface bypass systems operated 24 hours per day reduce forebay residence times of juvenile salmon and exposure to predators relative to the existing program of 12-hour spill at night. Optimum surface bypass performance is attained when the entrance is at a location known to attract large numbers of fish. Marginal results from other locations indicate that site conditions may limit performance at some locations. Weir entrances have proven more successful at passing juvenile salmon and steelhead migrants than deep-slot entrances, especially when flow velocity through the weir increases gradually. Discharge through the entrance should be sufficient to create a strong velocity flow-field upstream, but high discharge alone will not overcome a marginal entrance location. Additional passage facilities such as a floating forebay curtain or turbine intake occlusion can help improve fish guidance through surface bypass systems. To achieve high passage efficiency and survival, successful surface bypass designs need to address guidance into the entrance, passage through the collector, discharge into the tailrace, and egress through the immediate tailrace. Recent data indicate that impact velocities up to 15.2 ms^{-1} (50 fts^{-1}) are safe for juvenile salmonids for discharges greater than 1,000 cfs.

Estimates of juvenile fish survival through turbines ranged from 0.81 to 0.92 for studies conducted from 1939 through 1968. Studies conducted under contemporary conditions using radiotelemetry and PIT-tag methodologies suggest mean survival through turbines generally remains in this range. Therefore, turbines continue to represent a significant source of mortality for juvenile salmon migrating through the FCRPS. Indirect mortality can be a significant component of total estimated turbine mortality, which future efforts to improve survival through turbines must address.

While a statistically significant relationship between juvenile salmonid survival and peak turbine efficiency has not been demonstrated, operation of units within 1% of peak efficiency most often encompassed the operation that produced the maximum estimated survival. Therefore, continued operation within 1% of peak turbine efficiency is a reasonable operating guideline to follow until more detailed information on the relationship between unit operations and maximum fish survival for each family of turbines is developed. Development of more refined guidelines appears warranted because maximum survival can be as much as 3.2% higher than survival measured at peak efficiency, and this potential improvement in survival is large compared to that for other FCRPS configuration modifications.

Recent radiotelemetry estimates of adult salmonid survival from Ice Harbor Dam to Lower Granite Dam on the Snake River have generally exceeded 90%, and PIT-tag estimates of survival from Bonneville Dam on the lower Columbia River to Lower Granite Dam have been over 80% for spring/summer Chinook salmon and 70–80% for steelhead. For adult salmonids that do not fall back at dams, estimates of survival range from 3% to 5% higher than for those that do fall back. Adult survival above the hydrosystem to hatcheries or spawning grounds is variable and estimates from Ice Harbor Dam using radiotelemetry range from 54% to 77%. The effects of passage through the hydrosystem on reproductive success are unknown.

Overall travel times and migration rates of radio- and PIT-tagged adult salmonids from Bonneville Dam to upriver Columbia or Snake River sites vary by species/run, time of year, and environmental conditions. Recent estimates of travel times and migration rates are similar to times and rates observed during the early development of the FCRPS, when there were fewer dams and the Snake River was unimpounded. Delays in dam passage for upstream migrating adult salmonids are associated with high levels of spill ($\geq$40 kcfs), highly fluctuating spill, and high turbidity. In addition, elevated river water temperatures ($>$20°C) increase travel times between dams and through fishways at dams. Fishway entrance preferences are for deep/wide openings with significant attraction flow. The transition areas between collection channels and ladders are the fishway segments where adult Chinook salmon and steelhead are most likely to turn around and exit the fishway. Rates of fallback through a dam vary by species/run, environmental conditions, and dam. Higher rates are associated with high river flows, spill, and with adult fishway exit locations in close proximity to spillways. Fallback results in an overestimation of adult counts in fish ladders.

Estimates of repeat spawning rates for steelhead kelts that migrate downstream through the Snake and Columbia rivers are low ($\approx$2%). While potentially negative effects of the hydrosystem on iteroparity rates have been recognized, the magnitude of the effects on survival and reproductive success are not clearly understood. Higher flows and spill significantly reduce travel and passage times for downstream-migrating kelts.

Relative to salmonids, upstream passage at Columbia and Snake River dams is poor for adult Pacific lamprey (*Lampetra tridentata*) and white sturgeon (*Acipenser transmontanus*). American shad *(Alosa sapidissima)* have thrived in the FCRPS habitat, but their overall ecological impacts on anadromous salmonids have not been studied.

Acknowledgments

Acronyms and Abbreviations

AHTS. Advanced Hydropower Technology Study — a program funded by the U.S. Department of Energy to develop advanced turbine technologies and designs.

AWS. Auxiliary water supply — delivers supplemental attraction water to entrances used to attract adult salmon into adult fishways and ladders.

BiOp. Biological Opinion issued by NOAA Fisheries Service under the Endangered Species Act.

BPA. Bonneville Power Administration.

Cfs. Cubic feet per second.

COE. U.S. Army Corps of Engineers.

ESA. Endangered Species Act.

ESBS. Extended-length submersible bar screen.

ESU. Evolutionarily significant unit — an independent population that can be listed under the Endangered Species Act.

FCRPS. Federal Columbia River Power System.

FGE. Fish guidance efficiency — the proportion of fish entering the turbine intake of a powerhouse that are guided into a juvenile fish bypass system.

FPE. Fish passage efficiency — the proportion of fish passing a dam that pass the dam through nonturbine routes.

ft msl. Elevation above mean sea level in feet.

GBD. Gas bubble disease.

HVF. High-velocity flume.

ISAB. Independent Scientific Advisory Board — a regional body of scientists convened by NOAA Fisheries Service and the Northwest Power and Conservation Council to provide technical advice to researchers and decision makers.

Kcfs. Thousand cubic feet per second.

MGR. Minimum gap runner — a new turbine design that minimizes gaps between rotating and stationary turbine components.

NMFS. National Marine Fisheries Service, also called NOAA (National Oceanic and Atmospheric Administration) Fisheries Service.

NPPC. Northwest Power Planning Council, now called the Northwest Power and Conservation Council.

OPE. Orifice passage efficiency — the proportion of fish entering a turbine gatewell slot that exit the slot via an orifice and enter a juvenile fish bypass system within a specified amount of time, usually 22 or 24 hours.

PATH. Plan for Analyzing and Testing Hypotheses — an analytical process implemented in response to the NOAA Fisheries Service 1995 BiOp and the Northwest Power and Conservation Council Columbia Basin Fish and Wildlife Program and undertaken by a panel of regional scientists to evaluate hypotheses pertaining to mortality associated with passage through the FCRPS.

PIT tag. Passive integrated transponder tag.

RKm. River kilometer.

RPA. Reasonable and Prudent Alternative — a formal recommendation by NOAA Fisheries Service in a Biological Opinion to avoid jeopardy under the Endangered Species Act.

RSW. Removable spillway weir.

SAR. Smolt-to-adult return rate.

SBC. Surface bypass collector.

STS. Standard-length submersible traveling screen.

TDG cap. Total dissolved gas cap — the total dissolved gas limit that must be maintained in the tailrace or forebay of a dam.

TDGS. Total dissolved gas supersaturation.

TSP. Turbine survival program — a program funded by the U.S. Army Corps of Engineers to understand mechanisms of fish mortality within turbines and develop new turbine designs to improve juvenile fish passage survival through turbines.

VBS. Vertical barrier screen — a stationary screen between the upstream and downstream gatewell slots of a turbine intake that prevents fish that are guided into the upstream slot from reentering the turbine intake through the downstream slot.

Introduction

This technical memorandum synthesizes available information on the behavior and survival of juvenile and adult Pacific salmon (*Oncorhynchus* spp.) during passage through eight dams operated by the U.S. Army Corps of Engineers (COE) on the lower Snake and Columbia rivers (Fig. 1). These eight dams and their reservoirs constitute the mainstem component of the Federal Columbia River Power System (FCRPS). This synthesis focuses on those aspects of passage considered relevant by NOAA's National Marine Fisheries Service (NMFS or NOAA Fisheries Service) for judging FCRPS impacts upon specific salmon stocks listed under the Endangered Species Act (ESA), and in general, to all salmon stocks in the Columbia River that utilize habitats within or above the FCRPS.

The amount of information developed through studies conducted over the past 60 years is large, and reporting all the available information is beyond the scope of this document. This memorandum focuses primarily on information gathered under contemporary conditions, or as the dams have been configured and operated since the 1995 FCRPS Biological Opinion (BiOp) was implemented. Results from studies conducted earlier than this time frame are discussed for context. The focus is on passage issues associated with the dams as they exist, not effects on salmon that might accrue from major changes such as dam removal. Nonetheless, it is recognized that many of the adverse impacts from FCRPS dams on salmonids would be reduced or eliminated with the removal of dams.

The eight dams form back-to-back reservoirs over what was once a free-flowing riverine ecosystem. This stretch of river contained important habitat that salmon historically used for spawning, rearing, physiological development, preparation for seawater entry, growth, and shelter from predators. There is no doubt that this complex of eight dams and reservoirs has significantly changed this riverine ecosystem and affected the stocks of salmon that use and migrate through this habitat. The effects of the FCRPS on listed salmon evolutionarily significant units (ESUs) and potential benefits to these populations from actions undertaken in the hydropower system are being addressed elsewhere, such as in Williams et al. (2005), through life cycle modeling, and through technical deliberations of the Technical Recovery Teams. Here we present what is known about passage at the concrete under current conditions to provide a source of information that can be used by hydropower system and resource managers to judge actions and make decisions regarding dam operations and configurations in the future.

As the National Research Council (NRC) pointed out, the long-term persistence and productivity of salmon populations are dependent in part on maintaining conditions that allow salmon to retain their genetic variability and local adaptations (NRC 1996). More recently, a viable salmonid population has been defined as an independent population of any Pacific salmonid that has a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year time frame. Four key parameters are considered when evaluating population viability status: abundance, population

growth rate, population spatial structure, and diversity (McElhany et al. 2000). It is not possible at this time to assess potential impacts to the viability of salmon populations from specific dam operations or configurations. Perhaps this question will be addressed in the future when appropriate data are available to make such an assessment. For now, we simply recognize that factors considered to be important to the viability of salmon populations over time have been identified and are an important consideration in recovery planning.
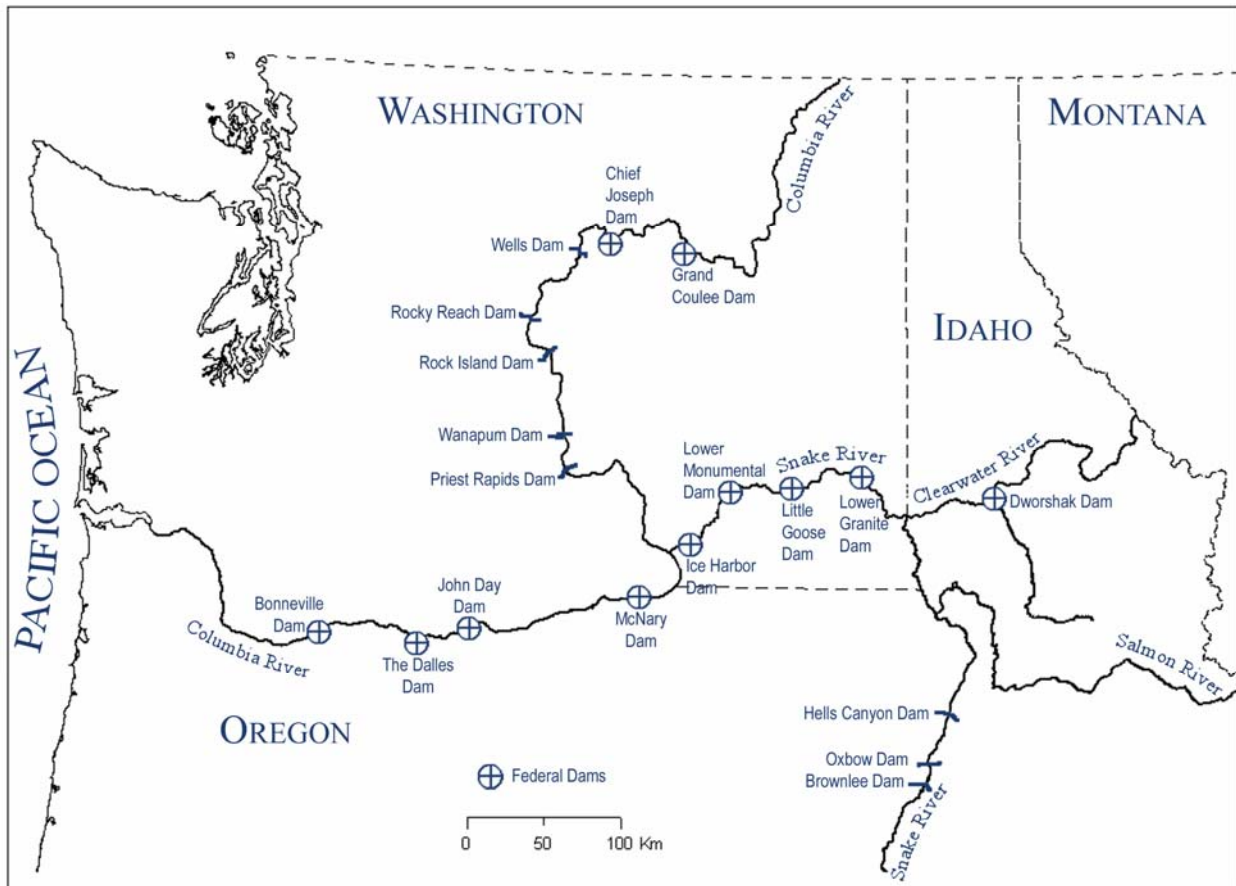
Figure 1.   Map of the federal Columbia River power system.

Juvenile Salmon Passage Through Spillways

Background

Downstream passage through spillways at major hydroelectric dams in the Columbia River Basin has been provided for many years to increase survival of juvenile salmonids. Historically, spill occurred operationally when project capacity or FCRPS generation needs were exceeded. However, as the hydroelectric system was developed, spill became less frequent. In 1988, a 10-year spill program under a Fish Spill Memorandum of Agreement was implemented to provide spill at projects to meet regional goals for fish passage through nonturbine routes. Continued declines in salmonid populations prompted the eventual development of the more aggressive spill management plans required under the NMFS FCRPS BiOps. Since implementation of the 1993 and subsequent BiOps, spill has generally been provided in accordance with the prevailing spill program in most years, but there has been considerable variation in spill among years for a variety of environmental (river flow), system (excess generation spill, excess hydraulic capacity spill, turbine failures, maintenance-related conditions), and fish research (test schedules) reasons.

The generally recognized benefits of spill include reductions in mortality, migrational delays (through forebays, tailraces, and reservoirs), exposure to predation risk, high water temperatures, and diseases (Whitney et al. 1997, Muir et al. 2001, Giorgi et al. 2002). Previous studies have found that passage survival for juvenile salmonids at Columbia and Snake River dams is generally highest for spillways, followed by bypass systems, then turbines (Schoeneman et al. 1961, Whitney et al. 1997, Muir et al. 2001).

The spillway passage route at all FCRPS dams includes a forebay, spill gates, ogee, stilling basin, and tailrace. The forebay typically consists of about 1 km of reservoir immediately above the spillway gates. Most spillway gates have a radial design with a 18.3-m radius and 15.2-m width (COE 1996b). Two dams (Bonneville and McNary) have vertically operated lift gates of similar width. The number of spillbay gates per spillway varies from 8 to 10 at lower Snake River dams to 18 to 23 at lower Columbia River dams.

The ogee sections provide a transition for flow from below the gates to the stilling basin. Flow deflectors that help reduce dissolved gas production at many FCRPS dams are located on the ogee sections at elevations designed for each project. Below the ogee, the spilled flow enters a stilling basin designed to dissipate turbulent energy in a confined, armored zone, thereby minimizing the threat to the spillway's structure. Beyond the stilling basin, the tailrace extends for about 1 km downstream. Spillway capacities are designed for the maximum probable flood and vary from 850 thousand cubic feet per second (kcfs) at lower Snake River dams to 2,290 kcfs at lower Columbia River dams.

Research Results and Discussion

Spill Efficiency and Effectiveness

Spill efficiency is the proportion of fish approaching a project that pass via the spillway. Spill effectiveness is spill efficiency divided by the proportion of total river flow passing over the spillway at the same time. Recent reviews of spill efficiency and effectiveness include Steig (1994), Giorgi (1996), Whitney et al. (1997), and Marmorek and Peters (1998). Estimates of spill efficiency vary by project.

Most spill efficiency estimates are based on hydroacoustic methodologies, which make estimates for the general population, not specific species or stocks. Hydroacoustic estimates are sensitive to transducer position, fish size, and fish orientation. During the summer, hydroacoustic estimates are particularly susceptible to error caused by the presence of nonsalmonids, such as juvenile shad (*Alosa sapidissima*), and high summer effectiveness values reported should be viewed with this in mind. Radiotelemetry studies have been successfully conducted at some dams, where yearling and subyearling Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*) were marked and released upstream, then their behavior and route of dam passage were noted. With sufficient receiving antennas and sample size, estimates of spill efficiency and effectiveness can be derived.

Steig (1994) reviewed studies at Snake and Columbia River dams conducted through 1992 and noted that there is considerable variability in daily and weekly spill effectiveness. However, he concluded that most results fall around a 1:1 relationship between the proportion of water spilled and the proportion of fish passed in spill (i.e., 1.0 spill effectiveness). Giorgi (1996) reviewed estimates of spill effectiveness published through 1993 and noted that effectiveness is poorly estimated for most species due to a combination of sparse observations, imprecise estimates, and the reliance of most estimates on hydroacoustic monitoring, which is unable to distinguish among species. He cautioned that the assumption of a spill effectiveness of 1.0 could not be justified in most cases and implied that a suite of estimates acquired with different methodologies should be considered when attempting to derive species-specific estimates at individual dams. Relying largely on Giorgi's (1996) review, the PATH Hydro Work Group (PATH 1997) concluded that a range of spill effectiveness from 1.0 to 2.0 should be incorporated into sensitivity analyses of passage at Snake and Columbia River dams.

Spill efficiency at Lower Granite, Little Goose, and Lower Monumental Dams can be estimated based on radiotelemetry observations for yearling Chinook salmon at Lower Granite Dam (Wilson et al. 1991), because of the similarity of the three projects. Combining the radiotelemetry observations with the assumptions that 0% of fish pass the spillway at 0% spill and 100% pass at 100% spill, the following relationship (Smith et al. 1993) can be applied:

$$P_f = 2.583P_w - 3.250P_w^2 + 1.667P_w^3$$

where: P_f is the proportion of fish passing over the spillway (spill efficiency),

P_w is the proportion of total river flow passing over the spillway, and

spill effectiveness is defined as $P_f \div P_w$.

The shape of this relationship is uncertain outside of the range of observations (20–40% spill), even though P_f must logically go to 0 and 1.0 at the extremes (ISG 1996).

Spill efficiency and spill effectiveness were examined in 1999 at Ice Harbor Dam using radiotelemetry (Eppard et al. 2000). The study concluded that spill efficiency was 82.6% and spill effectiveness was 1.8:1 during 2000 BiOp day conditions (approximately 45% spill). During nighttime BiOp conditions (approximately 100% spill), spill efficiency increased to 96.1%; however, spill effectiveness was closer to a 1:1 relationship. This suggests that there may be a level of spill at Ice Harbor Dam above which there is no gain in spill effectiveness at passing fish.

In 2003 radiotelemetry studies were conducted by Eppard et al. (2003) from 29 April to 6 June at Ice Harbor Dam to evaluate relative spillway passage and project survival for hatchery yearling Chinook salmon under a 50% voluntary spill operation versus the 2000 BiOp spill operation. Overall spill efficiency during this study was 88.5% (95% CI, 83.7–93.2) and spill effectiveness was 1.5 (95% CI, 1.4–1.6).

Hydroacoustic evaluations of these two spill conditions were also conducted at Ice Harbor Dam from 15 April to 15 July in 2003 (Moursund et al. 2003b). Spill efficiency for spring migrants was 75.9% and 51.7% for BiOp spill and 50% spill, respectively (significantly different), and spill effectiveness was 1.12 and 1.01, respectively. For summer migrants, spill efficiency was 70.8% and 64.4% for BiOp spill and 50% spill, respectively, and spill effectiveness was 10.7 and 1.27, respectively (significantly different). Moursund et al. (2003b) concluded that route-specific passage and project-wide performance metrics differed among operational treatments.

Passage of radio- and PIT-tagged yearling Chinook salmon was evaluated at McNary Dam in 2002 by Axel et al. (in prep. a) during day and nighttime operations from 5 May to 5 June. Nighttime spill remained relatively constant throughout this period (to 120% total dissolved gas [TDG] cap), but daytime spill increased considerably as of 21 May to accommodate increased river flows. Overall spill efficiency was 47% (95% CI, 27–67) and mean spill effectiveness (for Snake River migrants) was 1.12 (95% CI, 0.91–1.33).

Axel et al. (in prep. b) evaluated passage behavior for radio-tagged Columbia and Snake River yearling Chinook salmon at McNary Dam in 2003. During the first three weeks of the study period, from late April to early June, spill was at BiOp levels (nighttime spill to 120% TDG cap), but involuntary daytime spill began on 26 May due to increased river flows. Spill efficiency was 47% for Snake River fish and 49% for Columbia River fish; mean spill effectiveness for Snake River migrants was 1.43 and for Columbia River migrants was 1.32.

At John Day Dam in 1999 a constant nighttime spill of 60% was scheduled (but observed spill was actually 45%), and daytime spill was 0% and 30%, each scheduled for 3-day continuous periods within each replicate. During the spring at John Day Dam, Johnston and Nealson (1999) hydroacoustically measured nighttime spill efficiencies that were lower than reported previously, ranging from 1.4 to 1.7; however, there were potential problems with transducer placement. During the summer, 24-hour spill effectiveness ranged from 2.4 to 5.1,

while nighttime spill effectiveness ranged from 2.4 to 3.3. Both the spring and summer periods exhibited significantly higher 24-hour spill effectiveness during the 30% daytime spill treatment.

Using radio-tagged steelhead, Hansel and Beeman (1999) found spill efficiency was not significantly different between the two treatments, at 46% and 54%, although efficiency was higher during the treatment that included daytime spill. For radio-tagged yearling Chinook salmon, spill efficiency was 53% and 65%; the periods when daytime spill was provided were significantly different and greater than the treatments without daytime spill.

In the spring of 2000, Beeman et al. (2001) evaluated spill passage under two day/night treatments (0/60% and 30/60%) at John Day Dam using radio-tagged steelhead and yearling Chinook salmon. For Chinook salmon, spill efficiency was significantly higher for 24-hour spill (86% CI, 80–90) than for nighttime-only spill (75% CI, 68–81). For steelhead, spill efficiency ranged from 64% to 83% for 24-hour spill and 61% to 79% for nighttime-only spill (no CIs reported). Spill efficiency for Chinook salmon and steelhead passing through day spill was 92% and 68%, respectively. Spill efficiency for Chinook salmon passing through nighttime spill was 78% for both spill treatments, and for steelhead it was 72% during the 12-hour treatment and 78% during the 24-hour treatment. Also reported by Beeman et al. (2001), day spill at John Day Dam in 2000 was extremely effective for yearling Chinook salmon (2.9–3.2) and for steelhead. During nighttime spill (actual nighttime spill ranged from 50% to 54%), spill effectiveness ranged between 1.4 and 1.6 for Chinook salmon and 1.2 to 1.7 for steelhead.

The same two spill treatments were evaluated for passage of radio-tagged subyearling Chinook salmon at John Day Dam in the summer of 2000 (Beeman et al. 2002a). Spill efficiency was significantly higher for the 24-hour treatment (82%) than for the 12-hour treatment (54%), and spill effectiveness was greater for the 30% daytime spill (2.6–3.0) than for the 60% nighttime spill (1.1–1.6).

In 2002 Moursund et al. (2003a) conducted hydroacoustic studies at John Day Dam to estimate fish passage efficiency for 24-hour (30%) and 12-hour (0% day–60% night) spill treatments. With 12-hour spill in the spring, spill efficiency was 78% (± 6) and spill effectiveness was 2.90 (± 0.30); with the 24-hour spill treatment, efficiency was 72% (± 5) and effectiveness was 2.68 (± 0.26). With 12-hour spill in the summer, spill efficiency was 58% (± 11) and spill effectiveness was 2.10 (± 0.30); with the 24-hour spill treatment, efficiency was 61% (± 12) and effectiveness 2.30 (± 0.32). None of the differences for either the spring or the summer were statistically different.

Beeman et al. (2002a) also evaluated fish passage with 12- and 24-hour spill treatments at John Day Dam in 2002 and found no significant difference in spill efficiency between treatments for radio-tagged, hatchery yearling Chinook salmon. Nighttime-spill efficiency for yearling Chinook salmon was significantly higher during the 12-hour spill treatment than during the 24-hour treatment, whereas nighttime spill efficiency for wild steelhead was similar during the two spill treatments.

In contrast to all other dams, the powerhouse of The Dalles Dam is oriented nearly parallel to the natural course of the river, while the spillway is located on what was a shallow basalt bluff. The Independent Scientific Group (ISG 1996) suggests it is not surprising that this

project exhibits higher spill efficiency than many other projects, due to its unique configuration. Giorgi and Stevenson (1995) reviewed biological investigations that described smolt passage behavior at The Dalles Dam and discussed implications for future surface bypass research. They cited three investigations that indicated that spill effectiveness was near 2.0 when about 20% of the flow passed over the spillway. These investigations included Willis (1982), which describes a curvilinear relationship in which spill effectiveness is equal to or greater than 2.0 at spill below approximately 30% of total river flow, declining to about 1.4 at 60% spill, and further declining to 1.0 at 100% spill.

A 1996 radiotelemetry study at The Dalles Dam by Holmberg et al. (1998) supports this general relationship. Spill effectiveness for yearling Chinook was 2.3 at 30% spill and 1.25 at 64% spill in 1996. PATH (1997) reviewed the available information and suggested that a factor of 2.0 be applied at The Dalles Dam at spill levels up to 30%; when spill levels are above 30% spill, the relationship grades from 2.0 to 1.0. This relationship predicts a spill effectiveness of 1.5 at 65% spill.

Hansel et al. (2000) evaluated fish passage under two spill conditions (30% and 64%) at The Dalles Dam from 7 May to 29 May of 1999 using radio-tagged yearling Chinook salmon and steelhead. For 30% spill, they reported spill efficiencies of 51% (46–57) and 66% (60–71) for Chinook salmon and steelhead, respectively; for 64% spill, spill efficiencies of 79% (74–84) and 86% (82–90) for Chinook salmon and steelhead, respectively. Spill effectiveness at 30% spill was 1.7 for Chinook salmon and 2.2 for steelhead, and at 64% spill, 1.3 for Chinook salmon and 1.4 for steelhead.

Beeman et al. (2000a, 2000b) evaluated spill efficiency and effectiveness during the spring and summer at The Dalles Dam under constant conditions of 40% spill. From 28 April to 26 May, with river flows ranging from 217 to 305 kcfs, spill efficiency for radio-tagged yearling spring Chinook was 86% (95% CI, 83–89) during the day and 71% (95% CI, 67–76) at night; for steelhead, spill efficiency was 88% (95% CI, 84–91) during the day and 82% (95% CI, 77–85) at night. Spill effectiveness for both yearling spring Chinook and steelhead was 2.2 during the day, and 1.8 for Chinook and 2.1 for steelhead at night.

For the period 28 June to 20 July, under 39–40% spill (with river flows ranging from 107 to 281 kcfs) Beeman et al. (2000b) reported spill efficiency for radio-tagged fall Chinook was 85% (95% CI, 78–91) during the day and 73% (95% CI, 63–82) at night at The Dalles Dam. Spill effectiveness was higher during the day than at night, 2.15 vs. 1.86, respectively.

Moursund et al. (2001b) conducted a hydroacoustic evaluation of fish passage at The Dalles Dam in the spring and summer of 2000. For the period of 13 May to 6 June, under average spill of 32% (river flows averaged 234 kcfs), spill efficiency was 89% during the day and 83% at night. For the period of 6 June to 6 July, under average spill of 31% (river flows averaged 192 kcfs), spill efficiency was 75% during the day and 73% at night.

Ploskey et al. (2000) conducted a hydroacoustic evaluation of fish passage at Bonneville Dam in 2000. Under average spill of 33% in the spring, spill efficiency was 44% and spill effectiveness was 1.36. Under average spill of 48% in the summer, spill efficiency was 49% and spill effectiveness was 1.03.

Schilt et al. (2002) conducted a hydroacoustic study to provide project-wide estimates of fish passage efficiency (FPE), spill efficiency, and spill effectiveness for river-run fish passing Bonneville Dam in 2002. In the spring the spillway discharged about 44% of total river flow; spill efficiency averaged 57% and spill effectiveness was 1.17. In the summer 48% of river flow was spilled; spill efficiency averaged 42% and spill effectiveness was 0.97. For spill levels less than 100, 100–125, and greater than 125 kcfs, spring spill efficiency was 41%, 54%, and 66%, respectively; summer spill efficiency was 22%, 39%, and 50%, respectively.

For the spring of 2001 (unusually low river flow year) at Bonneville Dam, Evans et al. (2001b) reported spill efficiency for radio-tagged yearling Chinook salmon was 30% with 37% spill and 1% with 2% spill (adult attraction spill); spill effectiveness was 0.86 and 0.53 for the two spill levels, respectively. Overall for the spring season, spill efficiency was 16% with 22% spill and spill effectiveness was 0.7. During the summer, overall spill was 2.4% and spill efficiency for subyearling Chinook salmon was 2%.

Evans et al. (2002) evaluated passage behavior of radio-tagged yearling Chinook salmon and steelhead at Bonneville Dam in 2002. During the early 2 May to 9 June study period, 49% of the river flow was spilled. For three spill treatments (75 kcfs during the day, up to 137.6 kcfs during the day, and up to 141.0 kcfs during the night), spill efficiency for yearling Chinook was 42%, 63%, and 59%, respectively (57% overall); for steelhead, spill efficiency was 32%, 70%, and 49%, respectively (55% overall). Spill effectiveness for Chinook salmon during the three treatments was 1.3, 1.2, and 1.1, respectively (1.2 overall); for steelhead, spill effectiveness was 1.0, 1.4, and 1.0, respectively (1.1 overall).

Spill Survival

Whitney et al. (1997) reviewed 13 estimates of spill mortality for salmonids (3 steelhead and 10 salmon) published through 1995 and concluded that 0–2% is the most likely mortality range for standard spillbays. They also pointed out that local conditions, such as back eddies or other situations that may favor the presence of predators, may lead to higher spill mortality.

Some point estimates for mortality in spillbays with flow deflectors are higher than estimates for spillbays without deflectors. For example, the highest estimates of survival for yearling Chinook salmon and steelhead at Snake River dams were obtained from spillbays without flow deflectors, ranging from 0.984 to 1.000 (Muir et al. 1995b, 1996, 1998). Although lower survival estimates were obtained from spillbays with flow defectors (ranging from 0.927 to 1.000) (Iwamoto et al. 1994, Muir et al. 1995b, Muir et al. 1998), differences in survival between the two types of spillbays compared pairwise were not significant for steelhead at Little Goose Dam or yearling Chinook salmon at Lower Monumental Dam.

Eppard et al. (2003) also reported that at Ice Harbor Dam survival was lower over spillways with flow deflectors when flows were lower during the spring of 2002 and the summers of 2000 and 2002. These authors suggested that the lower survivals may have been due to hydraulic conditions in the stilling basin when flows were less than 90 kcfs. Survival of hatchery yearling Chinook salmon was 0.892 (95% CI, 0.841–0.943) in 2002. Survival of hatchery subyearling Chinook salmon was 0.894 (95% CI, 0.857–0.931) and 0.885 (95% CI, 0.856–0.915) in 2002 and 2000, respectively. In addition, for yearling Chinook salmon at Lower

Monumental Dam in 2003, relative survival was significantly higher (0.987 vs. 0.834) when submergence of flow deflectors increased as flows increased (Hockersmith et al. in prep.). Carlson and Duncan (2003b) suggested that turbulence over the spillway ogee increased with decreased flow over the spillway at Ice Harbor Dam, causing a higher probability of fish contacting spillway surface structures during lower discharge. Flow deflectors were originally designed to reduce gas supersaturation when involuntary spill was required during periods of higher flow and subsequent higher tailrace elevation. Hence, survival over spillways with flow deflectors may be reduced somewhat when spill occurs at lower flow levels.

Survival of juvenile salmonids passing through spillways at lower Snake River dams has been evaluated at least once at all projects (Table 1). Among the most recent studies, Plumb et al. (2004), in evaluating the performance of a removable spillway weir at Lower Granite Dam, reported a spillway relative survival estimate of 0.931 ($\pm$0.060) for yearling Chinook salmon during 12-hour nighttime spill to the gas cap. Hockersmith et al. (in prep.) estimated relative survival for radio-tagged hatchery yearling Chinook salmon released in spillbays 4 and 7 during daytime operations at Lower Monumental Dam from 29 April to 6 June of 2003. Relative spillway survival was 0.896 and 0.895 for spillbays 4 and 7, respectively. Spillway survival for releases after 23 May (when average total river flow doubled, average powerhouse discharge increased threefold, and average tailwater elevation increased 1.2 m) was significantly higher than for earlier releases (0.987 vs. 0.834).

In 2000, NOAA Fisheries Service estimated relative survival for river-run hatchery yearling and subyearling Chinook salmon passing through the spillway at Ice Harbor Dam (using PIT tags) was 0.978 and 0.885, respectively (Eppard et al. 2002a). The estimate for yearling Chinook salmon for this particular year was similar to previous spillway survival estimates at Little Goose (1.021) (Iwamoto et al. 1994) and Lower Monumental Dams (0.927 to 0.984) (Muir et al. 1995b). However, in 2002 at Ice Harbor Dam, analysis of PIT-tag data indicated spillway passage survival was 0.895 and 0.890 for hatchery yearling Chinook salmon under daytime and nighttime conditions, respectively; the overall survival estimate was 0.892. During the summer, spillway passage survival for hatchery subyearling Chinook was 0.876 and 0.915 for fish passing under daytime and nighttime conditions, respectively; the overall survival estimate was 0.894. In 2003, Eppard et al. (2003) evaluated relative spillway passage survival under 50% voluntary spill operation versus BiOp voluntary spill operation (45 kcfs spill during the day and up to 100% spill at night). Spillway survival for hatchery yearling Chinook salmon was 0.928 and 0.948 for the 50% test spill and BiOp spill operations, respectively. Also in the summer of 2003, Absolon et al. (2003a) evaluated the survival of PIT-tagged subyearling Chinook salmon passing through the spillway at Ice Harbor Dam under conditions of BiOp "bulk spill" (BiOp spill using only two or three spillbays rather than all spillbays); mean survival from spillbays 2, 3, and 4 to McNary Dam was 0.96.

Table 1. Location, species and run type, study year, fish-marking method, spillbay, test conditions, and survival estimates for spillway passage at dams on the lower Snake and Columbia rivers. Dam abbreviations: LGR-Lower Granite; LGO-Little Goose; LMO-Lower Monumental; IHR-Ice Harbor; MCN-McNary; JDD-John Day; TDA-The Dalles; BON-Bonneville.

Dam	Species/run type	Year	Method	Flow deflector	Location	Conditions	Survival	Reference
LGR	Steelhead	1996	PIT tags	no	Bay 1	3.9 kcfs	1.01	Smith et al. 1998
LGR	Yearling Chinook salmon	2003	Radiotelemetry	yes	Bays 2–8	BiOp night	0.931	Plumb et al. 2004
LGO	Steelhead	1997	PIT tags	no	Bay 1	4.9–10.0 kcfs	1.004	Muir et al. 1998
LGO	Steelhead	1997	PIT tags	yes	Bay 3	4.9–10.0 kcfs	0.972	Muir et al. 1998
LGO	Yearling Chinook salmon	1993	PIT tags	yes	Bay 3	3.8 kcfs	1.021	Iwamoto et al. 1994
LMO	Coho salmon	1973	Freeze brands	yes*	Bay 2	4.5 kcfs	0.970	Long and Ossiander 1974
LMO	Coho salmon	1973	Freeze brands	yes	Bay 4	4.5 kcfs	1.100	Long and Ossiander 1974
LMO	Steelhead	1974	Freeze brands	yes	Bay 7	4.5 kcfs	0.978	Long et al. 1975
LMO	Steelhead	1974	Freeze brands	no	Bay 8	4.5 kcfs	0.755	Long et al. 1975
LMO	Subyearling Chinook salmon	1972	Freeze brands	yes*	Bay 2	13.1 kcfs	0.831	Long et al. 1972
LMO	Subyearling Chinook salmon	1972	Freeze brands	yes*	Bay 2	2.8 kcfs	0.840	Long et al. 1972
LMO	Yearling Chinook salmon	1994	PIT tags	yes	Bay 7	4.4–4.8 kcfs	0.927	Muir et al. 1995a
LMO	Yearling Chinook salmon	1994	PIT tags	no	Bay 8	4.4–4.8 kcfs	0.984	Muir et al. 1995a
LMO	Yearling Chinook salmon	2003	Radio, PIT tags	yes	Bays 4, 7	2.0–11.5 kcfs	0.900	Hockersmith et al. in prep.
IHR	Yearling Chinook salmon	2000	PIT tags	yes	Bays 3, 5, 7	BiOp night	0.978	Eppard et al. 2002a
IHR	Subyearling Chinook salmon	2000	PIT tags	yes	Bays 3, 5, 7	BiOp night	0.885	Eppard et al. 2002a
IHR	Yearling Chinook salmon	2002	PIT tags	yes	All bays	BiOp	0.892	Eppard et al. 2002b
IHR	Subyearling Chinook salmon	2002	PIT tags	yes	All bays	BiOp	0.894	Eppard et al. 2002b
IHR	Yearling Chinook salmon	2003	Radiotelemetry	yes	All bays	BiOp	0.948	Eppard et al. 2003
IHR	Yearling Chinook salmon	2003	Radiotelemetry	yes	All bays	50% spill	0.928	Eppard et al. 2003
MCN	Subyearling Chinook salmon	1955	Tattoo	no	Not specified	Not specified	0.980	Schoeneman et al. 1961
MCN	Subyearling Chinook salmon	1956	Tattoo	no	Not specified	Not specified	1.00	Schoeneman et al. 1961
MCN	Yearling Chinook salmon	2002	Radiotelemetry	yes	All bays	BiOp	0.976	Axel et al. in prep. a
MCN	Yearling Chinook salmon	2003	Radiotelemetry	yes	All bays	BiOp	0.928	Axel et al. in prep. b
MCN	Yearling Chinook salmon	2003	Balloon tag	yes	Bay 5	Various	>0.98	Heisey et al. 2003
JDD	Subyearling Chinook salmon	1979	Freeze brands	no	Bay 16	4.3 kcfs	0.98–1.2	Raymond and Sims 1980
JDD	Yearling Chinook salmon	2000	Radiotelemetry	no	All bays	0/60% spill	0.986	Counihan et al. 2001
JDD	Yearling Chinook salmon	2000	Radiotelemetry	no	All bays	30/60% spill	0.937	Counihan et al. 2001
JDD	Steelhead	2000	Radiotelemetry	no	All bays	0/60% spill	0.988	Counihan et al. 2001
JDD	Steelhead	2000	Radiotelemetry	no	All bays	30/60% spill	0.905	Counihan et al. 2001
JDD	Yearling Chinook salmon	2002	Radiotelemetry	no	All bays	0/54, 30/30	.993, 1.000	Counihan et al. in prep. a

Table 1. Continued. Location, species and run type, study year, fish-marking method, spillbay, test conditions, and survival estimates for spillway passage at dams on the lower Snake and Columbia rivers. Dam abbreviations: LGR-Lower Granite; LGO-Little Goose; LMO-Lower Monumental; IHR-Ice Harbor; MCN-McNary; JDD-John Day; TDA-The Dalles; BON-Bonneville.

