Post-release Survival of Chinook Salmon and Steelhead Trout from an Experimental Commercial Fish Trap in the Lower Columbia River, WA.

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Abstract

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Gillnets and other conventional harvest techniques utilized in mixed-stock commercial salmon fisheries frequently result in bycatch mortality. In many parts of the U.S Pacific Northwest, bycatch and hatchery impacts impede the recovery of Endangered Species Act (ESA)-listed salmonids and constrain commercial fishing opportunities. For the benefit of wild salmon, threatened ecosystems, and coastal fishing communities, a post-release survival study was conducted on the lower Columbia River to evaluate the potential of an alternative commercial gear—an experimental pound net trap—as a live-capture, stock-selective harvest technique. Expanding upon a 2016 pilot study, a modified trap was constructed and operated under a variety of tidal stages, light levels, and weather conditions between August 26 and September 27, 2017. Utilizing a paired mark-release-recapture procedure with Passive Integrated Transponder (PIT) tags, post-release survival from the trap was estimated through the Cormack-Jolly-Seber method; catch-per-unit-effort (CPUE) and covariates of CPUE were analyzed through Generalized Linear Models (GLM). Results demonstrated that pound net traps can effectively target commercially viable quantities of hatchery reared fall Chinook (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*) while reducing cumulative bycatch mortality of ESA-listed species relative to conventional and alternative commercial gears. During the study period, 7,129 salmonids were captured and released. The ratio of wild to hatchery-origin salmonids captured was approximately 1:3. Cumulative survival to McNary Dam ranged from 94.4% for steelhead trout (*O. mykiss*) to 99.5% for Chinook salmon, warranting application of the gear as a stock-selective harvest tool in commercial salmon fisheries.

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Chapter 1: Background

1.1 The Bycatch Problem

In rivers, estuaries, seas, and oceans around the globe, the ecosystem hosts a variety of fish species and unique populations groups—commonly known as fish "stocks"—that coexist in sympatry (Knudsen et al. 2000). Commercial fishermen utilize specialized tools, or fishing gears, to target fish stocks that are deemed desirable through market forces for harvest and profit (NMFS 2011). In their efforts to capture specific stocks of commercial value, almost all fishermen encounter other species that are present within the ecosystem regardless of a gears specialized intent. These fisheries in which multiple stocks are encountered in a geographical region by a specified gear-type are labeled "mixed-stock" fisheries (Lloyd 1996; Knudsen et al. 2000).

Bycatch inevitably occurs in mixed-stock fisheries when fishermen capture non-target stocks or species that may "drop-out" during the fishing process or be intentionally discarded and returned to the ecosystem (NMFS 2011). Fishermen may choose to discard components of their catch if certain species, sizes, or sexes are not profitable, or if government regulations prohibit retention. In instances where a fishing gear inflicts little damage to species encountered or all stocks are of sufficient health to sustain fishery impacts, bycatch may not pose a substantial risk to a fishery or ecosystem. However, any mixed-stock fishery that contains a weak stock—a population or population group that is severely reduced from environmental, ecological, or anthropogenic pressures—may inflict detrimental impacts to an ecosystem if a fishing activity causes significant bycatch mortality (Lloyd 1996; Gayeski et al. 2018). The severity of a fishery's bycatch impact is the product of the quantity of bycatch encountered and the bycatch mortality rate inflicted by the gear in use. In some regions of the world where species or populations of evolutionary importance are threatened with the prospect of extinction, bycatch impacts may be significant enough to extinguish renewable resources, alter ecosystem dynamics, and close regional fisheries of substantial economic, cultural, and spiritual importance (Kappel 2005; Lichatowich 2013).

1.2 Pacific Northwest Salmonid Decline

Like many renewable resources throughout history, Pacific salmonids (genus *Oncorhynchus*) were once believed to be inexhaustible in waters of continental North America (Higgins 1928; Lichatowich et al. 1999). Five species of salmon—Chinook (*O. tshawytscha*), coho (*O. kisutch)*, chum (*O. keta)*, sockeye (*O. nerka*), and pink (*O. gorbuscha*)—and two species of trout—steelhead/rainbow (*O. mykiss*) and cutthroat (*O. clarki)*—once inhabited the west coast from southern California to northern Alaska in robust numbers (Lichatowich et al. 1999; Quinn 2005). Across the broad geographic range of Pacific salmonids, few rivers matched the natural productivity of the Columbia River; the largest river feeding the North American West Coast (Benke and Cushing 2005). Prior to the arrival of Europeans to the U.S Pacific Northwest, an estimated 8 to 35 million anadromous salmon and trout migrated annually from marine rearing locations in the Pacific Ocean up the Columbia River to spawn throughout the basin (Scholz et al. 1985; Chapman 1986). Indigenous peoples harvested these abundant resources for thousands of years with little impact on the resiliency of native fish populations (Arnold 2011).

The status of Pacific Northwest salmonids changed abruptly with the advent of the salmon canning industry in the 1860s. Leaving depleted New England waters behind, George and William Hume established the first salmon cannery in the Sacramento River in 1864 (Lichatowich 1999). In tandem with urbanization, mining, deforestation, agriculture, and the construction of dams, the salmon canning industry nearly extirpated wild salmonids of the Sacramento River in a matter of years from overfishing of targeted Chinook salmon. This encouraged expansion of the industry to the newly admitted State of Oregon and the banks of the Columbia River. With the arrival of the Hume brothers in Astoria, OR in 1866, the first major industry of the State was created, and the rapid decline of the region's salmonids began (Higgs 1982; Lichatowich et al. 1999; Arnold 2011).

From the construction of the first cannery and pack of 4,000 cases of salmon on the Columbia River in 1866, the industry grew unsustainably with little to no resource management (Licatowich et al. 1999). By 1883, there were 39 canneries packing 630,000 cases of salmon on the river (Cobb 1930; DeLoach 1939). With such intense fishing pressure and habitat loss from upriver mining, deforestation, and agriculture, settlers and managers of the region quickly noticed a marked decline of salmon and steelhead stocks within the river basin (Baird 1875).

The spring Chinook salmon stock was first to collapse due to market demand and laissez-faire resource management (Lichatowich 1999). Initially targeting spring Chinook salmon with gillnets, harvest efficiency grew with the development of new technologies adopted from indigenous peoples and other regions of the globe including fish wheels, seines, and fish traps (Figure 1-1) (Arnold 2011). Through the combined use of these fishing gears, overfishing of target stocks was rampant and bycatch mortality of less desirable stocks was high (Lichatowich 1999). Decimating spring Chinook salmon, the industry began targeting the next most profitable salmonid stocks of the river. Overharvest shifted to summer Chinook, then to steelhead, sockeye, fall Chinook, coho, and chum salmon. Throughout this era, pressure on wild salmonids mounted not only from overharvest, but from bycatch mortality and forces outside the fishery which reduced the quality and quantity of habitat for Pacific salmonids (Meehan 1991; NRC 1996).

Figure 1-1. Salmon brailed from a fish trap in the 1920s. Photo courtesy of UW Library Special Collections.

Once one the most productive salmon rivers in the world, the Columbia River was severely depleted by the 1930s. With construction of major dams on the mainstem in 1933 and 1937, the river experienced further marked declines in returns of diminishing anadromous salmonid stocks (Lichatowich 1999). By this time, various populations had been extirpated with the remaining runs likely representing less than 1/10th of historical abundance (Chapman 1986; Lichatowich 1999). Across the region, the story was nearly the same. Primarily a result of the excessive rise of industry and laissez-faire management from the 1870s through the 1930s, salmonids rapidly declined and were extirpated within nearly 40 percent of their historical range in the U.S Pacific Northwest (Nehlsen et al. 1991; Anderson 1993). Of the stocks that still remain today, many are now listed under the U.S Endangered Species Act (ESA) (NOAA 2014). Furthermore, once prosperous coastal fishing communities are constrained by low returns of salmon and ESA conservation and management measures.

1.3 Gear Conflicts and Politics Within the Fishery

From the onset of the industrial salmon fishery in the Pacific Northwest, fingers were pointed at competing users, placing blame on one group or another for the decline of the resource or inequitable distribution of economic benefits (Johnson et al. 1948; Higgs 1982). Gillnetters accused operators of fish traps and fish wheels for the loss of the resource. The great efficiency of these technologies, they claimed, was cause for alarm and strict regulation from management. On the other hand, operators of fish traps and fish wheels blamed the gillnetters for the problem (Johnson et al. 1948). With up to 2,500 gillnetters in operation on a given year and over 850 miles of net deployed in the river, fishing effort was likely excessive from this user group (Higgs 1982). Furthermore, gillnets outnumbered traps and fish wheels by a ratio greater than 10:1. Despite biased claims on all sides of the argument, it was the sheer intensity of total fishing effort, bycatch mortality, and waste at the canneries that was reducing wild salmonid stocks and fishery revenues after the peak of the fishery in 1883 (Cobb 1930; Higgs 1982; Lichatowich 1999). Throughout this period of rapid depletion, resource managers failed to address the open access resource problem through regulation and enactment of escapement goals, and instead focused on hatchery production to sustain the industry (Baird 1875; McDonald 1895; Cobb 1930).

In the midst of the salmon crisis, the gillnetting community was first to be heard by politicians in the newly formed states of Oregon and Washington. Through their large numbers and political might, the gillnetters (accompanied by the recreational fishing community) successfully framed the problem of salmon decline on greatly outnumbered operators of fish wheels and fish traps in both the Columbia River and Puget Sound—the new epicenter of Pacific salmon fishing by the 1890s (Higgs 1982). Increasingly, operators of traps and fish wheels suffered from taxes and regulations that began in 1893; gillnetters could operate freely without being subject to penalties or fees of any sort (Washington State Session Laws, 1893, pp. 15–18).

Figure 1-2. Salmon brailed from a fish trap in Puget Sound. Photo courtesy of UW Library Special Collections.

Hostilities between users of each gear mounted as fish traps were utilized as the primary tool for massive salmon canning corporations, including Pacific American Fisheries, Inc. (Radke et al. 2002; Arnold 2011). With decades of trial and error in salmon fisheries across the region, fish trap technology had evolved to become the most efficient method developed for the harvest of salmon and described by some as “the perfection of methods for catching fish” (Buschmann 1903, pg. 18-19). On multiple occasions, individual fish traps in Alaska harvested 1.2 million salmon in a single season (McDonald 1892; Hofstad 1939). Corporations adopted preference for fish traps to maximize efficiency and reduce labor costs across the West Coast (Radke et al. 2002). In the Columbia River from 1928 through 1934, catch-per-license of fish traps outnumbered that of the gillnetters by approximately 3:1 while generally requiring fewer hands on deck (Johnson et al 1948). Fishermen and the public began to view fish traps as a means for corporations to monopolize the fishery (Higgs 1982). In an era of trust-busting in the early 20th century and rising unemployment during the Great Depression, outnumbered fish trap operators became easy political targets and calls for an outright ban of the technology were heard in Washington, Oregon, and the Alaskan Frontier (Mackovjak 2013). During this era, “fish trap pirates”—typically gillnetters who robbed the catch of trap operators—were commonly regarded as heroes in the eyes of the public. Glorification of the illegal actions taken by fish trap pirates and tensions within the Pacific Northwest salmon fishing industry are exemplified in the Hollywood film, *The Spawn of the North* (Hathaway 1938). Throughout this time, trap operators frequently hired guardsmen equipped with rifles to defend their property; exchanges of gunfire and violence were common (Johnson et al. 1948; Arnold 2011).

A groundbreaking bill was passed by Washington State legislators in 1934 that changed the course of salmon fisheries in the Pacific Northwest for decades to come. Fish traps and all other fixed-gears were officially banned in Washington State waters; seines were dramatically limited to use in specific regions of the State (Washington State Session Laws, 1935, pp. 3-8). This landmark decision made the gillnet the primary legal gear for commercial harvest of salmon on the Washington-side of the Columbia River. It marked an attempt by legislators to conserve salmon and steelhead through the reduction of total commercial catch while improving prospects for gillnetters (Johnson et al. 1948). Momentum of the gillnet lobbyist followed in Oregon and Alaska (Figure 1-3) where fish traps were banned in 1948 and 1959 respectively (Johnson et al. 1948; Arnold 2011). In these decisions, legislators cast their vote for equity and conservation, but unwillingly fostered a growing interception externality, economic inefficiency, and bycatch mortality; furthermore, they failed to recognize that overfishing and the decline of the resource could not be adequately addressed in the absence of escapement goals (Johnson et al. 1948; Higgs 1982).

Figure 1-3. Campaign to ban fish traps. This controversy in-part led to Alaska's statehood in 1959 (Russ Dow Collection, University of Anchorage Alaska).

1.4 The Reign of Commercial Gillnetting and an Open Access Resource Problem

The eighty-year reign of commercial gillnetting began on the Washington-side of the Columbia River in 1935, but prospects for the State's fishermen continued to decline. With fish traps and fish wheels removed from the northern shore of the river, it was widely assumed that overall fishing effort would decrease, reducing pressure on wild salmonids and increasing catch for individual gillnet fishermen of the State (Johnson et al. 1948). Nevertheless, the results of the fixed-gear ban proved contrary to the intent of Washington State legislators. Analyzing data from the Washington State Department of Fisheries and Oregon Fish Commission from 1928 through 1946, Johnson et al. (1948) documented a gradual decline in salmon and steelhead escapement in the Columbia Basin. Furthermore, total catch on the Washington-side of the river plummeted by approximately 45% with gillnet landings remaining mostly unchanged. The primary result of Washington State's legislation was "benefit to the large gillnet fleet [of Oregon]…which compensated for the landings no longer made by Washington's fixed-gear" (Johnson et al. 1948, pg. 22). Further decreasing long-term profits within the fishery, the advent of the internal combustion engine and elimination of fixed-gears in terminal river and stream locations encouraged gillnetters and coastal trollers to move farther and farther away from the mouths of tributaries and rivers. This calculated effort to intercept another fisherman's catch was the logical profit maximizing decision of each fisher; however, the interception externality resulted in additional unnecessary costs and lost economic benefit to society. Overall, Higgs (1982, pg. 9) estimates that society could have saved about five-sixths of the total costs of salmon harvest annually since 1934 through the "simple expedient of outlawing the relatively unproductive gear [gillnets] rather than the relatively productive gear [fish traps]."