Dam	Species/run type	Year	Method	Flow deflector	Location	Conditions	Survival	Reference
JDD	Subyearling Chinook salmon	2002	Radiotelemetry	no	All bays	0/54, 30/30	.985, 1.003	Counihan et al. in prep. a
JDD	Steelhead	2002	Radiotelemetry	no	All bays	0/54, 30/30	.958, .938	Counihan et al. in prep. a
JDD	Yearling Chinook salmon	2003	Radiotelemetry	no	All bays	0/45, 0/60	.939, .934	Counihan et al. in prep. b
JDD	Subyearling Chinook salmon	2003	Radiotelemetry	no	All bays	0/60, 30/30	.901, .955	Counihan et al. in prep. b
TDA	Subyearling Chinook salmon	1997	PIT tags	no	All bays	64% spill	0.92	Dawley et al. 1998b
TDA	Coho salmon	1997	PIT tags	no	All bays	64% spill	0.87	Dawley et al. 1998b
TDA	Subyearling Chinook salmon	1998	PIT tags	no	All bays	64% spill	0.75	Dawley et al. 2000a
TDA	Subyearling Chinook salmon	1998	PIT tags	no	All bays	30% spill	0.89	Dawley et al. 2000a
TDA	Coho salmon	1998	PIT tags	no	All bays	64% spill	0.89	Dawley et al. 2000a
TDA	Coho salmon	1998	PIT tags	no	All bays	30% spill	0.97	Dawley et al. 2000a
TDA	Subyearling Chinook salmon	1999	PIT tags	no	All bays	64% spill	0.96	Dawley et al. 2000b
TDA	Subyearling Chinook salmon	1999	PIT tags	no	All bays	30% spill	1.00	Dawley et al. 2000b
TDA	Coho salmon	1999	PIT tags	no	All bays	64% spill	0.93	Dawley et al. 2000b
TDA	Coho salmon	1999	PIT tags	no	All bays	30% spill	0.96	Dawley et al. 2000b
TDA	Yearling Chinook salmon	2000	PIT tags	no	All bays	BiOp	0.91	Absolon et al. 2002
TDA	Yearling Chinook salmon	2000	Radiotelemetry	no	All bays	BiOp	0.92	Counihan et al. 2002
TDA	Subyearling Chinook salmon	2000	PIT tags	no	All bays	BiOp	0.897	Absolon et al. 2002
TDA	Subyearling Chinook salmon	2000	Radiotelemetry	no	All bays	BiOp	0.826	Counihan et al. 2002
TDA	Yearling Chinook salmon	2002	Balloon tags	no	Bays 4, 9, 13	40% spill	98.4, 98.9, 95.6	Normandeau Assoc. Inc. 2003
TDA	Subyearling Chinook salmon	2002	Balloon tags	no	Bays 4, 9, 11	40% spill	93.6, 93.3, 92.6	Normandeau Assoc. Inc. 2003
TDA	Chinook salmon	2002	Balloon tags	no	Bay 2	4.5,12 kcfs	0.949, 1.00	Normandeau Assoc. et al. 2004
TDA	Chinook salmon	2002	Balloon tags	no	Bay 4	4.5,12 kcfs	0.965, 0.995	Normandeau Assoc. et al. 2004
TDA	Chinook salmon	2003	Balloon tags	no	Bay 2	9–21 kcfs	0.931, 1.00	Normandeau Assoc. et al. 2004
TDA	Chinook salmon	2003	Balloon tags	no	Bay 4	9–21 kcfs	0.966, 0.999	Normandeau Assoc. et al. 2004
BON	Subyearling Chinook salmon	1974	Freeze brands	no	Bay 11	13 kcfs	0.958	Johnsen and Dawley 1974
BON	Subyearling Chinook salmon	1974	Freeze brands	yes	Bay 14	13 kcfs	0.868	Johnsen and Dawley 1974
BON	Subyearling Chinook salmon	1989	CWT/freeze brand	yes	Bay 5	6.8 kcfs	0.9604	Ledgerwood et al. 1990
BON	Yearling Chinook salmon	2002	Radiotelemetry	yes	All bays	75 kcfs/cap	0.969	Counihan et al. 2003
BON	Yearling Chinook salmon	2002	Radiotelemetry	yes	All bays	24-hour cap	0.98	Counihan et al. 2003

*Flow deflector included dentates.

Heisey et al. (2003) utilized the HI-Z balloon-tag-recapture technique to evaluate spillway survival in spillbay 5 at Ice Harbor Dam under various spill conditions and release depths in the spring and summer of 2003. Spring survival estimates for tagged yearling Chinook were greater than 0.98 for all conditions tested. One-hour summer survival estimates for tagged subyearling Chinook were 0.889 for bulk spill, shallow release; 0.923 for bulk spill, deep release; and 0.978 for dispersed spill, deep release.

A number of methodologies have been used to estimate spillway survival at lower Columbia River dams, including identification of test fish by fin clips, freeze brands, coded-wire tags and freeze brands, balloon tags, PIT tags, and radio tags. Most recently at McNary Dam, Axel et al. (in prep. a) using PIT- and radio-tag detections estimated 2002 spillway passage survival for hatchery yearling Chinook salmon was 0.976. In 2003 Axel et al. (in prep. b) estimated that spillway passage survival for radio-tagged hatchery yearling Chinook salmon at McNary Dam was 0.928.

Estimates of spillway passage survival at John Day Dam include a study conducted in 1979 by Raymond and Sims (1980), who found that spillway mortality relative to the tailrace was not different from 0. More recently, Counihan et al. (2001) evaluated spill survival of radio-tagged yearling Chinook and steelhead under two daytime/nighttime spill conditions at John Day Dam in 2000. For the two spill conditions (0/60% and 30/60%), survival was 0.986 and 0.937 for yearling Chinook, and 0.988 and 0.905 for steelhead, respectively.

At The Dalles Dam, Dawley et al. (1998b) released PIT-tagged subyearling Chinook and coho salmon (*O. kisutch*) in 1997, and estimated spillway survival of 0.87 and 0.92, respectively, with 64% spill. Results from a 1998 study (Dawley et al. 1999) show that relative survival rates during 64% spill were 0.88 for coho salmon and 0.76 for subyearling Chinook salmon, while during 30% spill, survival was 0.96 and 0.92 for coho and subyearling Chinook salmon, respectively.

Analyses of data from 1999 show that relative survival rates during 64% spill were 0.94 for coho salmon (*O. kisutch*) and 0.95 for subyearling Chinook salmon, while during 30% spill, survival was 0.96 and 1.00 for coho and subyearling Chinook salmon, respectively (Dawley et al. 2000b). In 2000, Absolon et al. (2002) estimated relative survivals at a 40% spill volume of 0.95 for yearling Chinook and coho salmon and 0.92 for subyearling Chinook salmon.

After 2000 spill research at The Dalles Dam shifted to causative mechanisms for the observed high mortality. Direct survival studies and studies of scale hydraulic models indicated that fish retention time in the spill basin might be the cause of the increased injury and mortality. The hydraulic model investigations had shown that flow from bay 4 moved directly out of the stilling basin, while flow from more southern bays like bay 9 tended to recirculate for a period of time before moving out of the stilling basin.

To evaluate the hypothesis that fish released in bay 9 would show higher mortality, Normandeau Associates Inc. (2003) evaluated direct survival by releasing HI-Z balloon-tagged yearling Chinook salmon directly into bays 4, 9, and 13 during the spring of 2002 (14–27 May), and subyearling Chinook salmon into bays 4, 9, and 11 during the summer (15–22 August). Control fish were released below bay 3. Spring survival for bays 4, 9, and 13 was 0.984, 0.989,

and 0.959, respectively. Bay 13 survival was significantly lower than the other two. For the summer in bays 4, 9, and 11, survival was 0.936, 0.933, and 0.926, respectively. Eye injuries were most prevalent, followed by scrapes and operculum damage. Injuries were thought to be mainly from shear and physical contact with solid objects.

In the fall of 2002 and the spring of 2003, Normandeau Associates et al. (2004) again used HI-Z balloon-tagged Chinook salmon to estimate direct passage survival through spillbays 2 and 4 under low (fall) and high (spring) tailwater conditions at The Dalles Dam. In the low tailwater test, 48-hour survival rates for spillbay 2 ranged from 0.949 (deep release at 4.5 kcfs) to 1.000 (releases at 12 kcfs). Survival estimates for spillbay 4 ranged from 0.965 to 0.995, at 4.5 and 12 kcfs, respectively. Spring survival estimates for spillbay 2 ranged from 0.931 (deep/edge release at 9 and 21 kcfs) to 1.000 (deep/mid release at 18 kcfs). Spillbay 2 deep/mid-release survival estimates were high (0.975–1.000) at all four volumes. Survival rates for spillbay 2 deep/off-center releases were greater than 0.99 for the 12 and 18 kcfs releases, but 0.931 for both the 9 and 21 kcfs spill volumes. Spillbay 4 survival estimates for four test conditions (all mid-releases) in the spring ranged from 0.966 to 0.999. The researchers concluded that safe fish passage was maximized at spillbay volumes of 12 and 18 kcfs with a total spill of 108–112 kcfs. Fish released at greater depth and off-center in spillbay 2 appeared to have higher rates of injury; eye damage or head and body bruises were generally the most prevalent visible injury types in both the fall and the spring.

At Bonneville Dam, Holmes (1952) estimated that subyearling Chinook salmon survival through the spillway was from 0.96 to 0.97, depending on how the data were analyzed. Johnsen and Dawley (1974) compared the survival of subyearling Chinook salmon passing through spillbays with and without flow deflectors, and found that relative survival was 0.868 and 0.958, respectively. Ledgerwood et al. (1990) found that survival of subyearling Chinook through spillbay 5 was not significantly different than for fish released downstream. Based on the balloon-tag methodology, the calculated survival probabilities for deflector and nondeflector spillways were both 1.0 at Bonneville Dam; however, fish passing through a spillbay without a spill deflector displayed a slightly higher injury rate (Normandeau Associates et al. 1996a).

Counihan et al. (2003) evaluated spillway passage at Bonneville Dam in 2002 with radio-tagged yearling Chinook salmon. The project was operated with powerhouse 2 priority, and spill was alternated between 75 kcfs day/gas cap at night and a test condition of gas cap for 24 hours; spillway survival was 0.969 and 0.980, respectively.

Causal Mechanisms Affecting Spill Survival

As pointed out in Table 1 and discussed in the Spill Survival subsection above, many point estimates of survival through spillways made in recent years under contemporary conditions using PIT- or radio-tag methodologies are less than 0.98, a value typically used in recent model studies of survival through the hydropower system (Marmorek and Peters 1998, NMFS 2000). In general, these lower survival estimates apply to Lower Monumental, Ice Harbor, Lower Granite, and The Dalles Dams, and suggest that conditions juvenile salmonids experience during spillway passage may differ among dams and within a dam spillway under various conditions. Here we summarize information developed to date on spillway passage survival at these dams.

Ruggles and Murray (1983) reviewed fish responses to passage through spillways at Canadian dams and concluded that spillways were not a serious source of fish mortality. They pointed out that fish may be injured by rapid pressure change, rapid deceleration, shear forces, turbulence, abrasion, and the force of striking water in free fall. They noted that spillway deflectors have been used successfully to reduce air entrainment associated with deep plunging jets, and that injuries begin to occur when the impact velocity of a fish striking the water surface exceeds 16.2 ms^{-1}. Based on a need to design spillway flow deflectors for gas abatement, the COE initiated a literature review of physical injury through spillways through their Dissolved Gas Abatement Study. The study considered three injury types: immediate mechanical injury, short-term delayed injury, and longer-term delayed injury. It concluded that results from studies of impacts from gas abatement structures (such as flow deflectors) have been inconsistent, and effects from gas abatement structures could not be separated from effects associated with passage through the total spillway environment. Immediate direct mortality was the injury type most studied and best understood, and long-term delayed mortality was the least studied and least understood. The report noted that gas abatement structures can produce hazardous conditions, such as shear zones (R2 Resources 1997).

Several years of research conducted at The Dalles Dam was summarized by Ploskey et al. (2001). A spill discharge of 40% of total project discharge was adopted as the best operation and implemented in the spring of 2000. However, testing that year showed that survival was still low in six of eight conditions tested, which included two spillway release locations (north and south), and daytime and nighttime releases during both the spring and summer periods. Average survival ranged from 0.842 for summer, daytime, north spillway to 1.026 for spring, daytime, north spillway (Absolon et al. 2002). Subsequently a committee of engineers and biologists used a 1:80 scale general hydraulic model at the COE Engineering Research and Development Center in Vicksburg, Mississippi, to evaluate physical conditions associated with these elevated spillway mortality rates. They concluded that large volumes of spill through the northern half of the spillway, designed to enhance tailrace egress, resulted in flow from southern spillbays moving northward within the stilling basin. This increased retention time in the basin, along with it being half the depth of the next shallowest basin of the FCRPS dams, suggested that mechanical injury from strike and shear in the turbulent stilling basin was causing the reduced survival.

Results from more recent radiotelemetry (Beeman et al. 2003) and balloon-tag studies (Normandeau Associates Inc. 2003) corroborate these observations. Therefore, an extension of the pier wall between bays 6 and 7 downstream to the stilling basin end-sill was evaluated in the COE hydraulic model. In the model, the wall reduced retention time in the stilling basin when six northern-most spillbays were operated under uniform discharges of from 8 to 20 kcfs. The pier wall extension was installed at The Dalles Dam prior to 2004. Preliminary results based on balloon tagging indicate direct survival was greater than 0.98, and more than 96.8% of the test fish showed no signs of injury.[1]

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[1] M. Langeslay, COE, Portland, OR. Pers. commun., May 2004.

At Ice Harbor Dam, spillway survival of radio-tagged yearling Chinook salmon averaged 0.978 in 2000, and 0.892 in 2002 (Eppard et al. 2002a, 2003). The 2000 Biological Opinion called for nighttime spill of up to 100 kcfs (the gas-cap limit) and daytime spill of 45 kcfs during the spring at Ice Harbor Dam, with uniform spill discharge across bays. To further investigate survival associated with higher (plunging flow) and lower (skimming flow) spill volumes, BiOp and 50% spills were tested during the spring of 2003 using a 2-day-block study design. Relative spillway passage survivals were similar, with 0.948 (95% CI, 0.915–0.981) for the BiOp spill and 0.928 (95% CI, 0.860–1.000) for the 50% spill. Also that year, estimated survival of balloon-tagged yearling Chinook salmon released at two locations (elevations) in spillbay 5 was greater than 0.98 during all conditions tested (Heisey et al. 2003). Although fish injuries ranged as high as 22% for releases made at the deepest elevation and were 8% or less for the shallower release location (Heisey et al. 2003), these release locations appeared to represent a relatively low percentage of the overall population passing the spillbay (Moursund et al. 2003b). Efforts to resolve this issue are continuing. In 2004 preliminary results from balloon-tag studies with yearling migrants released at locations more representative of the overall population indicate survival was still high (>0.98), but injury rates were reduced to 2%.[2]

Survival of subyearling Chinook salmon at Ice Harbor Dam during the summer averaged 0.885 in 2000 and 0.894 in 2002 (Eppard et al. 2002a, 2003). In 2003 a bulk-spill pattern was developed where more than 10 kcfs was spilled through fewer bays, as compared to the pattern of uniform spill through all bays. Estimated survival of PIT-tagged subyearling Chinook salmon was 0.96 (95% CI, 0.90–1.10) (Absolon et al. 2003a). When compared to 2000 and 2002, these results suggest the bulk-spill pattern may reduce fish exposures to physical conditions that cause injuries. In contrast, 1-hour estimated survival under the bulk spill pattern based on balloon-tagged, river-run subyearling Chinook salmon was 0.89 and 0.92 for shallow and deep spillbay release locations, compared to an estimated 1-hour survival of 0.98 for fish released through a deep release under the uniform spill pattern (Heisey et al. 2003). However, the authors indicated there were problems related to high temperatures during this test, suggesting a need for caution when interpreting these data.

At Lower Monumental Dam, Muir et al. (2001) found that survival of PIT-tagged yearling migrants was lower though spillbays with deflectors, compared to standard bays. The study did not address differences between spillbay locations, discharges, or tailwater elevations. Prior to 2003, additional deflectors were installed in the end bays of the spillway, and with these in place, from 25 to 40 kcfs could be spilled while maintaining gas criteria at downstream locations under 24-hour spill. In 2003 Hockersmith et al. (in prep.) evaluated spillway survival based on radio-tagged yearling Chinook salmon where spill was distributed uniformly between spillbays. Relative survival through the spillway was 0.900 (95% CI, 0.843–0.961). Relative survival through spillbays 4 and 7 was similar, at 0.896 (95% CI, 0.779–1.031) and 0.895 (95% CI, 0.724–1.106), respectively.

However, spill discharge early in the season was in the gas cap range noted above when total river discharge was less than 75 kcfs, but was outside this range later in the season when

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[2] M. Shutters, COE, Walla Walla, WA. Pers. commun., May 2004.

total river discharge increased. Additional analysis of the data suggested there was a tailwater elevation effect, where survival was low when tailwater elevation was low (0.834 [95% CI, 0.78–0.90], 440 ft msl, 76 kcfs river flow), and high when tailwater elevation was high (0.987 [95% CI, 0.92–1.06], 444 ft msl, 150 kcfs river flow).[3]

In summary, fish survival through spillways at these dams is influenced by stilling basin depth and turbulence, hydraulic patterns in the basin, spillbay location within the spillway relative to the spill pattern being used, deflector elevation relative to tailwater elevation and thus total river flow, gate opening, and fish location when passing through the spillbay and under the control gate. Efforts to further explore the relationships between these factors are ongoing. Hydraulic model studies indicate that deflector submergence (tailwater elevation) directly influences the hydraulic behavior of the spill jet as it passes over the deflector. Under conditions of high discharge and low submergence of the deflector, the jet passes over the deflector and plunges abruptly into the stilling basin. This operation was tested during the summer of 2003 at Ice Harbor Dam and resulted in an estimated average survival of 0.96. Under conditions of moderate discharge and higher deflector submergence, the spillbay discharge jet is deflected upward, creating an undular, ramped, or submerged hydraulic jump. This operation was tested during the spring of 2003 and resulted in an estimated average survival of 0.99 at Lower Monumental Dam.

Under both conditions, the spillway discharge jet appears to dissipate relatively quickly and produces low impacts on juvenile salmonids passing through the discharge, even though jet velocities are high (>18.3 ms^{-1}). Conversely, under conditions of low discharge per spillbay, certain tailwater elevations, and uniform spill between bays, the discharge jet performs as designed and skims off the deflector and stays near the tailrace water surface for an extended distance downstream. This operation was tested during the spring of 2002 and resulted in an estimated average survival of 0.89 at Ice Harbor Dam, and 0.83 during the spring of 2003 at Lower Monumental Dam.

Based on this information, a possible explanation of the low survival associated with skimming flow off the deflector is that a shear zone is created on the underside of the jet and fish in the lower portion of the nappe are exposed to this zone, which results in injuries. The zone of injury occurs until a point downstream where jet velocity drops to below 15.2 ms^{-1}, a threshold considered safe based on field and laboratory studies (Ruggles and Murray 1983, Pacific Northwest National Laboratory et al. 2001). If this explanation is the case, lowering the elevation of the deflector to produce hydraulic jump conditions a greater percentage of the time, or changing to higher discharges through fewer spillbays ("bulk" spill), may reduce injury and mortality rates.

Clearly, additional field and hydraulic model studies will be needed to verify this explanation, understand the implications for gas abatement, and design solutions to these conditions. The observations of spillway survival at The Dalles, Ice Harbor, and Lower Monumental Dams suggest that survival of juvenile salmonids through spillways is influenced

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[3] E. Hockersmith, NWFSC, Pasco, WA. Pers. commun., January 2004.

by many physical factors. Lessons learned from these studies should be applied to other spillways that have deflectors and training spill associated with the removable spillway weir (RSW) at Lower Granite Dam and others weirs that may be installed in the future.

Forebay Passage and Predation

Beamesderfer and Rieman (1991) found that forebay populations of northern pikeminnow (*Ptychocheilus oregonensis*) and smallmouth bass (*Micropterus dolomieui*) were present in substantial numbers in the forebay of John Day Dam. Poe et al. (1991) reported that the diet of northern pikeminnow in the forebay of John Day Dam was 66% salmonid smolts. In addition, detection histories for radio-tags (from hatchery yearling Chinook salmon) recovered from the Crescent Island Caspian Tern colony near the mouth of the Snake River indicated that, at a minimum, terns from this colony forage from Ice Harbor Dam forebay to McNary Dam forebay (Axel et al. 2003). This suggests that delay of outmigrants in forebays could reduce survival due to increased predation, and project operations such as daytime spill that decrease forebay residence time could increase survival.

Axel et al. (2003) evaluated passage behavior at Ice Harbor Dam in 2001 under low-flow (69.5 kcfs), no-spill conditions and reported a median forebay residence time for radio-tagged hatchery yearling Chinook salmon of 7.3 hours. Schools of yearling Chinook salmon were observed holding within the immediate forebay of Ice Harbor Dam on a daily basis throughout the study period (8–28 May). A previous radiotelemetry study conducted at Ice Harbor Dam in 1999 by Eppard et al. (2000) reported that median forebay residence time for radio-tagged hatchery yearling Chinook salmon was 1.3 hours, with average river flows of 109.7 kcfs and spill averaging 45% (54 kcfs) during the day and 81% (94.2 kcfs) at night.

Radiotelemetry data reported by Axel et al. (2003) regarding passage of radio-tagged hatchery yearling Chinook salmon at McNary Dam in 2001 also indicated prolonged residence time within the forebay under low-flow, no-spill conditions (7.6 hours). Detection histories indicated that once fish entered the forebay, they constantly moved back and forth across the spillway and powerhouse before passing.

At Lower Granite Dam in 2003, median forebay passage times during removable spillway weir tests were 1.92, 1.72, and 2.28 hours for radio-tagged hatchery and wild steelhead, and hatchery Chinook salmon, respectively. For these same test groups, median passage times during spill-to-the-gas-cap tests were 7.37, 4.64, and 4.98 hours, respectively (Plumb et al. 2003).

Tailrace Passage and Predation

The concept of developing spill patterns at FCRPS dams specifically for fish passage was first addressed systematically in the 1960s to facilitate adult salmon passage into the adult fish collection systems. Junge (1967) observed improved adult salmonid passage under intermediate-to-large spill volumes if four or five gates at each end of the spillway were at low-volume settings. At large dams, this resulted in a tapered spill pattern near each end and a flat spill pattern across the central portion of the spillway. At smaller dams, this produced a "crowned" pattern across the entire spillway tailrace, with the highest discharge in the middle bays.

Junge evaluated adult salmon passage success by comparing ladder passage counts associated with various spill patterns. The spill patterns he developed that appeared best for adult passage conflict with what is thought today to be best for juvenile passage (high shoreline velocities), because Junge kept near-shore velocities low to facilitate adult migration and passage into fishway entrances located along shorelines.

Smolt residence time in spillway tailraces is likely influenced by spill volume and pattern. High spill volume and water velocity push water, and presumably juvenile salmonids, out of the immediate tailrace, and help redistribute piscivorous predators (northern pikeminnow) away from the immediate spillway tailrace, reducing potential predation opportunities (Faler et al. 1988). Shively et al. (1996) found that ambient river flow velocities of at least 1 ms^{-1} were necessary to keep northern pikeminnow from holding in areas near bypass outfalls, and that the degree by which water velocity eliminated northern pikeminnow holding increased as outfall distance from shore and water depth increased.

Hansel et al. (1993) found that hydraulic cover such as eddies and backwaters at velocities below this threshold were preferred northern pikeminnow feeding habitats, particularly when near primary smolt outmigration paths. Spill patterns that facilitate rapid juvenile egress from the spillway stilling basin through the tailrace likely increase juvenile survival. Current spill patterns are developed to increase the survival of juvenile fish through tailraces, by emphasizing minimizing hydraulic cover and maintaining high water velocities near spillway shorelines. To not interfere with daytime adult passage, these juvenile spill patterns are often employed during nighttime hours only (COE 2004).

As information is gained on the use and benefits associated with daytime spill to pass juveniles, greater consideration will likely be given to the use of juvenile spill patterns during the daytime. Spill patterns that attempt to satisfy both adult and juvenile passage criteria during the daytime have been developed for Lower Granite, Ice Harbor, McNary, John Day, The Dalles, and Bonneville Dams using scale hydraulic physical models at the COE Engineering Research and Development Center (ERDC) in Vicksburg, Mississippi. Patterns for each dam for daytime and nighttime periods are developed in coordination with salmon managers and incorporated into the Annual Fish Passage Plan (COE 2004).

Eppard et al. (2000) evaluated tailrace residence times of radio-tagged yearling spring Chinook at Ice Harbor Dam during the spring of 1999. Spill levels during the study were as high as 45% of total river flow during the day and up to 100% at night. Median tailrace residence times were 2.7 , 5.7, and 7.3 minutes for fish that passed through the spillway, juvenile bypass system, and turbines, respectively, during daylight hours. At night during 100% spill, median tailrace residence time was 3 minutes for spillway-passed fish. During the spring of 2001 at Ice Harbor Dam (low flow, no spill), median tailrace residence times were 9.0 and 14.7 minutes for radio-tagged yearling spring Chinook that passed the through the bypass system and a turbine unit, respectively.

At McNary Dam in the spring of 2002, radiotelemetry analysis of yearling Chinook salmon passage identified a statistically significant difference between median tailrace egress times for spilled fish (5.5 minutes) and bypassed fish (10.2 minutes) (Axel et al. in prep. a). While there was also a significant difference in tailrace egress times between fish bypassed

during the day (9.4 minutes) and night (11.4 minutes), there was no day/night difference observed for fish passing through spill. Within the spillway, slightly longer tailrace egress times were observed for fish passing through the end bays.

At McNary Dam in the spring of 2003, Axel et al. (in prep. b) also observed a significant difference between median tailrace egress times for radio-tagged yearling Chinook salmon passing through the spillway (5.8 minutes) and the bypass system (7.7 minutes). No diel differences in egress times were observed for spilled fish, but as in 2002, longer egress times were associated with fish passing through the end spillbays.

At John Day Dam, Liedtke et al. (1999) observed good juvenile egress through the spillway tailrace with a new spill pattern; marked yearling and subyearling fish passed through the first 0.7 km of tailrace in 5–10 minutes. They also observed slower passage times and higher predation rates on fish that passed through end spillbays.

In 2000 Duran et al. (2002) monitored tailrace egress at John Day Dam and concluded that spillway passage generally resulted in more rapid tailrace egress than bypass passage. Residence times for spilled fish were similar for radio-tagged fish released at daytime 30% spill and nighttime 60% spill. Residence times were longer for bypassed fish during the 60% nighttime than the 30% daytime spill. Spill-passed yearling Chinook exited the tailrace within about 10 minutes for all spill conditions and bypassed yearling Chinook took 6 minutes longer to exit tailrace (15 vs. 9 minutes) under the nighttime 60% spill treatment. Spill-passed steelhead tailrace exit times and relationships were similar, although the test was compromised by lower than requested spill percentages. South-spillway-passed fish took longer than north-spillway (12–14 vs. 7–8 minutes). For subyearling Chinook salmon, residence times for spill-passed fish were generally lower during 60% nighttime spill than for 30% daytime spill. North-spillway fish had the lowest residence times. Subyearlings released at the bypass had significantly longer tailrace egress times during 60% nighttime spill than during 30% daytime spill (74 vs. 17 minutes).

At The Dalles Dam in 2000, Allen et al. (2001) reported that radio-tagged yearling and subyearling Chinook salmon that passed through the powerhouse in the spring and summer had double the tailrace residence times of north-spillway-passed or control fish. Sluiceway and south-spillway fish tailrace-residence times were similar to powerhouse times.

Daily Spill Timing

The early spill programs based daily spill timing on hourly monitoring of smolt abundance at projects where this information was available. If hourly information was not available, daily spill timing was determined prior to each season and was based on the best available diel passage information. This diel information was obtained from smolt monitoring or research activities conducted primarily at powerhouses, rather than spillways, and it was common to apply diel information collected at one dam to another. However, more recent studies have shown that diel passage timing through spillways is different than through powerhouses. For example, while smolt monitoring information suggests 60–90% of the daily powerhouse passage at John Day and Lower Monumental Dams occurs from 1800 to 0600 hours (Ransom and McFadden 1987, Martinson et al. 1997), hydroacoustic studies indicate that both

day and night passage of juvenile migrants through the spillway is high at both dams during involuntary daytime spill (BioSonics Inc. 1999a, 1999b).

Observations at The Dalles and Bonneville Dams also indicate that spillway passage occurs at fairly constant rates, day and night, if daytime spill is provided (BioSonics Inc. 1997, Hensleigh et al. 1998). This suggests that juvenile migrations may be delayed if spill is managed based on powerhouse diel passage patterns. Studies conducted at John Day Dam in years with little daytime spill found that fish often milled in front of the dam when they arrived during the day and passed the dam at night, most often through the powerhouse (Giorgi et al. 1985, Sheer et al. 1997). Sheer et al. (1997) found that fish passing The Dalles Dam during 24-hour spill delayed very little, and most (85%) passed through the spillway. Hensleigh et al. (1999) and Liedtke et al. (1999) found a similar response at John Day Dam, where radio-tagged yearling and subyearling Chinook salmon and steelhead had relatively short forebay residence times and high (50–75%) spillway passage rates during years where high flows caused high levels of daytime spill.

In radiotelemetry studies conducted in the spring and summer of 1999 at John Day Dam during two day/night spill treatments (0/60% and 30/60%), Hansel et al. (1999a) reported forebay residence time was significantly shorter for yearling Chinook salmon passing during the daytime spill treatment (0.8 vs. 8.5 hours), and a similar result was observed for smaller (<200 mm) steelhead. With subyearling Chinook, there appeared to a greater delay for fish arriving during the day with spill than at night with spill (2.5 vs. 0.6 hours) during similar spill levels, suggesting a day vs. night effect that is independent of percent spill. Also, during the 12-hour spill treatment, 55% of the fish arriving during the day delayed to pass at night.

For similar daytime/nighttime spill treatments (0/60% and 30/60%) at John Day Dam in 2000, Beeman et al. (2001) reported that median forebay residence time was longer for radio-tagged yearling Chinook and steelhead when no daytime spill was provided, but this was only statistically significant for Chinook (4.6–12.3 hours without daytime spill, 1.4–3.2 hours with daytime spill). The researchers concluded that the use of 24-hour spill decreases forebay residence times of fish arriving during the day, increases spill efficiency of yearling Chinook, but does not significantly alter fish passage efficiency from values obtained during 12-hour spill. For radio-tagged subyearling Chinook, Beeman et al. (2002a) reported fish arriving at John Day Dam during the 24-hour treatment had shorter forebay residence times with less variability between day and night arrival periods than those arriving during the 12-hour treatment. Median residence time during the 24-hour spill for day arrival was 1 hour (range 0.7–1.6 hours) and for night arrival was 1.6 hours (1.2–2.6 hours). Median times for the 12-hour spill for day arrival was 4.8 hours (2.4–6.6 hours) and ranged from 0.8 to 4.2 hours for night-arriving fish. Project fish passage efficiency was significantly higher for the 24-hour spill treatment (91%) than for the 12-hour treatment (79%).

For two daytime/nighttime spill treatments (0/54% and 30/30%) at John Day Dam during the spring of 2002, Beeman et al. (2002b) reported that although median forebay residence times for radio-tagged salmonids tended to decrease with increased percent spill levels, median residence times for the 12- and 24-hour treatments overall were similar for yearling spring Chinook (1.7 vs. 1.6 hours) and wild juvenile steelhead (6.8 vs. 7.0 hours).

Seasonal Spill Timing

Historically, seasonal spill timing has been based on juvenile fish abundance. Early spill programs relied primarily on preseason planning dates and in-season estimates of cumulative fish passage as a trigger for the beginning and end of the spill season at each project. These programs were managed to provide spill for the middle 80% of the juvenile outmigration. These percentages were applied separately to the spring and summer migration periods and were based on actual fish sampling at each project via the smolt monitoring program (FPC 1995).

The NOAA Fisheries Service 2000 BiOp proposed that the actual dates of spill be determined annually by the regional Technical Management Team and based on the best available monitoring and evaluation data concerning project passage, spill, and system survival research. The 2000 BiOp established Snake River spill planning dates as 3 April to 20 June and 21 June to 31 August, and lower Columbia River dates as 10 April to 30 June and 1 July to 31 August.

Summer Spill

The efficacy of continuing the NOAA Fisheries Service 2000 BiOp summer spill program during late July and August has recently been questioned for a variety of reasons and discussed in various regional forums. Because the majority of ESA-listed Snake River fall Chinook salmon are currently transported around FCRPS dams, some in the region view the benefits of late summer spill as being small and easily replaced by alternative biological offsets. The spill reduction would increase revenue from power generation and has led to a great deal of discussion regarding the cost effectiveness of the spill program. Others in the region believe that the adverse biological impacts of reducing spill have been greatly underestimated, the potential benefits of alternative biological offsets have been greatly overestimated, and the biological benefits of spill throughout the summer are clear and of utmost importance for both listed and nonlisted populations of Columbia River Basin salmonids. Therefore it is important that we address what is known regarding summer spill passage.

In general, fewer spill studies have been conducted during the summer than during the spring migration period, as summer spill survival tests present several challenges to researchers. These include issues associated with handling fish during warm water temperatures, large sample sizes required to detect small differences in survival due to low detection probability, whether test fish represent the general population, and the availability of test fish for marking. Since 1997, summer spill studies have been conducted using uniquely identified fish that were PIT, radio, or balloon tagged (Table 1). Here we summarize recent survival information for passage through specific spillways and survival through river reaches during periods of summer spill based on analyses presented in Williams et al. (2005).

Dam-specific spillway passage survival has been estimated during the summer under contemporary spill conditions at Ice Harbor, John Day, and The Dalles Dams. Based on PIT tags, estimated mean survival through the Ice Harbor Dam spillway under the spill pattern called for in 2000 BiOp was 0.89 in 2000 and 2002 (Eppard et al. 2002a, 2002b). This compares to an estimated mean survival through the spillway of 0.96 in 2003 under a "bulk" spill pattern, where higher flow was concentrated in fewer spillbays (Absolon et al. 2003a). At John Day Dam in

2003, Counihan et al. (in prep. b) used radiotelemetry and estimated subyearling Chinook salmon survival through the spillway was 0.901 (95% CI, 0.877–0.922) under 0% daytime and 60% nighttime spill, and 0.955 (95% CI, 0.938–0.970) under 30% spill throughout the 24-hour period. In 2002, Counihan et al. (in prep. a) reported that estimated survival through the spillway was 0.985 (95% CI, 0.934–1.023) under 0% daytime and 54% nighttime spill, and 1.003 (95% CI, 0.983–1.078) under 30% spill throughout the 24-hour period. At The Dalles Dam, estimated mean survival ranged from 0.75 to 1.00 based on radiotelemetry and PIT tags, and was roughly 0.93 using balloon tags (Table 1). Some of these estimates are lower than expected. Potential explanations are discussed above in the Causal Mechanisms subsection on page 13.

From 1999 through 2002, NOAA Fisheries Service collected, PIT tagged, and released river-run subyearling Chinook salmon (mostly wild fish from the Hanford Reach) at McNary Dam to estimate survival below McNary Dam. Study designs and release sites varied from year to year, but a series of releases into the tailrace of McNary Dam was included each year. We calculated indices of exposure to the following river condition variables: river discharge ("flow")(kcfs), amount of flow over spillways (kcfs), percentage of flow that passed over spillways, water temperature (°C), and water clarity (Secchi disk reading). We used various graphical methods, pair-wise product-moment correlations, and simple and multiple linear regression modeling to explore relations among indices of exposure to selected environmental factors and survival and travel time.

Because of concomitant temporal trends in river conditions, exposure indices for release groups of PIT-tagged fish were generally highly correlated with each other and with release date. Correlation of such magnitude among predictor variables generally makes it difficult for multivariate statistical methods to distinguish the relative importance of the predictors' influence on the response variable. Nonetheless, we explored bivariate patterns, and used multivariate methods to shed light on relations. Samples were of sufficient size that correlations with relatively little explanatory power (e.g., $r^2 = 0.16$) were statistically significant ($P < 0.05$). One response to this is to lower the level required to declare a correlation significant. Our approach, however, was simply to focus on the amount of variability in the response variable that is "explained" by variability in the predictor (i.e., the r^2 value).

The data available from river-run subyearling Chinook salmon from 1999 through 2002 were insufficient to make definitive statements regarding potentially complex dynamics among travel time, survival, and environmental (river) conditions between McNary Dam and John Day Dam. For example, strong correlations among river conditions (flow, spill, temperature) made it impossible to determine which variable had the strongest influence on response variables. Estimated survival was significantly correlated to flow, temperature, and percent spill at McNary Dam, but not strongly with percent spill at John Day Dam. The correlation with spill percentage at McNary Dam was probably due to common correlation with flow, as spill at McNary Dam probably had little influence on survival because study fish were collected from the powerhouse at McNary Dam and released into the tailrace. Within most years there was a relatively narrow range in river condition values, with little overlap between years in some cases. Therefore, it was not possible to separate processes and relations that occur within migration seasons from potential annual differences due to generalized "year effects."

The relationships identified among spill, flow, temperature, travel time, and survival of subyearling fall Chinook salmon between McNary and John Day dams are consistent with those found in the Snake River (Connor et al. 2003a, Connor et al. 2003b, Smith et al. 2003). We note, however, that weekly estimates of juvenile fish survival between McNary Dam and John Day Dam tailraces ranged from 0.52 to 0.66 in 2001 and were narrower in range and lower than in other years; weekly estimates of survival ranged from 0.45 to 0.96 in 2002, from 0.55 to 2.00 in 2000, and from 0.75 to 1.03 in 1999. There was no spill at John Day Dam in 2001 during the period when the test fish used in this analysis passed the dam. In addition, river flow was lower and water temperatures were higher in 2001 than the other years. While we cannot identify the lack of spill in 2001 as the sole cause of the lower and narrower range in survival observed that year, based on the high estimates of survival through the John Day Dam spillway under 30% spill reported above, it is likely that lack of spill that year was at least partially responsible for the observed results.

We also found that average survival was nearly constant for water temperatures below 19.3°C, and nearly constant but considerably lower for water temperatures above 20.6°C. In laboratory studies, Marine and Cech (2004) found that exposure of Sacramento River fall Chinook salmon fry to water temperatures above 20°C resulted in decreased growth, increased osmoregulatory impairment, and increased vulnerability to predation. That temperature value was nearly the same temperature threshold we found to affect survival of subyearling fall Chinook salmon migrants in the river reach from McNary Dam to John Day Dam. Thus a temperature threshold may exist above which mortality in this reach is increased.

Dissolved Gas/Gas Bubble Disease

Dissolved Gas Standards

Recommended total dissolved gas standards for surface waterways were developed by the U.S. Environmental Protection Agency under the authority of the 1977 Clean Water Act amendment to the Federal Water Pollution Control Act of 1948. These standards were subsequently adopted by state environmental quality agencies; each Pacific Northwest state has slightly different statutes.

In the case of the mainstem Snake and Columbia rivers, however, a common standard of 110% total dissolved gas supersaturation (TDGS) was adopted. Each state has provisions for a short-term waiver of the standard. In 1994 NOAA Fisheries Service first applied for and was granted total dissolved gas waivers from Washington and Oregon. These waivers allowed total dissolved gas levels up to 115% in the forebay and 120% in the tailrace of each FCRPS dam. The 1995 FCRPS BiOp included these standard waivers in the spill program requirements. Since 1994, NOAA Fisheries Service has requested and received annual water quality waivers from each state.

In 1997, the state of Washington replaced the dissolved gas standard annual waiver with a "fish passage exemption" in the Water Quality Standards for Surface Waters of the State of Washington (Chapter 173-201A-060 (4)(b)), which mandates allowable dissolved gas limits identical to those in the NOAA Fisheries Service BiOp, with the addition of a 1-hour maximum of 125%. This exemption specifically states that it is intended for "spillage for fish passage" and

is "temporary." In qualifying for this exemption, NOAA Fisheries Service has met annual reporting and monitoring requirements.

Table 2 presents the estimated spill volumes which result in approximately 120% TDGS at tailwater gas monitors or 115% TDGS at forebay monitors. Spill volumes vary with forebay TDGS, powerhouse flow volumes, and spillway tailwater elevations.

Dissolved Gas Supersaturation

Soon after Bonneville Dam was completed in 1938, dead adult salmon were observed downstream by fishermen (Merrell et al. 1971). Unreconciled losses of adult salmon continued through the 1950s. Westgard (1964) first documented gas bubble disease (GBD) on the Columbia River in the McNary Dam spawning channel. Dissolved gas supersaturation was documented and unequivocally associated with spill at mainstem Columbia River dams in the late 1960s by Ebel (1969). By inference, Merrell et al. (1971) attributed earlier observed adult salmon losses to supersaturation. In 1967 run-of-the-river adult and juvenile salmon were observed with GBD and holding tests linked high dissolved gas and increases in temperature with mortality of juveniles (Ebel 1969).

Adult salmon mortality was documented in conjunction with high levels of supersaturation (125–135%) from Wells to Chief Joseph Dams from 1965 through 1969 (Meekin and Allen 1974). In 1968, when John Day Dam was completed but before turbines were in operation, dead adult Chinook and sockeye salmon (*O. nerka*) were found floating downstream and live fish with signs of GBD were captured (Beiningen and Ebel 1970). From 1968 to 1975, GBD in high-flow years contributed to high mortalities of juvenile salmonids migrating from the Snake River (Ebel et al. 1975).

Many studies on GBD and its effects on salmonids were conducted in an attempt to define the threat in the mainstem Columbia and Snake rivers. The severity of GBD was dependent upon species, life stage, body size, level of total dissolved gas, duration of exposure, water temperature, general physical condition of the fish, and swimming depth (Ebel et al. 1975). A thorough review of the literature on dissolved gas supersaturation and of recorded cases of GBD was compiled by Weitkamp and Katz (1980) and updated by Fidler and Miller (1993). Despite numerous studies, there were still questions regarding the TDGS that migrating salmonids can safely tolerate and how to evaluate impacts. In 1994 NOAA Fisheries Service and BPA convened a panel of experts to review dissolved gas conditions on the river and assess impacts to salmonids from GBD resulting from voluntary spill (GBD Panel 1996). The panel concluded that not enough was known to accept the hypothesis that aquatic organisms in the Columbia and Snake rivers were not impacted by dissolved gas from voluntary spill. This conclusion initiated regional research and monitoring efforts to increase the knowledge base needed for spill management decisions. These efforts were administered by the Gas Bubble Disease Technical Work Group (now called the Water Quality Team) and was originally chaired jointly by NMFS and the Environmental Protection Agency. A comprehensive review of total dissolved gas (TDG) levels and risk assessment associated with spill and can be found in Appendix E of the NMFS 2000 FCRPS BiOp.

Table 2.   Estimated spill caps.

Project	120% TDG Limit (kcfs)[a]	Practical TDG Spill Limit (kcfs)[b]
Lower Granite	45–50	45–50
Little Goose	60	40[c]
Lower Monumental	25–40	35
Ice Harbor	95–105	95–105
McNary	160–185	160–185[d]
John Day	85–160	150–160[e]
The Dalles	95–200	150
Bonneville	145–150[f]	120–150[g]

[a] Spill cap based on observed gas levels at the tailwater monitor in 2002 and 2003.
[b] Spill cap based on observed spill flows limited by the tailwater monitor or the next forebay monitor in 2002 and 2003.
[c] Because of entrainment of powerhouse flow into the spillway, the Little Goose spill levels are normally constrained by TDG levels in the forebay of Lower Monumental Dam.
[d] The McNary spill limit is an estimate because a new spill pattern will be implemented in 2004.
[e] The John Day limit depends on how the spill schedule is implemented as total river discharge increases. The spill pattern changes from concentrated to flat at about 108 kcfs spill. TDG levels actually decrease above this point then increase with increasing spill. We are assuming the operators will avoid the 108 kcfs limitation.
[f] The Bonneville spill limit is based on the new spillway tailrace monitor.
[g] The TDG spill limit at Bonneville is dependent on the monitor site used for management. The tailwater site provides consistent TDG results; however, the Camas site is highly variable due to local environmental conditions and may, at times, further limit spill levels at Bonneville Dam.