The fishery during these times roughly represented an open access resource problem—commonly described in the field of micro-economics—where public ownership and a lack of regulation results in overproduction and inefficiency (Gordon 1954; Hardin 1968). Where there is a lack of private ownership and multiple users of the commons, a condition of mutually imposed externalities develops in a market, causing underinvestment in the preservation of a resource and overexploitation. Since the fisher receives all the proceeds from the sale of additional harvests while the effects of overexploitation are shared by all agents, there is incentive for each individual agent to over-fish as long as the price exceeds private marginal cost (Hardin 1968). The net revenue loss for all fishers from additional harvest effort is not fully considered, as only the individual fisher's share is internalized. As technologies advance and even more agents enter the market, intercept one-another's catch, and expand operations, the fishery begins to decline from decreased ecosystem functionality and overexploitation. Even if fishers recognize the decline of the resource, it remains in their best interest to maximize harvest. In the long run, the fishery collapses making fishers, processors, and society worse off. Garret Hardin (1968) branded this economic outcome "the tragedy of the commons".

1.5 Hatchery Production

To mitigate dramatic declines in salmon and steelhead stocks and improve commercial fishing opportunities with few harvest restrictions, Washington and Oregon turned to a recent technological innovation to increase fish production within the river. Since the 1870s, resource managers had experimented with hatchery production with varying perceived levels of success

(Baird 1875). Hatcheries are concrete installments located on the banks of a river or stream equipped with protected pools and controlled environmental conditions for egg incubation and juvenile salmon rearing (Figure 1-4). Understanding that wild salmon survival is lowest during incubation and early freshwater rearing, hatchery managers could spawn chosen adult mates, incubate eggs in shallow temperature-controlled trays, and raise hatched juveniles in protected pools to increase salmon survival at the most vulnerable life-history stages (Naish et al. 2007). By increasing the number of salmon that survive to hatch and migrate downstream, resource managers believed greater numbers of adult salmon would then return as adults to rivers and streams and the immense fishing effort of the late 19th and early 20th centuries could remain nearly unregulated in perpetuity (Baird 1875; Lichatowich et al. 1999).

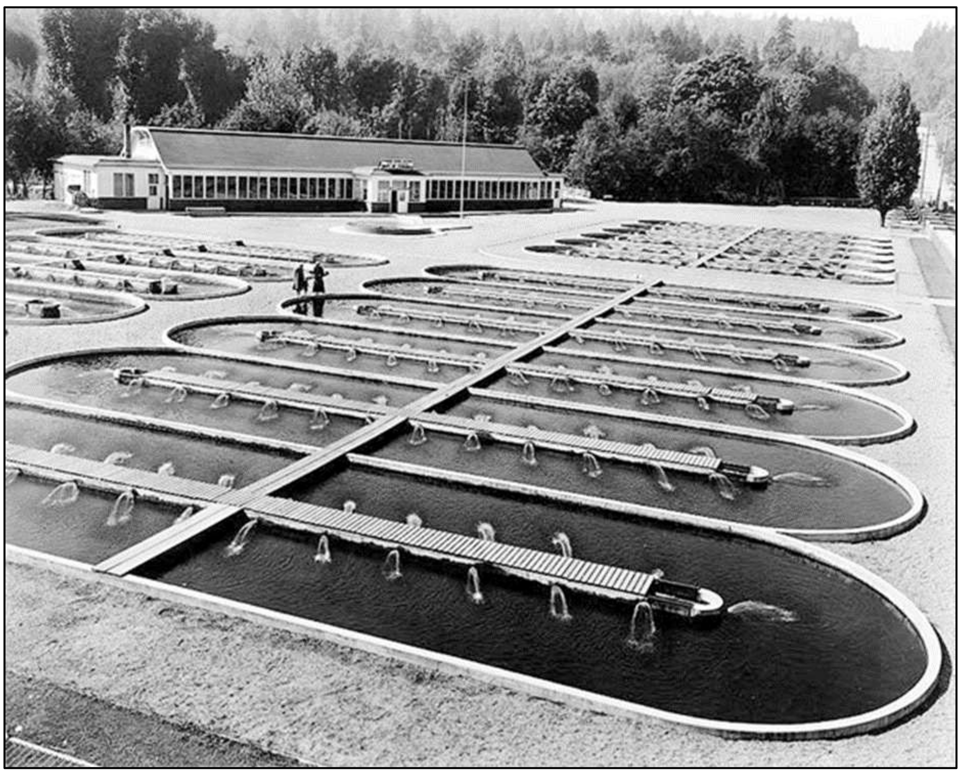

Figure 1-4. Salmon hatchery, 1950. Photo courtesy of Issaquah History Museums.

The promise of this technology was tempting to resource managers of the time. Regulations were unpopular in the fishery and private interests which cumulatively impacted salmon—agriculture, forestry, mining, and hydroelectric—were vast, powerful, and challenging to confront in the newly settled states of Washington and Oregon. As a result, U.S fisheries managers adopted hatcheries with haste, rather than take regulatory action to sufficiently correct the market inefficiency that had developed in the fishery (Cobb 1930; Lichatowich et al. 1999).

Hatchery production steadily increased throughout the early 20th century in the Columbia River and elsewhere in the Pacific Northwest (Cobb 1930). The technology was championed by resource managers, fishermen, and conservationist alike, despite the lack of scientific evidence warranting such investment. Where industrial harvest, habitat, and hydroelectric impacts to salmon were anticipated, hatcheries were generally accepted as a substitute for natural processes and utilized to mitigate the problem (Lichatowich et al. 1999). Founder of the University of Washington's Department of Fisheries, John Cobb, described management's "idolatrous faith in the efficacy of artificial culture of fish for replenishing the ravages of man'' (Cobb, 1930, p. 493). In this sense, the myth of hatchery success and the perceived benefits to Pacific Northwest fisheries enabled prolonged overharvest, loss of habitat, and construction of dams throughout the region (Lichatowich 1999).

Until the early 1990s, hatchery production for mitigation continued mostly unchallenged (Wahle and Vreeland 1978; Hilborn and Winton 1993; Lichatowich 2013). However, a publication by Nehlsen et al. (1991) garnered widespread attention to the Pacific salmon crisis that had unfolded in the 20th century and raised questions regarding hatchery and harvest management in the Pacific Northwest. Scientists began to realize how hatcheries had in part enabled loss of salmon habitat, overharvest, bycatch mortality, and construction of dams throughout the decades (NRC 1996). Furthermore, scientific literature explaining detrimental genetic, ecological, and fishery related effects of hatchery production on severely depressed wild salmonid populations became increasingly more convincing (Reisenbichler and McIntyre 1977; Chilcote et al. 1986; Wright 1993; Flagg et al. 1995; NRC 1996; Federal Caucus 1999; HSRG 2009).

To this day, an expanding body of research continues to validate that escapement of hatchery-origin fishes from commercial and tribal fisheries and genetic introgression with spawning wild salmon can significantly reduce the survival and reproductive success of depressed wild populations (NRC 1996; Chilcote et al. 2011; Naish et al. 2007; Rand et al. 2012; Christie et al. 2014). Within only a single generation of domestication selection in hatchery facilities, salmonids reared from local wild broodstock and their wild offspring exhibit reduced survival and reproductive capacity (Christie at al. 2012; Christie et al. 2014). As a result of the domestication process, hybridization of hatchery and wild fish reduces the fitness of wild salmon populations (Goodman 1990; Waples 1991; Levin et al. 2001; Christie et al. 2012). Beyond the genetic consequences, large releases of hatchery salmonids result in increased competition with wild salmonids for food and shelter in anthropogenically compromised freshwater, estuarine, and oceanic environments, and may reduce survival for all salmonid stocks where resources are limited (Beamish et al., 1997; Levin et al., 2001). Furthermore, hatchery salmon may directly predate wild salmonids, spread disease, and draw predators (such as marine mammals and birds), increasing mortality of wild salmonids (Lichatowich 1999; Orr et al. 2002; Taylor, 1999). In response, recommendations have been made for reductions in hatchery releases and changes in policy to reduce hatchery-wild interactions and genetic introgression (Goodman 1990; Hindar 1991; Krueger and May 1991; Hilborn 1992; NRC 1996; Lichatowich 1999; ISAB 2003; RSRP 2004; Beamesderfer et al. 2005).

1.6 Harvest and Hatchery Policy: Inherently Intertwined

Although genetic and ecological concerns surrounding hatcheries indeed have substantial merit, perhaps the most significant detrimental impact of hatcheries is caused indirectly by increasing fishing intensity and amplifying the “tragedy of the commons” in mixed-stock fisheries where non-selective commercial fishing gears are utilized (Wright 1993; Flagg et al. 1995). The impacts of harvest and hatcheries on wild salmonids are inherently intertwined as the presence of hatchery production in a river basin creates an ecological and economic need for harvest (NRC 1996; Lichatowich 2013). Producing millions of hatchery smolts annually in the Columbia Basin alone, managers generate an ecological problem that must be, in part, resolved through harvest. State, tribal, and federal agencies manage harvest to maximize catch of hatchery-origin fishes with minimal mortality to wild stocks to address the threat of genetic and ecological problems upstream from escapement of hatchery fish (WFWC 2009). However, lacking fishing gears that can selectively harvest targeted stocks (such as hatchery-origin fishes) in mixed-stock fisheries and release bycatch unharmed, fishing effort inevitably causes bycatch mortality (Wright 1993; Flagg et al. 1995; Gayeski et al. 2018).

The gillnet has long maintained a reputation for non-selectivity and bycatch mortality in mixed-stock salmon fisheries (Ricker 1976; ASFEC 1995). While mesh size regulations generally function well to restrict catch to the salmonid family, gillnet fisheries which involve mixed-stocks of salmon are greatly limited in harvest selectivity due to the principle of geometric similarity (Hamley 1975). In prominent fisheries such as the lower Columbia River salmon fishery, multiple salmonid species of both wild and hatchery origin may be caught. Selective harvest of hatchery stocks and release of non-target wild populations are essential to meeting conservation goals (WFWC 2009). Nevertheless, for decades, the non-selective nature of gillnets has compromised the survival of wild fishes which often become entangled, gilled, or wedged in commercial nets (Wright 1993; Flagg et al. 1995). Although mortality rates differ between species and fisheries across the west coast, bycatch mortality from gillnets commonly ranges from 35-70% (Buchanan et al. 2002; IFSP 2014; Teffer et al. 2017). Improving the ability of commercial fishing fleets to selectively harvest hatchery fish and release wild fish unharmed has been identified as a means to achieve both harvest and hatchery policy goals in Washington and Oregon for the recovery of wild salmonids and rejuvenation of stifled commercial fishing communities (WFWC 2009; WFWC 2013; WFWC 2015).

To increase the selectivity of commercial harvest and reduce bycatch mortality, fishers and managers must carefully consider the mechanisms through which commercial fishing practices impact salmonid survival. Recent decades of research have determined how entanglement in commercial gears causes lethal and sublethal physical and/or physiological impacts to salmonids (Davis 2002; Baker and Schindler 2009; Gale et al. 2011; Teffer et al. 2017). Physical injury can cause immediate mortality upon capture and removal; it can also result in sub-lethal infection that can reduce the survival probability of released fish (Baker and Schindler 2009). Furthermore, fish encountering commercial gears are susceptible to a range of physiological consequences that depend on the severity of entanglement and handling, the length of entanglement, ecological effects (e.g. predators, pathogens), environmental effects (e.g. river temperature, salinity), and the species and sex of the fish captured (Davis 2002; Cooke and Suski 2005; Donaldson et al. 2012; Nguyen et al. 2014; Teffer et al. 2017). Physiological impairment from conventional fishing practices increases disease infection rates, osmoregulatory imbalance, and anaerobic metabolism; it is also known to affect immune gene regulation (Raby et al. 2015; Teffer et al. 2017). Generally, fishing practices that reduce the severity and length of entanglement, handling, and air exposure will promote survival of bycatch in commercial salmon fisheries (Teffer et al. 2017).

1.7 Need for Research and Implementation of Alternative Fishing Gears

Since the U.S Endangered Species Act (ESA)-listing of various wild salmon stocks throughout the Pacific Northwest in the 1990s and 2000s, the urgent need for salmonid recovery has been widely recognized by scientists, resource managers, and fishermen alike (Lichatowich 1999; Federal Caucus 1999). By law, the ESA requires protection of threatened and endangered stocks and implementation of recovery plans to prevent extinction of species or distinct population segments (DPS). Furthermore, the 2007 reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) requires any fishery management plan to take appropriate conservation and management measures to prevent overfishing and minimize bycatch impacts (16 U.S.C. 1851; MSA § 301). Resource managers must gather, evaluate, and apply the best available science to meet requirements of the ESA and MSA to achieve recovery objectives and maximize benefit from fisheries (ESA; 16 U.S.C. § 1531 et seq). From the perspective of fishermen and commercial fishing communities, wild salmonid recovery and reduction of take during harvest activities is desirable to increase commercial fishing and related economic opportunities. Addressing these concerns, various attempts have been made to stimulate wild salmonid recovery through habitat restoration activities and adjustments to dam operations (Roni and Beechie 2012; Roegner et al. 2009; Laake et al. 2018). Nevertheless, habitat improvements are slow to come and limited in scope (Lackey 2017). Additionally, mainstem dam breaches in the Columbia River are unlikely to occur and additional flow management adjustments exhibit decreasing marginal benefits to salmon recovery.