GBD Monitoring and Research

Beginning in 1994, the annual TDGS waivers from the states were granted with the stipulations that proper monitoring would be conducted and results of research and monitoring would demonstrate minimal effects from GBD at temporarily increased levels of 115% in reservoirs and 120% in tailraces at dams. A formal plan was developed by NMFS (1997) to monitor TDGS and signs of GBD in the aquatic biota. Critical uncertainties of the monitoring program were identified (Biological Monitoring Inspection Team 1995), gas bubble disease research priorities were developed (NMFS 1996), and thresholds of 15% prevalence and 5% severe GBD signs were set for continued voluntary spill.

Through the 1990s, TDGS monitoring improved. By 1998 timely data distribution by COE included hourly dissolved gas levels and water temperatures from 41 monitoring sites in forebays and tailraces on the mainstem Columbia and Snake rivers (COE 1998b). Quality assurance and control measures, and improved gas measurement technology, increased data precision and provided confidence that fisheries management decisions regarding the threat from TDGS were based on sound data. However, concerns still existed over whether the monitoring sites appropriately represent all conditions experienced by salmonids (ISAB 1999). Additionally, intensive monitoring throughout reservoirs and in tailraces has allowed a series of models to be developed that relate tailwater TDGS to spill, total project release, and forebay TDG for all mainstem dams from Grand Coulee and Lower Granite to Bonneville Dams, as well as from Dworshak Dam (COE 1998a).

Through the 1990s, GBD monitoring of migrating juvenile and adult salmonids was also greatly improved. Numbers of fish, methods, locations, and times for sampling were adjusted to provide a representative network of samples. Quality assurance and control methods met regional requirements, providing confidence that fisheries management decisions were based on the best possible data given the present state of technology.

Uncertainties associated with the GBD monitoring are 1) whether the relationship between smolt/adult mortality and gas bubbles in fins, gills, and lateral line is known; 2) whether clinical signs change during collection and examination; 3) whether signs in sampled fish represent the river site over the entire 24-hour period; 4) whether samples are taken at representative locations, including those of high risk from GBD; 5) whether sample size is statistically adequate for required confidence limits; 6) whether key signs of GBD and their relative significance are known; and 7) whether the 15% threshold level for GBD prevalence can be tolerated by the juvenile migrant population. Research conducted to address these uncertainties suggests:

1. The relationship between prevalence and severity of GBD signs and mortality of juvenile salmonids is unresolved. Research has produced an equivocal relationship except at consistent 130% TDGS (Mesa et al. 2000a, Backman and Evans 2002).

2. Varying pressures encountered by juvenile salmonids during passage through the collection and bypass systems at dams appears to not substantially alter existing GBD signs nor induce new signs (Absolon et al. 1999). Collection protocols require fish be collected at the dewatering screens in the bypass conduit, held in river water at ambient temperature, and processed within 15 minutes of sampling (Maule et al. 1997). Prevalence of GBD signs in fish collected from bypass systems was somewhat greater than that of fish collected by seines from reservoirs (Backman et al. 1998). The increased GBD signs may be caused by passage delays in front of dams prior to bypass system entry. The ISAB (1999) called for more research to evaluate this difference.

3. Fish samples collected for GBD assessment of smolt monitoring facilities at Snake and Columbia River dams adequately represented fish passing through the smolt bypass systems 24 hours per day (NMFS 1999a).

4. Smolt monitoring sites appeared to provide samples that were generally representative of upstream and downstream locations.

5. Evaluation of sample statistics by Maule et al. (1997) showed that numbers of sampled fish were generally sufficient to document a species-specific GBD prevalence when impacted migrants reached the NOAA Fisheries Service threshold of 15% prevalence or 5% severe signs of GBD. An exception is at McNary Dam, where migrants from the Snake River cannot be differentiated from Columbia River migrants. Here a low prevalence in the total sample could be associated with a large prevalence within the Snake River subpopulation. However, at dissolved gas levels present during voluntary spill, this problem is insignificant.

6. The GBD signs used as a GBD index for smolt monitoring are externally visible subcutaneous emphysema. Research to identify other indicators of GBD such as blood chemistry changes was not pursued. Acoustic assessment of internal gas emboli was determined too expensive and not suitable for use in field locations (Carlson 1995).

Excision of gill lamella was investigated, but thought to be no better an index of GBD impacts than other less invasive methods.

7. Signs of GBD can only be used as a subjective tool. The GBD Panel (1996) stated that when GBD signs are observed in any fish, there should be regional concern about survival. Also, the greater the prevalence and severity of GBD signs, the more concerned managers should be about fish survival.

GBD Impacts

Results of GBD monitoring and research on juvenile salmonids in the 1990s show that incidents of high prevalence and severity of GBD signs and probable related mortality are associated with exceptionally high river flows or exceptional dam operation problems where powerhouse flows were limited. In 1990 a fire at John Day Dam caused 100% spill for an extended period, resulting in downstream TDGS levels greater than 130% and GBD prevalence up to 74% in steelhead and 38% in coho salmon.

In 1993 turbine outages at Lower Granite Dam caused TDGS to increase, resulting in 18% prevalence of GBD in migrants at Little Goose Dam. In 1995 and 1996, turbine outages at Ice Harbor Dam coupled with high river flows caused TDGS to exceed 130% during the spring freshet. PIT-tag interrogation data collected at Snake and Columbia River dams suggested losses of juvenile Chinook salmon coincident with the high levels of TDGS in the 67-km reach from Lower Monumental Dam to the Snake River mouth in 1995 (Cramer 1996a) and 1996 (Cramer 1996b, NMFS 1997). However, no losses of steelhead were measured in that same river reach for fish that prior to release at Little Goose Dam had been experimentally exposed to supersaturation sufficient to produce low-grade mortality (Monk et al. 1997a). Resident fish downstream from Ice Harbor Dam also showed high prevalence and severity of GBD signs in 1995 and 1996 (Schrank et al. 1997, 1998).

High river flows in 1996 and 1997 caused excessive spill at all mainstem dams, and TDGS exceeded 120% (the maximum allowed for voluntary spill) for extended periods throughout most of the lower Snake River and mid- and lower Columbia River. Results of GBD monitoring showed that prevalence of subcutaneous emphysema in juvenile salmonids was minor when TDGS was 120–125%, but was generally 10% or greater when TDGS was higher than 125% (NMFS 1998a). Under high river flow conditions, even when all possible system management actions are taken, TDG often cannot be kept below 120%.

In the early 1990s, river flow was low and often within powerhouse hydraulic capacities. When there was sufficient market for power, voluntary spill for fish passage and TDGS at 120% in tailraces and 115% in forebays could be regulated. Effects of GBD under these conditions appeared benign, signs were minimal or nonexistent, and there was no apparent GBD-related mortality (Maule et al. 1997, ISAB 1999). Even during periods of involuntary spill, GBD impacts appeared to be minor except when TDGS was over 120% (NMFS 1997, NMFS 1998a, NMFS 1999a, Backman and Evans 2002).

Impacts of GBD as measured by visual external signs in river migrants are greater at higher temperatures, particularly when the ambient water temperature of supersaturated water increases (Ebel 1969, Ebel et al. 1971). Steelhead have a lower tolerance to TDGS than other

species, based on Smolt Monitoring Program data (NMFS 1997, 1998a, 1999a) and laboratory tests (Fidler and Miller 1993).

Depth Compensation

Gas solubility increases with increasing pressure. For each meter of depth there is a 10% reduction in the TDG saturation level relative to the surface saturation (Weitkamp and Katz 1980). Dawley et al. (1975) and Weitkamp (1976) reported data suggesting depth compensation occurs with juvenile salmonids as gas solubility calculations would predict. Maule et al. (1997) observed that salmonids may migrate at protective depths and presented results suggesting that the depth of tagged fish would compensate for a surface TDG level of up to approximately 124%.

Employing depth-sensitive radio tags, Beeman et al. (1998, 1999) tracked juvenile salmonids between Ice Harbor and McNary Dams at depths of 1.8–2.5 m with a surface TDG level of 120%, and concluded the median depth of juveniles in the McNary pool was sufficient to protect fish from TDG levels of between 117% and 124%. The authors stated that a voluntary spill program with gas caps of 115% in forebays and 120% in tailraces can be expected to prevent gas bubble disease in juvenile Chinook and pose little threat to the more sensitive steelhead.

Dissolved Gas Abatement

Starting in 1994, a COE Dissolved Gas Abatement Study developed concepts for decreasing TDGS at the eight Snake and Columbia River dams. The COE is pursuing numerous potential structural modifications to spillways and stilling basins for dissolved gas abatement alternatives. While some of these alternatives may be implemented in the future, to date only flow deflectors have been implemented to reduce gas production at the FCRPS projects.

None of the gas reduction alternatives evaluated met biological, operational, and economic criteria, nor did they decrease gas levels to the 110% water quality standard for the 7-day, 10-year discharge events. Whitney et al. (1997) found that under appropriate operating conditions, flow deflectors generally lowered dissolved gas levels downstream from the projects by 10–20% for a given spill flow. Deflectors were installed at Ice Harbor and John Day Dams and have provided gas abatement benefits in the upper portion of this range. Additional deflectors were installed in the end spillbays of Lower Monumental, McNary, and Bonneville Dams. These additional deflectors have further improved the dissolved gas levels below these dams.

Spill Passage Conclusions

1. Estimates of spill efficiency and effectiveness for juvenile salmonids vary by dam, the percentage of river flow spilled, spill pattern, time of day, species (or run), and configuration of the dam. Spill effectiveness is generally between 1.0 and 2.0, but can range considerably higher at certain dams. There is a strong indication from studies at Ice Harbor Dam on the Snake River and The Dalles Dam on the lower Columbia River that, above a certain level of

spill, there is no gain in spill effectiveness and very likely a reduction. However, spill efficiency will continue to increase with the percentage of total river flow spilled.

2. Documented consequences of spill include increased fish passage efficiency (proportion of fish passing through nonturbine routes) and reduced migrational delays through forebays, tailraces, and reservoirs. Forebay residence time for downstream migrating juvenile salmonids is inversely related to spill.

3. Spill has the potential for reducing exposure risk to predation, high water temperatures, and disease. The presence of piscine and avian predators in forebays and detection histories of tagged fish in forebays suggest that delay of juvenile salmonids in dam forebays may lead to reduced survival.

4. Fish survival through spillways can be very high (near 1.00) and is often higher than turbine or bypass system survival when spill passage conditions are optimal. However, survival through spillways with deflectors or shallow stilling basins can be negatively influenced by stilling basin depth and turbulence, hydraulic patterns in the basin, spillbay location within the spillway relative to the spill pattern being used, deflector elevation relative to tailwater elevation and thus total river flow, gate opening, and fish location when passing through the spillbay and under the control gate. Survival through spillways with deflectors or shallow basins can be considerably less than 0.98, a value typically used in hydropower system modeling exercises. Observations and lessons learned from spillway survival studies at The Dalles, Ice Harbor, and Lower Monumental Dams should be applied to other spillways with deflectors to ensure that adequate spillway passage conditions are provided at these locations as well. Examples include Little Goose Dam, where survival may be lower than previously measured under some conditions, and possibly training spill associated with the removable spillway weir at Lower Granite Dam and other weirs that may be installed in the future.

5. Tailrace residence time for juvenile salmonids passing through spillways is influenced by spill volume and pattern. Generally, spillway passage results in more rapid tailrace egress than bypass system passage. Because piscine predators prefer hydraulic cover such as eddies and backwaters, spill patterns that facilitate rapid juvenile salmonid egress from the spillway stilling basin through the tailrace likely increase survival.

6. In general, research results indicate that when daytime spill is provided, spillway passage of juvenile salmonids occurs at fairly constant rates, day and night. Juvenile salmonid migrations may be delayed if spill is managed based on powerhouse diel passage patterns.

7. Relatively few studies have been conducted on subyearling Chinook salmon survival through spillways during the summer migration period under contemporary conditions. Estimates of mean spillway passage survival at Ice Harbor and The Dalles Dams since 1997 using traditional spill patterns ranged from 0.75 to 1.00. However, due to problems associated with stilling basin conditions at these dams, this wide range in spillway survival estimates may not represent other Snake and Columbia River dams. For example, estimated mean survival through the spillway for John Day Dam in 2002 and 2003 ranged from 0.96 to greater than 1.00 when 30% of river flow was spilled throughout the 24-hour period. In addition, estimated mean survival through the Ice Harbor Dam spillway was 0.96 in 2003 under a "bulk" spill pattern, where higher flow was concentrated in fewer spillbays.

8. Relatively few estimates of survival have been made through river reaches during periods of summer spill. The available information from the lower Columbia River as to whether summer spill improved reach survival was insufficient to make definitive statements regarding the relationships among travel time, survival, and environmental (river) conditions between McNary and John Day Dams. Strong correlations among flow, spill, and temperature made it difficult to predict the effect of any variable independently and impossible to determine which had the strongest influence on survival. This is also consistent with data on subyearling fall Chinook salmon in the Snake River. While we cannot identify the lack of spill in 2001 as the sole cause of the lower and narrower range in survival through the reach that year, it is likely that lack of spill was at least partially responsible for the observed results. We did find evidence that a temperature threshold of 20.6°C may exist in the reach from McNary Dam to John Day Dam, above which mortality increases.

9. Results of GBD monitoring and research on juvenile salmonids indicate that incidents of high prevalence and severity of GBD signs and probable related mortality are associated with exceptionally high river flows or exceptional dam operation problems where powerhouse flows are limited. Under TDG cap conditions of 115% (forebay) and 120% (tailrace), signs of GBD are minimal or nonexistent, with no apparent GBD-related mortality.

Juvenile Salmon Passage Through Mechanical Screen Bypass Systems

Background

In response to concerns for the mortality additional dams were expected to impart on juvenile anadromous salmonids during their seaward migrations in the late 1960s, NOAA Fisheries Service (then the Bureau of Commercial Fisheries) began to investigate means to mitigate the anticipated impacts. The development of systems to collect downstream migrants before they passed through turbines and the subsequent transport of the collected smolts around the remaining dams were two of the focal areas of original research during the earlier years, as they are currently. The original intent of the mechanical bypass systems then was to provide a method of collecting large numbers of downstream migrants at upstream dams for transportation to avoid the excessively high passage-related mortality of that period.

The contemporary mechanical screen bypass systems were developed over the next 30 years. The systems consist of a submersible fish screen within each turbine intake (for each turbine, there are three intakes and associated gatewells) that diverts migrants upward into a gatewell (Figure 2). Once in the gatewell, a vertical barrier screen (VBS) concentrates and further guides the fish into the upper gatewell area where they pass through a submerged orifice and into a collection channel that travels the length of the powerhouse. The channel conveys fish and orifice flow from all gatewells directly to the river, or to dewatering facilities that reduce flow to approximately 30 cfs. This reduced flow can then be routed directly to the river or to secondary dewatering/separation facilities for subsequent separation, sampling, holding (for delayed truck- or barge-loading), or direct loading onto barges.

Smolt monitoring facilities are located within key bypass systems and allow researchers to estimate species composition, fish condition, run timing, and passage indices. PIT-tagged fish can also be detected at these facilities, with time and date of passage noted and fish diverted for further evaluation, if desired. Monitoring at these facilities is continuous within the season and requires that fish be routed through the fish separators and sampling facilities.

At other dams, such as Ice Harbor Dam, monitoring is periodic and is done to determine if mechanical problems are causing poor fish condition. Based on 2001–2002 modifications at McNary Dam, secondary dewatering, separation, and sampling facilities can be bypassed completely by shunting collected fish to the dam's tailrace through a full-flow bypass pipe with PIT-tag detection capabilities. Similar systems are scheduled for installation at Ice Harbor and Lower Monumental Dams prior to the 2005 migration season.
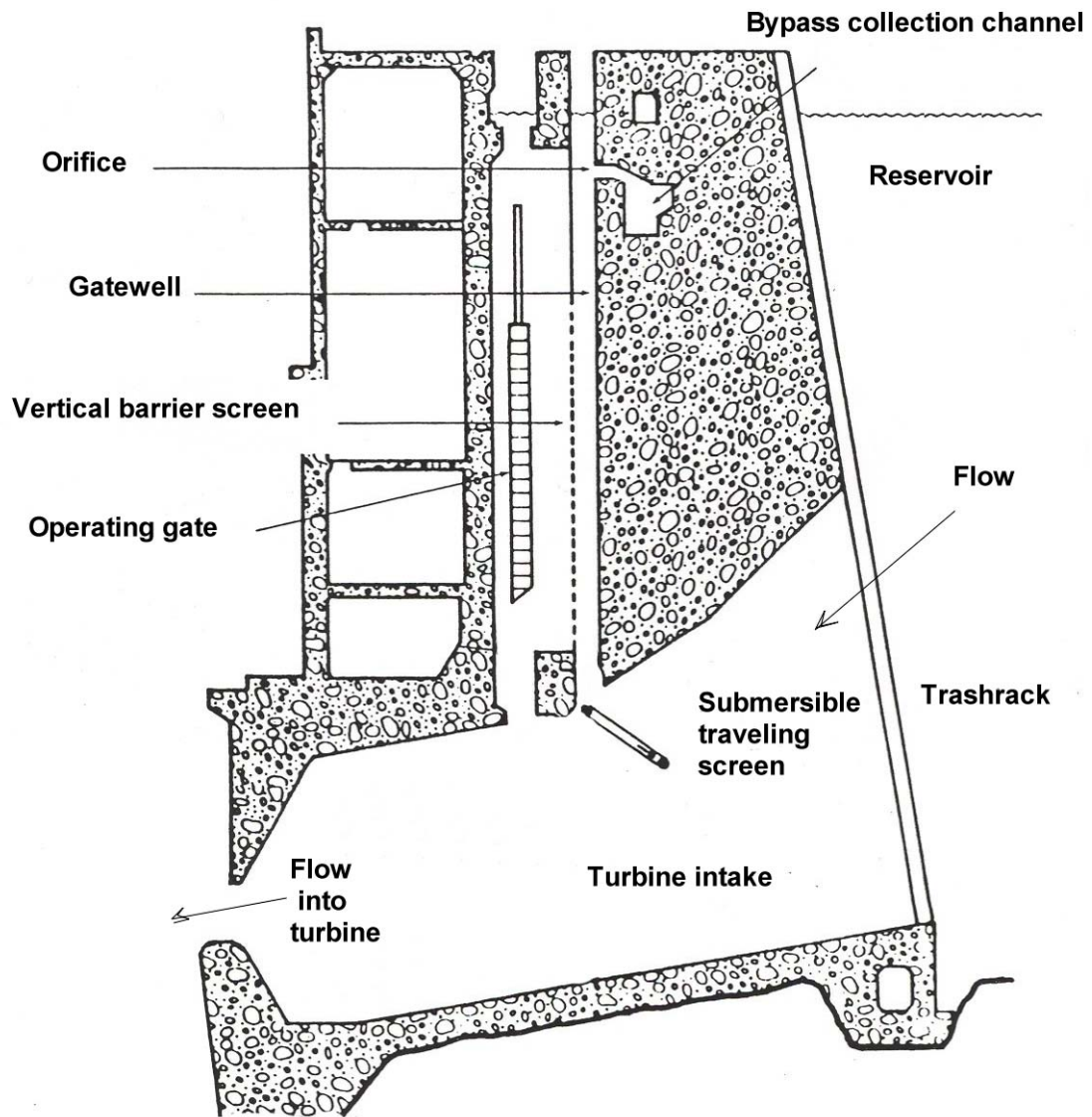
Figure 2.   Cross section of the mechanical screen bypass system at Lower Monumental Dam.

Two submersible fish-screen designs are now in use to guide fish away from turbine intakes and into juvenile bypass systems at Columbia River hydroelectric dams: a submersible traveling screen (STS) and an extended submersible bar screen (ESBS). The STSs utilize a monofilament mesh screen that rotates around large rollers at the top and bottom of the screen. The screen is rotated periodically to allow flow passing through the screen to flush the mesh surface clean of debris. STSs are currently installed at Lower Monumental, Ice Harbor, John Day, and Bonneville Dams. ESBSs are made of a fixed wedgewire screen material and have a brush sweep that is activated periodically to remove debris from the face of the screen. ESBSs are currently installed at Lower Granite, Little Goose, and McNary Dams. The Dalles Dam does not have a mechanical screen juvenile bypass system.

Mechanical screen bypass system design criteria are described in NMFS (2004), COE bypass system design memorandums (COE 1995a, 1996a), the COE Fish Passage Plan (COE 2004), and an intake design guidelines manual (ASCE 1995). NOAA Fisheries Service guidelines for locating and designing bypass outfalls are presented in NMFS (2004). Here we present essential fish passage information related to the development and operation of these systems.

Fish Guidance Efficiency

Definition and Methods

Fish guidance efficiency (FGE) is the proportion of smolts passing through turbine intakes that are guided upwards into gatewells by the submersible fish screens. FGE is calculated as the number of guided fish divided by the total number of fish (guided plus unguided) passing through a turbine intake during a period of measurement (Brege et al. 1992).

FGE estimates vary with sampling method, species, rearing history, stock, fish condition (disease and smoltification), project, time of day, day, season, turbine unit, fyke-net location, environmental conditions, and project operations. This spatial and temporal variability is likely related to complex interactions between biological and physical factors, such as the arrival of different stocks at a dam throughout the season. Williams et al. (1996) attempted to clarify some of the mechanisms affecting FGE. However, FGE data based on fyke-net estimates was generally limited to studies designed to compare FGE of existing versus modified (prototype) intake and vertical barrier screens. These typically involved different hydraulic conditions and little could be concluded. Gessel et al. (1991) suggest that factors such as water temperature, turbidity, flow, photoperiod, physiological development, and predation by northern pikeminnow may affect FGE.

There are four methods of estimating FGE: 1) intake fyke nets and gatewell dip baskets, 2) hydroacoustics, 3) probability of detecting PIT tags, and 4) radiotelemetry. Early FGE studies utilized fyke-net frames beneath STSs to collect unguided fish in intakes (Krcma et al. 1986), and dip baskets (Swan et al. 1979) to collect guided fish in gatewells. Beginning in 1993, streamlined fyke-net frames and nets were designed to test extended screens and placed in gatewell slots located downstream from the guidance screen (Brege et al. 1994). The newly designed fyke nets and frame and placement downstream from the guidance screen minimized water resistance and possible FGE bias, which may have previously resulted from larger fish

avoiding fyke collection. Gatewell dip-net recapture efficiency tests with yearling and subyearling Chinook salmon produced recapture efficiencies of 95–100% (McComas et al. 1994, Brege et al. 1997a, Brege et al. 1997b, Brege et al. 1998). More recently, FGE estimates have been obtained from the probability of detecting PIT-tagged fish (Smith 1997, Krasnow 1998, Anderson et al. 1998), and radiotelemetry (Adams et al. 1998a) and hydroacoustic (Ploskey and Carlson 1999) methods.

Differences exist among the FGE estimates derived from these methodologies, and each method has its advantages, disadvantages, and critical assumptions. For example, a strength of the fyke-net method is that it produces FGE estimates by individual species or life history type. On the other hand, the method generally measures FGE in only one gatewell of only one or two turbine units and for only relatively brief diel periods. The data tend to be highly variable and may not accurately represent even daily, much less seasonal, FGE over entire powerhouses. The method is probably better suited for comparing FGE during direct testing of various guidance technologies and configurations, with the possible exception of head-to-head comparisons between STSs and ESBSs. Recent tests of ESBSs at John Day Dam in 1999 suggested that gatewell dipping might not always recover all of the fish guided into a gatewell equipped with an ESBS (Brege et al. 2001). In this study, it was estimated that 20% of the guided fish were gatewell mortalities, and were not recovered by gatewell dipnetting because they had dropped out of the test gatewell prior to the commencement of dipnetting.

On the other hand, Ploskey and Carlson (1999) previously found strong seasonal correlations between counts derived from hydroacoustic and fyke-net methods for both guided and unguided fish at John Day Dam. However, hydroacoustics significantly underestimated both guided and unguided fish passage relative to the fyke-and-dip-netting method. The two methods produced similar FGE estimates, but hydroacoustics undercounted both groups nearly equally relative to the netting method. In addition, hydroacoustics does not provide species-specific estimates of FGE, can count nonsalmonid species and debris, and passage routes are generally not sampled completely. Hydroacoustic can, however, provide continual fish count estimates over long time periods.

The probability-of-detecting-PIT-tags method measures FGE of the entire powerhouse by species, but results are most reliable when groups of PIT-tagged fish pass during periods of near 0% spill. When groups pass during periods of relatively high spill, the method produces results dependent on the questionable assumption that spill index and detection probability have the same linear relationship throughout the range of observed spill values (Smith 1997).

Radiotelemetry has also been used to estimate FGE. Radio tags are implanted either gastrically or by surgery in fish that are generally larger than the population they represent. Both Adams et al. (1998a, 1998b) and Martinelli et al. (1998) concluded that surgical implantation was the preferred method for radio tags in most studies, though gastric implantation could be used in studies of short duration. Hockersmith et al. (2003) indicated that both tagging techniques would provide data comparable in quality to that derived from PIT tags in studies lasting 6 days or less. Radio-tagged fish are generally released well upstream of a particular dam and are therefore normally distributed relative to other fish by the time they arrive at the dam.

Williams et al. (1996) compared the probability of PIT-tag detection with fyke-net estimates of FGE at McNary Dam and suggested a pressure field created by the fyke nets located under the STSs biased yearling Chinook FGE estimates upward. PATH (1998) developed a correction factor to adjust STS FGE estimates based on fyke-net position to improve the accuracy of the PATH retrospective model analysis for yearling Chinook salmon. The correction factor was the ratio of FGEs derived at McNary Dam with the nets in the downstream slot compared to the upstream slot (i.e., directly beneath the STS). Krasnow (1998) estimated the correction factor to be 0.82. The validity of applying the FGE correction factor for yearling Chinook salmon at McNary Dam to other intakes and species is unknown.

For example, McNary Dam has a large, low-velocity intake compared to most other powerhouse intakes. Monk et al. (1999a) acknowledged the PATH FGE adjustment but chose not to apply it to the data for Bonneville Dam Second Powerhouse. In their view, the correction factor was not considered applicable to STS FGE at the Second Powerhouse because the intakes at McNary Dam and Bonneville Dam Second Powerhouse are hydraulically different, nor would the correction have affected the conclusions of their analysis.

Estimations of FGE are inherently variable, regardless of the method used. This inherent variability makes it difficult to develop accurate and precise single-point estimates of FGE, and to use a typical scientific format that would express FGE as a mean with a standard deviation. Even with this variability, single-point estimates of FGE are often used for modeling or management purposes.

Results

Table 3 presents values commonly used by PATH (Anderson et al. 1998) and NOAA Fisheries Service (Krasnow 1998) in modeling analyses. They reviewed the available data and developed point estimates of FGE for each project and certain species and rearing history.

NOAA Fisheries Service developed a spreadsheet model (SIMPAS) to evaluate benefits to fish survival associated with various FCRPS alternatives. FGE values used in SIMPAS differ slightly from Krasnow (1998). No single estimate of FGE by species and dam is universally accepted, and others in addition to those presented in Table 4 have been developed. To characterize the variability in the data, Table 4 also presents ranges in the actual daily FGEs measured through various sampling methodologies. Point estimates of FGE that are used to model the hydropower system should generally fall within these ranges.

Project operations can also affect FGE variability. For example, at Lower Granite Dam in 2002, Plumb et al. (2003) radio-tagged hatchery yearling Chinook salmon and hatchery and wild steelhead to estimate passage metrics when the removable spillway weir (RSW) was either on or off. Operation of the RSW affected FGE differently for the three groups of fish. For hatchery yearling Chinook salmon, total powerhouse FGE was 69% regardless of RSW operation. However, for hatchery steelhead, FGE dropped from 90% when the RSW was operating to 79% when it was not. The opposite trend occurred for wild steelhead, with the 77% FGE measured when the RSW was operating increasing to 83% when it was not. For the entire powerhouse, seasonal FGE was 68% for yearling Chinook salmon, 87% for hatchery steelhead, and 75% for wild steelhead. These values are similar to those reported for yearling Chinook

salmon and steelhead at this dam in Table 3. However, during these studies, some components of the prototype surface bypass collector were still in place, which may have affected hydraulic conditions and study results.

During 2003 Plumb et al. (2004) again radio tagged the same groups of fish at Lower Granite Dam to estimate passage metrics with the RSW either on or with the dam spilling to the gas cap and with all surface bypass components removed from the forebay. In this study, seasonal FGEs were 82%, 89%, and 96% for yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively, and are higher than those reported in Table 3, especially for wild steelhead. During the RSW treatment, FGEs were 80%, 88%, and 98% for yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively. During the gas cap spill treatment, FGEs were slightly higher for yearling Chinook salmon and hatchery steelhead, but slightly lower for wild steelhead, with measurements of 85%, 90%, and 95% FGE, respectively.

Despite the variability discussed above, some general FGE trends have been observed. Side-by-side fyke-net estimates of FGE with STSs and ESBSs generally indicate that FGE is statistically significantly higher with ESBSs (McComas et al. 1993, Brege et al. 1994). FGE for yearling Chinook salmon at Lower Granite Dam generally increases over time throughout the migration season (Swan et al. 1990). A positive correlation was found between FGE and the level of smolt development exhibited by the migrant population (Giorgi et al. 1988).

With subyearling Chinook salmon, an opposite trend appears to occur; FGE generally decreases as the season progresses (Brege et al. 1988a, Monk et al. 1999a). In the lower Columbia River, FGE of coho salmon is generally high and similar to steelhead, and sockeye FGE is generally lower than that of all other yearling migrants. FGE has been increased by a variety of adaptations to the basic STS, including extending the length of the screen from 6.1 to 12.2 m (STS vs. ESBS), raising the operating gate to allow more flow up the gatewell, lowering STS screens to open the throat area, and using an inlet flow vane with ESBSs to direct more flow into gatewell slots.

Recent PIT-tag data also indicate that smaller Snake River yearling Chinook salmon and steelhead smolts tend to be selected for guidance into bypass systems by submersible fish screens at higher rates than their larger cohorts, and that smaller individuals are also more prone to guidance multiple times as they migrate downstream through the FCRPS (Zabel et al. in prep., Williams et al. 2005). Zabel and Williams (2002) showed that smaller PIT-tagged fish returned as adults at lower rates than their larger cohorts for some years. Together, these results indicate that mechanical screen bypass systems have, to varying degrees, the potential to be selective for species, life history types, and other attributes such as size. Therefore, system managers should consider dam operations that utilize multiple routes to pass fish around dams to minimize any potential risk of reducing biodiversity.

Table 3. Fish guidance efficiency (FGE) at Columbia and Snake River dams for 1999 configuration.[a]
These values are commonly used by PATH and NOAA Fisheries Service in modeling
exercises.

Site (Screen type)	Species	PATH FGE (%)[b]	NOAA FGE (%)[c]
Lower Granite Dam (ESBS)	Yearling Chinook	78	78
	Subyearling Chinook	-	53
	Steelhead	-	81
Little Goose Dam (ESBS)	Yearling Chinook	82	82
	Subyearling Chinook	-	45
	Steelhead	-	81
Lower Monumental Dam (STS)	Yearling Chinook	61	61
	Subyearling Chinook	-	49
	Steelhead	-	82
Ice Harbor Dam (STS)	Yearling Chinook	71	71
	Subyearling Chinook	-	46
	Steelhead	-	93
McNary Dam (ESBS)	Yearling Chinook	95	95
	Subyearling Chinook	-	62
	Steelhead	-	89
John Day Dam (STS)	Yearling Chinook	67	64
	Subyearling Chinook	-	34
	Steelhead	-	85
The Dalles Dam (None) [d]	Yearling Chinook	46	46
	Subyearling Chinook	-	46
	Steelhead	-	40
Bonneville Dam			
Powerhouse One (STS)	Yearling Chinook	41	38
	Subyearling Chinook	-	16
	Steelhead	-	41
Powerhouse Two (STS)	Yearling Chinook	43	44
	Subyearling Chinook	-	18
	Steelhead	-	48

[a] These estimates likely have range, but that range changes with a number of factors and is not easily estimated.
[b] Based on report to PATH from Anderson et al. (1998).
[c] Based on NMFS sensitivity run #1 (assumes $FGE_{ESBS} > FGE_{STS}$ for wild yearling Chinook salmon).
[d] FGE values for The Dalles Dam are based on passage through the ice and trash sluiceway.

Table 4. Daily estimates of fish guidance efficiency (FGE) for existing conditions. Hydroacoustic estimates are not species- or rearing-history-specific. Estimates come from a variety of sources and include adjustments for fyke-net position to yearling Chinook salmon FGE with STSs.

Project (Screen type)	Species	Fyke net Min	Fyke net Max	Hydro-acoustics Min	Hydro-acoustics Max	Radio-telemetry Min	Radio-telemetry Max
Lower Granite (ESBS)	Yearling Chinook			68[a]	93[a]	56[b]	62[b]
	Steelhead			68[a]	93[a]	84[b,c]	86[b,c]
	Subyearling Chinook					42[b]	53[b]
Little Goose (ESBS)	Yearling Chinook	67[d]	87[d]	17[e]	100[e]		
	Steelhead	88[d]	95[d]	61[e]	100[e]		
	Subyearling Chinook						
Lower Monumental (STS)	Yearling Chinook	43[f]	83[f]				
	Steelhead	58[f]	95[f]				
	Subyearling Chinook	30[g]	40[g]				
Ice Harbor (STS)	Yearling Chinook	53[h]	85[h]			50[i]	100[i]
	Steelhead	65[h]	95[h]				
	Subyearling Chinook						
McNary (ESBS)	Yearling Chinook	70[j]	96[j]	13	100		
	Steelhead	75[j]	97[j]	12	100		
	Subyearling Chinook	45[j]	86[j]				
John Day (STS)	Yearling Chinook	54[k]	78[k]			53[l]	84[l]
	Steelhead	65[k]	87[k]			67[l]	86[l]
	Subyearling Chinook	13[k]	55[k]				
The Dalles	NA						
Bonneville Dam — First Powerhouse	Yearling Chinook	21[m]	65[m]	15[n]	100[n]		
	Steelhead	27[m]	87[m]	15[n]	100[n]		
	Subyearling Chinook	2[m]	43[m]	0[n]	100[n]		
Bonneville Dam — Second Powerhouse	Yearling Chinook	22[m]	65[m]	0[n]	100[n]	29[o]	30[o]
	Steelhead	15[m]	70[m]	0[n]	100[n]	51[o]	52[o]
	Subyearling Chinook	19[m]	62[m]	0[n]	100[n]		

[a] Johnson et al. (1999).
[b] Adams et al (1998c).
[c] Includes range estimates for wild and hatchery steelhead.
[d] Gessel et al. (1995).
[e] Anonymous (1996).
[f] Ledgerwood et al. (1987) and Gessel et al. (1993); range includes FGEs adjusted for fyke-net-frame position per Krasnow (1998).
[g] Ledgerwood et al. (1987).
[h] Brege et al. (1988b); range includes FGEs adjusted for fyke-net-frame position and raised operating gate per Krasnow (1998).
[i] Brad Eppard, NWFSC, Pasco, WA. Pers. commun., February 2000.
[j] McComas et al. (1995).
[k] Krcma et al. (1986) and Brege et al. (1987); range includes yearling migrant FGEs adjusted for fyke-net-frame position per Kransow (1998).
[l] Hansel et al. (1999a).
[m] Bonneville First: Monk et al. (1992a, 1993); Bonneville Second: Monk et al. (1994, 1995).
[n] Ploskey et al. (1998).
[o] Hansel et al. (1999b).

Conclusions

1. There is no single best way to estimate FGE. Each of the four methods used has advantages and disadvantages, and each is better suited to certain applications or conditions than others.

2. Estimates of FGE are inherently variable, regardless of the method used. This variability makes it difficult to develop accurate and precise single-point estimates of FGE with high confidence. Even so, best estimates by dam have been developed. They are reported above in Tables 3 and 4 and are used in regional modeling exercises.

3. Mechanical screen bypass systems have the potential to be selective for species, life history types, and other attributes such as size. Therefore, system managers should consider dam operations that utilize multiple routes to pass fish around dams and thus minimize the risk of reducing biodiversity.

Orifice Passage Efficiency

Definition and Methods

Once downstream migrants have been guided into gatewells by submersible fish screens, they exit into a bypass channel via an underwater orifice. It is important that fish not linger in gatewells for extended periods. Residence in this area can result in a variety of stresses, including those from delay and crowding (particularly in gatewells equipped with STSs), and excessive descaling and injury (particularly in gatewells equipped with ESBSs).

Orifice passage efficiency (OPE) is the percentage of guided juvenile salmonids which exit a gatewell via the orifice in a given time period (usually 24 hours). The most recent estimates of OPE at Lower Granite, Little Goose, McNary, and John Day Dams and at Bonneville Dam First Powerhouse have been conducted by releasing either fin-clipped or PIT-tagged fish into the gatewell with the orifice open. With the fin-clipped fish, any remaining study fish are removed from the gatewell and counted after 24 hours. The percentage that left the gatewell in 24 hours is the OPE. With the PIT-tagged fish, OPE is calculated using the number exiting the gatewell as recorded by PIT-tag detectors in the bypass system.

The regionally accepted minimum level for OPE with STSs installed is 70%. However, because of the increased flows and higher turbulence in gatewells associated with ESBSs, OPE levels approaching 90% are probably more appropriate for gatewells with these guidance devices.

Results

At Columbia and Snake River dams, where OPE has been estimated with either an ESBS or STS in place, estimates have generally been greater than 70% for yearling and subyearling Chinook salmon and steelhead, with the exception of yearling Chinook at McNary Dam (Brege et al. 1998) and steelhead at Lower Monumental Dam (Gessel et al. 1993), both of which had lower OPE estimates with ESBSs and with much greater variability (Table 5).

Table 5. Most recent orifice passage efficiency (OPE) estimates for existing conditions (and prototype testing) at Snake and Columbia River dams.

Project	Species	OPE[a] (%)	Reference
Lower Granite Dam	Yearling Chinook	95	Monk et al. 1997b
ESBS, 25-cm orifice	Steelhead	97	
Little Goose Dam	Yearling Chinook	97	Gessel et al. 1996
ESBS, 25-cm orifice	--	--	--
Lower Monumental Dam[a]	Yearling Chinook	88	Gessel et al. 1993
STS, 30-cm orifice	Steelhead	66	
McNary Dam	Yearling Chinook	69	Brege et al. 1998
ESBS, 30-cm orifice	Subyearling Chinook	79	
John Day Dam (ESBS w/ devices)	Yearling Chinook	99	Brege et al. 2004
Existing STS, 35-cm orifice	Subyearling Chinook	97	
Bonneville Dam First Powerhouse	Yearling Chinook	80	Monk et al. 1999b
Existing STS, 30-cm orifice	Subyearling Chinook	98	
Bonneville Dam First Powerhouse	Yearling Chinook	90	Monk et al. 1999b
Prototype ESBS, 30-cm orifice	Subyearling Chinook	97	
Bonneville Dam Second Powerhouse[b]	Yearling Chinook	97	Monk et al. 2002
Modified screen system	Yearling Chinook	94	
Bonneville Dam Second Powerhouse[b]	Subyearling Chinook	100	Monk et al. 2002
Modified screen system	Subyearling Chinook	99	
Existing STS, 30-cm orifice			

[a] All estimates (except Lower Monumental) developed using releases of fin-clipped fish; Lower Granite estimates are results of PIT-tagged and fin-clipped releases averaged together.
[b] Estimates made using PIT-tagged fish.

Discussion

Although FGE and OPE are higher in gatewells equipped with ESBSs than those equipped with STSs, hydraulic conditions are also considerably more turbulent in the ESBS-equipped gatewells due to higher water velocities caused by more water being forced up the gatewells by the longer ESBSs. Additionally, the degree of gatewell turbulence at different dams varies with turbine discharge and gatewell volume. In recent studies at John Day Dam (which has relatively severe gatewell conditions), Brege et al. (2001) found that FGE and OPE were both high and within acceptable ranges in gatewells equipped with ESBSs. However, in a different test where fish were exposed to gatewell conditions over a longer period, gatewell-released marked fish were recovered at a high rate (99%), but also experienced 8% descaling and 2% mortality.

These results suggest that at some sites, improvements in FGE and OPE in gatewells equipped with ESBSs relative to STSs might be offset by increases in descaling and injury for the small percentage of fish that do not exit the ESBS gatewells rapidly. The fate of these fish is uncertain and may require additional attention, especially at McNary Dam where a 50% increase in turbine capacity is currently being evaluated. The primary issue with ESBSs is not delay but the condition and survivability of the small percentage of fish that do not exit the turbulent ESBS-equipped gatewells quickly. Therefore, there may have been a trade off between the two

guidance systems. STSs have lower FGE and OPE than ESBSs, but might be better for those fish that are guided in terms of overall condition.

However, to correct the above problems at John Day Dam, the COE modified the porosity of VBSs and, as a contingency, added a gatewell flow-damping feature (which later evaluations in 2002 showed were not required). These adjustments were successful, as descaling and mortality rates returned to acceptable levels, averaging 4.0% and 0.1%, respectively, for yearling Chinook salmon and 1.0% and 0.1% for subyearling Chinook salmon, respectively (Brege et al. 2004).

There is no indication that fish diverted with ESBSs at Lower Granite Dam, where high numbers of fish are collected, are in poorer condition or suffer higher mortalities than expected. Descaling and injury of fish sampled by the smolt monitoring program and overall facility mortality are low. In addition, adult returns of PIT-tagged fish are consistently higher for fish collected and bypassed at Lower Granite Dam than from other dams, including those equipped with STSs (Williams et al. 2005).