With continuation of salmonid hatchery programs, implementation of alternative stock-selective fishing gears for improved targeting of hatchery-origin fishes and reduction of bycatch impacts has been recognized as a necessary means for recovering ESA-listed salmonids and sustaining participation of fishing communities (WFWC 2009; WFWC 2013; WFWC 2016). Removal of the adipose fin from hatchery-origin fish—a practice developed in the 1980s and expanded in the 1990s—enables visual differentiation between otherwise identical wild and hatchery stocks (Ashbrook 2008). To capitalize on advancements in stock identification and meet ESA recovery objectives, the WA Fish and Wildlife Commission's (WFWC's) Hatchery and Fishery Reform Policy Decision (2009) initiated the commercial selective gear implementation program to "develop and implement alternative fishing gear to maximize catch of hatchery-origin fish with minimal mortality to native salmon and steelhead" (WFWC 2009). WFWC strives to "phase out use of non-selective gill nets in the mainstem Columbia" and transition to stock-selective gears to meet management and conservation objectives and maximize utilization of fisheries allocations (WFWC 2013). If hatchery production and harvest are to continue in the Columbia Basin, viable stock-selective gears must be developed, tested, and implemented to prevent further decline of ESA-listed stocks and maximize commercial, recreational, and tribal fishing opportunities.

1.8 Previous Alternative Gear Research in the Lower Columbia River

Bycatch and bycatch mortality in commercial salmon fisheries are leading conservation issues as they can contribute to the degradation of a population or species (Kappel 2005; Coggins et al. 2007; Davies et al. 2009). In the presence of ESA-listed species, bycatch impacts also constrain commercial, tribal, and recreational fisheries via management and conservation measures required for recovery, including time and area closures, fleet reductions, and take restrictions (Vander Haegen et al. 2004). Estimates of bycatch and mortality are included as components of overall fishing mortality during stock assessment, status evaluation, estimation of ESA-fishery and research impacts, and determination of Magnuson-Stevens Act (MSA)-required annual catch limits (16 U.S.C. 1852; MSA § 302). As a result, precise estimates are essential to effective management of our Nation's marine resources (NMFS 2011).

Since the late 1990s, various researchers have assessed bycatch mortality in commercial salmon fisheries to identify sustainable alternatives to conventional fishing practices (Farrell et al. 2001; Vander Haegen et al. 2004; Ashbrook 2008; WDFW 2014). Primary metrics in these investigations are 1) quantity of bycatch encountered; 2) immediate survival (the number of fish that survive from capture to release); and 3) post-release survival (survival from release to spatially and temporally distant detection points). Of these metrics, post-release survival is of particular concern. In mixed-stock salmon fisheries of the Pacific Northwest—which frequently consist of both hatchery and wild origin fishes—a significant component of catch is typically discarded, often with the hope that released fish survive to spawn (Chopin and Arimoto 1995; Vander Haegen et al. 2004). Nevertheless, in many instances, fish released from conventional commercial fishing operations perish before they can reproduce from physical or physiological impacts incurred during capture (Chopin and Arimoto 1995; Davis 2002; Teffer et al. 2017).

To address bycatch mortality in Columbia River commercial salmon fisheries, various alternatives to conventional gillnets were proposed for ecological evaluation including tangle nets (a type of modified gillnet), beach seines, and purse seines (LCFRB 2004). These gears were recognized for their potential to minimize physical and physiological damages to captured fishes by reducing gilling, wedging, overcrowding, air exposure, and handling. Since hydrological and geomorphological conditions of the Columbia River have changed dramatically since the construction of mainstem dams from the 1930s through 1970s, researchers and contracted fishermen have generally taken a two-step approach to assessment of alternative gears, involving 1) a pilot study to determine immediate mortality and the feasibility of each alternative fishing tool under modern conditions of the Columbia River; and 2) an assessment of post-release survival (WDFW 2009; WDFW 2014).

Post-release survival in commercial salmon fisheries has been estimated primarily through post-capture confinement in artificial pens or mark-recapture methodologies since the 1970s (Vander Haegen et al. 2004). Thompson et al. (1971) estimated short-term post-release survival of fish after capture from gill nets in Washington State. Observing sockeye salmon released into confined net pens, short-term post-release survival was nearly zero (Thompson et al. 1971). Utilizing on-board revival boxes equipped with constantly circulating sea water, Farrel et al. (2001) documented substantial improvements in post-release survival from gillnets in British Columbia; approximately 97% coho salmon captured and held in confinement for a 24-h period survived. Nevertheless, evaluations of post-release survival through artificial confinement are likely prone to biases as fish are not subject to selection pressures from predatory effects and many challenges associated with post-release migration from commercial fishing operations (Farrell et al. 2000; Vander Haegen et al. 2004). Confinement periods less than 5-12 days post-release may be too short to fully observe latent mortality (Teffer et al. 2017). Furthermore, transport to net pens from the site of capture, site-specific water quality conditions, pathogens, sustained presence of marine mammals, and the effects of confinement also can cause physiological stress to animals, biasing survival estimates from post-capture confinement studies (Davis 2002; Donaldson et al. 2011; WDFW and ODFW Joint Staff 2018).

In the 2000s, Vander Haegen et al. (2004) and Ashbrook (2008) developed mark-recapture protocols in the lower Columbia Basin that have been mirrored ever since to maintain consistency for comparison between alternative gear studies. With various mainstem dams positioned throughout the Columbia River, paired mark-release-recapture tagging studies are well suited to the study region enabling resighting of tagged fish post-release and estimation of survival through the Ricker relative recovery method and the Cormack-Jolly-Seber method (Burnham et al 1987; Seber 1982). These methods estimate survival through comparison of upstream detections of control and treatment groups of captured, tagged, released, and resighted fish. Utilization of mark-release-recapture helps rid of biases associated with post-capture confinement if all model assumptions are met (Vander Haegen et al. 2004).