Two locations where increased gatewell flow may impact fish condition are McNary Dam and the Bonneville Dam Second Powerhouse. To address the potential for increased accumulations of fine debris and injury problems not anticipated with clean VBSs, methods to removed debris from the VBSs are being designed and tested at these locations.

Conclusions

1. At dams where OPE has been estimated with either an ESBS or STS in place, OPE estimates have generally exceeded the regionally acceptable values for yearling and subyearling Chinook salmon and steelhead. Two exceptions were OPE for yearling Chinook salmon at McNary Dam and for steelhead at Lower Monumental Dam. In both cases, OPE was lower than acceptable with an ESBS as the guidance device.

2. While OPE is generally higher for ESBSs than STSs, some earlier studies indicated that overall condition of juvenile fish might be poorer with the former. More recent studies on modified ESBSs suggested otherwise, as does some adult-return PIT-tag data. Thus ESBSs appear to guide higher percentages of smolts without concomitant increases in descaling and injury rates when appropriately designed VBSs are coupled with ESBSs at specific dams.

Separators and Separation Efficiency

Background and Methods

Stress indices have repeatedly been shown to be elevated in yearling Chinook salmon smolts when in the presence of steelhead smolts (Park et al. 1983, Congleton et al. 2000). Moreover, steelhead smolts, particularly those raised in hatcheries, are much larger and more physical than yearling Chinook salmon smolts. Separation of these species at collection facilities is likely necessary to maximize survival of yearling Chinook salmon smolts.

Separation of juvenile migrant salmonids by size is now an integral part of the juvenile bypass operations at FCRPS dams. Separators at Columbia River juvenile fish bypass facilities

sort fish using an array of appropriately spaced bars oriented parallel to flow passing through a tank. This allows smaller fish to sound and pass between the bars, while excluding larger fish.

The strategy for using bars for separation falls into two categories: single- and two-stage, depending on the objective of the process. Single-stage separators are in use at Ice Harbor, John Day, and Bonneville Dams, where the primary purpose of separation is to monitor the condition of juvenile salmon and steelhead passing through the project bypass system. These units remove adult salmonids, large incidental species, and debris using "wetted" separation in which smaller fish (including salmonid smolts) fall between separation bars along with incidental flow, while large animals and most debris are carried across a short section of exposed bars into another flume for eventual return to the river. Smolts can then be sampled without concern for injury induced by larger adult and incidental fish and debris in the sample holding area. At John Day Dam, the adult fish separation bars are wetted by small low-pressure jets along their upper surface to facilitate fish movement along the bars. Following the separation and sampling process, facilities using single-stage separators bypass all fish to the river downstream from the dam. The Dalles Dam has no juvenile fish bypass system or separator.

Two-stage separators are used at McNary, Lower Monumental, and Little Goose Dams and are intended to separate large steelhead from smaller Chinook salmon smolts, as well as remove adult salmonids, large incidentals, and debris prior to collection for transport. Size separation also allows selective bypass or transport of one or both smolt size classes. The separators at these sites rely on "wet" separation by keeping the fish completely submerged throughout the process. Wet separators depend on sounding behavior as an avoidance response to shallow conditions above the separation bars to achieve separation (Gessel et al. 1985). All fish are introduced to the upstream compartment along with transport flow from the bypass channel (Katz et al 1999). The smaller fish are filtered between bars in the upstream compartment, while larger smolts are removed in the second section. Large incidentals, adults, and debris pass through the unit to the third compartment for return to the river.

Results

Because fish are continually submerged, wet separation was thought to be less stressful to fish. However, separation efficiency of operational wet separators has been usually less than 70% for small fish and varies considerably among years This is due to small fish passing over the bulkhead from the small to large fish section of wet separators where they are subsequently held or transported with larger fish. In addition, some fish hold under the bars for extended periods rather than exit expeditiously from separator units. With existing wet separator designs, especially those at Lower Monumental and McNary Dams, fish that delay in the wet separators become fatigued as a result of swimming in resistance to the hydraulic conditions within the unit.

The separator at Lower Granite Dam is unique in that a single-stage wet separation process is employed to segregate adult salmonids, large incidental species, and debris from smaller fish. Presently, smolts are not separated by size. This system is scheduled for modification when the entire Lower Granite Dam juvenile fish bypass facility is upgraded.

A research program was initiated in 1996 to evaluate potential improvements to existing wet separators, and to develop new separator concepts (Katz et al. 1999, McComas et al. 2001).

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These studies evaluated the effects of many physical separator designs, including separation bar length, slope, color, and spacing as well as effects of transport flow, water velocities through the separator, depth over separation bars, and exit orifice orientation and configuration (McComas et al. 1998, 2000, 2001). Other work considered parameters independent of physical design which may affect separation potential, such as ambient light and fish density.

Design criteria from those research efforts resulting in positive effects on separation efficiency and separator exit efficiency were incorporated into a separator insert used for comparison to the operational separator at McNary Dam (McComas et al. in prep. b). In addition to enhancements to increase separation efficiency, the insert included a perforated-plate false bottom to reduce the volume available for residence under the separation bars, and a more efficient submerged exit-orifice configuration.

Evaluation of the insert against the McNary Dam operational separator over a series of 2-day replicates resulted in measurably higher separation efficiency values using the insert than for the McNary Dam operational separator condition, though the difference was only significant for large fish groups and the total salmonid catch. Using lights above the separator enhanced separation efficiency for all small salmonid groups, although the relationship was significant only for the total small salmonid catch and for the total salmonid catch. Because use of the insert did not result in a biologically significant increase in descaling or adverse stress levels compared to the McNary Dam operational condition, it was installed for use in the Lower Monumental Dam operational separator at the beginning of the 2002 juvenile migration.

In addition to studies aimed at improving the function and design of existing operational separators, a second approach explored alternatives to the existing separator design. The most promising alternative concept to emerge from interagency brainstorming sessions was the high-velocity flume (HVF) design. Under this strategy, smolts enter a section of open flume directly after transport from the bypass channel. While traveling at velocities not normally present in current operational separator designs (1–2 ms^{-1}), smaller smolts could sound between appropriately spaced separation bars within the flume, effecting separation from larger smolts unable to fit between the bars. Both groups would continue to different holding areas without the interruption caused by abrupt velocity reduction of incoming transport flow, and without migration timing delays, stress, and fatigue induced by combating flows within the separator.

Results using an evaluation HVF separator at McNary Dam during the 1997 and 1998 juvenile salmonid migration periods indicated that over 80% separation could be achieved for the total catch of all species combined (McComas et al. 2000, 2001). Based on these observations, a full-scale prototype HVF separator was constructed at Ice Harbor Dam.

Results from cumulative evaluations of design and operational criteria using the prototype HVF separator from 1999 through 2001 suggest that separation efficiency values of over 80% are possible for the total salmonid smolt catch, with moderate (5.3%) overall descaling, and virtually no timing delays in the separator unit (McComas et al. in prep. b). However, most research to date has been conducted when fish abundance was relatively low. Studies are now proposed to test HVF separators under conditions of high fish abundances at Ice Harbor Dam. In addition, other studies have proposed a method for removing adults, large incidental fish, and debris prior to entering the smolt section of the HVF separator (McComas

et al. in prep. a). Although no HVF separators are in operational use at juvenile bypass facilities in FCRPS dams, work is ongoing to complete the development of this concept as a possible option to, or replacement for, the current separator design.

Conclusions

1. Small yearling Chinook salmon smolts appear to be stressed when in the presence of larger steelhead smolts. Separation may be necessary to reduce extended holding of small fish with larger fish, especially in the context of transportation.

2. Past research has demonstrated that separation efficiencies in excess of 80% can be attained with a high-velocity flume separator concept. Research continues to further refine this method of separation for potential use in mechanical screen bypass systems.

Water Temperature Effects

Background

During the summer, juvenile salmonids migrating through the Columbia and Snake rivers encounter water temperatures that often exceed the 20°C (68°F) threshold mandated in the Washington State Clean Water Act (WAC 1992). Of greater concern, Columbia and Snake River temperatures exceeding the 25°C (77°F) upper incipient lethal water temperature for salmonids (Brett 1952) have been documented during the summer juvenile migration period. In recent years, considerable effort has been expended to investigate the effects of and to identify potential measures to decrease high summer river temperatures.

The NMFS 2000 Biological Opinion (NMFS 2000) includes three Reasonable and Prudent Alternative (RPA) actions pertinent to the temperature issue and juvenile salmonids. The first, RPA Action 141, calls for the action agencies to investigate the relationships among elevated water temperature, fish disease, and fish mortality during critical migration periods. The second, RPA Action 142, directs COE to identify and implement measures to address juvenile fish mortality associated with high summer water temperatures at McNary Dam. The third, RPA Action 143, calls for the action agencies, in consultation with other regional entities, to develop and coordinate a plan to model the effects of alternative Snake River operations (flow augmentation).

Results

Several studies have addressed RPA Action 141 so far (effects of temperature on fish disease and mortality). In the summer of 2001, researchers sampled subyearling Chinook salmon at McNary Dam on the Columbia River to assess the prevalence of the bacterial pathogens *Flexibacter columnaris* and *Renibacterium salmoninarum* (*Rs*; the causative agent of bacterial kidney disease) and to measure relative levels of heat-shock proteins in liver tissue (Tiffan et al. 2003a). These authors reported:

1) of 181 fish sampled, 2 tested positive for *F. columnaris*;

2) of 61 fish necropsied, 8 had kidneys characteristic of infection by *Rs*;

3) of 120 fish from which kidney samples were collected and analyzed by an enzyme-linked immunosorbent assay (ELISA), none were positive for *Rs*; and

4) results from heat-shock protein sampling indicated that exposure to water temperatures above 22°C (71.6°F) evoked a strong peak in this physiological parameter.

In the summer of 2002, Haskell et al. (2003) conducted a second year of study at McNary Dam, again assessing the prevalence of bacterial pathogens and levels of heat-shock protein. These authors reported:

1) no F. columnaris was observed in 599 fish randomly sampled, although 3 fish found during smolt monitoring operations tested positive for the pathogen;

2) low Rs levels were detected in 4 of 96 fish sampled; and

3) a mid-August peak in heat-shock protein levels coincided with a water temperature exceeding 23°C (73°F).

Tiffan et al. (2003a) and Haskell et al. (2003) characterized the incidences of bacterial pathogens as low and recommended laboratory studies be conducted to quantify short- and long-term effects of elevated heat-shock protein levels. In laboratory tests conducted by Mesa et al. (2002), effects of a single, acute thermal stress on survival, predator avoidance, and physiology of juvenile fall Chinook salmon obtained from the Hanford Reach of the Columbia River were studied. They found that the thermally-stressed fish showed little direct mortality and no increases in vulnerability to predation. Marked physiological responses of fish to the temperature stress were noted, including transient increases in plasma cortisol, glucose, and lactate, as well as dramatic and persistent increases in levels of liver heat-shock protein 70.

In response to a catastrophic loss of subyearling Chinook salmon at the McNary Dam juvenile fish facility in the summer of 1994, the NOAA Fisheries Service 1995 BiOp required the action agencies to take measures to reduce the potential of a recurrence of the incident (NMFS 1995). In mid-July 1994, the water temperature at the McNary Dam juvenile fish facility reached 20°C (68°F). A few days later, the facility experienced a catastrophic loss of subyearling Chinook salmon over a period of several days, coincident with water temperatures exceeding 21°C (70°F). In prior years, large thermal gradients between operating and nonoperating turbine units were observed and thought to be associated with instances of fish mortality. In the days leading up to the 1994 incident, although thermal profile data showed elevated water temperatures, no large gradients were present (Hurson et al. 1996). In a later analysis, Coutant (1999) suggested that the cause of acute summer mortalities at McNary Dam was cumulative exposure to high temperature water received over a period of several days. Occurrences of acute mortality on the scale of the 1994 loss over several days have not recurred at McNary Dam. For example, total juvenile fish facility mortality during the summer at McNary Dam was 2.2% during both 1997 and 1998, with a one-day peak mortality rate of approximately 8.0% in early July 1998 (Hurson et al. 1999). For the most recent 5-year period (1999 through 2003), summer fish facility mortality has averaged 1.0%, 0.6%, 0.7%, 0.7%, and 1.0% for the respective years, with no notable daily mortality peaks (data available at www.fpc.org).

Continuing concern over the McNary Dam temperature issue resulted in inclusion of RPA Action 142 in the NOAA Fisheries Service 2000 BiOp (NMFS 2000). RPA Action 142 directs the COE to identify and implement measures to address juvenile fish mortality associated with high summer temperatures at McNary Dam. Enhanced temperature monitoring has been in place at McNary Dam since the 1990s to monitor temperatures in the forebay, across the powerhouse, in gatewells, at locations within the collection channel, and at the fish monitoring facility. Temperature gradients within the system are known to occur when turbine units at the south end of the powerhouse are put into operation following an off-line period. Surface waters are warmer in the south forebay than in the forebay upstream from turbine units at the north end of the powerhouse. Turbine units at the south end of the forebay draw this warmer surface water into the bypass system via the gatewells, creating temperature differentials in the collection channel and increasing temperature in downstream portions of the bypass system.

To the extent possible, COE limits occurrence of temperature increases through operational means, that is, by restricting operation of the south end turbines (Units 1, 2, and 3), and by loading the powerhouse from the north end. Among the recommendations in RPA Action 142 is investigating the feasibility of developing a hydrothermal computational fluid dynamics model of the McNary forebay. Model development is currently underway through a contract with the University of Iowa.[4] The completed model will allow different operations (especially the 50% McNary turbine capacity increase) and potential structural mitigation measures to be evaluated.

In 2001 two large industrial mixers were installed in the forebay upstream from turbine units 1 and 2. Evaluation of the cooling effect of mixers on water column temperatures was conducted by Tiffan et al. (2003a). Results indicated that the mixers were ineffective, decreasing temperature inside the mixer plume and in gatewells by only $0.5°C$ ($0.9°F$) and $0.1°C$ ($0.2°F$), respectively. In 2002 Haskell et al. (2003) operated turbine unit 1 experimentally to determine effects of turbine operation on water temperatures. Results of this study (Haskell et al. 2003) showed that turbine unit 1 gatewell temperatures increased by up to $3°C$ ($5°F$) upon starting the unit, followed by a gradual temperature decline lasting two or more hours. Haskell et al. (2003) released subyearling Chinook salmon tagged with temperature-sensing radio transmitters into the bypass system. Tagged fish were exposed to thermal gradients of up to $3°C$ ($5°F$) when turbine unit 1 was started, tagged fish resided in gatewells for a mean time of 7.8 hours, and fish tended to hold in water less than 5 m deep within the gatewells. Haskell et al. (2003) concluded that fish condition, prior stress, and acclimation temperature could affect capacity to respond to temperature increases such as those observed in the gatewells of turbine unit 1 and recommended additional work to explore the effects on survival and performance.

Because juvenile migrant salmonids experience high summer water temperatures during migration through the lower Snake River, the effect of elevated water temperatures on ESA-listed Snake River fall Chinook salmon is a primary regional concern. This issue has been addressed through a number of biological field studies examining effects of flow augmentation and through RPA Action 143 of the NOAA Fisheries Service 2000 Biological Opinion (NMFS

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[4] P. Ocker, COE, Portland, OR. Pers. commun., November 2003.

2000). Williams et al. (2005) provides additional information on the effects of water temperature on the survival of juvenile migrants in the FCRPS.

Dauble et al. (2003) estimated that Snake River adult fall Chinook salmon are currently restricted to about 20% of the Snake River habitat that was available prior to dam construction. Brownlee Dam, a high water storage dam in Hells Canyon, has altered annual temperature cycles below it in the lower Snake River. Connor et al. (2002) noted that the presently available spawning and rearing habitat is cooler and has lower growth potential than areas favored by the species prior to hydroelectric development. They suggested that slower development of fall Chinook salmon to migrational size leads to delayed seaward migration, resulting in an unnatural exposure of smolts to elevated summer water temperatures in the Snake River.

A number of studies have investigated the biological consequences and temperature effects of flow augmentation. Flow augmentation using relatively cool water from Dworshak Dam is intended to increase migration speed and survival of smolts through the Lower Granite Dam reservoir. Connor et al. (1998) PIT-tagged subyearling Chinook salmon in the Snake River from 1992 to 1995 and monitored PIT-tag detections at Lower Granite Dam. They concluded that detection rate at Lower Granite Dam was positively related to mean summer flow and negatively related to maximum summer water temperature, supporting the concept that summer flow augmentation is a beneficial interim recovery measure for improving the survival of subyearling fall Chinook salmon in the Snake River.

From 1992 to 2001, Connor et al. (2003a) used mark-recapture data to study migrational behavior of wild subyearling Chinook salmon in the Snake River. Results suggested that flow augmentation increases the rate of seaward movement during one of four migrational phases and that augmentation likely prevents reversion to parr. Connor et al. (2003b) evaluated the results from wild subyearling Chinook salmon tagged and released in the Snake River from 1998 to 2000 and concluded that survival of these fish generally increased with increasing flow and decreased with increasing temperature.

Tiffan et al. (2003b) tagged naturally produced subyearling Chinook salmon with temperature-sensing radio tags and tracked the fish in the Little Goose Dam reservoir on the Snake River during the summers of 1998 and 1999. They determined that temperatures selected by juvenile migrants were not significantly different from mean water temperatures and no areas of thermal refugia existed in the reservoir. Further, they suggested that augmentation with cold-water releases from Dworshak Dam may increase survival by lowering water temperatures by up to 4°C (7°F). Smith et al. (2003) studied survival and travel time of Lyons Ferry Hatchery subyearling Chinook salmon from 1995 to 2000. These researchers determined that survival from release locations in the free-flowing reach of the Snake River to the tailrace of Lower Granite Dam was significantly correlated with flow, water clarity, and temperature. Survival decreased as flow decreased, as water clarity increased, and as temperature increased.

Smith et al. (2003) were unable to determine which of the three factors were most important, because the factors were highly correlated among themselves. They concluded that augmenting Snake River summer flow with relatively cool water will increase discharge while decreasing temperature of the river and will likely increase the speed of seaward migration, thereby benefiting the recovery of ESA-listed Snake River fall Chinook salmon.

RPA Action 143 directs the action agencies, in consultation with other regional entities, to develop and coordinate a plan to model the effects of alternative Snake River operations (flow augmentation). Implementation of RPA Action 143 has occurred through the NOAA Fisheries Service Regional Forum's Water Quality Team. The Water Quality Team technical workgroup addressing RPA Action 143 is a cooperative effort, open to all regional entities. The RPA 143 Workgroup has met since March 2002 and has recently produced a comprehensive report detailing their progress toward meeting RPA Action 143 (NMFS 2003). Included in the report, and of particular interest to those considering the biological implications of Snake River flow augmentation, is an evaluation of biological needs including various life cycle stages of ESA-listed anadromous salmonids in the Snake River from Hells Canyon Dam downstream to the Columbia River confluence and in the Clearwater River from Dworshak Dam downstream to the Snake River confluence.

Other studies have also addressed RPA Action 143. Connor et al. (2000) described a method to forecast survival and cumulative percent passage for subyearling Chinook salmon at a dam, thereby allowing managers to effectively time water releases. Cook et al. (2003) monitored thermal conditions at the confluence of the Clearwater and Snake rivers and in the Lower Granite Dam reservoir and applied the results to a three-dimensional hydrodynamic and water quality model. They also applied a two-dimensional, laterally-averaged hydrodynamic water quality model to the three Snake River reservoirs downstream from Lower Granite Dam.

Conclusions

1. During the summer, juvenile anadromous salmonids migrating through the Columbia and Snake rivers encounter water temperatures that often exceed the 20°C (68°F) threshold mandated in the Washington State Clean Water Act. Temperatures exceeding the 25°C (77°F) upper incipient lethal water temperature for salmonids have also been documented during the summer juvenile migration period.

2. High summer water temperatures caused a catastrophic mortality within the juvenile collection facility at McNary Dam in the summer of 1994. The mortality was thought to have been the result of operating turbine units at the south end of the powerhouse, which are supplied with warmer water. Since then, modifications to powerhouse operations and improved temperature monitoring have presumably prevented a recurrence, with average mortality rates during the most recent five summers ranging between 0.6% and 1.0%.

3. The two common fish pathogens, *F. columnaris* and *R. salmoninarum*, appear to have been of little concern relative to survival of subyearling Chinook salmon within the bypass system at McNary Dam during two recent summers.

4. To date, a single laboratory study found that thermally-stressed subyearling Chinook salmon showed little direct mortality and no increase in vulnerability to predation, even though test fish responded physiologically to the single, acute thermal stress.

5. In the Snake River, colder than normal water temperatures in spring delay development of subyearling Chinook salmon, resulting in later-than-normal summer migrations that coincide with increasing water temperatures. However, once migrating, these fish appear to respond positively in terms of their migration and survival rates due to flow augmentation from Dworshak Dam.

Effects of Bypass Systems on Smolt Condition and Survival

Background

Over the last 30 years, many improvements have been incorporated into existing juvenile bypass systems at Columbia and Snake River dams (Williams and Matthews 1995), and new systems have also been constructed. This work intensified during the 1990s, with existing bypass systems at Little Goose (1990), McNary (1994), and John Day Dams (1998) replaced with upgraded facilities, and completely new facilities built at Lower Monumental (1993) and Ice Harbor Dams (1996), and at the Bonneville Dam Second Powerhouse (1999). The Lower Granite Dam facility, which was completed as a state-of-the-art facility in 1975, received major structural improvements during the early 1980s, but was never completely upgraded or rebuilt. Each of these new facilities was tested to locate and correct any areas within the structures, flumes, and pipes that might cause descaling, injury, or mortality to juvenile fish passing through them. In addition, the region's state and federal fisheries agencies monitor daily samples of the fish for descaling, injury, and mortality to ensure that these facilities continue to be operated in a manner safe for fish. Starting in 1993, all hatchery spring/summer Chinook salmon from the Snake River were adipose-fin clipped, which allowed hatchery fish to be distinguished from wild fish. Fish condition at the Snake River dams and at McNary Dam on the Columbia River is detailed in Hurson et al. (1996) and Hurson et al. (1999). More recent fish condition information is collected as part of the annual juvenile fish transportation program at each of the Snake River dams and at McNary Dam on the Columbia River.[5] This information is summarized below.

Results

Generally, passage through the fish bypass systems on the Snake River resulted in high levels of fish condition, based on observations of low rates of descaling or direct mortality (Table 6). Descaling ranges from a low of 1.5% for wild yearling Chinook salmon at Lower Granite Dam to a high of 10.1% for the more easily descaled sockeye salmon at Little Goose Dam. It is noteworthy, however, that overall descaling is similar for the older Lower Granite Dam facility and the new Lower Monumental Dam facility and is lower at both of these facilities than at the recently upgraded Little Goose Dam facility. Mortality during passage through the Snake River facilities follows a similar pattern as descaling, ranging from a low of less than 0.1% for hatchery and wild steelhead at Lower Granite Dam to a high of 6.1% for subyearling Chinook salmon at Little Goose Dam. As with descaling, overall mortality tends to be higher at Little Goose Dam than at Lower Granite or Lower Monumental Dams.

The juvenile bypass system at Ice Harbor Dam was completed and evaluated in 1996. Hatchery steelhead and yearling Chinook salmon released at two locations in the collection channel were recaptured and examined. Descaling and mortality for both species were less than 0.2% (Table 6) (Gessel et al. 1997). At this facility, the mechanical screen bypass system operates continuously throughout the migration season, while the sample facility operates intermittently to determine whether descaling and injury rates have increased or are excessive.

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[5] D. Hurson, COE, Walla Walla, WA. Pers. commun., November 2003.

While this provides a cursory examination of descaling and mortality annually, the small numbers of fish examined minimize the usefulness of any data collected.

Recent descaling and mortality rates for fish passing through bypass systems on the lower Columbia River are presented in Table 7. As at the Snake River dams, negative effects of passage through these systems appear to be minor in most cases. At McNary Dam, the first dam that fish encounter after exiting the Snake River, overall descaling rates appear to be slightly higher than for the Snake River dams. Conversely, direct mortality appears to be similar between the two areas.

At John Day Dam, measures of bypass system survival were comprised of mortality observations at the sampling facility (Martinson et al. 2003) and data gathered during a bypass system evaluation in 1998 and 1999 (Absolon et al. 2000a, 2000b). As part of this evaluation in 1998, hatchery Chinook salmon yearlings and steelhead were released at various points within the system. In 1999, three groups of Chinook salmon fry were also released into the upstream end of the elevated flume and recaptured in the sample tank. Based on measurements taken at the sampling facility, descaling at John Day Dam was similar to that measured at McNary Dam and at the Snake River dams. Based upon data gathered during the bypass system evaluation, direct mortality was also similar to that measured at the other dams. For the Chinook salmon fry releases in 1999, no descaling or injuries were observed on any of the recaptured fish.

At Bonneville Dam, descaling in both the First and Second Powerhouses was again within the ranges observed at all of the upstream dams. Direct mortality at sampling facilities in the First and Second Powerhouses was noted by Martinson et al. (2003). Again, these values were similar to values observed at upstream dams.

Although these estimates give some measure of the upper limit of descaling and direct, immediate bypass system passage mortality, they cannot reflect mortality through the entire systems because sampling locations are typically some distance upstream from the outfalls. Also, some mortality observed within a system may have resulted from prior injuries, and, conversely, some live fish observed in samples may die of passage effects later.

Table 6. Average descaling and mortality rates for juvenile salmonids passing through fish bypass systems at Snake River dams from 1998 to 2002.

| | Descaling | | | | | | | Mortality | | | | | | |
| | **Chinook** | | | **Steelhead** | | **Other** | | **Chinook** | | | **Steelhead** | | **Other** | |
Facility	Yearling hatchery	Yearling wild	Subyearling	Hatchery	Wild	Sockeye	Coho	Yearling hatchery	Yearling wild	Subyearling	Hatchery	Wild	Sockeye	Coho
Lower Granite	2.7	1.5	2.8	4.1	1.7	5.7	3.0	0.3	0.3	1.2	<0.1	<0.1	1.6	0.3
Little Goose Dam	6.2	4.9	3.2	6.7	3.5	10.1	6.4	0.6	0.7	6.1	0.3	0.2	2.0	0.7
Lower Monumental Dam	3.1	2.5	2.3	4.5	2.7	3.7	3.9	0.2	0.2	2.3	0.3	0.2	0.7	0.1
Ice Harbor Dam	*	*	*	*	*	*	*	*	*	*	*	*	*	*

* Descaling and mortality for all species and rearing types measured were less than 0.2% for the 1996 juvenile bypass system evaluation at Ice Harbor Dam.

Table 7. Average descaling and mortality rates for juvenile salmonids passing through fish bypass systems at lower Columbia River dams from 1998 to 2000.

| | Descaling | | | | | | Mortality | | | | | |
| | **Chinook** | | **Steelhead** | | **Other** | | **Chinook** | | **Steelhead** | | **Other** | |
Facility	Yearling	Subyearling	Hatchery	Wild	Sockeye	Coho	Yearling	Subyearling	Hatchery	Wild	Sockeye	Coho
McNary Dam	6.4	2.9	5.7	3.7	9.6	3.9	0.2	1.0	0.4	0.3	0.4	0.4
John Day Dam	3.9	1.1	6.1	1.9	8.2	2.5	0.5	0.2	0.3	0.1	0.8	0.2
Bonneville Dam First Powerhouse	5.0	1.7	9.6	4.2	20.3	3.6	0.2	0.4	0.1	0.1	0.5	0.2
Bonneville Dam Second Powerhouse*	2.5	0.5	5.5	1.9	6.6	1.0	0.8	0.5	0.4	0.3	2.0	0.9

* The figures for Bonneville Dam Second Powerhouse are for the years 2000–2002, because the juvenile bypass system modifications became operational in 2000.

Discussion

Today's juvenile bypass systems are vastly improved compared to those that existed during the 1970s and early 1980s. The descaling and injury and system mortality rates for all of the above facilities are substantially lower than those reported by Williams and Matthews (1995), which were reported primarily for fish collected at the then state-of-the-art facility at Lower Granite Dam. This is in spite of the fact that the criteria used to measure descaling are far more comprehensive today than during the earlier years. In addition, recent post-handling and -marking mortality rates are greatly reduced compared to those reported by Williams and Matthews (1995). Marsh et al. (1996a, 1996b, 1997, 2000, 2001, 2003, 2004) and Harmon et al. (2000) reported annual 24-hour post-handling and -marking mortality rates ranging between 0.2% and 1.6%, with most values below 1.0%. These values differ little from those reported as bypass system mortality rates for fish that were bypassed, but not handled and marked.

Passage survival studies conducted by NOAA Fisheries Service at Bonneville Dam from 1987 through 1990 and in 1992 involved releases of differentially marked subyearling Chinook salmon into various passage routes at Bonneville Dam, including the Second Powerhouse bypass system (1987 to 1990) and the First Powerhouse bypass system (1992). These studies used fish identified with freeze brands and marked with coded-wire tags. Relative short-term survival for treatment groups was based on data from recapture of juvenile test fish downstream at Jones Beach (RKm 74) during seaward migration. Relative long-term survival for test fish using the various passage routes was to have been based on recoveries of coded-wire tags from fisheries, hatcheries, and spawning ground surveys.

Results from the above survival studies were reported in Dawley et al. (1988, 1989), Ledgerwood et al. (1990, 1991, 1994), and Gilbreath et al. (1993). Results from Second Powerhouse tests conducted from 1987 to 1990 indicated that relative survival of bypass-released groups averaged 8.3% less than reference groups released in the tailrace immediately downstream from the dam, 7.6% less than turbine releases, and 17.4% less than groups released approximately 3.2 km downstream from the dam (Dawley et al. 1998a). Using a similar methodology, subyearling Chinook salmon were released at the First Powerhouse in 1992 (Ledgerwood et al. 1994). Results indicated that relative survival of bypass-released groups was 11.8% less than turbine-released groups and 28.3% less than groups released approximately 3.2 km downstream from the dam.

Due to the unexpected low survival of bypass-released groups, a separate evaluation was conducted at the Second Powerhouse bypass system from 1990 to 1992 (Dawley et al. 1998a). In these studies, referred to as direct assessments of passage survival, a trap-net attached to the submerged outfall was used to recapture study fish immediately upon exit from the bypass system. Results indicated that the bypass system was not responsible for the considerable survival differences noted in the previous paragraph, although descaling, injury, and stress among release groups did increase as the distance traveled through the bypass system increased. It was hypothesized that increased stress and fatigue of fish traveling through the bypass system, in combination with a poor outfall location, resulted in high predation rates on juvenile salmonids exiting the bypass system. Hence outfall location and configuration are important considerations for maximizing survival of bypassed juvenile salmonids.

Based on the results of these studies, the COE modified the bypass system at the Second Powerhouse in 1999. The collection channel was modified to improve hydraulic conditions, a new dewatering facility that uses floor and wall screens was installed, a new 2.7-km-long conveyance pipe was installed, and outfalls were constructed at Hamilton Island, well downstream of Bonneville Dam. Gilbreath and Prentice (1999) evaluated the new bypass system. They found that none of the hatchery-reared steelhead that traversed the system died or were descaled or injured. For run-of-the-river steelhead, none showed adverse signs of passage effects other than a 0.3% mortality rate. Median passage time from the upstream end of the collection channel to Hamilton Island was about 3 hours for hatchery-reared steelhead and about 1 hour for run-of-the-river steelhead. The median passage time through the length of the collection channel was about 1 hour for hatchery-reared steelhead, compared to only a few minutes for run-of-the-river steelhead. For hatchery-reared yearling Chinook salmon, none of the fish released were injured or descaled. However, mortality was 0.1%, which the researchers attributed to handling and tagging. For run-of-the-river yearling Chinook salmon, descaling was 2.0% and mortality was 0.5% after passage through the new system.

Median passage time from the upstream end of the collection channel to Hamilton Island was about 46 minutes for both hatchery-reared and run-of-the-river Chinook salmon. The median passage time through the length of the collection channel was about 4 and 7 minutes for hatchery-reared and run-of-the-river Chinook salmon, respectively. During the summer smolt migration period, run-of-the-river subyearling Chinook salmon were released to pass through the system. Descaling and injury was nil and mortality was 0.1%. Median passage time was 5 minutes through the length of the collection channel and 42 minutes from the upstream end of the collection channel to Hamilton Island.

Finally, recent PIT-tag data indicate that passage through multiple or individual bypass systems can reduce smolt-to-adult return rates (SARs) of hatchery-reared, but not wild, smolts compared to those that were never detected in any bypass system during downstream passage. The exception is for smolts that were only detected at Lower Granite Dam (Sandford and Smith 2002). These fish consistently returned as adults at equal or higher rates than those not detected in any of the bypass systems during downstream passage. There is also evidence that juvenile bypass systems tend to select fish that are smaller than those represented in the general population (Zabel et al. in prep.), and that smaller fish survive at lower rates than larger fish (Zabel and Williams 2002). This confounds the causal relationship between decreased SARs and detection history at dams. A more detailed review of this subject is presented Williams et al. (2005).

Conclusions

1. Contemporary mechanical screen bypass systems are vastly improved compared to the original systems that operated during the 1970s and early 1980s. Recent very low descaling, injury, and system-mortality rates are testimonials to the improved conditions for juvenile migrants using these passage routes.

2. Outfall location and configuration are important considerations for maximizing survival of juvenile salmonids that are bypassed back to the river below dams.

3. SAR rates for PIT-tagged in-river migrants are related to bypass history. Smolts detected at dams other than Lower Granite Dam often return as adults at lower rates than fish not detected at any dams. For hatchery (but not wild) smolts, detection frequency is inversely related to SAR. A confounding factor is that smaller smolts with lower SARs tend to be more easily guided into bypass systems than larger smolts with higher SARs.

Effects of Bypass Systems on Blood Chemistry

Background

Stress agents, or stressors, are external and internal stimuli that induce quantifiable biochemical responses in higher animals, including fish (Hane et al. 1966, Grant and Mehrle 1973). These physiological responses prepare animals for the "fight or flight response," which is the first stage of the general adaptation syndrome otherwise known as stress (Selye 1956). Stress, then, is an adaptive response by an animal to any of an innumerable number of environmental stressors. The response enhances the probability of survival; therefore, it is not necessarily negative, unless the stress is chronic in nature or the animal has been physically injured or forced to exhaustion. The primary stress response involves changes in blood plasma concentrations of cortisol and adrenaline, which influence levels of secondary indicators including lactate, glucose, liver glycogen, leucocyte count, free fatty acids, and the balance of various electrolytes (Mazeaud et al. 1977).

Results

Several stressors related to passage through fish bypass facilities at hydroelectric dams have been shown to alter concentrations of stress indices in juvenile salmonids under experimental conditions. For example, elevated plasma cortisol and glucose levels have been associated with crowding and handling (Wedemeyer 1976, Congleton et al. 1984), descaling (Gadomski et al. 1994), acclimation temperature (Barton and Schreck 1987a), and confinement (Strange et al. 1978).

Under field conditions, Maule et al. (1988) among others demonstrated that juvenile Chinook salmon plasma cortisol and glucose concentrations could increase cumulatively as the fish passed successive points in a bypass system, corroborating similar results from laboratory work with sequential handling (Barton et al. 1986, Congleton et al. 1999). However, Congleton et al. (1999) found that, generally, peak plasma cortisol levels in fish following handling experiments did not appear to be affected by repeated stressful experiences. They point out it is possible that fish in the multiple stress groups experienced some form of chronic, low-level stress, which was masked by the elevated pre-stress plasma cortisol levels in these groups.

Lactate, also used as a stress index, has been shown to increase in response to the stress of exertion (Mesa and Schreck 1989, Gadomski et al. 1994). As with glucose and cortisol, blood plasma concentrations of lactate can rise dramatically with exposure to this stressor, returning to preexposure levels after several hours following suspension of, or acclimation to, the stimulus.

Blood plasma indicators have been used routinely as stress indices during the fish bypass system evaluation process for upgraded facilities constructed at FCRPS projects since 1990. At

each dam, blood samples were collected from Chinook salmon and steelhead juveniles at successive points within the bypass system. Because blood plasma indicator levels in fish captured from a gatewell are similar to levels found in hatchery-reared (native) smolts (Congleton et al. 1984), results were compared to samples taken from smolts collected concurrently from gatewells. This process was repeated several times through the smolt migration to include populations originating from different places and changes in other environmental factors. In all cases, blood plasma concentrations of cortisol, glucose, and lactate were measured as indices of stress resulting from passage through the facility. Mean levels of indicators (averaged across the entire smolt migration) are included in Table 8 by facility for each species evaluated.

Results from these studies varied by site. For yearling Chinook salmon, there were significant increases in plasma cortisol and glucose concentrations at Little Goose (Monk et al. 1992b), Lower Monumental (Marsh et al. 1995), and John Day Dams (Absolon et al. 2000a) as fish passed from the gatewells through the sample tanks, while no significant differences were reported for any indicator levels from Ice Harbor (Gessel et al. 1997) and McNary (Marsh et al. 1996a) Dams. Lactate levels increased significantly only at Little Goose Dam. Subyearling Chinook salmon were evaluated only at McNary Dam and showed a change only for mean lactate values, which declined significantly before returning to gatewell levels in raceways.

Steelhead plasma cortisol concentrations increased significantly through all bypass systems except at McNary Dam, and significantly elevated levels of plasma glucose were recorded at Little Goose, Lower Monumental, and McNary Dams. Mean lactate concentrations in steelhead showed significant increases only at Little Goose and Lower Monumental Dams.

Discussion

In general, blood plasma stress indices are within the normal range of expected values, as fish pass through current bypass systems. In cases where levels were measured following the bypass process (e.g., from raceways), stress indicator concentrations returned to near gatewell levels within a few hours of raceway residence.

Results to date suggest that the physiological effects of passage through fish bypass facilities on juvenile Chinook salmon and steelhead are nominal, as measured by blood plasma stress indices, and follow a typical sequence for fish subjected to an acute stressor followed by acclimation to or removal of the stress (Wedemeyer 1976, Mazeaud et al. 1977). Several researchers have reported that effects of stress as measured by hepatic plasma indicators diminish within 24 hours following removal of the stressor in a protected environment (Gadomski et al. 1994, Schreck and Knoebl 1997, Sharpe et al. 1998). For example, transported fish showed an initial increase in plasma cortisol levels immediately after loading onto a barge, but the effect abated to preloading levels after 3 hours (Maule et al. 1988).

Table 8.   Recent juvenile Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*) mean blood plasma stress indicator concentrations by juvenile fish facility sample location for COE-operated hydroelectric projects on the Columbia and lower Snake rivers.

| Hydroelectric project | Evaluation year | Authority | Species | Sample location | Blood plasma indicator concentration | | |
					Cortisol (ng/ml)	Glucose (mg/dl)	Lactate (mg/dl)
John Day	1998	Absolon et al. (2000a)	Yearling Chinook salmon	Gatewell	103.2	63.2	91.2
				Pre-separator	151.6	76.2	82.6
				Pre-sample tank	160.1	73.2	85.7
			Steelhead	Gatewell	98.8	88.2	98.6
				Pre-separator	192.7	81.4	89.3
McNary	1994	Marsh et al. 1996a	Yearling Chinook salmon	Pre-sample tank	179.0	107.5	105.5
				Gatewell	92.5	93.9	62.1
				Post-dewaterer	97.5	97.1	74.8
				Separator	102.4	91.6	65.3
				Raceway (0-h)	89.5	87.7	66.9
				Raceway (2-h)	98.4	93.1	61.5
				Raceway (4-h)	84.7	86.2	57.1
				Raceway (6-h)	79.8	88.3	52.3
				Raceway (10-h)	92.3	103.2	61.7
			Steelhead	Gatewell	83.2	153.3	59.3
				Post-dewaterer	84.4	108.2	49.9
				Separator	90.5	109.7	59.9
				Raceway (0-h)	101.2	132.7	72.4
				Raceway (2-h)	108.0	137.3	62.4
				Raceway (4-h)	86.9	132.7	59.6
				Raceway (6-h)	75.8	131.2	53.4
				Raceway (10-h)	100.9	161.5	62.6

Table 8.   Continued. Recent juvenile Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*) mean blood plasma stress indicator concentrations by juvenile fish facility sample location for COE-operated hydroelectric projects on the Columbia and lower Snake rivers.

Hydroelectric project	Evaluation year	Authority	Species	Sample location	Blood plasma indicator concentration		
					Cortisol (ng/ml)	Glucose (mg/dl)	Lactate (mg/dl)
McNary, cont.	1994	Marsh et. al 1996a	Subyearling Chinook salmon	Gatewell	57.2	92.7	122.8
				Post-dewaterer	82.6	85.1	75.5
				Separator	75.9	88.2	77.7
				Raceway (0-h)	84.4	73.1	114.2
				Raceway (2-h)	70.8	82.0	82
		Marsh et al. 1996a	Subyearling Chinook salmon	Raceway (4-h)	75.6	80.9	78.4
				Raceway (6-h)	68.1	87.0	77.3
				Raceway (10-h)	88.3	89.2	87
Ice Harbor	1995	Gessel et al. 1997	Yearling Chinook salmon	Gatewell	140.3	105.9	53.7
				Pre-separator	140.7	95.2	59.2
				Pre-sample tank	135.9	97.2	60.2
			Steelhead	Gatewell	101.0	131.4	51.4
				Pre-separator	165.5	117.7	71.1
				Pre-sample tank	188.8	112.9	65.2
Lower Monumental	1993	Marsh et al. 1995	Yearling Chinook salmon	Gatewell	115.7	62.5	114.5
				Post-dewaterer	144.6	68.9	73.4
				Pre-separator	144.7	81.5	66.8
				Raceway (0-h)	152.3	103.1	57.9
				Raceway (2-h)	127.2	110.4	49
				Raceway (4-h)	121.4	111.7	50.4
				Raceway (6-h)	110.3	108.3	55.9
				Raceway (10-h)	166.6	104.3	57.5
			Steelhead	Gatewell	84.4	119.8	54.6
				Post-dewaterer	154.2	116.3	60.6
				Pre-separator	184.0	127	72.7
				Raceway (0-h)	138.6	147	52.1

Table 8.   Continued. Recent juvenile Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*) mean blood plasma stress indicator concentrations by juvenile fish facility sample location for COE-operated hydroelectric projects on the Columbia and lower Snake rivers.

| Hydroelectric project | Evaluation year | Authority | Species | Sample location | Blood plasma indicator concentration | | |
					Cortisol (ng/ml)	Glucose (mg/dl)	Lactate (mg/dl)
Lower Monumental, cont.				Raceway (2-h)	173.1	147.2	44.9
				Raceway (4-h)	69.3	149.7	55.7
				Raceway (6-h)	103.2	138	54.8
				Raceway (10-h)	174.1	142.8	59.8
Little Goose	1990	Monk et al. 1992b	Yearling Chinook salmon	Gatewell	75.7	96.5	56.4
				Post-dewaterer	77.3	87.5	75.1
				Pre-separator	112	86.7	72.6
				Raceway (0-h)	140.7	107.8	71.0
				Raceway (2-h)	160.5	164.9	59.5
				Raceway (4-h)	129.4	155.6	55.7
				Raceway (6-h)	85.5	154.2	58.2
				Raceway (10-h)	81.7	160.4	72.2
				Pre-barge	79.4	109.2	55.6
			Steelhead	Gatewell	42.2	133	36.6
				Post-dewaterer	114.5	126.3	67.5
				Pre-separator	142.3	119.5	74.8
				Raceway (0-h)	125.1	109.9	68.7
				Raceway (2-h)	91.2	150.1	44.7
				Raceway (4-h)	61	142.1	46.8
				Raceway (6-h)	65.3	129.1	46.3
				Raceway (10-h)	103.5	147	53.2
				Pre-barge	125.4	140.8	51.4

The relationships between physiological indicators of bypass-induced stress and subsequent survival during seaward migration are not well-documented or understood. There is evidence that short-term survival may be directly impaired as a result of stress in poor-quality Chinook salmon smolts. Under controlled conditions, Barton et al. (1986) demonstrated that multiple acute disturbances, similar to those encountered during the bypass process, resulted in cumulative physiological responses. Plasma indicator levels of healthy animals returned to control levels within 30 hours with no mortality after three stress events, while the same treatment for unhealthy fish resulted in 50% mortality within 3–6 hours after the third event.

Indirectly, bypass stress may also somewhat impair survival during subsequent migration. Barton and Schreck (1987b) found a positive relationship between metabolic rate and the plasma cortisol level in stressed fish. They concluded that even relatively minor events can reduce available energy stores in fish by as much as one quarter, leaving the animal with substantially fewer reserves to cope with future environmental challenges such as temperature adaptation, disease, and demands on swimming stamina. On the other hand, Mesa et al. (2000b) found that imposition of an acute stressor on fish heavily infected with *Rs* did not lead to higher infection levels or increased mortality relative to infected but unstressed fish.

As with all animals, weaker, more vulnerable fish are likely targets for predators. For example, Congleton et al. (1984) demonstrated a significant positive correlation between impaired predator avoidance and crowding stress resulting in plasma cortisol levels of 75–150 ng/ml. In another study, Gadomski et al. (1994) could not find a significant relationship between juvenile salmonid descaling and predation by northern squawfish (*Ptychocheilus oregonensis*), but noted there was evidence of increased predation of descaled fish compared to controls at higher descaling rates.

Conclusions

1. Passage through components of mechanical screen bypass systems induces physiological changes related to the general adaptation syndrome, or stress, in juvenile anadromous salmonids.

2. However, the physiological effects of passage through fish bypass facilities are nominal for juvenile Chinook salmon and steelhead, as measured by blood plasma stress indices, and follow a typical sequence for fish subjected to an acute stressor followed by acclimation to or removal of the stressor. Unless exhaustion or physical injury are involved, the majority of bypassed fish are probably minimally affected by at least a single, and possibly several, stressful experiences.

3. There is both pro and con evidence that multiple stress experiences can cumulatively diminish survivability, particularly in animals of inferior quality. Some multiple stress events may inhibit survival while others may not. Post-stress survival likely depends upon which of an innumerable array of stressors are involved in combination with each fish's particular sensitivity to that stressor in terms of its overall condition, health, and environment at the time.

Diel Passage and Timing

Background

Beginning in the late 1960s, sampling from turbine intake gatewells and at smolt monitoring facilities within powerhouses of Columbia and Snake River dams consistently showed that the majority of smolts passed through the dams' powerhouses at night (Long 1968, Sims and Ossiander 1981, Brege et al. 1996), and that the timing of this pattern could be affected by holding behavior in gatewells and collection channels. Since then, hydroacoustic investigations and radiotelemetry data (for example, Johnson and Dauble 1995, BioSonics Inc. 1996b, BioSonics Inc. 1998, Ploskey et al. 1998, Vendetti et al. 2000) from most dams confirmed this as a consistent diel passage pattern and provided more details regarding specific hours of peak passage into turbine intakes.

The more recent studies also showed that peak passage hours differed among passage routes. For example, passage through sluiceways at Bonneville and The Dalles Dams generally peaked in early morning. Diel passage through spillways is more variable, paralleling powerhouse patterns at some dams, but showing high daytime passage and morning peaks at others. In the following subsections, we review diel passage by project.

Lower Granite Dam

Since 1994 both radiotelemetry and hydroacoustic studies have been conducted in the forebay of Lower Granite Dam to assess fish behavior and efficiency relative to a surface bypass collector (SBC) in front of turbine units 4, 5, and 6. Johnson et al. (1998b) found that fish passage into the SBC and spillway was higher during daytime hours than nighttime hours. However, passage at the powerhouse pier nose and inside the turbine intake was the reverse (Figure 3). In 1999 spill occurred from 1800 to 0600, and highest peak spillway passage occurred at 2300. However, the diel distributions of fish passage were similar for the SBC and turbines, with peak passage occurring between 0600 and 1700 (Anglea 1999) (Figure 4). Anglea et al. (1999) found that nighttime spill efficiency was 57% and downstream passage was slightly lower at night, while abundance of smolt-sized fish was higher during the day.

Adams and Rondorf (1999) tracked radio-tagged spring Chinook salmon smolts as they approached the SBC. In 1998 from 66% to 78% of all radio-tagged fish entered the fish bypass (via the turbine intakes) between 1900 and 0600. With the exception of juvenile hatchery spring Chinook salmon, most fish also entered the SBC during early evening and nighttime hours.

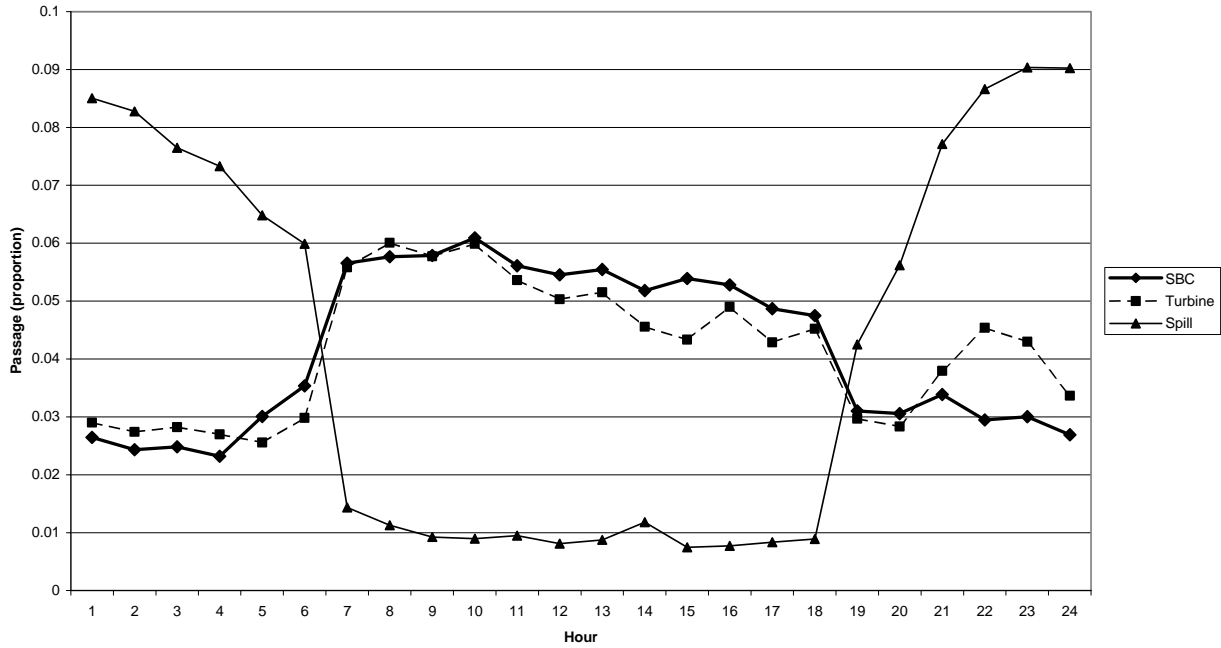Figure 3.   Diel distributions for surface bypass collector (SBC), pier nose, spillway, and within-turbine sample locations at Lower Granite Dam, 1998, based on fixed-location hydroacoustics (adapted from Johnson et al. 1998b).
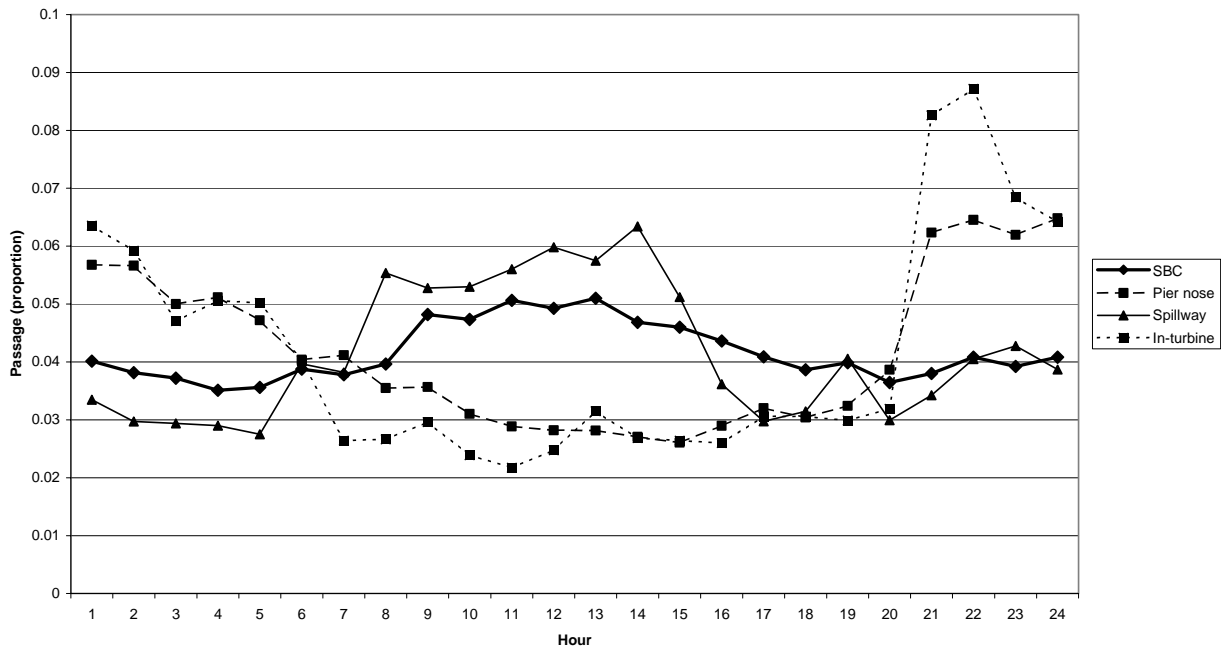
Figure 4.   Diel distribution of fish passage for surface bypass collector, powerhouse, and spillway with fixed-location hydroacoustics at Lower Granite Dam, 1999 (adapted from Anglea 1999).

Seventy-six percent of wild steelhead and wild fall Chinook salmon and 80% of hatchery steelhead entered the SBC between 1900 and 0600. A summary of all radiotelemetry and hydroacoustic studies conducted to evaluate the effectiveness of the SBC, including all information related to diel passage timing and movement, can be found in Anglea et al. (2002).

Adams et al. (1997) and Adams and Rondorf (1999) compared forebay residence times of hatchery Chinook salmon, hatchery steelhead, and wild steelhead for the period of 1994–1998. Flow and spill varied throughout the period, were generally highest in 1997, and the peak spill in May was approximately 115–120 kcfs. Comparing forebay residence times for high flow years (1996–1998) to the lower flow years (1994 and 1995) suggests that in general, high flows and increased spill volume and hours reduced forebay residence time. For example, hatchery steelhead median residence times were 14.2 hours and 26.7 hours in 1995 and 1994, respectively, compared to 0.7–4.0 hours from 1996 to 1998. Hatchery yearling Chinook median residence times were 6.8 and 5.0 hours in 1995 and 1994, respectively, compared to 1.0–1.9 hours from 1996 to 1998. Wild steelhead median residence times showed little difference between years and ranged from 0.7 to 3.5 hours between 1995 and 1998.

In an attempt to improve surface bypass at the dam in 2001, the COE installed a removable spillway weir (RSW) in spillway 1. Using radiotelemetry in spring 2002, Plumb et al. (2003) reported that route of passage at the dam varied depending upon the time of day (Table 9). Higher proportions of salmon smolts passed through the RSW during the daytime than during the nighttime.

Table 9.   Routes of passage in spring 2002, determined via radiotelemetry (adapted Plumb et al. 2004).

	Daytime (0500–1959)			Nighttime		
Stock	RSW	Spill	Turbine routes	RSW	Spill	Turbine routes
Yearling Chinook	60%	20%	20%	49%	26%	25%
Hatchery steelhead	73%	6%	21%	38%	22%	40%
Wild steelhead	69%	14%	17%	42%	25%	33%

In 2003 Plumb et al. (2004) confirmed the above trends. As they approached the dam, smolts were distributed shallower during daytime and deeper during nighttime. Hence they are more apt to pass the dam through the deeper passage routes afforded by turbines and spillways at night than during day. Performance of the RSW was optimal during the day, regardless of species or rearing type.

Little Goose Dam

In three years of research by Vendetti et al. (2000), passage of radio-tagged juvenile subyearling Chinook salmon showed a slightly different diel pattern for passage through the powerhouse at Little Goose Dam (Figure 5). Peaks in the number of detections were observed between 0500 and 0600 hours and again in the evening between 1800 and 2000 hours, with the evening peak being the larger and longer of the two.

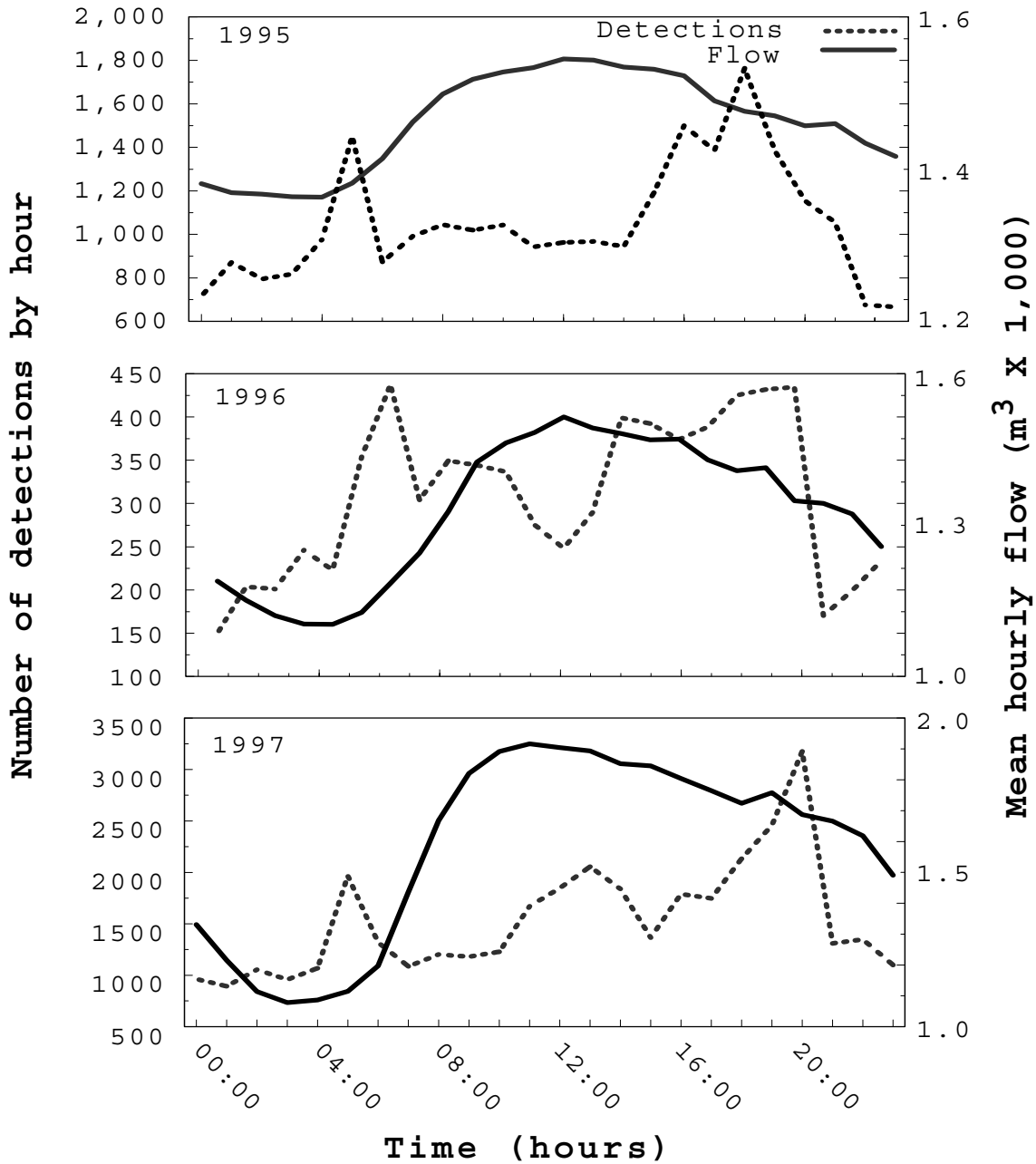Figure 5.   Total number of detections over the entire study period vs. mean hourly discharge at Little Goose Dam, 11 July–24 August 1994, 17 July–24 August 1995, 17 July–26 August 1996, and 15 July–23 August 1997 (adapted from Vendetti et al. 2000).

Lower Monumental Dam

To assess the downstream migrational patterns of smolts passing the dam and to estimate the effectiveness of spill in passing migrants, hydroacoustic investigations were conducted at Lower Monumental Dam from 1986 through 1989 (Wright et al. 1986, Ransom and McFaden 1987, McFaden 1988, Ransom and Sullivan 1989). As at other dams, turbine passage rates were highest during nighttime, with a peak at 2300 hours. During the four years of study, nighttime project passage (powerhouse and spill) ranged from 70% to 84%.

Spillway operation varied over the four years of tests. In 1986 spill varied from just nighttime spill (1800–0600 hours) to a constant 24-hour spill. Examination of diel passage during these different flow regimes suggested that both fish behavior and dam operations influenced the timing of spillway passage. During a period when spill was relatively constant throughout the day, a pronounced peak in passage occurred at 2200 hours. However, this evening peak was enhanced during periods when spill was off during daylight hours and started at 1800. In studies from 1987 to 1989, the spillway was only operated from 1800 to 0600 hours, and highest passage rates occurred at 2300 hours and declined steadily from 2300 to 0600 hours.

Ice Harbor Dam

Three years of hydroacoustic investigations at Ice Harbor Dam revealed diel passage patterns similar to other Columbia and Snake River dams (Johnson et al. 1983, Ransom and Ouellette 1988). Most migrants passed the dam at 2300 hours. Sluiceway diel passage rates were highest from 0600 to 1300 hours. Turbine passage rates were highest from 2100 to 0600 hours.

During all three years of study, hourly passage rates through the spillway were more variable than through the turbine units or sluiceway. Generally, spillway passage rates were low in the early morning and then increased steadily to a peak at 1200 hours. The rate then declined rapidly, reaching a low point at 1700 hours, followed by a secondary peak at 2100 hours (only slightly lower than the peak at 1200 hours).

In 1999 a radiotelemetry study of yearling Chinook salmon passage was conducted at Ice Harbor Dam to determine tailrace egress and routes of passage under varying levels of spill and powerhouse flow (Eppard et al. 2000). At 1800 hours each day, powerhouse flow was reduced while spill was increased to the maximum level, based on dissolved gas levels, until 0600 hours the following day. The test fish were released in the tailrace of Lower Monumental Dam each morning (0800 hours) via the bypass outfall. Receivers were positioned 1 km above Ice Harbor Dam and across the powerhouse and spillway, in the juvenile bypass channel, and on each submersible traveling screen. Each individual fish could be timed from the study entrance line through the passage route. Of the 580 fish detected 1 km above the dam, diel passage was fairly evenly distributed. A total of 302 and 278 fish passed the project during the day and night, respectively, although release location may have affected these results. Spilled fish had a lower forebay residence time, and fish first detected after dark had a lower forebay residence time than those first detected during daylight hours.

McNary Dam

No hydroacoustic or radiotelemetry studies have been conducted that provide a robust source of diel passage information at McNary Dam. Studies of OPE using an orifice trap provide information on hourly passage from the gatewell(s) sampled. Using an orifice trap, McComas et al. (1997) found that passage from gatewells into the juvenile bypass channel appeared heaviest within a few hours of dawn and dusk, with the dusk peak having generally larger numbers of fish. Orifice trap data does not provide information on when fish first arrived in the forebay or entered the gatewell.

John Day Dam

Diel passage patterns have been reported as part of the smolt monitoring program at John Day Dam since the 1970s. This sampling, done mainly in gatewell slots 3A and 3B, has shown that all species of juvenile salmonids tended to pass through the John Day Dam powerhouse during hours of darkness (Brege et al. 1996). In more recent years, further observations have confirmed that passage occurs predominantly from 1800 to 0600 hours (Martinson et al. 1998) (Table 10).

Table 10. Percent night passage (1800–0600 hours) for each season at John Day Dam, 1985 to 1997 (Martinson et al. 1998).

Year	Yearling Chinook	Subyearling Chinook	Wild steelhead	Hatchery steelhead	Coho	Sockeye
1985	83.2	83.7	N/A	N/A	91	86.8
1986	75.5	80.1	N/A	N/A	95.9	81.9
1987	84.5	85.4	93.6	85.6	95	94.9
1988	80	80.7	80.8	70.3	83.9	87.1
1989	86.4	86.2	73.6	79.4	93	79
1990	79.7	84.4	76.3	94.8	95.6	85
1991	89.9	77	91	92.3	96.2	83.6
1992	82.8	78.7	95.3	91.5	96	94.9
1993	83.3	87.8	83.4	80.7	95.1	86.5
1994	80.9	68.1	91.6	81.4	92.2	94.5
1995	80.7	79.7	87.9	75.8	91.5	79.5
1996	68.6	70	81.6	74.7	80.2	65.6
1997	62.6	73.1	67	70.6	73.7	59.6
Average	79.8	79.6	83.8	81.5	90.7	83
Min.	62.6	68.1	67	70.3	73.7	59.6
Max.	89.9	87.8	95.3	94.8	96.2	94.9

Giorgi and Stevenson (1995) reviewed all gatewell sampling, hydroacoustic, and radiotelemetry data from 1980 to 1989. They concluded that even though none of the reports reviewed offered robust estimates of diel passage, there was consistent agreement among all evaluation tools and across many years. The data indicated that smolts at John Day Dam exhibit a strong tendency to sound and pass through deep passage entrances during nighttime hours.

Both the hydroacoustic and radiotelemetry data corroborated the gatewell sampling and also showed that timing of spillway passage through the deep spill intakes (14.3–17.7 m below normal operating pool) paralleled powerhouse passage.

One factor that could influence passage timing measured at the dam is the release location of the radio-tagged fish during the earlier and even some more recent studies. In 1996, Holmberg et al. (1998) saw diel passage patterns similar to those summarized by Giorgi and Stevenson (1995). However, the test fish were released only a short distance upstream (8 km) from the dam. In 1997 Hensleigh et al. (1999) observed a similar pattern for fish also released a short distance upstream from the dam. In contrast to previous years, they saw no apparent diel pattern for fish released at McNary Dam. They also reported that, with high spill volumes in 1997, 50–75% of radio-tagged fish passed through the spillway, as compared to roughly 40% in previous years.

In 1997 BioSonics Inc. (1999a) reported higher numbers of fish passing the dam at night based on hydroacoustic studies. They estimated that 54–89% of the fish passed through the spillway. Also, during the spring when spill level was similar day and night, they observed that fish passed the spillway at a rather uniform rate. In 1999 Hansel and Beeman (1999) saw differences in species response to dam operations. Steelhead generally passed at night regardless of spill discharge. Further analysis of the data suggests that steelhead smaller than 200 mm (primarily wild) did readily pass the spillway throughout the diel period.[6] Yearling Chinook salmon appeared to readily pass the project if daytime spill was provided. Information on John Day Dam spill effectiveness and efficiency is discussed in the Spill Efficiency and Effectiveness section.

Time to pass the project is another important consideration. In 1995 Sheer et al. (1997) estimated that forebay residence time of spring Chinook was 10 hours during conditions of low spill (<14 kcfs). For 1996 Holmberg et al. (1998) reported delays of less than 1 hour during moderate spill conditions (47–125 kcfs). In 1997 Hensleigh et al. (1999) observed forebay residence times of 0.3 hour when spill discharge exceeded 300 kcfs.

To summarize, data collected during the 1980s suggested there was a strong and consistent tendency for fish to pass the project at night. However, it is difficult to compare results from recent and earlier studies, due to differences in sample sizes, release points for tagged fish, and radio-tagged fish release timing. Based on radiotelemetry and hydroacoustic study results since 1997, it appears that yearling migrants, especially yearling Chinook, will readily pass the spillway during the day if spill volumes are 30% or greater. Daytime spill volumes other than 30% have not been tested using recent release protocols for radio-tagged fish.

More recently, Hansel and Beeman (1999) found that juvenile steelhead generally passed the dam at night regardless of the spill discharge when they arrived. Yearling Chinook salmon passed similarly when arriving during a daytime spill discharge rate of 68%, but during 30% daytime spill, they passed at that time. Fish arriving at night generally passed at night.

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[6] G. Fredricks, NOAA Fisheries Service, Portland, OR. Pers. commun., March 2000.

According to Anglea et al. (2001), radiotelemetry studies conducted in 1999 and 2000 and hydroacoustic studies conducted after 1999 were the only studies that collected sufficiently reliable passage and survival data for studies conducted at John Day Dam. They also reported that it appeared a majority of juveniles passed the dam at night (especially for steelhead, where often over 90% passed at night, regardless of treatment). In addition, they reported that the addition of daytime spill increased the proportion of juvenile salmonids passing the spillway during the day (especially yearling Chinook salmon in 2000).

The Dalles Dam

Diel passage through the powerhouse at The Dalles Dam is similar to that seen at other Columbia and Snake River dams. Both gatewell dipping (Long 1968) and hydroacoustics (Magne et al. 1983, Steig and Johnson 1986, Johnson et al. 1987, BioSonics Inc. 1996a) documented that the majority of all salmonids entered gatewells between 1900 and 0700 hours. In studies conducted by BioSonics Inc. (1996a), both spring and summer migrants exhibited peak passage during hours of darkness.

As with Bonneville Dam, diel passage through the sluiceway does not seem to follow the nighttime passage pattern. Early studies by Nichols (1979) and Nichols and Ransom (1980, 1981) reported that sluiceway passage peaked during daylight hours, typically around midday. More recent hydroacoustic studies have also found that average daylight passage through the sluiceway was the predominant pattern at The Dalles in the spring. During the summer period, passage rates through the sluiceway also showed low nighttime passage, but the peak was during the afternoon rather than morning. After reviewing all available literature, Giorgi and Stevenson (1995) concluded that, regardless of the sampling approach employed, during the spring juvenile salmonids pass via the sluiceway at The Dalles Dam through most of the 24-hour period, with peak passage occurring during daylight hours.

There are few measures of diel passage at The Dalles Dam spillway, because in most of the studies conducted there spill was only provided at night and did not span a 24-hour period. However, in studies conducted in 1996 by BioSonics Inc. (1996a), spill levels were maintained for 24-hour periods and slightly higher morning passage rates via spill were found during the spring. This morning peak was even more pronounced during the summer migration. In 1999 Ploskey et al. (1999a) found that diel distribution of fish passage during the spring through the spillway was similar between the 30% and 64% spill treatments. The proportion of spillway fish passage was relatively uniform, except for a substantial peak between 2000 and 2100 hours. During the summer, fish passage through the spillway was again fairly uniform, except for peaks at 0600–0700 hours and 2000–2200 hours during 30% spill, and from 1800 to 1900 hours during 64% spill.

More recently, Hansel and Beeman (1999) examined FPEs and spill and sluiceway passage efficiencies of yearling Chinook salmon and steelhead at The Dalles Dam. With the exception that the FPE of juvenile steelhead was significantly greater during the day than night during the 30% spill discharge (95% vs. 86%, respectively), neither FPE nor spill passage efficiency differed significantly between day and night for either species. However, differences between diel passage conditions did affect spillway passage location. During the adult spill pattern, radio-tagged fish passed mostly through the southern end of the spillway, whereas

during the juvenile spill pattern, they passed predominantly through the northern end. Median residence time in the near-dam forebay area was generally less than 0.4 hour for both species, but juvenile steelhead during the daytime 30% spill discharge had a median residence time of 2.3 hours.

Using radio tags, Beeman et al. (2000a) found that residence times were significantly longer for steelhead arriving during the day than at night, and that steelhead greater than 200 mm in length had significantly shorter residence times than smaller steelhead. Presumably, this reflects a behavioral difference between wild and hatchery steelhead, because the former tend to be smaller. In a similar study on subyearling Chinook salmon, Beeman et al. (2000b) reported total project diel passage was 59% and 41% during day and night, respectively.

With hydroacoustics, Moursund et al. (2001a, 2001b) used relative passage rates between the major routes of turbine, sluice, and spillway to examine temporal differences in passage. Turbine passage peaked at night in the spring, but that peak shifted to afternoon by summer. Spillway passage showed a peak in the morning hours, and this trend remained through the summer. There was a pronounced trend for fish to preferentially pass through the sluice during the day rather than at night. This trend was clear in both the spring and the summer.

Bonneville Dam

Passage patterns at Bonneville Dam are more complex than at other dams because there are three separate structures: two powerhouses and an unattached spillway. There are also two passage routes at each powerhouse: the turbines and ice/trash sluiceway. For this reason, the available information is provided separately for each powerhouse and the spillway.

First Powerhouse

Gessel et al. (1986) dipped gatewells over 24 hours during six periods throughout the spring migration and observed peaks in passage shortly after dawn and dusk, with the evening peak being typically much higher. From 1992 to 1995 the Smolt Monitoring Program sampled juvenile migrants moving through the downstream migrant channel on a 24-hour basis (Martinson et al. 1999). During this time, passage for all species increased at dusk, at about 2000 hours, and peaked at 2200 hours (Table 11). Since 1995 the Smolt Monitoring Program has sampled for 8 hours per day only (from 1600 to 2400 hours) and has reported the typical dusk peak in turbine passage observed at Snake River and lower Columbia River dams (Martinson et al. 1996, 1997, 1998).

In 1998 Martinson et al. (1999) observed that the 8-hour passage patterns of tule and bright stocks of subyearling Chinook salmon followed the same general pattern as spring migrants with only slight differences. Increases in tule passage began earlier, at about 1800 hours, had a smaller peak at 2200 hours, then averaged a lower percent of total passage for the remaining hours. Upriver bright passage resembled the spring migrants, with a larger, more abrupt increase at 2200 hours and a gradual decline in passage thereafter.

Table 11. Percent night passage (1800–0600 hours) for 1992–1995 at Bonneville Dam First Powerhouse (Martinson et al. 1999).

Year	Yearling Chinook	Subyearling Chinook	Wild steelhead	Hatchery steelhead	Coho	Sockeye
1995	52.8	65.8	68.3	62.9	70.7	56.2
1994	49.6	52.4	52.2	53.1	66.7	74.6
1993	43.2	56.2	67.1	62.4	68.1	63.6
1992	52	44	62.3	61.3	60	69
Median	50.8	54.3	64.7	61.9	67.4	66.3
Min.	43.2	44	52.2	53.1	60	56.2
Max.	52.8	65.8	68.3	62.9	70.7	74.6

Hydroacoustic data from Ploskey et al. (1998) also showed that mean hourly smolt passage into turbines at the First Powerhouse was higher during night hours than during the day in 1996 and 1997. The data were more variable in the spring than in the summer. The authors also observed that diel passage through the sluiceway at the First Powerhouse varied from turbine passage. During both the spring and summer smolt migrations, the majority of passage through the sluiceway occurred during the early morning hours with a peak at approximately 0300 hours. Passage was reduced during daytime hours, with a secondary peak shortly after sunset.

Second Powerhouse

The patterns of diel variation in fish passage rates during the spring migration, seen at the First Powerhouse, do not seem to be as strong or consistent at the Second Powerhouse. Studies by BioSonics Inc. (1998) showed little diel variation at unit 11A (both guided and nonguided), with no peak at dusk. Spring studies by Ploskey et al. (1998) showed a nighttime peak for a couple of treatment days, but the majority of days had highly variable data, with no consistent pattern.

During summer migration studies by BioSonics Inc. (1998), subyearling Chinook salmon exhibited a minor diel variation in unit 11A with highest fish passage between 2100 and 2200 hours. However, Ploskey et al. (1998) observed a pronounced nighttime peak from 2200 to 2300 hours during summer smolt passage through the turbines at the Second Powerhouse.

Similar to the First Powerhouse sluiceway, spring-time fish passage through the sluice chute at Second Powerhouse occurred during early daytime, between 0600 and 1300 hours (BioSonics Inc. 1998). A second, smaller increase in passage rates was observed between 1800 and 2100 hours. During the summer migration, the peak passage rates through the sluice chute occurred between 0600 and 1800 hours. The hours of lowest passage through the sluice chute were from 2100 to 2200 hours, coinciding with the hours of highest fish passage of guided fish into unit 11A.

Spillway

A limited amount of hydroacoustic data is available pertaining to diel timing of juvenile salmonids through the spillway. BioSonics Inc. (1998) found that fish passage rates were higher during nighttime at the spillway in the spring. These results were not adjusted for changes in spill levels; however, because of high river flows, spillway discharges showed little variation during the spring. During the summer, fish passage rates were also highest at night at the spillway; however, this was in part due to spillway operations, when spillbay 5 (with transducer) was typically closed between 0500 and 2200 hours.

In 1999 Plumb et al. (2001) found that regardless of release site, more fish were first detected at the spillway (37%) than at Bonneville Dam First (31%) or Second Powerhouse (29%). Median forebay residence times for hatchery steelhead were 5.6, 3.9, and 0.3 hours for the two powerhouses and spillway, respectively. Corresponding times for yearling Chinook salmon were 1.0, 0.2, and less than 0.1 hour. Additionally, fish from all releases passed the dam throughout the diel period, with slightly higher numbers during nighttime passage (2000–0659 hours).

During the low-flow year of 2001, Evans et al. (2001a) found that passage rates were highest for yearling Chinook salmon during the day at the spillway and at the First Powerhouse, but nighttime passage was the highest at the Second Powerhouse. Overall diel passage was generally flat. A higher proportion of the total daily passage occurred during the day (0500–2059 hours), but hourly passage was highest at night, with peak hourly passage at sunset (2100 hours). Also, the median residence times in the forebays of Bonneville Dam ranged from 8 minutes to 9.7 hours, depending on forebay area. In a similar study on subyearling Chinook salmon, Evans et al. (2001b) found that median forebay residence time was shortest for the Second Powerhouse (0.7 hour), compared to 2.4 hours at the First Powerhouse and 1.5 hours at the spillway. Diel passage was very similar to that reported above for yearling Chinook salmon.

Diel Passage Conclusions

1. Passage timing for each route varies with dam configuration, flow volume through the route of passage, species, and life history type. Most studies were not designed to evaluate 24-hour passage behaviors for each species and life history type, making it difficult to precisely describe passage timing for each dam under varying conditions.

2. In general, juvenile migrants that arrive during daytime tend to not pass through powerhouse routes (turbines and bypass) on arrival, but will readily pass through surface-oriented outlets and daytime spill if provided.

3. In general, for juvenile migrants that arrive at night, passage is more equally distributed amongst available routes.

Juvenile Salmon Passage Through Sluiceways and Surface Bypass Systems

Many of the older FCRPS dams were constructed with surface-oriented ice and trash sluiceways to pass ice and debris from the forebay to the tailrace. These include (year completed) Bonneville Dam First Powerhouse (1938), McNary Dam (1954), The Dalles Dam (1957), and Ice Harbor Dam (1962). The sluiceways included adjustable weir gates above turbine intakes that passed water from the forebay into a longitudinal channel around the dam to the tailrace. In addition, the Bonneville Dam Second Powerhouse was constructed in 1983 with a surface outlet located in one corner of the powerhouse to pass ice and debris to the tailrace.

Juvenile yearling salmon generally migrate in the upper portion of the water column and approach FCRPS dams near the surface (Dauble et al. 1999, Johnson et al. 2000). However, most turbine intakes and all spill gate sill elevations are greater than 12.2 m below forebay water surfaces. Thus juvenile migrants have to follow flow lines downward into turbine and spill intakes, a behavior that is counter to their normal surface orientation. Also, juveniles can accumulate in the immediate forebay during the day at sites where spill is limited or nonexistent, and pass the dam at night. Delay of juvenile migrants in forebays may increase losses to predation (Raymond and Sims 1980). Recognizing these behaviors, researchers began to evaluate the sluiceways for their potential to pass juvenile salmonids.

The Ice Harbor Dam sluiceway was evaluated in 1995 prior to installation of a juvenile bypass system in the sluiceway channel. Combinations of wide, surface-overflow entrances (6.1 m wide, 1.8 m deep, and 2.3 ms^{-1} entrance velocity) and deeper, slotted entrances (up to 1.8 m wide, 12.2 m deep, and up to 1.2 ms^{-1} entrance velocity) were studied. The wide, surface-overflow entrances had the highest passage rates based on hydroacoustics (BioSonics Inc. 1996b) and radiotelemetry, where a total of 57% of the radio-tagged Chinook salmon used sluice gate 2B, which was a surface-skimming flow (Swan et al. 1996).

The Dalles Dam sluiceway is located along the entire length of the 22-turbine-unit powerhouse, which is oriented parallel to the river channel centerline. Surface flow passes through weir-type chain gates into a channel that has a hydraulic capacity of 5,000 cfs. Typically, three gates above turbine unit 1 at the west end of the powerhouse are opened to pass 3,200 cfs through the sluiceway. During periods of no spill, sluiceway passage was estimated at 40–55% (Giorgi and Stevenson 1995). However, sluiceway passage is reduced during periods of spill. For example, in 2000, estimated sluiceway passage was 6% and 7% during the spring and summer, respectively, during 40% spill (Moursund et al. 2001b).

The Bonneville Dam First Powerhouse is perpendicular to the river channel centerline. Estimates of sluiceway efficiency have ranged from 83% for steelhead (Willis and Uremovich 1981) to 13% for subyearling Chinook salmon (Krcma et al. 1982), and vary highly by species,

flow, time of day, estimation method, forebay elevation, the number of turbines operating, and which sluiceway weirs are open. Although the capacity of the northern-most sluice gates was reduced in the early 1980s by installation of a juvenile fish bypass system, total sluiceway capacity was not reduced, and remains at approximately 1,000 cfs. In 2002 estimated sluiceway passage was 33% (95% CI, 32–34%) for spring and 29% (95% CI, 28–30%) for summer with gates 7A and 10C operating, under conditions of 24-hour spill, and the Second Powerhouse being operated as the priority powerhouse. Sluiceway passage during both the spring and summer was higher during the day than at night (Ploskey et al. 2003).

The Bonneville Dam Second Powerhouse sluice chute is located immediately south of turbine unit 11, the southernmost of eight turbines at the powerhouse. The entrance is 4.6 m wide by 6.1 m deep and discharges approximately 3,000 cfs. Strong lateral flows toward the north and south ends of the powerhouse concentrate juvenile fish at these locations (Monk et al. 1999a). In 1998 with the six southernmost turbine intake extensions removed (over turbine units 11–14), 52% of the radio-tagged steelhead and 36% of the radio-tagged yearling Chinook salmon approaching the powerhouse passed through the sluice chute, compared to 21% of the tagged steelhead and 14% of the tagged yearling Chinook salmon detected in the juvenile bypass system. When the chute was closed, 50% of the steelhead and 30% of the yearling Chinook salmon were detected in the bypass system (Hensleigh et al. 1998). The combined efficiency of the sluice chute and juvenile bypass system relative to passage through the chute and turbine units 11–13 was 90% during both spring and summer periods, based on hydroacoustics. When the chute was closed, guidance into the juvenile fish bypass system was 55% and 30% of the total fish passing turbine units 11–13 during the spring and summer, respectively. For the sluice chute alone, chute efficiency (proportion of fish passing the chute relative to passage through turbine units 11–13 and the chute) was 83% and 81% in the spring and summer, respectively. The effectiveness of the sluice chute was high; about five times more fish passed the chute than would be expected based on the proportion of water passing the chute (Ploskey et al. 1999b).