Vander Haegen et al. (2004) conducted the first alternative gear mark-recapture study in the lower Columbia River from 2001 through 2002. To evaluate post-release survival of spring Chinook salmon from tangle nets and gillnets, treatment groups were captured, jaw tagged, and released approximately 32 km downstream of Bonneville Dam. A control group was jaw tagged and released upstream from the test fishing location at the Bonneville Dam AFF. Jaw tagged fish from treatment and control groups were recovered throughout the Columbia River Basin in recreational fisheries, commercial fisheries, hatcheries, and spawning grounds in four different locations: 1) below Bonneville Dam; 2) between Bonneville and McNary Dams; 3) above McNary Dam; and 4) above Ice Harbor Dam (Vander Haegen et al. 2004). Pooled recaptures and detection probabilities of the two treatment groups were compared to that of the control group through the Ricker relative recovery method to estimate the post-release survival rate from each gear type (Ricker 1958). The results of the study in 2001 demonstrated that approximately 91.2% (CI ($47.0 \leq S \leq 100.0\%$) = 0.95) of spring Chinook salmon captured and released from tangle nets survived in total to be recovered at a designated recovery site (often referred to as *cumulative survival*, which is the product of immediate and post-release survival); in contrast, the conventional gillnet demonstrated only 52.5% (CI ($47.1 \leq S \leq 57.9\%$) = 0.95) cumulative survival (Vander Haegen et al. 2004; Ashbrook et al. 2004). Despite promising results from the alternative gear evaluation in 2001, repetition of the study in the year following produced conflicting findings as cumulative survival from tangle nets was estimated at only 67.6% (CI ($61.3 \leq S \leq 73.3\%$) = 0.95) (Vander Haegen et al. 2004; Ashbrook et al. 2004). This discrepancy was likely caused by chance. The small sample size used for control and treatment groups and poor jaw tag detection efficiency resulted in low precision, and the 2002 point-estimate fell within the broad confidence interval associated with the 2001 survival point estimate (Ashbrook 2008).

Ashbrook (2008) modified the methods of Vander Haegen et al. (2004) to improve precision and reduce potential biases in post-release survival estimates for alternative gear evaluations. Passive Integrated Transponder (PIT) tags were utilized to increase tag retention and upstream detection probability for treatment and control groups (therefore increasing statistical power). Mainstem Columbia River dams with PIT tag arrays were used as "recovery" sites to determine upstream detection probabilities. All unique detection histories were attained through PTAGIS—an online database containing all PIT tag detections throughout the Columbia Basin (PTAGIS 2017). Furthermore, the location of release for the control group was modified to a location approximately 21 km downstream of Bonneville Dam near the test fishing and treatment tagging site to better control for predatory effects. Through the Ricker relative recovery method, post-release survival was estimated, with detection at any upstream dam or PIT tag array counting as surviving the post-release experience. This study improved upon the precision of previous studies, with cumulative survival of spring Chinook to pooled upstream detection points estimated at 87.2% (CI ($84.5 \leq S \leq 89.8\%$) = 0.95) for tangle nets (Ashbrook 2008).

Mirroring methods developed by Ashbrook (2008), WDFW investigated the potential of purse and beach seines as alternative fishing gears in the lower Columbia River—two gears used historically in the river prior to 1935. Once again, a mark-recapture procedure utilizing PIT tags and the Ricker relative recovery method were used to estimate post-release survival of fall Chinook, coho, and steelhead by comparing upstream recovery probabilities of treatment and control groups to discrete detection points at Bonneville and McNary Dams, representing "short-term" and "long-term" post-release survival respectively. A pilot study was conducted in 2009 to determine the feasibility of each gear-type under modern conditions of the river. Demonstrating adequate promise to capture commercially viable quantities of salmon with high immediate survival, two years of mark-recapture research were performed from 2011 through 2012. These studies demonstrated high cumulative survival of steelhead from purse and beach seines to McNary Dam; however, survival was surprisingly low for fall Chinook and coho salmon (Table 1-1).

Table 1-1. Best performance estimates of cumulative survival for beach and purse seines from release at river mile 140 to McNary Dam (WDFW 2014).

Gear	Chinook Survival	Coho Survival	Steelhead Survival
Beach Seine	0.750 (0.710 – 0.790)	0.620 (0.460 – 0.810)	0.920 (0.820 – 1.000)
Purse Seine	0.780 (0.720 – 0.850)	0.770 (0.620 – 0.940)	0.980 (0.930 – 1.000)

Although improvements in cumulative bycatch survival have been documented relative to conventional gillnets, all alternative gears tested in the lower Columbia River (including tangle nets, beach seines, and purse seines) have resulted in mortality rates that may be detrimental to wild salmonid recovery objectives. For Chinook salmon, the IFSP (2014) has recommended a 21% mortality rate be applied to tangle nets; beach seines and purse seines inflict an estimated 25% and 22% mortality respectively (WDFW 2014). Progress in lower Columbia River fisheries is evident as new tools have been identified to address bycatch mortality concerns in both spring and fall fisheries (Table 1-1); nevertheless, stock-specific mortality rates exhibited by seines and tangle nets will continue to constrain fishing opportunities in the region and limit the success of basin-wide hatchery programs. As a result, testing of additional alternative gears and adoption of best harvest practices have been recommended by scientists and resource managers alike (Teffer et al. 2017; NOAA 2017).

1.9 Fish Trap Research in the Lower Columbia River

Recognizing the limitations of previously evaluated alternative commercial gears in reducing stock-specific bycatch mortality rates, fish traps were proposed for evaluation in salmon fisheries (LCFRB 2004; Ashbrook 2008; Arnold 2011). Traps are a form of fixed gear, meaning that the tool remains deployed in one place to passively capture fishes. Three separate forms of traps historically existed in Pacific Northwest commercial salmon fisheries:

1) Pile/pound net traps: constructed of stout wood pilings driven into benthic sediment of rivers and estuaries with high-current or foul weather;

2) Hand/stake traps: constructed of wood stakes/poles in shallow estuaries or small rivers;

3) Floating traps: anchored with concrete and chain in deeper, more protected waters.

Consisting of a series of pilings, stakes, or anchors and attached web fences that extend from the high-water mark toward the river or estuary bottom, fish traps passively funnel returning adult salmon from the shoreline "lead"—positioned perpendicular to shore—to a maze of walls and compartments (including the "heart" and "tunnel"). The final compartment, the "spiller", enables fish to swim freely until removal upon selective harvest or passive release (Mackovjak 2013). Salmon that enter the spiller are captured without tangling of teeth or the operculum, reducing physical injury arising from gillnets (Baker and Schindler 2009). Furthermore, when regulated and operated with a conservation-minded approach, there is potential to lessen sub-lethal physiological effects by reducing air exposure, overcrowding, entanglement, and handling of fishes (Davis 2002; Teffer et al. 2017).