Sluiceway survival is a function of direct mortality that occurs during passage through the sluiceway and plunge into the immediate tailrace, and delayed mortality that occurs downstream. Estimated survival through sluiceways varies with life history type, time of day, and spill conditions. For example, estimated survival through The Dalles Dam sluiceway in 1998 was 0.960 (95% CI, 0.874–1.054) for coho salmon under 30% spill during the daytime, while estimated survival of subyearling Chinook salmon was 0.889 (95% CI, 0.806–0.980) during 30% daytime spill. In 2000 estimated survival of yearling Chinook salmon was 0.945 (95% CI, 0.895–0.998) during daytime and 0.940 (95% CI, 0.889–0.995) at night during 40% spill. Estimated survival of subyearling Chinook salmon was 0.955 (95% CI, 0.849–1.074) during daytime and 0.972 (95% CI, 0.864–1.094) at night during 40% spill (Absolon et al. 2002).

These sluiceway evaluations suggest that downstream migrants use nonturbine, surface-oriented passage routes, especially during daytime and periods of nonspill. Relatively high passage rates resulted in the sluiceways being highly effective, given their low hydraulic capacity relative to total river discharge. However, their low hydraulic capacity also limited further increases in the percentage of fish that could be passed through these routes.

Wells Dam on the upper Columbia River (completed 1967) is configured differently than all other run-of-the-river dams on the Snake and Columbia rivers. Powerhouse turbine intake

ceilings are 21.3 m deep, with 11 spillway gates located over and between the 10 turbines. Each of the five turbine pairs has one surface bypass entrance 4.9 m wide by 22.3 m deep. The surface bypass entrance velocity is approximately 0.6 ms^{-1}, and each surface bypass slot passes up to 2,200 cfs. Bypassed fish and flow pass through the surface entrances into an afterbay between the entrance and the spill gate, and are discharged to tailwater through the spillway gates, which control bypass entrance flow and velocity. Surface bypass flow is 5–7% of the hydraulic capacity of each pair of turbines (Johnson 1996). Evaluations of a surface bypass system at Wells Dam demonstrated that surface-oriented fish will pass in large numbers through the surface route with a relatively small flow. Surface bypass entrance efficiency (ratio of fish passing into the bypass relative to fish passing both the bypass entrance and turbine pair) is approximately 90% (Johnson et al. 1992), and ranges from 84.3% to 95.0% for spring migrants and 76.5% to 97.0% for summer migrants (Skalski et al. 1996).

Prototype Surface Bypass Systems

Based on the successful performance of the Wells Dam surface bypass system, prototype surface bypass systems have been tested at a number of locations in the Pacific Northwest to determine whether fish behavior principles observed at Wells Dam could be applied to dams with appreciably different and more typical powerhouse and spillway configurations, but with similar fish passage performance results. Evaluations of prototype surface bypass systems commenced in 1995 at Wanapum and Rocky Reach Dams, at Lower Granite Dam in 1996, at the Bonneville Dam Second Powerhouse in 1997, and the Bonneville Dam First Powerhouse in 1998. Prior to the 1998 evaluation, the Lower Granite surface bypass collector prototype was deepened to simulate the Wells Dam intake, and a behavioral guidance device was installed. A J-block occlusion device was installed at the west turbine units of The Dalles Dam in 2001, and a removable spillway weir was installed at Lower Granite Dam in 2002.

In 1995 a 16.8-m-deep prototype surface collector was installed upstream of turbine units 7–10 at Wanapum Dam. A single slot entrance, 4.9 m wide by 15.2 m deep, with a hydraulic capacity of 1,400 cfs was located directly above turbine unit 8. The structure occluded the upper portion of the turbine intakes. Due to poor performance of the slot entrance, reduced turbine entrainment associated with the turbine intake occlusions, and increased entrainment through turbines without intake occlusions, the prototype collector was extended in 1996 to also occlude turbine units 4–6. Additional collector entrances were not added. Mean collector efficiency (percent of fish entering the prototype collector relative to the total passing into the collector plus the number passing into reference units) was 30% in 1995 relative to turbine unit 8, and averaged 12% relative to turbine units 7–10.

Horizontal distribution across the powerhouse was skewed toward turbine units not covered by the occlusion device when compared to distribution in years before the prototype collector was installed (Ransom et al. 1996). Furthermore, spillway efficiency during surface bypass prototype testing was higher (43%) than in previous years (30%) (Kumagai et al. 1996). This suggests the prototype surface collector may have reduced entrainment at the occluded units, and fish passed over the spillway, increasing spillway effectiveness. The prototype collector was again extended in 1997 to occlude turbine units 1–3, and mean collector efficiency relative to turbine unit 8 averaged 15%, and was 0.9% relative to the entire 10-unit powerhouse during the spring (Kumagai et al. 1997).

Rocky Reach Dam has an 11-unit powerhouse oriented parallel to the river channel centerline, similar to The Dalles Dam. Juvenile salmon accumulate at the downstream half of the powerhouse between the powerhouse and a nonoverflow wall connecting the powerhouse to the west shoreline (Dauble et al. 1999). Evaluations of various prototype surface collection configurations occurred from 1995 through 2002. In 2003 a permanent surface bypass facility was constructed consisting of a single entrance, 12.2 m wide by 15.2 m deep, located immediately downstream from turbine unit 1 with a hydraulic capacity of 6,000 cfs. Most of this flow is screened and pumped back to the forebay. Fish from the surface collector and screened juvenile fish bypass system in turbine units 1 and 2 are routed across the downstream face of the powerhouse and spillway to monitoring facilities and an outfall located on the east shore. Estimated fish passage efficiency (proportion of fish passing through nonturbine routes including the surface bypass, intake screen bypass, and spillway) was 66%, 68%, 31%, and 42% for yearling Chinook salmon, steelhead, sockeye salmon, and subyearling Chinook salmon, respectively (Mosey 2003).

Lower Granite Dam has a six-unit powerhouse located immediately to the south of eight spillbays. In 1996, a prototype surface bypass collector was installed immediately upstream from turbine units 4–6, consisting of a channel 6.1 m wide by 18.3 m deep, the bottom of which was at the same elevation as the turbine intake ceiling. Attraction flow passed through various entrance configurations and was conveyed through a longitudinal channel to an adjacent spillbay. Maximum discharge through the collector was 3,500 cfs, and flow control was provided by the spillbay gate.

From 1996 through 2000, 11 different entrance configurations were tested at the Lower Granite Dam surface bypass collector to evaluate the effects of entrance characteristics, such as discharge, shape, orientation, area, velocity, and change in rate of accelerations (Anglea et al. 2002). In 1998 based on hydraulic model studies, a 4.6-m extension was installed on turbine units 4–6 to emulate hydraulic conditions observed at Wells Dam. The upper 17% of the turbine intakes was occluded, which decreased the magnitude of downward velocities at mid-depths in the forebay upstream of the collector (Johnson et al. in prep.). Also in 1998, a 330-m-long floating curtain was installed to affect the horizontal distribution of juvenile salmon in the forebay and divert them from turbine units 1–3 to units 4–6 and the collector. The height of the curtain varied from 17 to 24 m. Hydroacoustic evaluations were used to evaluate passage for the general population and compare collector to noncollector passage routes. Radiotelemetry was used to evaluate the responses of yearling Chinook salmon, hatchery and wild steelhead, and subyearling Chinook salmon to the various configurations tested.

In 1997, 12% of fish passing the dam did so through the collector, based on hydroacoustics (Johnson et al. 1998a). Similarly, from 6% to 10% of radio-tagged hatchery and wild steelhead and hatchery spring Chinook passed through the collector as compared to the total passing the dam, and from 43% to 65% of the radio-tagged fish that passed within 6 m of the entrances passed through the collector (Adams et al. 1998c). In 1998 the objective was to determine whether horizontal and vertical distribution could be modified by installation of the forebay curtain and collector modifications to increase the number of fish near the entrances and passing through the collector. More fish were observed in front of the collector and fewer fish were entrained into turbine units 4–6. Passage efficiency was from 14% to 34% depending on species, and from 14% to 51% of fish within 10 m of the entrances passed through the collector.

A total of 92%, 61%, and 67% of radio-tagged hatchery steelhead, hatchery spring Chinook salmon, and wild steelhead, respectively, that approached turbine units 1–3 stayed to the north of the forebay curtain (Adams and Rondorf 2001). Additional research was conducted in 1999 and 2000 with similar results.

Research conducted from 1966 to 2000 showed that surface flow bypass is a valid concept for Lower Granite Dam (Johnson et al. in prep.). Both spillway efficiency and passage of fish through nonturbine routes increased. For example, performance of the surface bypass collector increased spillway fish passage efficiency from 30% to 57% in 1998 (Johnson et al. 1998b). It also improved passage through nonturbine routes at the powerhouse. In 1998 an estimated 83% of the downstream migrants were guided into the bypass system relative to total powerhouse passage, and this increased to 90% when passage through the surface bypass collector was included. Also, concentrating inflow in a single surface entrance produced the optimum configuration. Finally, surface bypass efficiency alone was not high enough for it to be a stand-alone bypass system, but when used in combination with spill and the juvenile bypass system, passage past the dam through nonturbine routes exceeded 90% with more than half of these fish using the collector (Johnson et al. in prep.).

The Bonneville Dam First Powerhouse consists of 10 turbines and is separated from the spillway by Bradford Island. A prototype surface collector in front of turbine units 3–6 was installed in 1998 and evaluated to determine whether occluding the upper intake and passing high flows through a deep-slot entrance would successfully attract fish. The bottom of the collector was from 11.0 to 12.8 m below the water surface and flow through the entrances was routed into the turbine intake. Both 1.5- and 6.1-m-wide entrances were tested, which passed approximately 1,000 and 3,000 cfs, respectively.

Ploskey et al. (1999b) estimated that collector entrance efficiency was approximately 90% for both spring and summer migrants for both entrance configurations, based on hydroacoustic targets that passed into and under the prototype collector. However, while 45% of the radio-tagged steelhead and 40% of the radio-tagged yearling Chinook salmon detected upstream from the First Powerhouse passed near the prototype collector, 67% of these fish did not enter the collector and instead moved south along the face of the structure, and initially held upstream from turbine units 1 and 2 (Hensleigh et al. 1998).

In 2000 the prototype collector was expanded southward through turbine unit 1 to attempt to collect fish moving laterally (as observed in 1998). Estimated collection efficiency (the number entering the collector relative to the total number passing turbine units 1–6) was 83%, 78%, and 81% for radio-tagged steelhead, yearling spring Chinook salmon, and subyearling spring Chinook salmon, respectively. Also, 71% and 59% of the radio-tagged steelhead and yearling Chinook salmon approached more than one entrance before passing (Evans et al. 2001a). Multibeam hydroacoustic evaluations conducted to evaluate behavior in front of the collector found extensive milling of fish within 5 m of the entrances (Johnson et al. 2001).

The Dalles Dam forebay has strong lateral flows toward the west end of the powerhouse and spillway that concentrate fish into these areas. Weir entrances located at the very western end of the powerhouse effectively pass fish into the sluiceway. The combination of flow patterns and high sluiceway fish passage efficiency, in combination with the knowledge from Wanapum

and Lower Granite Dams where passage of juvenile salmon into turbines was reduced by occluding the upper portion of the turbine intakes with prototype surface bypass structures, suggested that intake occlusion at The Dalles might reduce turbine entrainment rates. In 2002 the upper 13.7 m of the turbine intakes of two fish attraction units and four main turbine units at the west end of the powerhouse were outfitted with "J" shaped occlusions. The long stem of the "J" blocked the intake and the nearly horizontal leg extended 6.1 m upstream, then upwards 3.0 m at a 90° angle. In 2002 there was no difference in turbine entrainment with the occlusions installed or removed during the spring, and turbine entrainment was actually higher during periods with the "J" device installed during the summer (Johnson et al. 2003).

Turbine priorities were different in 2002 compared to previous years, which resulted in every other turbine unit being operated. In 2002 spill passage efficiency dropped to 45% and 38% during the spring and summer, respectively, which was lower than in previous years and may have been affected by the turbine operations that year. For example, in 2000, spill efficiency was 86% and 74% during the spring and summer, respectively (Moursund et al. 2001b). The turbine operations in 2002 may have influenced turbine entrainment and the performance of the occlusions.

A new concept in surface bypass collectors, an RSW, was installed in the southernmost spillbay at Lower Granite Dam in 2001. The RSW is nearly 15.2 m wide and has a 5.9-m-long ramp extending upstream. The approach ramp was incorporated into the design based on Haro et al. (1998), where juvenile Atlantic salmon passed more readily over weirs with gradual velocity increases upstream of the crest as compared to sharp-crested weirs. The weir crest is from 3.4 to 4.6 m below normal water surface elevations and transitions to the existing ogee of the spillbay. Discharge through the weir is 7,000 cfs at minimum operating pool. The weir is designed to rotate 90° in the upstream direction to a lowered position to not reduce spill capacity during high river discharge.

In November 2001, a preliminary study using balloon-tagged hatchery yearling Chinook salmon found no significant differences in survival or injury rates between fish released into the RSW and into an adjacent spillbay that was partially-opened and outfitted with a deflector. Estimated 1- and 48-hour survival probabilities for fish passing through the RSW were 0.992 (90% CI, 0.983–1.000) and 0.981 (90% CI, 0.966–0.995), respectively. Estimated 1- and 48-hour survival probabilities for fish passing through the adjacent spillbay were 1.00 and 1.00, respectively, and the survival probabilities between the two routes were not significantly different ($P > 0.05$) (Normandeau Associates et al. 2002).

The RSW was evaluated in 2002. During the tests, spill of 7,000 cfs was provided through the weir, spill through adjacent bays was provided to establish satisfactory tailrace egress conditions, and total spill was 15,000 cfs and 23,000 cfs. During spill-to-the-gas-cap tests, spill of approximately 45,000 cfs was provided though seven spillbays 12 hours at night. Surface bypass collector components remained in turbine units 4–6, the floating curtain was attached to the southern end of the surface collector, and surface collector components north of the powerhouse that had routed collector flow to the spillway were removed.

In 2002 the bypass efficiency of the RSW (the ratio of fish passing the weir to fish passing through all routes) was 56%, 62%, and 61% for radio-tagged hatchery yearling Chinook

salmon, hatchery steelhead, and wild steelhead, respectively. The bypass efficiency of the RSW and the adjacent training spill was 78%, 73%, and 78% for radio-tagged hatchery yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively. In comparison, spill passage efficiency during the spill-to-gas-cap tests was 62%, 66%, and 54% for radio-tagged hatchery yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively. While training spill adjacent to the RSW was greater than 50,000 cfs for part of the evaluation period because of high river flow, the combined bypass efficiency of the RSW and training spill during these periods was similar to the combined bypass efficiency observed when adjacent training spill was 8,000 and 16,000 cfs (Plumb et al. 2003).

In 2003 the surface bypass collector was removed, which eliminated the intake occlusions in turbine units 4–6, and the floating curtain in the forebay was stored well upstream from the dam and had no influence on fish behavior and distributions. Flow through the RSW was again 7,000 cfs and training spill through adjacent spillbays totaled 12,000 cfs. Due to a more stable hydrograph, flow through spillbays adjacent to the RSW was more consistent than in 2002. The bypass efficiency of the RSW was 58%, 69%, and 67% for radio-tagged hatchery yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively. The bypass efficiency of the RSW and the adjacent training spill was 66%, 74%, and 71% for radio-tagged hatchery yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively. In comparison, spill passage efficiency during the spill-to-gas-cap tests was 52%, 59%, and 54% for radio-tagged hatchery yearling Chinook salmon, hatchery steelhead, and wild steelhead, respectively.

Passage through the RSW was greatest during daylight hours. Median forebay passage times during RSW tests were 1.92, 1.72, and 2.28 hours for radio-tagged hatchery and wild steelhead, and hatchery Chinook salmon, respectively. For these same test groups, median passage times during spill-to-the-gas-cap tests were 7.37, 4.64, and 4.98 hours, respectively. The estimated relative survival of radio-tagged hatchery yearling Chinook salmon passing through the RSW was 0.980 (95% CI, 0.957–1.03), compared to estimated survival through the spillway under spill-to-the-gas-cap spill of 0.931 (95% CI, 0.871–0.991). There were no significant differences between the treatments (Plumb et al. 2004).

In 2002 and 2003, acoustically tagged hatchery yearling Chinook salmon and hatchery and wild steelhead generally stayed more than 4.6 m away from the upstream face of the 24.4-m-deep forebay curtain, and tagged fish in the upper 4.6 m of the water column generally stayed upstream of the 1.2-m-deep trash shear boom. In 2002, 80% of the tagged wild steelhead that passed within 70 m, 80% of tagged hatchery steelhead that passed within 40 m, and 80% of the tagged hatchery Chinook salmon that passed within 20 m of the RSW passed through the surface weir (Cash et al. 2003).

Discussion

Surface bypass performance is a function of forebay collection, safe passage through the surface bypass entrance conveyance, and safe and rapid tailrace egress. Testing of prototype surface bypass systems during the past decade focused primarily on measuring how many fish were guided out of the forebay and into the bypass, an issue considered the greatest challenge at most sites. Results indicate that bypass guidance efficiency varied with location, and where

successful, no single feature or operation was responsible. Rather, a combination of factors appear to influence both forebay hydraulic conditions and related fish behavioral responses. These include:

1) selecting a location where fish accumulate (naturally or artificially through guidance structures);

2) developing surface-oriented entrance(s) with unimpeded surface drawdown, and open, natural lighting beyond the weir crest;

3) having a gradual, increasing velocity as flow approaches the weir crest;

4) using a floating trash boom or guidance curtain, which may improve forebay collection performance;

5) creating larger bypass flows, which increase passage efficiency but do not overcome other deficiencies or compromises; and

6) increasing discharge through turbines and the spillway, which reduces passage through surface bypass systems.

Deep-slot entrances at Wells Dam were highly effective, passing approximately 90% of yearling and subyearling migrants. Also, surface-oriented entrances with weir-type flow have historically passed a large number of fish with small quantities of flow. Surface bypass facilities that have proven successful are the Bonneville Dam Second Powerhouse sluice chute and the Lower Granite Dam RSW. The weir entrance appears to collect surface-oriented fish, but also some deeper fish move vertically to pass into the entrance (Cash et al. 2003). Furthermore, there are no signs of fish holding for extended periods upstream of the RSW as compared to observations in front of deep-slot entrances. These observations suggest that volitional passage of juvenile salmonids is more attainable over weirs than through deep-slot entrances. However, fish respond differently to hydraulic conditions in the weir entrance and approach to the weir, and favor gradual rather than sharp velocity increases (Haro et al. 1998).

In addition, data from most sites suggest that deep-slot entrances do not perform as well as surface-oriented weirs. Fish appeared to delay and avoid entering multiple, deep-slot entrances with lower velocities that ranged from 0.3 to 1.2 ms^{-1} at Lower Granite Dam and the Bonneville Dam First Powerhouse, and many radio-tagged fish moved back upstream. Multi-beam hydroacoustic studies also identified fish holding immediately upstream from the deep-slot entrances, suggesting a lack of volitional movement into this type of entrance. Also, many radio-tagged fish moved laterally rather than directly into, or under, the prototype surface collector entrances at turbine units 3–6 at the First Powerhouse in 1998. Fish behavioral investigations have not been conducted upstream of the surface bypass entrances at Wells Dam, and therefore it is not known whether delay is associated with this type of deep-slot entrance.

Surface bypass system performance is related to the proportion of flow through the system. Johnson et al. (in prep.) suggest that surface bypass flow needs to be greater than 7% of project flow to establish a large enough flow net in the forebay to be effective, based on studies at Wells Dam. However, the configuration of the Wells Dam surface bypass system is unlike any other dam. Therefore, this criterion should be used as a guide and adjusted to meet unique conditions associated with each potential surface bypass location. Performance also appears

related to total project flows. For example, at Lower Granite Dam when average project discharge was 85,000 cfs, approximately 60% of the tagged juvenile fish passed through the RSW or spillway, and this dropped to 40% when average project discharge was 110,000 cfs.

While the forebay guidance curtain at Lower Granite Dam appeared to influence horizontal distribution of fish at the powerhouse in 1998, the potential benefit appears dependent on total project discharge and project operations, and it remains to be determined as to when and under what conditions guidance curtains perform best. For example, the RSW performed better with the guidance curtain removed in 2003 than when deployed in 2002, and both shallow (1.2-m-deep trash shear boom) and deep (24.4-m-deep curtain) devices have been shown to alter fish behavior.

During the mid- and late-1990s, surface bypass designs and implementation decisions were made based on the best possible synthesis of biological and hydraulic observations. Hydraulic observations were largely from physical hydraulic models and fish behavioral information was primarily from hydroacoustic and radiotelemetry studies. Radiotelemetry provided an indication of fish location in two dimensions and to the nearest 10 m. Recently, however, an emphasis has been placed on developing the ability to track fish in three dimensions at finer spatial scales to gain an improved understanding of fish behavior; computational fluid dynamics models (with visualization software) were developed that allowed coarse- and fine-grid observation of forebay hydraulic conditions. Integrating acoustic tag and hydraulic model outputs was necessary to identify fish behaviors and optimize surface bypass system designs and performance.

Thus three-dimensional evaluations at Lower Granite Dam using acoustically tagged fish were conducted to assess fish movement under various operations and forebay hydraulic conditions and identify any previously undetected fish responses to forebay hydraulic conditions (Cash et al. 2003). Fish behaviors near both the guidance curtain and trash shear boom were noted in 2002, where fish appeared to avoid zones immediately upstream of both the 24.4-m-deep guidance curtain (first observed in 1998) and the 1.2-m-deep trash shear boom. Investigation of hydraulic conditions in the zones immediately upstream of both devices suggests these are areas where strain, as defined as the derivative of the vertical velocity component, can be detected. Nestler and Goodwin (in prep.) developed a model to analyze potential future configurations and operations using fish behavioral responses to hydraulic conditions. Preliminary analyses suggest the model can be used at larger scales, such as designing the appropriate length, depth, and location of forebay guidance curtains.

A successful surface bypass system also requires safe passage through the structure at the point where the surface bypass discharge jet enters the tailrace and during egress through the tailrace. Routing a large surface bypass discharge directly to the tailrace results in high impact velocities and potential losses associated with mechanical effects (shear, strike, abrasion) or predation. Northern pikeminnow and gulls (*Larus* spp.) may accumulate at sluiceway or bypass outfalls and shallow areas directly downstream (Hansel et al. 1993, Ward et al. 1995, Jones et al. 1997, Snelling and Mattson 1998).

In addition, northern pikeminnow can respond quickly in tailraces to changes in project operations (Faler et al. 1988), emphasizing the need to design good egress conditions in the

immediate tailrace. Surface bypass system outfalls are generally located where bypass flow quickly passes downstream to minimize fish exposure to eddies and stagnation zones. For example, radio-tagged spring Chinook migrated past a transect 1.2 km downstream of Lower Granite Dam in 19–23 minutes when passage was through the RSW with training spill, compared to some fish taking up to five times that long when passing during normal spill conditions (Plumb et al. 2004).

Development of surface bypass systems at collector dams required evaluating the feasibility of dewatering large flow volumes if surface-bypassed fish were to be transported. Mechanical screening systems of unprecedented size would be required to achieve uniform through-screen velocity distributions, and reliable debris cleaning and removal capability would have to be included in the design. Therefore, a study of high-flow dewatering and high-flow outfall alternatives was conducted for the Bonneville Dam First Powerhouse (ENSR et al. 1998). Based on the options evaluated, NOAA Fisheries Service and other salmon management agencies recommended use of high-flow outfalls to convey flow to tailraces and that further high-flow dewatering investigations be curtailed.

The high-flow outfall alternative, however, required confirmation that discharges greater than 1,000 cfs and impact velocities greater than $7.6\ \text{ms}^{-1}$ ($25\ \text{fts}^{-1}$ NMFS criteria) would not result in elevated injury rates. Based on laboratory studies, impact velocities of up to $15.8\ \text{ms}^{-1}$ at the point of jet entry produced safe conditions for juvenile salmon. Similarly, field studies at the Bonneville Dam Second Powerhouse sluice chute indicated no mortality and injury rates of 2% associated with tailrace impact velocities of $14.6\ \text{ms}^{-1}$ (Pacific Northwest National Laboratory et al. 2001). These data suggest that surface bypass system outfalls with discharges greater than 1,000 cfs and impact velocities of up to $15.2\ \text{ms}^{-1}$ can be safely designed.

Passage Through Surface Bypass Systems Conclusions

1. Results from studies of the Lower Granite Dam removable spillway weir and Wells Dam deep-slot surface bypass systems demonstrate that surface bypass is a viable concept that can produce high rates of nonturbine fish passage with a relatively small percentage of project discharge. However, poor or marginal results from other studies indicate that satisfactory results are not always achieved with surface bypass systems, and site conditions may limit performance at some locations. Also, it can take several years of testing various configurations under different flow regimes to determine the best design and project operations.

2. Optimum surface bypass performance is attained when the entrance is at a location known to attract large numbers of fish. Weir entrances have proven more successful at passing juvenile salmon and steelhead migrants than deep-slot entrances. Fish appear to pass weir-type entrances most readily if flow velocity into and through the weir increases gradually. Discharge capacity should be sufficient to create a strong upstream velocity flow-field, but high discharge alone will not overcome a marginal entrance location.

3. Surface bypass systems operated 24 hours per day reduce forebay residence times of juvenile salmon and potential exposure to predators relative to the existing program of 12-hour spill at night. At Lower Granite Dam, forebay residence times of radio-tagged juvenile salmon were

from two to four times lower with a removable spillway weir operating compared to spill-to-gas-cap operations, potentially reducing mortality due to predation in the forebay.

4. A surface bypass entrance by itself may not be enough to achieve high rates of dam passage through nonturbine routes. Additional passage facilities, such as a floating forebay curtain or upper turbine intake occlusion device, have been successful in guiding fish.

5. Successful surface bypass designs need to address guidance into the entrance, passage through the collector, discharge into the tailrace, and egress through the immediate tailrace to achieve passage performance goals of high passage efficiency and survival. Recent data indicate that for discharges greater than 1,000 cfs, impact velocities up to 15.2 ms^{-1} are safe for juvenile salmon.

6. Tracking acoustically tagged fish in three dimensions and integrating these data with hydraulic conditions from computational fluid dynamics modeling improves the understanding of fish behavior upstream of large hydroelectric dams and may increase the potential for achieving fish passage objectives.

Juvenile Salmon Passage Through Turbines

Early studies of juvenile salmon passage at FCRPS dams reported high levels of mortality (8–19%) for fish passing through turbines (Holmes 1952, Schoeneman et al. 1961, Long et al. 1968). As a result, salmon managers have focused on providing safer, nonturbine passage routes for juvenile fish to improve survival. Nevertheless, substantial numbers of juvenile fish continue to pass through turbines, and reducing turbine-related mortality remains an important factor in improving the survival of fish migrating through the hydropower system.

Salmon managers need to know current levels of mortality for different species, runs, and life history stages that NOAA Fisheries Service will use to evaluate hydropower system effects on listed stocks. Also, it is important to inform managers about whether survival through turbines can be increased by modifying current turbine operations, and whether new turbine designs hold promise for improving physical conditions within the turbine environment. In this section we discuss the methods available to estimate turbine passage survival and injury, provide estimates of fish survival through turbines made under various operations, and present results from studies conducted under COE and U.S. Department of Energy (DOE) programs to improve turbine designs and operations for fish passage.

Methods

Turbine Survival Estimates Based on Juvenile Versus Adult Returns

Holmes (1952) conducted releases of juvenile fish at Bonneville Dam from 1939 to 1945 and adult recoveries from 1940 to 1950 to test for differences in survival between spillway and turbine passage routes. He found a somewhat greater loss of fingerling Chinook salmon associated with turbine passage, and by assigning varying weights to individual observations, estimated that mortality was from 0.11 to 0.15 through the turbines and spillway mortality was from 0.03 to 0.04. However, the data were highly variable and he concluded that the observed differences between the turbines and spillway could also have been due to "chance variation in sampling where no real differences existed."

Schoeneman et al. (1961) used inclined plane traps at various locations from 32.2 to 80.5 km downstream from McNary Dam to recapture tagged juveniles and evaluate mortality. Their results were similar to Holmes', where estimated spillway mortality was 0.02 (95% CI, 0–0.04). Estimated turbine mortality was 0.13 at 80% wicket gate opening and 0.08 at 75% of gate opening. Overall, based on studies conducted at both McNary Dam and Big Cliff on the Santiam River, estimated mortality through turbines was 0.11 (95% CI, 0.09–0.13).

From 1987 to 1992, NOAA Fisheries Service released 1.5–2.2 million differentially marked upriver bright and tule stock juvenile fall Chinook salmon each year through various passage routes at Bonneville Dam to estimate short-term survival based on recovery of test fish

in the estuary, and long-term survival based on salmon caught in fisheries and adult returns to hatcheries. Sample sizes were established to detect differences of 4–5% between release locations based on an estimated adult return rate of 0.5%. However, adult returns of fall Chinook salmon to the Columbia River were much lower than expected during the test period. Recoveries of test fish averaged 0.24% for fish released in 1987, and in all other years average recoveries ranged from 0.02% to 0.04% (Gilbreath et al. 1993).

These poor return rates increased the range of detectable differences to 12–40% and eliminated our ability to make meaningful comparisons among treatments based on adult recovery data. Based on analyses of juvenile recapture data, Ledgerwood et al. (1990) found no significant difference between fish released into the turbine and immediate tailrace; however, survival of fish released through turbines was significantly different from, and approximately 91% that of fish released 2.5 km downstream from the Second Powerhouse.

These studies demonstrate the strengths and weaknesses of studies based on adult versus juvenile recapture data. Adult returns require long release and recapture periods to obtain results and include additional portions of the salmonid life cycle where biological and environmental effects can result in low returns, weakening the power to test for differences between specific treatments. To obtain tight confidence bounds about return rates for different treatments, they also require large juvenile releases because of low adult return rates. Studies where adult returns are used as the basis for comparison or adult data are used for trend analysis are presented in Williams et al. (2005). Studies based on juvenile recaptures can produce statistically valid results based on recaptures the same year, if recapture rates and sample sizes are sufficient.

Results of studies based on juvenile recapture data (Schoeneman et al. 1961, Ledgerwood et al. 1990, Muir et al. 1998, Bickford and Skalski 2000, Muir et al. 2001, Absolon et al. 2003b) indicate that juvenile recoveries a significant distance downstream from the release site can be used to estimate turbine mortality, because indirect effects from turbine passage expressed downstream of the turbine to the point of recapture are incorporated into the estimate. When treatments through turbines, spillways, bypass systems, and sluiceways are combined into a single test, relative comparisons of survival can be made across the treatments (passage routes) using juvenile recapture data.

Turbine Survival Estimates Based on Balloon-Tag, Radiotelemetry, and PIT-Tag Methodologies

Mortality associated with turbine passage has two components: direct mortality, which is expressed immediately, and indirect mortality, which is expressed downstream of the powerhouse.

In recent years, investigators have used three types of tagging methods to evaluate behavior, injury, and survival associated with turbine passage: 1) balloon tags, which are attached externally and inflate so the test fish can be recaptured by netting in the tailrace, 2) telemetry tags (acoustic or radio) that are surgically implanted into the body cavity or inserted in the esophagus, and 3) passive integrated transponder (PIT) tags, which are injected into the body cavity. Both the PIT and telemetry methods require subsequent detection downstream from the treatment area, but do not require subsequent handling once tagged.

Each method has potential biases, such as whether fish selected for tagging represent the general population (species, size, condition, etc.) and whether fish are released such that their redistribution represents the general population. In addition, researchers need to address any bias related to a specific tagging method, such as effects on fish behavior and fate (e.g., vulnerability to predation).

Looking at the relative range in turbine passage survival estimates at a variety of dams based on PIT- and radio-tag methodologies and described below, we note that in general, PIT- and radio-tag based estimates are similar. For example, Absolon et al. (2002) estimated that relative survival of PIT-tagged yearling Chinook and coho salmon through turbines at The Dalles Dam was 0.790 and 0.830 in 2000 during the daytime and evening, respectively, and the 24-hour survival estimate was 0.810 (95% CI, 0.760–0.850). This compares with an estimated yearling Chinook salmon survival of 0.869 (95% CI, 0.804–0.933) in 2000, based on radiotelemetry (Counihan et al. 2002). There was no statistically significant difference between the estimates generated by the two methods.

Comparisons of data from balloon- and radio-tag evaluations indicate differences in the survival estimates derived by these two methods may exist. At McNary Dam in 2002, NOAA Fisheries Service used radio-tagged yearling Chinook salmon to evaluate fish behavior and survival through two different turbine operations: a discharge of 11,200 cfs, which is at the upper end of the range in turbine efficiency defined as within 1% of peak efficiency, and a discharge of 16,400 cfs, which is outside the 1% range and at the position where the turbine blades are at their maximum angle to the flow. We found no statistically significant differences in fish survival between the two treatments. Estimated relative survival was 0.858 (SE = 0.034) and 0.871 (SE = 0.016) for the lower discharge, and 0.814 (SE = 0.037) and 0.856 (SE = 0.011) for the higher discharge for two detection locations 46 and 15 km downstream, respectively. These estimates were based on radio tags and included both direct and indirect effects from passing the turbines to the point of detection downstream.

Concurrent with our radiotelemetry study, Normandeau Associates et al. (2003) conducted a survival study using balloon-tagged yearling Chinook salmon. They also found no significant differences between the two operating conditions. For fish released during May, they estimated that turbine survival was 0.930 (90% CI, 0.900–0.970) and 0.946 (90% CI, 0.915–0.981) at the 11,200 cfs and 16,400 cfs operations, respectively. Test fish were released into the turbine intake and recaptured in the immediate tailrace shortly thereafter. Survival estimates using balloon tags include direct effects from passing the turbines that are expressed immediately within the tailrace prior to recapture or in the first 24 or 48 hours of post-recapture holding.

Comparison of the two methodologies was facilitated in this case because identical test conditions, time periods within the juvenile salmon migration, and release apparatus and locations were used in the studies, along with similar test fish (yearling Chinook salmon collected at McNary Dam). Comparing the two estimates provides insights into potential turbine effects expressed in the reach of river between the immediate tailrace where balloon-tagged fish were recaptured and 46 km downstream based on detections of radio-tagged fish. The balloon-tag estimates of survival (Normandeau Associates et al. 2003) were 6–7% higher than our radio-tag estimates for the 11,200 cfs operation, and from 9% to 13% higher for the 16,400 cfs operation (Ferguson et al. in prep.).

Bickford and Skalski (2000) analyzed 102 replicate releases of salmonid smolts from 53 survival investigations conducted from 1971 to 1996 at Snake and Columbia River dams and found similar results. Average survival that incorporated both direct and indirect effects from turbine passage was 0.873 through Kaplan turbines. Estimated turbine survival based on nine balloon-tag studies of direct mortality was 0.933.

In contrast, Muir et al. (1996) compared balloon-tag and PIT-tag estimates of survival for yearling Chinook salmon released at identical locations within a turbine at Lower Granite Dam. Estimated survival was 0.927 (SE = 0.027) using PIT tags and 0.940 (SE = 0.023) based on balloon tags. Thus there was no difference between the estimates derived by the two methods.

Single- and Paired-Release Protocols for Turbine Survival Estimates

Two protocols have been developed to estimate smolt passage survival. One is the release of a single experimental group of uniquely identified fish that can be detected at downstream sites. Complete capture history permits survival to be estimated based on the assumption that upstream detections have no influence on detection downstream or survival probability. This method requires detections at a minimum of two sites downstream from the treatment area. The single-release protocol allows fish tagged from the general population to redistribute after release and approach the study site in a distribution that represents the general population. Both radio- and PIT-tagged fish can be released using this protocol. Because the passage route distribution of radio-tagged fish can be determined, estimates of turbine passage survival using this methodology reflect the general population.

A second protocol is to use paired treatment and control releases, or upstream and downstream releases. Radio-, balloon-, and PIT-tagged fish all can be released using this protocol. For radio- and PIT-tagged fish, the protocol assumes the groups mix downstream and thereafter have equal survival and recapture probabilities. A release device (hose, pipe, or canister) is required for both the treatment and control releases, which forces passage distribution within the turbine to reflect the location of the release mechanism.

This is a positive aspect if specific portions of the turbine environment are being evaluated, and can be a negative aspect if the test release is designed to represent the distribution of the general population but the release mechanism is placed in a location that either does not represent the general population or the distribution of the general population is unknown. Also, careful attention should be paid to the design of the release pipe or hose. For example, the exit velocity should match the velocity of the receiving water, and test fish must be released into a hose or pipe without injury or stress, otherwise the estimate will be biased. Placement of control releases is another consideration so these releases are not preyed upon differentially by predators. Typically, control fish are released into the turbine boil "front roll" via a hose or in the immediate tailrace via a boat.

Test fish released through hoses or pipes are surface acclimated. Abernathy et al. (2001) found that juvenile salmonids acclimated to surface pressure showed little direct mortality, but injury rates increased due to expansion of gas in the swim bladders when fish were acclimated to depth. Therefore, results from studies that used surface-acclimated test fish may not incorporate the full range of mortality effects associated with turbine passage.

Results of Survival Estimates for Juvenile Fish Passing Turbines

Turbine passage survival estimates from each study conducted at each dam are provided in Table 12. Survival estimates were categorized as being "direct" if test fish were recaptured immediately (balloon tags or turbine-outlet fyke nets), and "total" if test fish were allowed to proceed downstream before detection (PIT tags or radiotelemetry) such that any delayed effects from turbine passage up to the point of detection could be incorporated into the estimate.

Operation of Existing Turbines

All mainstem FCRPS turbines are six-bladed, vertical-axis Kaplans, except for Bonneville Dam turbines which have five blades due to lower head (difference between forebay and tailwater elevations) (Figure 6). Curves relating efficiency to unit performance (power output or flow) are developed for any given head.

Turbine efficiency is thought to have a direct effect on fish passage survival. Inefficient turbine operation is a result of a poor blade-to-wicket gate relationship, where efficiency drops due to turbulence that results from the rotating machinery (hub and blades) being misaligned with the hydraulic flow field coming off the stationary but adjustable wicket gates. The relationship between survival of juvenile fish passing through Kaplan turbines is thought to be positively correlated with unit efficiency, based largely on a visual interpretation of the figures provided by Oligher and Donaldson (1966). Bell (1981) reviewed all the available data and recommended making every effort to operate turbines at peak efficiency during periods of peak fish passage to minimize fish mortality.

NOAA Fisheries Service recognized the relationship between turbine efficiency and fish survival and adopted this recommendation as part of their FCRPS BiOps (NMFS 1995, 1998b, 2000). Since 1992 turbines at COE projects have been operated within 1% of peak unit efficiency during the juvenile migration period (NMFS 1995). The COE provides the turbine operating points necessary to meet the BiOp requirement in their annual Fish Passage Plan, and any updates are coordinated with Bonneville Power Administration and salmon managers through the COE Fish Facility Operation and Maintenance Team.

Juvenile salmon mortality associated with passing mainstem Columbia River turbines is potentially caused by several mechanisms, including strike, pressure, cavitation, and shear (Ferguson 1993, COE 1995b, Cada et al. 1997). The turbine environment through which juvenile salmon pass varies physically with load, head, runner speed, and fish location within the intake. It also varies over time with environmental conditions such as temperature and fish condition. While efforts are underway to explore the relationships between these various factors and fish survival (for example, Carlson and Duncan 2003a), turbines are currently operated within 1% of peak efficiency based on the assumption that fish survival and hydraulic efficiency are correlated.

Table 12. Turbine passage survival estimates for Snake and Columbia River dams. Abbreviations: PIT = Passive Integrated Transponder, B = Balloon, C = Coded-wire tag, R = Radio, SHT = steelhead, SYCS = subyearling Chinook salmon, YCS = yearling Chinook salmon, S = Single release, P = Paired release.

Year	Report	Tag type	Survival model	Test fish	Treatment release type/location	Reference release type/location	Turbine operation	Direct survival	Total survival
Lower Granite Dam									
1988	Giorgi et al. 1988	PIT	P	YCS	Point; turbine intake	Downstream from Unit 3 turbine boil	Normal load response	Not estimated	0.831 (95% CI, 0.741–0.922)
1993	Iwamoto et al. 1994	PIT	P	YCS	Point; turbine intake	Lower tailrace midriver off juvenile bypass outfall	Normal load response	Not estimated	0.823 (SE 0.025)
1994	RMC Environmental et al. 1994	B	P	YCS	Point; turbine intake elevation 623 ft msl	Draft-tube exit	Normal load response	0.946 (90% CI, 0.955–0.992) 1 hr survival	Not estimated
1995	Normandeau Associates et al. 1995	B	P	YCS	Point; turbine intake elevation 603 ft msl Point; turbine intake elevation 603 ft msl Point; turbine intake elevation 603 ft msl Point; turbine intake elevation 603 ft msl Point; turbine intake elevation 603 ft msl Point; turbine intake elevation 603 ft msl Pooled	Draft-tube exit	18 kcfs Discharge 13.5 kcfs Discharge 19 kcfs Discharge	0.975 (90% CI, 0.955–0.992) 1 hr survival 0.975 (90% CI, 0.955–0.992) 1 hr survival 0.953 (90% CI, 0.928–0.973) 1 hr survival 0.972 (90% CI, 0.949–0.989) 1 hr survival 0.946 (90% CI, 0.922–0.965) 1 hr survival 0.949 (90% CI, 0.925–0.979) 1 hr survival 0.961 (90% CI, 0.951–0.969) 1 hr survival	Not estimated
1997	Muir et al. 2001	PIT	P	YCS	Point; turbine intake	Lower tailrace mid-river downstream of juvenile bypass outfall	Normal	Not estimated	0.927 (SE 0.027)
Little Goose Dam									
1993	Iwamoto et al. 1994	PIT	P	YCS	Point; turbine intake	Lower tailrace mid-river off juvenile bypass outfall	Normal load response	Not estimated	0.920 (SE 0.025)
1997	Muir et al. 1998	PIT	P	SHT	Point; turbine intake	Lower tailrace mid-river downstream of juvenile bypass outfall	Normal load response	Not estimated	0.934 (SE 0.016)
1997	Muir et al. 1998	PIT	P	YCS	Point; turbine intake	Lower tailrace mid-river off juvenile bypass outfall	Normal load response	Not estimated	0.920 (SE 0.025)

Table 12. Continued. Turbine passage survival estimates for Snake and Columbia River dams. Abbreviations: PIT = Passive Integrated Transponder, B = Balloon, C = Coded-wire tag, R = Radio, SHT = steelhead, SYCS = subyearling Chinook salmon, YCS = yearling Chinook salmon, S = Single release, P = Paired release.

Year	Report	Tag type	Survival model	Test fish	Treatment release type/location	Reference release type/location	Turbine operation	Direct survival	Total survival
Lower Monumental Dam									
1997	Muir et al. 2001	PIT	P	YCS	Point; turbine intake	Lower tailrace mid-river downstream of juvenile bypass outfall	Normal load response	Not estimated	0.865 (SE 0.018)
Ice Harbor Dam									
1968	Long et al. 1968	--	P	Coho	Units 1–3	Frontroll	Normal load response	0.810–0.900	Not estimated
2003	Absolon et al. 2003a	PIT	P	YCS	Unit 1	Frontroll	Normal load response	Not estimated	0.89 (95% CI, 0.84–0.94)
2003	Absolon et al. 2003a	PIT	P	YCS	Unit 3	Frontroll	Normal load response	Not estimated	0.86 (95% CI, 0.81–0.90)
2003	Absolon et al. 2003a	PIT	P	SYCS	Unit 1	Frontroll	Normal load response	Not estimated	0.89 (95% CI, 0.85–0.94)
McNary Dam									
1955	Schoeneman et al. 1961	Tattoo	P	YCS	Hose-unit not specified	Control-not specified	0.80% wicket gate opening	Not estimated	0.87
1956	Schoeneman et al. 1961	Tattoo	P	YCS	Hose-unit not specified	Control-not specified	0.75% wicket gate opening	Not estimated	0.92
1999	Normandeau Associates et al. 1999	B	P	YCS	Turbine intake upstream of wicket gate Stay vane-runner tip Stay vane-runner hub	Stay vane- mid runner blade	12.0 kcfs	0.98 (90% CI, 0.955–1.005), 1hr survival 0.98 (90% CI, 0.955–1.005), 1 hr survival 0.978 (90% CI, 0.952–1.004), 1 hr survival	Not estimated
2002	Normandeau Assoc. et al. 2003	B	P	YCS	Point release all three intake bays	Draft-tube exit	8.0 kcfs 11.2 kcfs 16.4 kcfs	0.944 (90% CI, 0.914–0.977), 1 hr survival-April 0.955 (90% CI, 0.931–0.982), 1 hr survival-April 0.930 (90% CI, 0.900–0.970), 1 hr survival-May 0.944 (90% CI, 0.914–0.977), 1 hr survival-April 0.945 (90% CI, 0.945–0.964), 1 hr survival-April 0.953 (90% CI, 0.915–0.994), 1 hr survival-April	Not estimated

Table 12. Continued. Turbine passage survival estimates for Snake and Columbia River dams. Abbreviations: PIT = Passive Integrated Transponder, B = Balloon, C = Coded-wire tag, R = Radio, SHT = steelhead, SYCS = subyearling Chinook salmon, YCS = yearling Chinook salmon, S = Single release, P = Paired release.

Year	Report	Tag type	Survival model	Test fish	Treatment release type/location	Reference release type/location	Turbine operation	Direct survival	Total survival
McNary Dam (Continued)									
2002	Absolon et al. 2002	R	P	YCS	Unit 9 point release all three intake bays	Tailrace 2 km below dam;	11.2 kcfs 16.4 kcfs	Not estimated	To 15 km downstream: 0.871 (SE 0.016) To 46 km downstream: 0.858 (SE 0.034) To 15 km downstream: 0.856 (SE 0.011) To 46 km downstream: 0.814 (SE 0.037)
2003	Peery and Bjornn 2003	R	P	SYCS	Unit 9 point release	Frontroll	Normal load response	Not estimated	0.816 (95% CI, 0.755–0.877
John Day Dam									
2002	Counihan et al. in prep. a	R	P	SHT	Rock Creek	Tailrace 1 km downstream from dam	Normal load response	Not estimated	Powerhouse 0.90 (95% CI, 0.81–0.97) Spill: 30 day/ 30 night 0.93 (95% CI, 0.85–1.00) Spill: 0 day/60 night
2002	Counihan et al. in prep. a	R	P	SYCS	Rock Creek	Tailrace 1 km downstream from dam	Normal load response	Not estimated	Powerhouse 0.97 (95% CI, 0.89–1.03) Spill: 30 day/30 night Powerhouse 0.87 (95% CI, 0.80–0.93) Spill: 0 day/60 night
2002	Counihan et al. in prep. a	R	P	YCS	Point; turbine intake U15	Tailrace 1 km downstream from dam	Normal load response	Not estimated	0.778 (SE 0.051) Spill: 0% day/ 60% night 0.832 (SE 0.042) Spill: 30% day/30% night
2003	Counihan et al. in prep. b	R	P	YCS SYCS	Point; turbine intake U4 and U15	Tailrace 1 km downstream from dam	Normal load response	Not estimated	0.820 (SE 0.043) Spill: 0% day/60% night 0.764 (SE 0.046) Spill: day/45% night 0.719 (SE 0.024) Spill: 0% day/60% night 0.722 (SE 0.024) Spill: 30% day/30% night

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Table 12. Continued. Turbine passage survival estimates for Snake and Columbia River dams. Abbreviations: PIT = Passive Integrated Transponder, B = Balloon, C = Coded-wire tag, R = Radio, SHT = steelhead, SYCS = subyearling Chinook salmon, YCS = yearling Chinook salmon, S = Single release, P = Paired release.

Year	Report	Tag type	Survival model	Test fish	Treatment release type/location	Reference release type/location	Turbine operation	Direct survival	Total survival
The Dalles Dam									
2000	Counihan et al. 2002	R	P	YCS	Point; several turbine intakes	Downstream of dam at proposed bypass outfall	Normal load response	Not estimated	0.869 (95% CI, 0.718–1.020)
2000	Absolon et al. 2002	PIT	P	YCS and Coho	Point; several turbine intakes	Downstream of dam at proposed bypass outfall	Normal load response	Not Estimated	0.790 (95% CI, 0.748–0.834) day 0.830 (95% CI, 0.785–0.878) night
				SYCS					0.791 (95% CI, 0.703–0.890) day 0.889 (95% CI, 0.790–1.000) night

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Table 12. Continued. Turbine passage survival estimates for Snake and Columbia River dams. Abbreviations: PIT = Passive Integrated Transponder, B = Balloon, C = Coded-wire tag, R = Radio, SHT = steelhead, SYCS = subyearling Chinook salmon, YCS = yearling Chinook salmon, S = Single release, P = Paired release.

Year	Report	Tag type	Survival model	Test fish	Treatment release type/location	Reference release type/location	Turbine operation	Direct survival	Total survival
Bonneville Dam First Powerhouse									
1939-1945	Holmes 1952	Fin Clip	P	SYCS	Various, upstream OR, WA, spillway, turbine	Various tailrace locations	Normal load response	Not estimated	0.88
1999-2000	Normandeau Associates et al. 2000	B	P	YCS	Stay vane-blade tip	Draft-tube exit	Original	0.947 (SE 0.0164)	Not estimated
					Stay vane-mid-blade		Kaplan	0.964 (SE 0.0144)	
					Stay vane-blade hub		6.2 kcfs	0.986 (SE 0.019)	
					Stay vane-blade tip	Draft-tube exit	Original	0.933 (SE 0.0166)	
					Stay vane-mid-blade		Kaplan	0.959 (SE 0.0137)	
					Stay vane-blade hub		7.0 kcfs	1.009 (SE 0.077)	
					Stay vane-blade tip	Draft-tube exit	Original	0.963 (SE 0.0145)	
					Stay vane-mid-blade		Kaplan	0.986 (SE 0.0106)	
					Stay vane-blade hub		10.5 kcfs	0.968 (SE 0.0106)	
					Stay vane-blade tip	Draft-tube exit	Original	0.909 (SE 0.0189)	
					Stay vane-mid-blade		Kaplan	0.968 (SE 0.0139)	
					Stay vane-blade hub		12.0 kcfs	1.004 (SE 0.0063)	
					Stay vane-blade tip	Draft-tube exit	MGR	0.955 (SE 0.0155)	
					Stay vane-mid-blade		Kaplan	0.981 (SE 0.0116)	
					Stay vane-blade hub		6.2 kcfs	0.986 (SE 0.018)	
					Stay vane-blade tip	Draft-tube exit	MGR	0.949 (SE 0.0149)	
					Stay vane-mid-blade		Kaplan	0.963 (SE 0.0134)	
					Stay vane-blade hub		7.0 kcfs	0.974 (SE 0.0144)	
					Stay vane-blade tip	Draft-tube exit	MGR Kaplan 10.5 kcfs	0.977 (SE0.0122)	
					Stay vane-mid-blade			0.977 (SE 0.0123)	
					Stay vane-blade hub			0.986 (SE 0.0119	
					Stay vane-blade tip	Draft-tube exit	MGR	0.947 (SE 0.0153)	
					Stay vane-mid-blade		Kaplan	0.977 (SE 0.0124)	
					Stay vane-blade hub		12.0 kcfs	0.980 (SE 0.0132)	
2002	Counihan et al. 2003	R	P	YCS	Point; turbine intake MGR unit	Tailrace downstream of turbine discharge frontroll	Normal load response	Not estimated	1.06 (95% CI, ± 0.057)
						Tailrace downstream of PH2 JBS outfall		Not estimated	1.01 (95% CI, ± 0.031)

Table 12. Continued. Turbine passage survival estimates for Snake and Columbia River dams. Abbreviations: PIT = Passive Integrated Transponder, B = Balloon, C = Coded-wire tag, R = Radio, SHT = steelhead, SYCS = subyearling Chinook salmon, YCS = yearling Chinook salmon, S = Single release, P = Paired release.

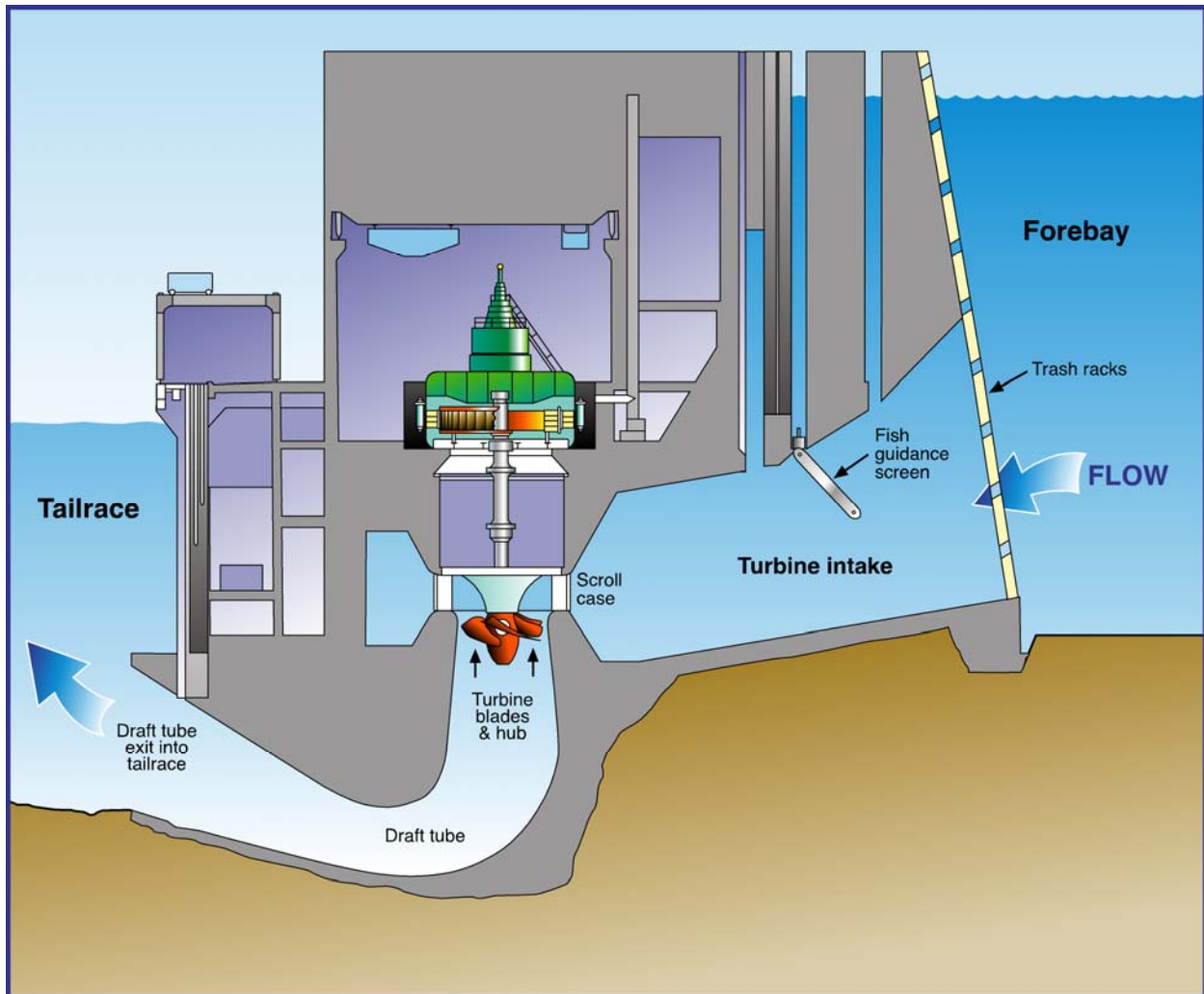| Year	Report	Tag type	Survival model	Test fish	Treatment release type/location	Reference release type/location	Turbine operation	Direct survival	Total survival
Bonneville Dam Second Powerhouse									
1988, 1989	Ledgerwood et al. 1990	CWT and Cold Brand	P	SYCS	Upper turbine — 1 m below gate slot	Tailrace 2.5 km downstream	Normal load response	Not estimated	0.91
					Lower turbine — 1 m below tip of STS				0.91
					Upper turbine — 1 m below gate slot	Frontroll			0.98
					Lower turbine — 1 m below tip of STS				0.97

Figure 6. Cross-sectional view of a Columbia River dam showing turbine passageway components (adapted from Ploskey and Carlson 2004).

Results from some recent studies do not match this basic assumption. Mathur et al. (1996, 2000) used balloon-tagged yearling Chinook salmon to evaluate turbine survival at Lower Granite Dam. They found no significant differences ($P > 0.05$) between survival estimates for fish passing through a turbine operated at the low end of the 1% efficiency range, near peak efficiency, and at a high-flow level outside the 1% range that caused cavitation. Maximum survival occurred at the low end of the 1% range. While we report the findings from several balloon-tag studies here, we have also noted above that balloon tags may not capture the indirect mortality that may occur in a tailrace and reservoir after turbine passage.

Normandeau Associates et al. (2000) evaluated direct smolt survival through Bonneville Dam First Powerhouse turbines under four discharge levels and at three different release locations within the turbine (tip, mid-blade, and hub) using balloon tags. Direct turbine survival varied more with fish distribution (release location) than discharge, and maximum survival did

not occur at peak efficiency. Normandeau Associates and Skalski (1996) evaluated direct smolt survival through Rocky Reach Dam turbines under three discharge levels and at two different release locations within the turbine (3.0 and 9.1 m below the intake ceiling). Peak survival was observed to occur within the 1% range, but did not occur at peak efficiency. In contrast, in a similar study conducted at Wanapum Dam, Normandeau Associates et al. (1996b) concluded that maximum survival occurred outside of the 1% range in efficiency.

In 2002 NOAA Fisheries Service conducted an evaluation of fish survival through McNary Dam turbines. We found no difference in fish survival between two treatments. Estimated relative survival to detection locations 46 and 15 km downstream was 0.858 (SE=0.034) and 0.871 (SE=0.016) for the lower discharge (11,200 cfs, which was within the 1% range), and 0.814 (SE=0.037) and 0.856 (SE=0.011) for the higher discharge (which was 16,400 cfs and outside the 1% range), respectively (Ferguson et al. in prep.).

Normandeau Associates et al. (2003) conducted a series of balloon-tag tests in April 2002 to evaluate direct turbine survival of Chinook salmon under four different operating levels of a single unit at McNary Dam. This evaluation provided direct survival estimates (90% CI) of 0.944 (0.914–0.977), 0.955 (0.931–0.982), 0.983 (0.957–1.000), and 0.945 (0.925–0.964) for operating levels of 8,000, 11,200, 14,000, and 16,400 cfs, respectively. None of the estimates were significantly different. Of the four operating levels, 8,000 and 16,400 cfs were outside the 1% range, while 11,200 and 14,000 cfs were within 1% of peak turbine efficiency.

Skalski et al. (2002) conducted a retrospective analysis of recent balloon-tag studies at four Snake and Columbia River dams and concluded that peak passage survival did not occur at peak turbine efficiency. They found as much as a 3.2% difference between the maximum survival and survival at peak turbine efficiency. Furthermore, a meta-analysis using results from 11 balloon-tag survival studies conducted at different sites found no association between survival and efficiency ($r^2 = 0.0311$, $P = 0.2640$).

Operating near peak efficiency offers project operators guidelines that are convenient for management purposes, but as Skalski et al. (2002) pointed out, a statistical relationship between turbine efficiency and fish survival has not been observed. They recommended that turbines be operated where fish passage survival is maximum, and not solely on peak operating efficiency. While this recommendation is valid based on their analyses, currently, operations that correspond with maximum fish survival for turbines at COE dams on the Columbia River have not been determined.

Thus given the complex and variable nature of the relationship between fish survival and turbine operations, guidelines for project operators are needed that provide the highest level of fish survival possible given the current information. Skalski et al. (2002) noted that operation of units within 1% of peak efficiency most often encompasses the turbine operation that corresponds with maximum turbine survival, which was the case at McNary Dam in 2002 (Absolon et al. 2003b).

Studies Related to New Turbine Designs

Since 1997 the COE has evaluated juvenile fish passage through Kaplan turbines with an emphasis on identifying turbine structures and operations responsible for injuring fish (COE 2003). Their Turbine Survival Program (TSP) has focused on hydraulic model studies, engineering evaluations, and biological investigations to relate survival and injury to physical conditions encountered during turbine passage.

Under the TSP, Carlson et al. (2002) compared three-dimensional tracks of juvenile steelhead and Chinook salmon with neutrally buoyant drogues passing through a turbine at McNary Dam under two different operations. They found a significant difference in the trajectories and time to pass the turbine intakes between the juvenile salmon and drogues and recommended that particles placed in physical models cannot be used to simulate the elevation component of the trajectories of smolts passing through turbines.

Carlson and Duncan (2003a) released sensor fish through turbines at McNary Dam in 2002 in conjunction with Absolon et al. (2003b) and Normandeau Associates et al. (2003). They observed several differences in the physical conditions fish were exposed to under the two different operations. Exposure to draft tube turbulence was lower in both duration and magnitude and less dependent on intake (bay) location at a turbine discharge of 16,400 cfs compared to 8,000 cfs. These observations agreed with physical model data. When compared to similar exposure metrics for spill at Bonneville and The Dalles Dams, draft tube turbulence at McNary Dam under low turbine discharge may not be a major source of injury. Acceleration impulses believed to be associated with the stay-vane-wicket-gate cascade were more frequent at the lower discharge, possibly indicating increased potential for strike, scraping, and shear.

A minimum gap runner (MGR) is a Kaplan turbine design that minimizes the clear openings between the rotating blade tip and stationary discharge ring, and between the base of the blade and the hub. This reduces the opportunity for fish to be entrained in potentially harmful areas of high velocity discharge through narrow gaps (Eicher Associates Inc. 1987, COE 1995b). This is accomplished by incorporating a spherical hub, a specially shaped runner cone that allows water passing the blades to exit with a constant velocity and without excessive turbulence, and a spherical-shaped discharge ring to close the gap at the blade tips. Minimum gap runners are being installed in all 10 units of the Bonneville Dam First Powerhouse. The efficiency curves of an MGR design are steeper and have more of a "peak" than standard Kaplan designs. Operating within 1% of peak efficiency for fish passage using this steeper curve reduces the turbine operating range.

Fish survival through a MGR turbine at Bonneville Dam First Powerhouse was tested during the winter of 1999–2000 using balloon tags (Normandeau Associates et al. 2000). The MGR (Unit 6) was compared to a standard unit (Unit 5). The blade-to-wicket-gate relationships for the standard unit were adjusted or "tuned" prior to the test. Survival of juvenile Chinook salmon through the MGR turbines was evaluated for three different release locations (tip, mid-blade, and hub) under four different discharge levels: 6,200, 7,000, 10,500, and 12,000 cfs. Survival varied more with fish distribution (release location) than discharge. For both types of turbines, survival was higher for fish released near the hub and mid-blade locations than for fish released at blade tips (Table 13).

Table 13. Ranges in estimated survival for three turbine release locations; each range represents the four turbine loads tested (Normandeau Associates et al. 2000).

Release location	MGR	Unit 5
Hub releases	0.974–0.986	0.968–1.009
Mid-blade releases	0.963–0.981	0.959–0.986
Tip releases	0.947–0.977	0.909–0.963

Maximum survival occurred at peak unit efficiency for the blade-tip and mid-blade releases. Maximum survival for the hub releases did not occur at peak efficiency, and in fact, the lowest survival was at peak efficiency. Skalski et al. (2002) conducted additional analyses of the Bonneville Dam MGR data and found no significant relationship between salmonid survival and turbine efficiency at any of the three release locations, nor between survival and average head, blade angle, discharge, or power generated. Survival and injury comparisons between the MGR unit and the standard unit indicated that MGR survival was equal to or better than the standard unit. Absolute survival for fish passing near the MGR blade tip was up to 3% higher than for the same location in the standard unit. Overall, survival was not significantly different between the units; however, the incidence of fish injury was lower for fish passing through the MGR unit (0.014) compared to the standard unit (0.025).

Since 1994 the DOE has funded studies under the Advanced Hydropower Turbine Systems Program (AHTS) to develop environmentally friendly turbines, with the goal of improving hydropower's environmental performance. Ploskey and Carlson (2004) modeled the probability of blade strike for the Bonneville Dam First Powerhouse turbines and compared these predictions to empirical data collected in 1999 and 2000 by Normandeau Associates et al. (2000). Blade-strike models predicted that the probability for injury from strike increases with decreasing unit discharge and with increasing distance from the center of the runner where the fish passes the blade. However, the predicted injury from strike was from two to five times higher than that observed in the empirical data and there was not a strong relationship between injury and discharge in the empirical data. They pointed out that the researcher's inability to assign causal mechanisms to observed injuries hinders relating injury or mortality to discharge. They concluded that efforts to empirically determine fish distributions at the turbine intake trashracks, wicket gates, and runner blades are needed before models can be used more effectively to identify turbine operations and designs that reduce injury and mortality rates.

To investigate the relative importance of pressure as a source of turbine passage mortality and injury, Abernathy et al. (2001) examined the responses of fish both acclimated to and not acclimated to gas supersaturation to rapid pressure changes. They concluded that if gas saturation is not a problem, exposure of juvenile salmonids to areas of low pressure associated with passing a turbine runner causes little direct mortality. Under high saturation levels, passage through areas of low pressure could produce injury and mortality if fish responded to supersaturation by seeking greater depths, because if fish are acclimated to depth, injury rates increase due to expansion of gas in the swim bladder.

Coutant and Whitney (2000) conducted a literature-based evaluation to develop biological information regarding normal behavioral patterns of fish for use in engineering designs to improve the survival of fish passing through turbines. They suggested that models of

fish trajectories should not assume neutral buoyancy for the period fish are passing through a turbine. This was supported by empirical results from Carlson et al. (2002), discussed at the beginning of this subsection.

Discussion

The primary question that salmon managers and the action agencies need answered is whether existing operations provide the best protection for juvenile salmon migrating downstream, or whether alternative operations are warranted based on the available data. The data in Table 12 reflect the variability associated with survival estimates through turbines relative to specific operating loads, conditions, and methods used to make the estimates. The data and analyses reported in the literature support two points: 1) operating within 1% of peak unit efficiency is a reasonable surrogate operation to maximize fish survival until additional, detailed information on the relationship between fish survival and turbine unit operations is developed, and 2) additional, rigorous biological testing will be required to determine the optimal turbine operations for each species and life history type of salmon and each set of turbines.

The need for rigorous evaluations of fish survival associated with each set of turbines presents a dilemma to salmon managers and hydro operators. The sample sizes required to distinguish small differences between test conditions can only be achieved at a high cost in terms of the number of fish required. For example, Perry et al. (2003) estimated that 244,000 subyearling Chinook salmon would be required to detect a difference between treatments of 1% using a 2-tailed test when $\alpha=0.05$ and $\beta=0.80$. This number is reduced by use of a 1-tailed test, and is significantly reduced by use of a significant t-test and by adjusting α, β, or the difference between treatments, but still remains high. For example, when $\alpha=0.05$ and $\beta=0.80$ and using a 1-tailed test, more than 21,000 subyearling Chinook salmon would be required to detect a difference of 2% between treatments with a significant t-test. Based on comparisons of balloon-tag and radiotelemetry data from studies conducted at McNary Dam in 2002 (Ferguson et al. in prep.) and by Bickford and Skalski (2000), indirect mortality can be a significant component of total estimated turbine mortality and should be incorporated into any new study designs.

Releases of treatment groups made upstream of the dam for these studies represent the distribution of the general population, but also incorporate forebay and reservoir mortality into the treatment unless these are partitioned out of the estimate. Point releases made into turbine intakes allow for specific areas of the passage environment to be evaluated, and can represent the overall population passing through the turbine if an assumption is made that the point release represents the overall turbine environment. Data presented in Table 12 present few opportunities to directly compare estimates made within the same year using both release strategies. Use of the paired-release protocol requires that reference groups be released downstream of the test turbine, typically either in the front roll of the turbine boil or 1–2.5 km downstream from the immediate tailrace.

Although few studies made both daytime and nighttime releases, the available information suggests that tailrace predation may be a component of turbine mortality if the tailrace is included in the treatment. We base this statement on trends observed in a study of survival at The Dalles Dam in 2000. Absolon et al. (2002) estimated that relative survival of

PIT-tagged yearling Chinook and coho salmon through turbines (turbine releases compared to tailrace releases made approximately 2 km downstream from the project) was 0.790 (95% CI, 0.748–0.834) for daytime releases and 0.830 (95% CI, 0.785–0.879) for nighttime releases. Estimated relative survival of subyearling Chinook salmon passing through turbines during the summer was 0.791 (95% CI, 0.703–0.890) and 0.889 (95% CI, 0.790–1.000) during the day and night, respectively. Although nighttime survival was higher during both seasons, these differences were not statistically significant.

An important finding from the COE TSP is that draft tubes were designed for specific turbine loadings, and hydraulic performance of the draft tube increases with discharge. This is part of the reason that turbulence through turbines and passage times of neutrally buoyant beads are higher at lower discharges. This observation may be of considerable importance and turbine investigators will likely focus on this in the future. In addition, the most severe hydraulic conditions occur at the trailing edges of wicket gates and runner blades, within the hub "rope," and near the leading edge of draft tube splitter walls.

Potential negative effects from fish diversion screens installed in turbine intakes on turbine passage survival is an aspect of passage through turbines that should be addressed. The survival data presented in Table 12 include screens installed in turbine intakes in most cases, the exceptions being the early studies before screens were installed starting in the 1970s and The Dalles Dam. Therefore, most recent studies incorporated effects from screens into the reported survival estimates. Because the potential effect from screens has not been partitioned from the overall estimates, little can be discussed regarding this issue. However, we note that engineering studies have shown that intake screens reduce turbine unit efficiency, which may affect juvenile salmonid survival.

Passage Through Turbines Conclusions

1. Estimates of survival through turbines from studies conducted during the period of early FCRPS development, from 1939 through 1968, ranged from 0.81 to 0.92 (Holmes 1952, Schoeneman et al. 1961, Long et al. 1968) (Table 12). Studies conducted under contemporary conditions using radiotelemetry and PIT-tag methodologies suggest mean survival through turbines generally remains in this range, but means as low as 0.72 have been reported. Therefore, turbines continue to represent a significant source of mortality for juvenile salmon migrating through the FCRPS.

2. Balloon-, PIT-, and radio-tag methodologies all have valid applications, depending on the study objectives. Radio or PIT tags should be used when the goal of the study is to incorporate both direct and indirect mortality into the survival estimate. The balloon-tag methodology is a useful diagnostic tool to examine test fish for injuries and correlate injury types with physical parameters experienced during passage, but it is not a good estimator of overall survival rates in all cases. All studies require that careful consideration be given to the use of release mechanisms, selection of representative test fish, location of treatment and control releases, and verification of test assumptions.

3. We do not recommend obtaining estimates of turbine survival based on adult returns due to large sample sizes required to detect small differences between treatments. The action agencies and salmon managers should rely on juvenile recapture data to judge turbine

passage effects and differences between treatments, as no apparent differences exist in measurements of turbine survival between studies based on juvenile recaptures or adult returns.

4. Indirect mortality can be a significant component of total estimated turbine mortality. Future efforts to improve survival through turbines have to address this component if improvements are to be made to the turbine passage route.

5. A statistically significant relationship between juvenile salmonid survival and Kaplan turbine peak efficiency for Snake and Columbia River dams has not been demonstrated. This is because it is difficult to detect small differences between test conditions under highly variable conditions. Operation of units within 1% of peak efficiency most often encompassed the operation that produced the maximum estimated survival. Thus continued operation within 1% of peak turbine efficiency is a reasonable operating guideline to follow until more detailed information on the relationship between unit operations and maximum fish survival for each family of turbines at FCRPS dams is developed. To develop more refined guidelines, additional rigorous biological testing for each set of turbines will be required. These tests appear warranted because maximum survival can be as much as 3.2% higher than survival measured at peak efficiency. This potential improvement in survival is large compared to other potential FCRPS configuration modifications to improve survival.

6. Since 1997 a significant amount of effort has gone into understanding the causal mechanisms of mortality associated with fish passage through turbines. However, we have not seen a significant reduction in turbine passage mortality associated with any new turbine designs tested to date. Additional studies will be needed and new designs developed before turbine passage survival is improved to levels similar to alternative passage routes. There are no easy solutions to improving fish passage survival through this challenging route, and efforts to do so should be viewed as a long-term process.

Adult Passage

Background

All eight Columbia and Snake River mainstem dams, from Bonneville through Lower Granite, provide upstream passage for adult salmon and steelhead through one or more fishways. The fishway systems are operated according to criteria developed by the COE, NOAA Fisheries Service, and state and tribal fishery co-managers. Each criterion is based on results of biological testing to determine the hydraulic conditions that maximize fish attraction and minimize delay. The COE annually coordinates with salmon managers and publishes the Fish Passage Plan (FPP). The FPP provides detailed operating procedures and criteria for adult fish passage facilities and special operations to accommodate research. This includes criteria on water depth and head on the entrance gates, powerhouse collection channels, floating orifices, ladder flow, counting windows, and fishway exits.

There is a significant backlog of unfunded maintenance and repair projects on entrance gates and gate lifting machinery at Snake and Columbia River dams. Also, at some Snake River project entrance gates, the auxiliary water supply (AWS) systems cannot supply sufficient attraction water to meet the required minimum of 0.3 m head differential between the adult collection channel and tailrace. This situation occurs during low tailrace conditions, when river flows are low and the downstream project is operated within the lower foot of the operating range of the reservoir.

At Lower Granite, Little Goose, and Lower Monumental Dams on the Snake River, all fishway entrances are served by a single AWS pump station. Fishway entrances located on the opposite side of the spillway from the pump station cannot satisfy the established flow criteria. During high river flow conditions, submergence and head differential criteria for south entrances at Lower Monumental and the north entrances at Little Goose and Lower Granite Dams are not met as pumps cannot get enough water into the AWSs. During low river flow, and with reservoirs at minimum operating pool, submergence criteria for south entrances at Lower Monumental Dam and the north entrances at Little Goose and Lower Granite Dams are not met.

The NMFS 2000 BiOp requires the COE to provide an emergency source of water at each project to satisfy fishway criteria if the main AWS system fails. Engineering studies have been completed for all projects except Little Goose and Lower Granite Dams. The Walla Walla District has selected a contractor to complete the design studies for these projects. John Day and McNary Dams were able to meet the BiOp requirement with existing facilities. Engineering alternatives for meeting emergency AWS capability at The Dalles Dam and Bonneville Dam Second Powerhouse are being evaluated.

Research Results and Discussion

Dam Passage

Migration behavior of adult salmonids in the Columbia and Snake River drainages has been documented using uniquely coded radio transmitters. The transmitters are implanted in the fish's esophagus or surgically inserted into the abdominal cavity and are detected using an extensive array of fixed-site antennas. Matter and Sandford (2003) compared the upstream travel times of PIT-tagged and radio-tagged spring/summer Chinook salmon from Bonneville to Lower Granite Dams in 2000. They found no evidence that radio tagging negatively affected Chinook salmon behavior.

Salmonids moving up through the Columbia and Snake rivers tend to stay near the shore. This seems to hold true for all species and runs. This trend is documented by tracking radio-tagged fish, and data from individual fish tracks indicate crossing between sides of the river or reservoir occurs at what appear to be random locations. Natural migration behavior is a consideration when designing fishway entrances and exits, and Keefer et al. (2003a) showed that existing fishway entrances located near shorelines had high net entrance rates (number of fish entering orifices minus number of fish leaving), and high fishway entrance efficiency (entrances that produce passages).

For upstream migrating adult salmonids, the first approach to a dam changes as the proportion of river discharge changes from powerhouse to spillway. With no spill, most fish approach the dam on the shoreline adjacent to the powerhouse, and as spill begins a portion of those fish move to the shoreline adjacent to the spillway. At moderate spillway discharge, more fish approach the dam at the junction of the powerhouse and the spillway. Under rare high-flow and high-spill events, when turbidity is also usually high, upstream movement of adults can slow considerably for a few days until the event subsides. Radiotelemetry studies at lower Snake River dams have indicated passage delays associated with high spill (≥40 kcfs) and highly fluctuating spill (Turner et al. 1983, 1984).

Once at the dam, fish search across the downstream face looking for passage routes. Entrance preferences are for deep/wide openings with significant attraction flow. Shallower/ smaller openings tend to have low numbers of entrances and negative net entrance rates. Median times for Chinook salmon to first enter fishways ranged from 1.9 to 2.6 hours at Ice Harbor and Lower Granite Dams in 1993 and 1994 (98.7% and 98.9% entered in less than 5 days). At Lower Monumental and Little Goose Dams, 98.5% and 98.2% entered in less than 5 days, and median times to first entries were 4.6 and 3.9 hours, respectively (Bjornn et al. 1998d).

Radiotelemetry studies in 1993 and 1994 have shown that adult fish both enter and exit the floating orifices in the powerhouse collection channels of Snake River dams. The net entry rate indicated that for fish entering the collection channel via the orifices, more fish left via the orifices than stayed in the collection channel (Bjornn et al. 1998d). The COE Fish Passage Operations and Maintenance (FPOM) committee concluded that closing the floating orifices at Snake River dams would improve the operation of the adult fishways because more water would be available for the main fishway entrances, and maintenance would be improved because the bulkhead could be sealed and the collection channels dewatered on a more frequent basis.

First entry times in the lower Columbia River are comparable to those at the three lower Snake River dams. At Bonneville Dam, Chinook salmon median first entry times were 2.0 and 2.2 hours in 1996 and 1997, respectively; steelhead median first entry times were 1.9 and 0.3 hours in 1996 and 1997, respectively. First entry times ranged from 0 (when the first at-dam record was in the collection channel) to 20 days for Chinook salmon and 17.8 days for steelhead.

Bjornn et al. (1998b) concluded that when powerhouses are not at full load, changing the end of the powerhouse where turbines were operating had little if any influence on the time for steelhead to approach, enter, or pass fishways at Snake River dams. They saw a slight change in first-approach sites, but not first-entrance locations. Bjornn et al. (1998f) did not detect any differences in Chinook and sockeye salmon or steelhead entrance locations or passage times when turbine unit 1 (and its associated discharge located near the south shore adult entrance) at John Day Dam was operated at 150 and 100 megawatts.

Fishway fences installed adjacent to the north powerhouse entrances at Little Goose and Lower Granite Dams in 1991 to reduce fallouts as fish moved upstream in the collection channel were not effective (Bjornn et al. 1995). Funneling the down-channel moving fish away from the entrances at the downstream end of the collection channels at Little Goose Dam in 1994 improved the net entrance rate at those entrances (Bjornn et al. 1998d).

A high rejection rate of the transition area between the collection channels and the fish ladders has been documented for spring/summer Chinook salmon and steelhead (Stuehrenberg et al. 1995, Bjornn et al. 1998d). For steelhead, the rejection rates range from 46% to 71% for their first approach to the transition area at Snake River dams. Similar rates were found for Chinook salmon at both Columbia and Snake River dams in 1996 (Keefer et al. 2003a).

In 1994 the University of Idaho Cooperative Fish and Wildlife Research Unit installed radiotelemetry antennas in the transition pools of the four lower Snake River projects and tracked the progress of 220–246 steelhead at each project. From 36% to 61% of the fish turned around in the transition pools, moved downstream, and exited the fishway at least once, and an additional 8–27% moved downstream in the fishway but did not exit (Bjornn et al. 1998a).

From 4% to 35% of spring/summer Chinook salmon turned around in the transition pool and headed downstream at McNary Dam and Ice Harbor Dam. At Bonneville Dam and Lower Granite Dam, 53–55% turned around in the transition pool (Keefer et al. 2003a). Similarly, fall Chinook entering fishways at lower Columbia River dams turn downstream in the transition pool more so than any other fishway segment.[7]

Several hypotheses have been offered to explain this transition and junction pool behavior: 1) velocities are too low and unsteady, 2) flow rates and velocity are inadequate to attract fish to the submerged section of the ladders and through the orifices at the base of the submerged ladder weirs, 3) seasonal and intermittent temperature gradients exist between the ladder flow and the diffuser flow, 4) high flow rates through large floor diffuser areas obscure

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[7] B. Burke, NWFSC, Seattle. Pers. commun., October 2003.

attraction flow at the base of the ladders, and 5) fish may be wary and reluctant to move into confined ladder pools.

Jumping within and from fishways is another adult passage issue. This behavior is typically associated with the transition from overflow weirs to vertical slot weirs, and with diffuser flow at the top of fishways (Dresser and Stansell 1998). This occurs primarily with steelhead at both John Day Dam fishways. However, at Lower Granite Dam faster passage times and more direct passage routes were observed for both Chinook salmon and steelhead when weir panels were in the down position (Naughton and Peery 2003).

Migration Rates

The earliest study of migration rates in the Snake River prior to impoundment was conducted from 1954 to 1957 (Oregon Fish Commission 1960). Chinook salmon migration rates averaged 17.7–24.1 km/day. Steelhead migration rates varied with the season; spring rates averaged 11.3–16.0 km/day, whereas fall rates averaged 8.0–9.7 km/day. Steelhead tagged at McNary Dam in January, February, and April migrated at average rates of 3.2, 3.9, and 12.2 km/day, respectively. Sockeye averaged 19.3 km/day to a weir at Redfish Lake in Idaho.

The effect of season and water temperature on steelhead migration rates was also seen in studies conducted from 1969 to 1971, where steelhead had summer migration rates of 10.7–16.7 km/day, and late fall migration rates as low as 0.5 km/day. Bjornn and Peery (1992) concluded that in the Snake River prior to impoundment, salmon migrated from 18 to 24 km/day, and steelhead migration rates varied with season and water temperature and ranged from 11 to 17 km/day during the spring and summer to as low as 0.5 km/day during the late fall.

Spring/summer Chinook salmon migration rates through the Snake River reservoirs in 1991–1993 ranged from 31 to 65 km/day, while migration rates through free-flowing river sections above Lower Granite Dam ranged from 10 to 30 km/day (Bjornn 1998c). Travel through the Columbia River hydrosystem was slower, with median rates of 13–33 km/day for spring Chinook in 1996–2000. Migration times from Bonneville Dam to McNary Dam ranged from 6 to 20 days for individual fish. For all groups, migration rate variation between years was correlated to mean flow and temperature; the date of migration was the most influential predictor of migration rates, followed by river discharge (Keefer et al. in press).

The median steelhead migration rate through Snake River reservoirs in 1993 was 30 km/day, while migration rates through free-flowing river sections were generally less than 11 km/day (Bjornn 1998c). Based on the passage times at the dam and faster passage in the reservoirs than in free-flowing rivers, Bjornn et al. (1999) estimated that median time for salmon to pass the four dams and reservoirs in the lower Snake River in 1993 was the same or less with dams as without the dams.

In 1998, 38 hatchery steelhead of known Snake River origin (based on PIT tags) were detected at both Bonneville and Lower Granite Dams (through 31 December 1998). Their median time for passage between the two dams was 32 days, or 14.4 km/day (ranged from 5.7 to 40.7 km/day for the 460 km reach). Also in 1998, 22 spring/summer Chinook salmon of known Snake River origin (based on PIT tags) were detected at both Bonneville and Lower Granite

Dams. The median time for passage between the two dams (a total of 460 km) was 16.4 days, or 28.0 km/day (ranged from 12.4 to 47.9 km/day).

Also in 1998, 38 fall Chinook salmon of known Snake River origin (based on PIT tags) were detected at both Bonneville and Lower Granite Dams. Of these fish (1 wild and 37 hatchery fish), the median time for passage between the two dams was 12.0 days, or 38.3 km/day (ranged from 13.9 to 51.1 km/day).[8] This suggests that the 1998 migration rate of PIT-tagged steelhead through the FCRPS falls within the range observed in the Snake River prior to construction of dams, and the migration rate of PIT-tagged spring/summer Chinook salmon slightly exceeds the range reported by Bjornn and Peery (1992) for Chinook salmon (run-type was not specified).