1.10 Research Objectives

To develop an innovative and effective fishing technology for the reduction of bycatch and hatchery impacts to ESA-listed salmonids and benefit of U.S fishermen and fisheries, I designed, constructed, and monitored the performance of a modified fish trap in the lower Columbia River from 2016-2017 with the non-profit Wild Fish Conservancy (WFC) and local commercial fisherman Jon Blair Peterson. Specifically, objectives were to determine the effectiveness of the gear in capturing hatchery-origin Chinook and coho salmon and reducing post-release and cumulative mortality of wild fall Chinook salmon and summer steelhead trout relative to the performance of previously tested commercial gears in the lower Columbia River. Environmental and biological covariates, CPUE, capture conditions, bycatch, immediate survival, and post-release survival of fish were assessed. Methods developed by Vander Haegen et al. (2004), Ashbrook (2008), and WDFW (2014) for experimental seine and tangle net operations were utilized to maintain consistency for comparison of results between studies, with minor alterations to improve precision and reduce bias of survival estimates. Similar to previous alternative gear tests, this study intended to achieve three major goals:

1) Test and refine deployment and operation of a pound net trap under modern conditions of the Columbia River;

2) Determine effectiveness of the harvest method in capturing salmon relative to previously tested alternative gears. Directly estimate species-specific catch-per-unit-effort (CPUE) and CPUE covariates;

3) Evaluate the ability of a pound net trap to protect non-target species through identification of capture and release conditions, immediate survival, and post-release survival of fall Chinook salmon and summer steelhead trout.

Assessing CPUE from the experimental trap and employing the Cormack (1964)-Jolly (1965)-Seber (1965) method for estimation of survival through paired mark-release-recapture, this study investigated the effectiveness of the alternative gear in capturing targeted stocks with improved survivorship of released fishes relative to previously tested commercial gears. Providing precise and unbiased estimates of cumulative survival to fisheries managers may enable commercial implementation of viable stock-selective harvest tools for the rejuvenation of working waterfront economies and the reduction of bycatch and hatchery related impacts to wild salmonids across the region.

1.11 Research Questions and Hypotheses

<u>Questions</u>:

- How do cumulative survival estimates from an experimental trap compare to other commercial gears tested in the lower Columbia River?
- How does stock-specific CPUE from the modified 2017 trap compare to the performance of the trap in 2016 and other commercial gears used in the lower Columbia River?
- What environmental covariates explain CPUE at the trap site?

Null-Hypotheses:

A) Cumulative survival of fall Chinook salmon and steelhead trout from the experimental trap is equal to or less than that of previously tested gears in the lower Columbia River.
B) CPUE of fall Chinook and coho salmon from the experimental trap is equal to or less than that of conventional gears used in the lower Columbia fall fishery. CPUE cannot be explained by environmental covariates.

Alternative Hypotheses:

A) Cumulative survival of fall Chinook salmon and steelhead trout from the experimental trap is greater than that of previously tested gears in the lower Columbia River.
B) CPUE of fall Chinook and coho salmon from the experimental trap is greater than that of conventional gears used in the lower Columbia fall fishery. CPUE can be explained in part by environmental covariates.

Chapter 2: Methods

2.1 Pilot Study

2.1.1 Initiation of the Project and Selection of the Study Location

In 2013, commercial fisherman Jon Blair Peterson initiated development of the first fish trap prototype in Washington State waters in nearly eighty years. Peterson initially strived to identify a new tool for monitoring steelhead run-timing and stock-composition in the lower Columbia River. However, due to the passive nature of the proposed gear and potential to reduce entanglement of fishes, air-exposure, and handling, the non-profit WFC and WDFW took interest in the trap as an alternative commercial fishing gear in July 2013 and worked with Peterson to repurpose the project. Recognizing the tool's potential to reduce bycatch and hatchery impacts in commercial salmon fisheries for the recovery of ESA-listed stocks, WFC dedicated volunteer labor to assist the fisherman in constructing the trap in August 2013.

Figure 2-1. Pound net trap site located approximately 2 miles upstream from Cathlamet, WA.

Based on historical blueprints of Columbia River traps, untreated wood pilings (16'' diameter) were driven for the prototype design in the Cathlamet Channel of Wahkiakum County, WA at river kilometer (rkm) 70 where pound net traps were once common prior to the 1934 fixed-gear ban (Figure 2-1). This study site was selected by the fisherman for a variety of reasons: 1) the site was historically successful and was utilized by the fisherman's grandfather in the early 20th century, and 2) the location was locally known for high densities of steelhead trout, enabling the trap to more effectively function as a steelhead run-timing and stock-composition monitoring tool. In this developmental season, necessary modifications were identified. Ultimately, layout of the pilings proved insufficient and limited funding availability prevented the required changes to be made to make the trap fully operational.

2.1.2 Trap Design and Construction in 2016

In 2016, I initiated a pilot year evaluation of a substantially modified trap with WFC, WDFW, and Jon Blair Peterson. Based upon experiences in 2013 and extensive research of historical trap designs, photographs, and anecdotes from 1880s through the 1930s, pilings were repositioned approximately 3 to 5 m apart at rkm 70, resulting in an extended lead (~90 m), addition of a jigger to increase capture efficiency, and modified heart and spiller compartments (Figure 2-2). Given the altered piling layout, bathymetry of the riverbed, and tidal range of the site, we designed the nets for the various components of the trap (e.g. lead, jigger, heart, tunnel, and spiller). Knotted black nylon mesh with a stretch of 3-1/8'' (7.94 cm) was selected for application in the lead, jigger, heart, and tunnel sections of the trap to minimize both entanglement of fishes and drag within the water column (influenced by river flow and tidal effects at the project site). The spiller was designed of a mixture of 2-1/2'' (6.35 cm) knotless black nylon mesh at the bunt end and 3-1/8'' mesh on the sides. These mesh sizes were selected to reduce potential injury to fishes during lift and minimize drag in the water column during the soak period. Net designs were submitted to Christensen Net Works (Everson, WA) for construction.