Matter and Sandford (2003) compared the upstream travel time of PIT- and radio-tagged adult spring/summer Chinook salmon from Bonneville Dam to Lower Granite Dam in 2000. Travel time for adult Chinook salmon outfitted with PIT tags (15.9 days) was significantly longer than passage time for fish outfitted with radio transmitters (14.1 days). The authors concluded that this difference was probably related to differences in study design or data complications.

Raymond (1964) compared the median migration timing of sockeye and Chinook salmon past Bonneville and Rock Island Dams between 1938 and 1950 when no other dams existed in the hydropower system corridor. The mean passage time between Bonneville and Rock Island Dams for sockeye salmon was 16.5 days (range 7–27 days). NOAA Fisheries Service computed the same statistic for the period between 1985 and 1999 and found a mean difference in passage time of 15 days (range 11–19 days). Quinn et al. (1997) also studied migration rates of Columbia River sockeye salmon. They found that travel time decreased from 1954 to 1994 between Bonneville and McNary Dams, but was unchanged between McNary and Rock Island Dams.

Fallback

A substantial percentage of adult salmon and steelhead passing dams have been observed to fall back through spillways and turbines at certain dams under certain conditions (Bjornn and Peery 1992). High fallback rates are usually associated with high river flows and spill, as well as the location of fishway exits relative to the spillways. For example, the south fishway at Bonneville Dam exits into the forebay at Bradford Island in close proximity to the spillway and has been associated with the highest fallback rates on the Columbia River. Bjornn et al. (1998e) found that fallback rates were 2.5–3.7 times higher for spring/summer Chinook salmon that passed via the Bradford Island fishway than the Washington shore fishway, and 16–20 times higher for sockeye salmon. When only fallbacks within 24 hours of passage are considered, 92–97% of all fish that fell back at Bonneville Dam between 1996 and 1998 had used the Bradford Island fishway (Bjornn et al. 1998e). In 1996 almost all steelhead that fell back passed via the Bradford Island fishway.

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[8] L. Stuehrenberg, NWFSC, Seattle. Pers. commun., September 1999.

Bjornn and Peery (1992) presented fallback rates from over 50 radiotelemetry studies of Chinook and sockeye salmon and steelhead at various dams from 1966 through 1985. Keefer and Bjornn (1999) estimated (based on radiotelemetry) that adult counts at Bonneville Dam were overcounted by 13.5–19.3% for spring/summer Chinook salmon from 1996 through 1998, by 4.7–8.2% for steelhead from 1996 through 1997, and by 12.6% for sockeye in 1997 when fallback and reascension were taken into account. Similarly, from 1996 to 2001, passage overestimation at lower Columbia and Snake River dams ranged from 1% to 16% for spring/summer Chinook salmon, 1% to 38% for fall Chinook salmon, and 1% to 12% for steelhead (Boggs et al. 2004).

Between 1996 and 2001, fallback rates for spring/summer Chinook salmon in the lower Columbia River ranged from 2% to 20% and from 1% to 14% at Snake River dams. Fallback rates for steelhead during that same time period ranged from 5% to 13% at lower Columbia River dams and from 2% to 9% at Snake River dams. From 1998 to 2001, fallback rates for fall Chinook ranged from 2% to 12% at lower Columbia River dams and from 3% to 58% at Snake River dams (Boggs et al. 2004).

In addition to fish that fall back within 24 hours of exiting the fishway, a large number of fallbacks occur after fish have migrated significant distances upstream. Dam operating procedures or environmental conditions at individual dams are unlikely to affect fallback associated with this behavior. The cause of this wandering behavior has not been determined. Potential causes include difficulty locating natal tributaries, natural survival adaptation, the use of hatchery broodstock not native to the drainage, and some level of homing impairment from having been transported. Mendel and Milks (1996) attributed much of the fall Chinook salmon fallback they observed to poor entrance conditions at the Lyons Ferry Hatchery and extensive wandering up and down the Snake River and into tributaries. They estimated that the fallback at Lower Granite Dam was 16–39% in 1993 and 30–41% in 1992. They noted that over 80% of the radio-tagged fall Chinook salmon that entered Lyons Ferry Hatchery each year had reached Little Goose Dam before descending downstream to the hatchery.

Keefer and Bjornn (1999) reported that for all seven dams studied in 1996, the median passage time at a dam was higher for spring/summer Chinook salmon that fell back one or more times. The impacts of increased passage times due to fallback on adult delayed mortality and reproductive success are unknown.

Survival

Losses can be broken down into two segments: 1) losses between dams, defined as the difference between fish counted at consecutive dams minus those accounted for between those dams (harvest and tributary turn off), and 2) losses between the upper dam and the spawning grounds or hatchery.

Losses between dams that are based on dam counts are susceptible to counting errors and assumptions. For example, the largest portion of overcounts at dams is likely due to multiple counts of fish that have fallen back over the dam and reascended the ladder. Because fallback rates vary with each dam, environmental conditions, and project operations, bias or error associated with dam counts is variable. Radiotelemetry provides a more accurate assessment of

actual passage rates and behavior than dam counts, although use of radiotelemetry assumes that tagged fish represent the general population and that tagging and release do not affect the behavior of the fish. Furthermore, estimates of survival based on radiotelemetry may be conservative because they include both actual mortality and unaccounted-for losses, such as tag regurgitation, unreported harvest, and tributary turnoff that is missed or not monitored. Survival estimates based on detection of PIT-tagged adults is also accurate and increasingly possible as detectors are installed at more dams.

With respect to losses between the upper dam and the spawning grounds or hatchery, little early information is available. Survival-to-spawning for spring/summer Chinook salmon was estimated at 55% for fish counted at Ice Harbor Dam from 1962 to 1968 when only one Snake River dam was present, and 46% from 1975 to 1988 when all four dams were in place, based on relating counts at Ice Harbor Dam with redd counts in Snake River tributaries (Bjornn et al. 1998g).

More recently, with all four Snake River dams in place, radiotelemetry has been used to estimate the survival of spring/summer Chinook salmon from Ice Harbor Dam to spawning grounds or hatcheries. Bjornn et al. (1995) estimated survival through this reach was 77% in 1993, 73% in 1992, and 54% in 1991 for fish that were radio tagged at John Day Dam in 1993, and Ice Harbor Dam in 1992 and 1991. Caution should be used in comparing survival between the pre-development and post-development periods because of differences in technique and methodology.

For spring/summer Chinook salmon radio tagged at Ice Harbor Dam, survival from Ice Harbor Dam to Lower Granite Dam was 80.8% and 74.4% in 1991 and 1992, respectively. For spring/summer Chinook salmon radio tagged at John Day Dam in 1993, survival from Ice Harbor Dam to Lower Granite Dam was 85.9% (Bjornn et al. 1999).

For spring/summer Chinook radio tagged at Bonneville Dam in 1996, 1997, and 1998, survival from Ice Harbor Dam to Lower Granite Dam was estimated two ways: first, for only those fish that passed the top of Ice Harbor Dam and moved upstream, and second, for all fish that approached Ice Harbor Dam (whether or not they passed the dam). Survival from Ice Harbor Dam to Lower Granite Dam for fish that passed the top of Ice Harbor Dam was 96.4%, 97.7%, and 98.4% in 1996, 1997, and 1998, respectively. Survival from Ice Harbor Dam to Lower Granite Dam for fish that approached Ice Harbor Dam was 92.8%, 95.2%, and 98.4% in 1996, 1997, and 1998, respectively (in 1998 both survival estimates were 98.4% because all fish that approached Ice Harbor either passed the top of the dam or were harvested or detected downstream.

Estimates of survival through the Snake River based on tagging conducted at John Day and Bonneville Dams probably reflect a more accurate estimate of survival for fish passing Snake River dams as they do not include effects from trapping, tagging, and release because these likely occurred prior to entering the lower Snake River.

Bjornn et al. (1998g) found that the survival of spring/summer Chinook salmon from Ice Harbor Dam to over Lower Granite Dam from 1991 to 1993 was similar between fish marked at Ice Harbor Dam with radio transmitters (83–87%) and unmarked fish (70–92%), based on

comparing radiotelemetry data with dam counts. This suggests that there was no effect on survival estimates from trapping fish at Ice Harbor Dam.

The installation of a complete PIT-tag detection system at Bonneville Dam in 2001 combined with additional detection capabilities upriver has allowed adult survival from Bonneville Dam to Lower Granite Dam to be estimated based on PIT tags. Data from transportation studies conducted by NOAA Fisheries Service shows that for spring/summer Chinook salmon adults, survival varied between 80% and 85%, with no effects of transportation as juveniles. Steelhead survival for this section was 70% for adults that were transported as juveniles and 80% for nontransported fish.

Assigning mortality associated with fallback to dam operations or behavior is difficult because some fish may have "overshot" and then returned to lower river tributaries. Bjornn (1998a) observed fallback mortality of 8% for sockeye salmon at Bonneville Dam (a species with no spawning below Bonneville Dam). Mendel and Milks (1996) estimated fall Chinook fallback mortality at 26% and 14% in 1993 and 1994, respectively, for fish that fell back through one or more of the four lower Snake River dams. This higher mortality for fall Chinook occurred during periods of no spill, when the fallback was assumed to have been through turbines. Keefer and Bjornn (1999) used radio-tagged fish known to have passed Bonneville Dam in 1996 to estimate survival to tributaries or the top of Priest Rapids Dam. Steelhead and spring/summer Chinook salmon that did not fall back over Bonneville Dam had survival rates that were 3.8–5.2% higher than fish that did fall back. Fish that did not fall back over any dam had survival rates from 3.0% to 5.4% higher than fish that did fall back.

Marine Mammal Predation

When considering all causes of salmonid decline in the Columbia River Basin, predation by marine mammals, while not considered a dominant regional cause, can be a significant local factor, especially when salmonid runs are low (NRC 1996). Marine mammals prey on salmonids near man-made structures such as dams or fish passage facilities where fish congregate, and the presence of marine mammals in the lower Columbia River and estuary during adult salmonid migrations raises serious concern for already depressed populations. However, the current precision of estimates of the percentage of salmonid spawner escapement consumed by marine mammals in the Columbia River Basin is poor. This is due in part to high temporal variability (interannual, seasonal, weekly, and day versus night foraging) and spatial variability in salmonid consumption, and the difficulties in sampling sufficiently to capture all major sources of variability.

Three species of pinnipeds—harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and Stellar's sea lions (*Eumetopias jubatus*)—occur in the Columbia River. Harbor seals are the most abundant pinniped species in the lower Columbia River, with peak haul-out counts exceeding 2,000 at Desdemona Sands, a tidal sandbar adjacent to Astoria, Oregon, which is the largest seal haul-out. However, harbor seals in the Columbia River can forage many kilometers away from haul-outs, including upriver areas and coastal regions outside the river.

Markings (including apparent tooth and claw abrasions) commonly observed on returning adult salmon are indicative of marine mammal predation. Harmon et al. (1994) estimated annual injury rates of 14–19.2% for spring and summer Chinook salmon and 5.4–14.2% for steelhead that arrived at Lower Granite Dam from 1990 to 1993. For the years 1994 through 2001, the injury rate for adult spring and summer Chinook salmon at Lower Granite Dam ranged from 15.7% to 28.6%.[9] Based on the size of some of the teeth marks, it is believed they were made by harbor seals. A NOAA Fisheries Service report to Congress (NMFS 1999b) under the Marine Mammals Protection Act concluded that pinniped populations are now abundant, increasing, and widely distributed along the West Coast, where there is a high potential for impacts on salmonid populations.

Zero Flow Operations

Powerhouse discharge can be reduced to near zero during hours of darkness to preserve water for periods of higher power demand. Although the proportions of adult fish recovered were small, Bjornn et al. (1998c) found no statistically significant evidence that reducing flows to near zero at night affected adult steelhead migration rate, the proportion of fish passing dams, the proportion of fish captured in the fishery, or the proportion of fish returning to hatcheries in 1991, 1992, and 1993. A consistent pattern of slower steelhead migration in the early and late portions of the runs was found, which was associated with warm and cold water temperatures, respectively. Adult fishways were operated continuously throughout the tests. McMaster et al. (1977) also found no difference in the migration rates of Chinook salmon and steelhead when discharge was reduced to near zero at night.

Water Temperature

Bell (1991) described the preferred temperature range and upper and lower lethal temperature limits for most salmonid species. For example, for Chinook salmon the preferred range is 7–14.5°C (45–58°F), with a lower and upper lethal limit of 0°C and 25°C (32°F and 77°F). Summer water temperatures in the Snake and Columbia rivers often exceed 21°C (70°F), and water temperatures in adult fishways can be even higher than ambient river temperatures.

River water temperatures greater than 20°C increase travel times between dams for migrating salmonids (Peery et al. 2003). Affected fish may abort their migrations or seek cooler water that may not be in the direct migration route to their spawning site. Snake River fall Chinook salmon and steelhead often slow their migration through the Columbia River and delay entering the Snake River when water temperatures are high (Stuehrenberg et al. 1978). These fish seek temporary refuges in lower Columbia River tributaries en route to their final destinations at upriver locations or hold in cooler Columbia River water off the mouth of the Snake River until Snake River water temperatures cool.

Of 125 Chinook salmon recorded at Ice Harbor Dam in 1996, 20% had been recorded in lower Columbia River tributaries (Bjornn et al. 2000). For steelhead, 65% of fish passing Lower

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[9] J. Harmon, NWFSC, Lower Granite Research Station, Clarkston, WA, and D. Marsh, NWFSC, Seattle. Pers. commun., December 2001.

Granite Dam were observed in one or more lower tributaries (Keefer et al. 2002). In both cases, peak tributary use coincided with peak mean daily temperatures at dams (approximately 22°C), while recorded temperatures in the tributaries were less than 20°C. Delays associated with high water temperatures can subsequently affect reproductive success. Of 71 radio-tagged sockeye salmon that held in the Columbia River until the Okanogan River cooled in 1992, only 24 (33.8%) survived to the spawning grounds (Swan 1994). Delay caused by high water temperatures could also impact the reproductive success of fall Chinook salmon, which spawn in the fall. The effect of delays on steelhead, which spawn in the spring, are unknown.

The NMFS 2000 BiOp requires the COE to provide water temperature control in adult fishways. The University of Idaho Cooperative Fish and Wildlife Research Unit began monitoring temperatures in the forebays and adult fishways at Ice Harbor and Lower Granite Dams in 1995. Temperature changes along the length of the fish ladders were found to be slight: less than 0.5°C with differential increases to 2°C found on a few occasions (Keniry and Bjornn 1998). Temperature differences upstream and downstream from diffusers were also minimal.

However, the use of warm surface water from reservoirs affects temperature in the entire fishway, and a discontinuity may occur at the transition pools at the base of fish ladders. Here cooler water is added to augment flow at fishway entrances. At Lower Granite Dam, this has resulted in temperatures up to 4°C warmer in transition pools than at fishway entrances (Peery et al. 2003). The COE also monitors water temperatures in the fishways at John Day Dam, which exhibits similarly steep thermal gradients near the bottom of its south fish ladder. Salmonid passage times at John Day Dam are longer as a result, and more fish exit fishways (interrupting passage) as temperatures increase, which further contributes to extended passage times (Keefer et al. 2003b).

Since 1992 Dworshak Dam has been operated to provide cold water for lower Snake River temperature control to benefit outmigrating juvenile fall Chinook salmon. Up to 1.2 million acre feet of water at temperatures from 5.6°C to 13.3°C (typically 8.9°C to 11.1°C) have been used to augment flows and reduce high water temperatures. These releases, typically made during July and August, are managed by regional salmon and reservoir managers to improve migration conditions for migrating smolts and adults.

Karr et al. (1998) estimated that because of water releases from Dworshak Dam in 1994–1996, water temperatures were 3.1–5.3°C cooler for 18–32 days at Lower Granite Dam and 2.4–3.1°C cooler for 14–43 days at Ice Harbor Dam. A literature review by McCullough et al. (2001) of the effect of temperature on salmonids found that laboratory and field studies show warmer water temperatures (>13–15.6°C) cause a detrimental effect on the size, number, or fertility of eggs. Warm water temperatures during the summer, therefore, have the potential to reduce reproductive success of salmonids.

Dauble and Mueller (1993) suggest that reducing mainstem water temperatures to below 21°C could reduce risk to populations of migrating adult salmon. It appears likely that operation of the Hells Canyon Complex has decreased late winter/spring temperatures and increased fall

temperatures in the Snake River (Ebel and Koski 1968, Graves[10]). Bjornn[11] found a similar trend when he plotted mean water temperatures at Ice Harbor Dam from August through October, and compared two periods: 1962–1968 and 1975–1989. The 1975–89 period was cooler during August, but warmer in September and October than the 1962–68 period.

Recently, comparisons have been made of the migration timing of steelhead and fall Chinook salmon from Bonneville Dam to Ice Harbor Dam, based on radiotelemetry studies conducted from 1996 to 1998 and dam counts from 1991 to 1998. Plots were made comparing date, cumulative proportional count at Ice Harbor Dam, and river temperature (turbine scroll case).

The plots indicated several general trends. First, steelhead passage at Ice Harbor Dam is slightly later than fall Chinook salmon. Second, a high proportion of fall Chinook salmon move into the Snake River when water temperatures exceed 20°C. This was particularly apparent in 1998 when over 80% of the fall Chinook salmon crossed Ice Harbor Dam before water temperatures dropped below 20°C. Third, fish appeared to move rapidly upstream through the system relative to spring/summer Chinook (also noted in Matter and Sandford 2003). If fall Chinook salmon migrations over Ice Harbor Dam are later than normal, it appears their migration dates are delayed from Bonneville Dam upstream, rather than being delayed by specific warm water conditions at the mouth of the Snake River.[12]

Dissolved Gas Supersaturation

Mortality of adult salmon from GBD has occurred in the Columbia River intermittently since the first dams were constructed. However, effects of GBD on adult salmonids are less understood than are the effects on juveniles. Relationships between TDGS exposure, GBD signs, and mortality are not defined. Depth distributions of upstream migrants are poorly documented; thus, the mitigative effects of hydrostatic compensation at ambient TDGS is unknown. Also, insufficient data exists for determining a threshold TDGS level below which successful spawning is known to occur (NMFS 1997).

Following implementation of voluntary spill to enhance dam passage for juvenile salmonids, monitoring was initiated to examine upstream migrating adult salmonids for GBD signs. Beginning in 1994, adult fish were examined annually at Bonneville and Lower Granite Dams and intermittently at Ice Harbor and Priest Rapids Dams. Under voluntary spill conditions where reservoir and tailrace TDGS is limited to 115% and 120%, respectively, signs of GBD were not seen and effects were assumed to be benign. Monitoring was also done at Three Mile Dam on the Umatilla River, Oregon. However, facilities to capture and examine adult salmonids at this site are limited. Thus the ability to obtain representative samples from locations where the greatest impacts may occur, for example downstream from McNary Dam, is limited. Also, spawning success in relation to TDGS has not been monitored.

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[10] R. Graves, NOAA Fisheries Service, Portland, OR. Pers. commun., January 2000.
[11] T. Bjornn, Idaho Cooperative Fish and Wildlife Research Unit, Moscow, ID. Pers. commun., January 2000.
[12] See footnote 11.

The only period in the 1990s when GBD signs were readily observed on adult salmonids was during the spring freshet in 1997 (the highest runoff volume year in recent history) (NMFS 1998a). However, GBD signs were documented at only a few sites and no estimates of impact could be made. During that period, TDGS downstream from Bonneville Dam exceeded 135% for 16 days and 130% for 24 days. TDGS exceeded 125% for extended periods at other river reaches.

At Bonneville Dam Second Powerhouse, daily prevalence of GBD was high for sockeye salmon (14–100% for more than 3 weeks) and steelhead (6–50% for 2 weeks). Chinook salmon during the same period showed relatively few signs, with 0–6.5% prevalence. No samples were collected from fish traversing the Bonneville Dam spillway tailrace, where TDGS was highest. No GBD signs were observed at Lower Granite Dam; however, fish were not examined until they had spent several hours in the low TDGS conditions of the fishway; GBD signs may have disappeared during fishway passage. At Priest Rapids Dam, sampling took place after TDGS decreased to moderate levels of 113–124%. In 1996, prior to spillway deflectors being installed, average TDGS downstream from Ice Harbor Dam was very high for almost one month, generally exceeding 135%. Adults examined at Lower Granite Dam showed no signs of GBD. However, this site is 172 km upstream from Ice Harbor Dam. Furthermore, TDGS in the Little Goose reservoir (59 km long) was less than 125% for all but 3 days.

Head burn (exfoliation of the skin and underlying connective tissue) on the top or sides of the heads of adult salmonids has been observed in the Snake River during periods of high spill and was thought to be a sign of GBD at one time. Head burn was considered by the Gas Bubble Disease expert panel held by NOAA Fisheries Service in 1995, but given a low research priority ranking (NMFS 1996). Efforts to address the cause of head burn were transferred to the COE Fish Passage Operations and Maintenance Coordination Team, which has discussed the potential causes of head burn. Elston (1996) conducted clinical evaluations of fish with typical head burns from Lower Granite Dam and suggested that head burns were caused by mechanical abrasion and laceration, rather than necrosis associated with subcutaneous emphysema from GBD.

Monitoring for head burn has been conducted at Lower Granite Dam by NOAA Fisheries Service since the early 1990s. From 1993 through 1999, the percentage of adult Chinook salmon with head burn ranged from 0% to 9.8%. Monitoring for head burn has been conducted at Bonneville Dam by the Columbia River Inter-Tribal Fish Commission since the mid-1990s. From 1997 through 1999, the percentage of adult Chinook salmon with head burn was less than 1%.[13] This indicates that the incidence of head burn likely is occurring during passage through the hydropower system. The exact cause of head burn remains unknown. Monitoring will continue to document the rate of injury. Research on the migration behavior of fish with head burn will be needed to relate the injury to survival. For example, Bjornn et al. (1995) found that of 66 radio-tagged Chinook salmon that were noted as having head scrapes or injuries, 38% did not migrate to known spawning areas and were classified as possible prespawning moralities.

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[13] L. Basham, Fish Passage Center, Portland, OR. Pers. commun., February 2000.

Kelts

Post-spawning, downstream migrations of steelhead (kelts) have been documented throughout the existence of the FCRPS (Long and Griffin 1937, Whitt 1954, Evans and Beaty 2000, Evans and Beaty 2001, Wertheimer et al. 2001, Wertheimer et al. 2002) and the potentially negative effects of the hydropower system on iteroparity rates have been recognized (NPPC 1986, ISG 1996). However, effects of the hydropower system on steelhead kelt survival and reproductive success are poorly understood. Only recently have studies been directed at conclusively identifying kelts (vs. prespawners or fallbacks) and documenting their downstream timing and project-specific passage. (Recent information for upper Snake River Basin areas indicates that repeat spawning rates in the Snake River probably average less than 2% [Evans and Beaty 2000].)

In 2000 the COE initiated a program to study the downstream passage behavior of kelts at Bonneville Dam. Overall, 39.5% of radio-tagged kelts passed through the spillway, 39.5% through First Powerhouse, and 21% through Second Powerhouse. At First Powerhouse, 83% (43/52) kelts passed through turbine units, 8% (4/52) passed through the prototype surface collector into the ice and trash sluiceway, and 10% were guided by traveling screens into gatewells and the juvenile bypass system. Median forebay residence times were longest at First Powerhouse (8 hours, 39 minutes), followed by Second Powerhouse (4 hours, 34 minutes), and were significantly shorter at the spillway (15 minutes) (Wertheimer et al. 2001).

In 2001 low-flow conditions provided an opportunity to evaluate kelt passage under low- or zero-spill conditions. Sluiceway passage efficiencies for kelts of 69% and 89% were estimated at The Dalles and Bonneville Dams, respectively (Wertheimer et al. 2002), and only 3% (7/212) of kelts released at Lower Granite Dam were detected below Bonneville Dam.

Studies in 2002 focused on evaluating the abundance of kelts at McNary and John Day Dams (Wertheimer et al. 2003). An estimated 14,057 kelts were present in McNary reservoir and an estimated 13,081 kelts passed John Day Dam during a 9-week period. Passage through John Day Dam under 24-hour spill (30%) and nighttime spill (0% day / 54% night) were evaluated (Table 14). At The Dalles Dam, 30%-spill kelt-passage efficiency was 99% and sluiceway efficiency increased from 64% to 89%.

In spring 2003, Boggs and Peery (2003) examined 1,838 steelhead (using ultrasound) collected from the Lower Granite Dam juvenile bypass separator and reported 93% were kelts, of which 17% were males and 83% females; 50% of the kelts were wild. They also reported that return rates for kelts PIT tagged and released in 2002 were 2.7% for kelts transported from Lower Granite Dam to the estuary, and 0.8% for kelts released in the Lower Granite Dam tailrace to migrate in river.

In 2003 mean migration rates of radio-tagged kelts were slower in Snake River reaches (32.4 km/d) than in lower Columbia River reaches (55.2 km/d) (Boggs and Peery 2003). Migration rates were positively correlated with river flow and generally increased in progressively downstream reaches ($P < 0.0001$, $R^2 = 0.63$). Of the 112 kelts released in the Lower Granite Dam tailrace, 55% were detected in the tailrace of Ice Harbor Dam and 34% were detected in the tailrace of Bonneville Dam (with no difference in the migration rates or

hydrosystem survival between hatchery and wild kelts). This compares with detections below Bonneville of 4.1% and 17% in 2001 and 2002, respectively.

Table 14. Project kelt passage (PE), guidance (GE), sluiceway (SLE), and spillway (SPE) efficiencies, sluiceway (SLF) and spillway effectiveness (SPF), at John Day (JDD), The Dalles (TDA), and Bonneville (BON) dams, 2001–2002 (Wertheimer et al. 2003).

Project	Year	N*	Spill (%)	PE (%)	GE (%)	SLE (%)	SLF	SPE (%)	SPF
JDD	2002	58	0:54	90	50	NA	NA	79	1.5:1
JDD	2002	97	30:30	95	58	NA	NA	88	2.9:1
JDD	2002	209	NA	93	46	NA	NA	87	NA
TDA	2001	28	0	64	NA	64	16.8:1	NA	NA
TDA	2001	70	30	99	NA	89	19.3:1	87	2.9:1
TDA	2002	207	37	95	NA	52	20.0:1	90	2.4:1
BON	2001	73	2	58	53 (B2)	87 (B1)	124.3:1	NA	NA
BON	2001	68	37	88	55	100	349.9:1	54	1.6:1
BON	2002	207	45	90	58	100	250.0:1	67	1.5:1

* N = number of kelts.

However, migration conditions in 2001 consisted of low flows and little spill, conditions in 2002 included BiOp spills and below-average flows with a sustained late runoff, and conditions in 2003 included BiOp spills, below-average flows, and a large freshet in late May/early June when kelts were in the lower Snake and Columbia rivers. Because migrating kelts predominately choose spill and sluiceway routes of passage when available and migrate faster with higher flows, spill, sluiceway operation, and flow augmentation appear to substantially improve kelt survival to below Bonneville Dam.

Other Adult Passage Issues

The use of individually coded radiotelemetry tags for adult fish has greatly increased the precision associated with studies of migration behavior and survival at dams and through the mainstem corridor. Individual fish have been uniquely tagged, their approach behaviors and passage over dams and through reservoirs monitored, and their run histories reconstructed. A large amount of information has been obtained and reported, but a number of uncertainties associated with adult passage at dams and through the hydropower system remain.

Adult passage through the lower river and passage efficiency at Bonneville Dam have not been fully investigated. Flow control has altered the lower river and estuarine hydrograph and potential adverse impacts include delay or increased mortality in the estuary. In addition, only fish captured in Bonneville Dam fishways are tagged and used to assess passage efficiency at this, the most downstream obstacle to passage. Studies are needed to confirm that passage efficiency estimates at Bonneville are not biased due to the exclusive use of "successful" fish (i.e., those that make it into the fishways where they are captured and tagged). This information is particularly critical if passage recorded at Bonneville Dam PIT-tag detection systems is used to compare relative SAR rates.

Limited information is available regarding the effects on adult Pacific salmon from passing through turbines. Bell et al. (1967, 1981) used mathematical formulas to estimate injury and mortality associated with turbine passage. In their formulas, the probability of strike increases with fish length. Thus we assume that mortality rates for adults passing through turbines will be higher than for juveniles. Several studies have shown mortality rates from 22% to 57% for summer steelhead passing through Kaplan turbines at Foster and Lower Monumental Dams (Wagner and Ingram 1973, Liscom et al. 1985, Buchanan and Moring 1986). The 22% and 41% mortalities observed in 1969 and 1970 by Wagner and Ingram (1973) represented only those fish with observable injuries, indicating direct strike; as such, these estimates are minimums. Theoretical strike calculations by Chet Scott, consulting engineer in the Rock Island Settlement hearing, showed 41% and 49% mortality for 0.64- and 0.76-m fish, respectively (NOAA Hydro Division memorandum, December 5, 1985[14]).

The COE and Battelle Pacific Northwest National Laboratory conducted a turbine passage survival workshop, which included a panel discussion of adult passage (Carlson 2001). Effects on adults associated with passage through turbines can affect upstream migrants of any species that "fall back" through turbines because they overshot their natal tributary or from behavior associated with searching for that stream. It can also affect steelhead kelts that have spawned and are migrating downstream to return to the Pacific Ocean. The topic has two components: what percentage of adults fall back through turbines, and what are the injury and mortality rates associated with this route of passage?

Although the rate of adult fallback at dams has been documented, the proportion that can be attributed to dam operations versus migratory behavior is not clear. A high proportion of fallback behavior is associated with the Bonneville Dam Bradford Island exit during spill. Adult salmon are shoreline-oriented and follow the Bradford Island forebay shoreline around to the spillway where they are entrained by spill. However, a substantial number of fish fall back from several miles upstream and this behavior appears unrelated to dam operations. Mobile tracking of radio-tagged fish has been conducted in the First Powerhouse forebay. This information has been modeled in conjunction with physical models of hydraulic conditions and dam operations to further understand the causes of fallback at the Bradford Island fish ladder exit. Fallback over Ice Harbor Dam of fish destined to the upper Columbia River appears more related to searching for the appropriate tributary than dam operations. Similarly, fallback of fall Chinook in the Snake River appears related to fish overshooting the entrance to Lyons Ferry Hatchery.

Despite limited information regarding cause, fallback of adult spring/summer Chinook past dams (Bonneville, The Dalles, John Day and Ice Harbor) during spill significantly reduced escapement (Keefer and Peery 2004); fallback (at Bonneville and Ice Harbor Dams) of steelhead similarly reduced escapement. On average, escapement was lower for fallback fish by 0.065 for spring/summer Chinook salmon ($P < 0.05$), 0.195 for fall Chinook salmon ($P < 0.005$), and 0.133 for steelhead ($P < 0.005$).[15] The difference in escapement multiplied by system-wide fallback

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[14] C. Ross, NMFS, Northwest Regional Office, Portland, OR. Pers. commun., August 2004.

[15] M. Keefer and C. Peery, Idaho Cooperative Fish and Wildlife Research Unit, Moscow, ID. Pers. commun., June 2004.

proportions showed reductions in overall run escapements of 1.30% (range: 0.46–2.27%), 2.26% (range: 1.32–2.91%), and 2.84% (range: 1.34–4.02%) for spring/summer Chinook salmon, fall Chinook salmon, and steelhead, respectively.

Project-specific data can show high mortality from fallback. Mendel and Milks (1996) estimated fall Chinook salmon fallback mortality was 26% and 14% in 1993 and 1994, respectively, at Lower Snake River dams; these fallbacks occurred during periods of no spill.

System-wide information shows no difference in escapement of spring/summer Chinook salmon and steelhead due to fallback during spill and no-spill periods in 2001.[16] Escapement estimates for adult steelhead from Bonneville Dam to Lower Granite Dam in 2000, 2001, and 2002, adjusted for harvest, were very similar (87.6%, 87.7%, and 88.8%) even though 2001 had very little spill at dams compared to 2000 and 2002. No differences (P < 0.05) in escapement were found for fallback of spring/summer and fall Chinook salmon with and without spill for all years (1996–2002) pooled (Keefer and Peery 2004). These similar conversion rates with and without spill may be due to so few fish falling back during nonspill periods.

There may also be increased energetic costs associated with migration delays and re-ascending dams may impact reproductive success. For example, Brown et al. (2002) showed a 62% increase in energetic demand for spring Chinook salmon in the tailrace of Bonneville Dam when compared to the forebay and a 26% increase compared to fishways.

Survival of spring and summer Chinook salmon from Ice Harbor Dam to spawning grounds or hatcheries varied between years, and was estimated to be as high as 77% and as low as 54% in 1993 and 1991, respectively. Further studies are required to resolve the variability of these preliminary observations and improve our understanding of losses that occur above Lower Granite Dam.

The potential effects of migration through the hydropower system on adult reproductive success is unknown. Successful reproduction requires a migration of both sexes to the spawning grounds, appropriate lipid reserves to carry out the necessary reproductive behavior (nest building and defense, and spawning), high gamete quality, proper embryonic development, and survival of offspring for the downstream migration. Delays, excessive energy consumption, and exposure to higher water temperatures during the migration are factors that could lead to reduced reproductive function, disease, and low quality and quantity of gametes. A literature review of the effect of temperature on salmonids by McCullough et al. (2001) found that laboratory and field studies show warmer water temperatures (>13–15.6°C) cause a detrimental effect on the size, number, or fertility of eggs.

Test fish for radiotelemetry studies in the past were selected from the entire population crossing a given dam, without knowing the source, origin, or evolutionary significant unit (ESU) of the fish tagged. This was problematic when using radio-tagged fish to determine inter-dam losses, because the origin of the fish lost between release and upstream sites was unknown. In recent years, the number of PIT-tagged adults returning to the Columbia River has increased and

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[16] See footnote 15.

this, along with the development of adult PIT-tag detection systems at key locations, will enable more precise estimates of survival and inter-dam losses for known-source fish.

Adult counting occurs at all mainstem dams to ensure, among other purposes, that fish passage facilities are operating properly. Partial hourly counts are expanded, and little counting occurs during hours of darkness. Counts include all adults passing each dam and are upwardly biased by any fish that fall back and reascend. The present counting schedule and systems meet this intended purpose. However, additional counts during winter months are now necessary for management of winter steelhead passage at Bonneville Dam and steelhead passage in the lower Snake River. The ISAB noted a number of problems associated with the precision and accuracy of adult counts at mainstem dams, especially if the data are used to make fisheries management decisions (Bisson et al. 1999).

Straying of steelhead into the Deschutes River, Oregon, has been observed during recent radiotelemetry studies. The natal origin of these fish was unknown. The behavior was observed during periods when the water of the Columbia River was warmer than the Deschutes River. Possible causes for this behavior include fish seeking the cooler water, straying behavior associated with transportation, and an evolutionary adaptation that enhances survival. Additional studies of known-source fish will be required to better understand the cause of this observed behavior.

The potential impacts adult fish passage systems have on nonsalmonid species is an uncertainty associated with salmonid passage. Concurrent with the decline of salmonid populations, Columbia River Pacific lamprey populations have also declined. Indigenous peoples from the Pacific coast to the interior Columbia River have harvested adult lamprey for subsistence, religious, and medicinal purposes for many generations (Close et al. 2002). In the Columbia River drainage, adult Pacific lamprey support fisheries that have recently experienced dramatic declines, and unprecedented regulation and concern about the status of Pacific lamprey has resulted in a recent petition to list this species under the Endangered Species Act (Moser and Close 2003).

As an anadromous species, lamprey must pass through the same hydropower system as salmonids. However, they have different migration behaviors and lack the physical swimming capabilities of salmon. Passage success studies show that although approximately 90% of the lamprey released downstream from Bonneville Dam from 1997 to 2000 reapproached the dam, less than 50% successfully passed and required a median passage time of 4–6 days (Moser et al. 2002). Radiotelemetry studies also indicated that lamprey had the least success in areas with confusing flows or structural obstacles, primarily where gratings exist on the floor of the fishway system, and at the serpentine weirs near the tops of ladders. Additional studies are needed to address whether the hydropower system is impacting lamprey recruitment, homing, and life histories.

White sturgeon are an important recreational, commercial, and cultural resource in the Columbia River Basin. However, white sturgeon populations vary considerably in abundance and age structure throughout the basin (Beamesderfer et al. 1995). The abundance and density of fish is greatest in the unimpounded river downstream from Bonneville Dam and this area supports one of the largest and most productive sturgeon populations in the world. However, the

white sturgeon population from the Kootenai River in northern Idaho has been listed as endangered since 1994, and populations in the Snake River downstream from Hells Canyon Dam appear to be persisting, but at a lower abundance than prior to impoundment (De Vore et al. 1995).

This variation in population status can be attributed to a number of factors including differences in exploitation rates and recruitment success, access to marine food resources, and suitability of hydrologic conditions and available habitats (Beamesderfer et al. 1995, DeVore et al. 1995). In particular, construction and operation of hydroelectric dams on the Columbia and Snake rivers has directly affected white sturgeon populations in several ways. Spawning habitats are reduced (Parsley and Beckman 1994), upstream and downstream passage is limited, and substantial numbers of juvenile and adult fish can be entrained and killed during dam maintenance activities. White sturgeon seldom ascend the existing fishways at the hydroelectric projects, presumably because the fish passage facilities for upstream migrating fish at Columbia River Basin dams were designed primarily for anadromous salmonids (Warren and Beckman 1993).

Little is known about sturgeon downstream passage through dams. Fisheries sampling done by the Oregon and Washington Departments of Fish and Wildlife and angler returns of tagged fish have resulted in several recaptured fish that had moved downstream past one or more dams. North et al. (1995) reports that since 1987, the Oregon Department of Fish and Wildlife has recovered 4 tagged white sturgeon that passed upstream and 74 fish that passed downstream through one or more dams. Thus construction of hydroelectric dams on the Columbia and Snake rivers may restrict movements of white sturgeon to impounded reaches, effectively resulting in a series of individual landlocked populations with potentially reduced genetic diversity (North et al. 1993). Prior to dam construction in the Columbia River Basin, white sturgeon likely responded to seasonal changes in food and habitat availability by ranging extensively between freshwater, estuarine, and marine environments. Because the physiochemical and biotic characteristics of reservoirs vary greatly within the basin, individual impoundments may not contain optimal conditions for all life stages of white sturgeon (Parsley and Beckman 1994, Beamesderfer et al. 1995). Identifying the specific factors currently influencing or limiting white sturgeon movement at dams and improving passage could enhance the productivity of white sturgeon populations.

American shad are not indigenous to the Columbia River Basin, but this species has successfully exploited the reservoir habitat currently available in the hydropower system. An estimated total of 4.6 million shad passed Bonneville Dam in 2003. A fishery on shad at The Dalles Dam resulted in an increased passage time for Chinook salmon as well as an increased rate of fallback (Jepson et al. 2003). Potential biological interactions between shad and salmonids exist but have not been studied. In particular, adult salmonid behavior in fishways may be altered when shad passage peaks occur during late spring and physical crowding in the ladders occurs due to high shad densities. Examinations of fine-scale salmonid behaviors in and around the fishways during these periods are needed to assess potential interspecific interactions.

Adult Passage Conclusions

1. Recent radiotelemetry estimates of adult salmonid survival from Ice Harbor to Lower Granite Dam on the Snake River have been generally well over 90%. Recent PIT-tag estimates of survival from Bonneville Dam on the lower Columbia River to Lower Granite Dam have been over 80% for spring/summer Chinook salmon and 70–80% for steelhead. For adult salmonids that do not fall back at dams, estimates of survival range from 3% to 5% higher than for those that do fall back.

2. Delays in dam passage for upstream migrating adult salmonids are associated with high levels of spill (≥40 kcfs), highly fluctuating spill, and high turbidity. Fishway entrance preferences are for deep/wide openings with significant attraction flow. The transition areas between collection channels and ladders are the fishway segments where adult Chinook salmon and steelhead are most likely to turn around and exit the fishway.

3. Overall travel times and migration rates of radio- and PIT-tagged adult salmonids from Bonneville Dam to upriver Columbia or Snake River sites vary by species/run, time of year, and environmental conditions. While delay of adult salmonid passage at Columbia and Snake River dams has been documented, recent estimates of travel times and migration rates are similar to times and rates observed during the early development of the FCRPS when there were fewer dams and the Snake River was unimpounded.

4. Fallback rates vary by species/run, environmental conditions, and dam. Higher fallback rates are associated with high river flows and spill, and with adult fishway exit locations in close proximity to spillways. Fallbacks result in an overestimation of adult counts at fish ladder count windows.

5. Powerhouse zero-flow operations at night do not appear to affect migration rates and passage of adult salmonids.

6. For adult salmonids, elevated river water temperatures (>20°C) increase travel times between dams and passage through fishways at dams. Warm water temperatures during the summer, which often exceed 20°C in the Columbia and Snake rivers, also may reduce reproductive success of salmonids.

7. Although monitoring of adult salmonids for signs of GBD has occurred at Columbia and Snake River dams since 1994 (including periods of very high TDGS), very few signs have been observed. The relationships between TDGS exposure, GBD signs, mortality, and spawning success are not well understood.

8. Estimates of repeat spawning rates for steelhead kelts that migrate downstream through the Snake and Columbia rivers are low (≈2%). While potentially negative effects of the hydrosystem on iteroparity rates have been recognized, the magnitude of the effects on survival and reproductive success are not clearly understood. Higher flows and spill significantly reduce travel and passage times for downstream-migrating kelts.

9. Adult survival above the hydrosystem to hatcheries or spawning grounds is variable and estimates from Ice Harbor Dam using radiotelemetry range from 54% to 77%. The effects of passage through the hydrosystem on reproductive success are unknown.

10. The impacts of predation by marine mammals on Columbia River Basin salmon populations are not well understood. In general, very little is known regarding how the existence and operation of the FCRPS may have altered the historical habitat and salmonid consumption rates of marine mammals.

11. Relative to salmonids, upstream passage at Columbia and Snake River dams is poor for adult Pacific lamprey and white sturgeon. American shad have thrived in the hydrosystem habitat, but the effects of the biological interactions between adult shad and salmonids (in particular with regard to extremely high fish densities in the fishways during late spring) have not been studied.

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