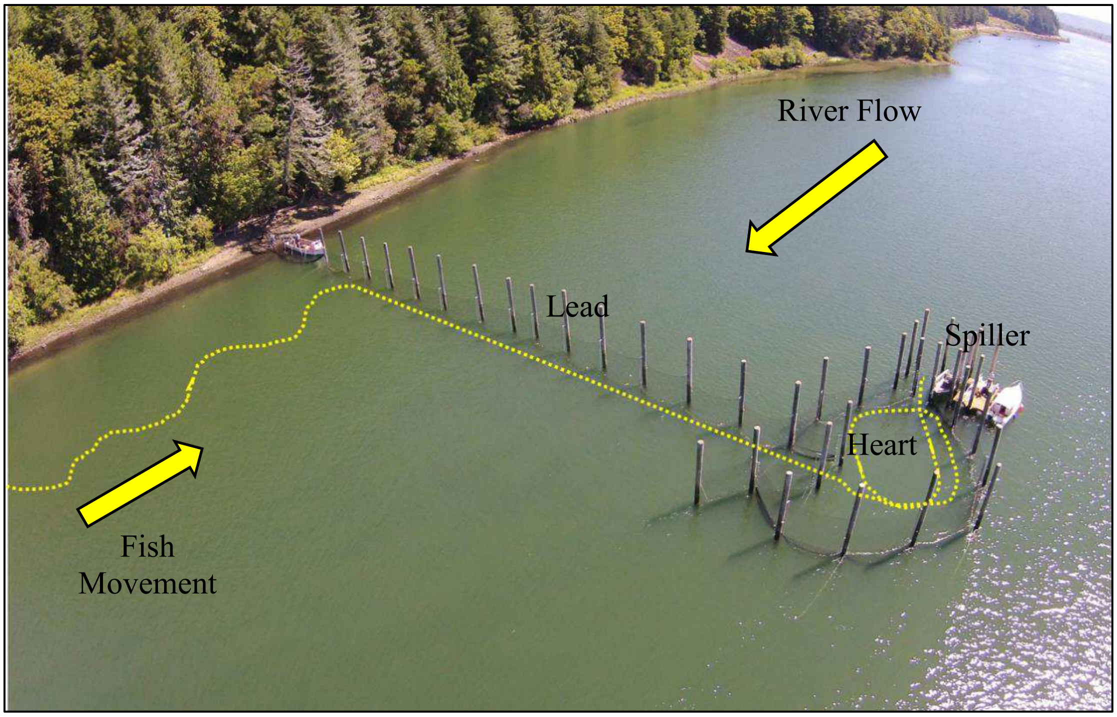

Figure 2-2. The 2016-17 pound net trap. Photo courtesy of Jamie Glasgow (WFC 2018).

From August 2 - 21, all nets and hardware were applied to the trap (Figure 2-3). Nets were secured at the downstream side of the pilings, creating a smooth mesh wall to migrating fish (in contrast to netting attached on the upstream side, where fish would encounter exposed pilings and may be deterred). All nets were hung approximately 0.60 m above the high-water line and deployed to the river bed with attached steel weights and lead line. Cylindrical steel weights were engineered to slide downwards with gravity around 2-3/4'' (6.99 cm) aluminum poles, which contained the net on the downstream side of the pilings from billowing with river flow and the ebb tide. Nets could be lifted in all trap compartments with line and pulley—connecting each net-piling attachment point above the high-water line to its associated steel weight at the river bottom. This strategy enabled occasional maintenance of the nets and fish passage during times when test fishing was not occurring.

Figure 2-3. Hanging nets on the lead of the 2016 trap in preparation for deployment. Photo courtesy of Aaron Jorgenson (WFC 2018).

The spiller/tunnel complex was engineered to deploy to the river bottom in a similar fashion to the lead and heart nets. Aluminum poles (6.99 cm diameter, 9.14 m length) distanced nearly 30 cm inside each associated spiller piling gave a frame to the cube-shaped spiller compartment. Escape rings attached the spiller net to each aluminum pole, enabling the spiller compartment to move upwards with force by line and pulley. Four pulleys were secured at the top of the corner aluminum poles 9.14 m above the riverbed with line attaching to the bottom four corners of the spiller. Steel weights at the bottom four corners of the compartment enabled gravity to draw the mesh flush to the river bottom during each soak period. A solar powered winch was installed 9.75 m above the river bottom on a platform near the top of the pilings to pull the bottom mesh of the spiller upwards with line and pulley in the water column during each haul to allow all captured fishes to be accessed. Adjacent to the spiller, a 4.88 m pontoon dock equipped with a perforated aluminum framed live-well (2.13 m X 0.61 m) enabled fish transferred from the spiller compartment to be sorted within the confines of a live-well (a much smaller and more manageable space for sorting than the spiller). Within this compartment (Figure 2-4), all fish remained free-swimming and submerged with continuously circulating river water. With the completion of a set, a small door to the live-well could be opened allowing all captured fish to swim upstream with minimal handling.

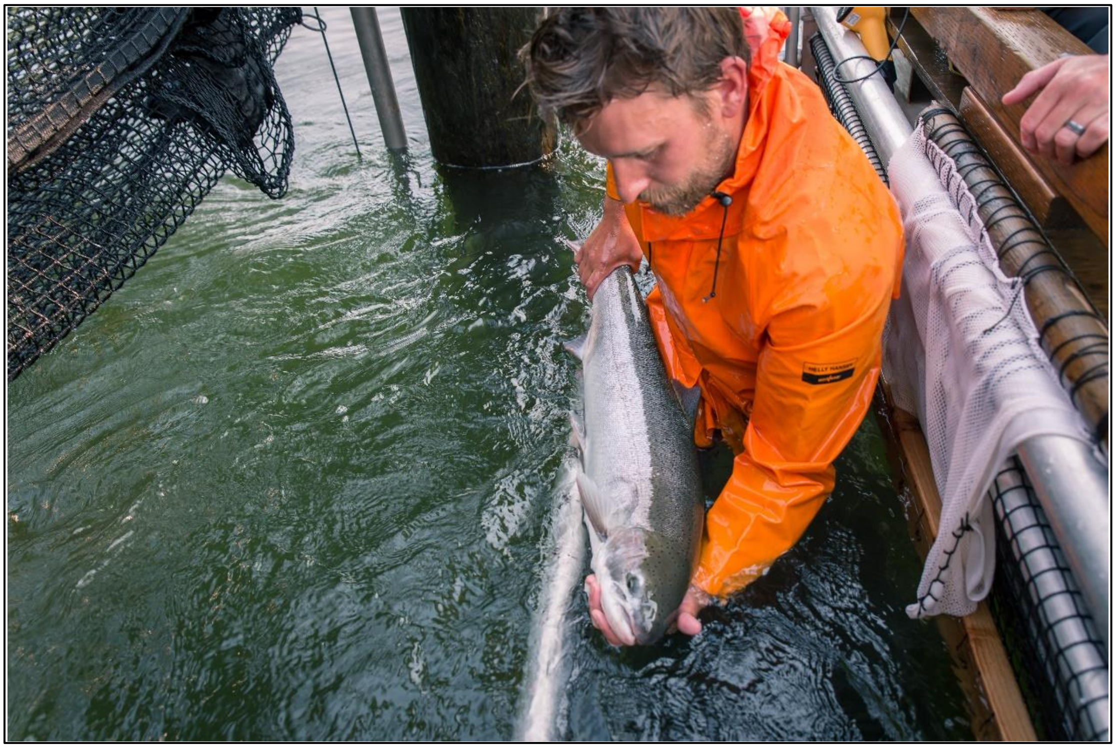

Figure 2-4. A hatchery-origin steelhead is released from the live-well. Photo courtesy of Aaron Jorgenson (WFC 2018).

2.1.3 Target Species for Research

The study was conducted at the Cathlamet, WA trap site at rkm 70 from August 26 through September 29, 2016 and from August 26 through September 27, 2017. This late-summer to early-fall period represents the peak of fall Chinook salmon, coho salmon, and steelhead trout upriver migration in the lower Columbia River (Healey 1991; Fish Passage Center 2016). Hatchery origin Chinook and coho salmon are commercially lucrative target stocks within the lower Columbia fall fishery. Wild-origin summer steelhead trout, fall Chinook salmon, and coho salmon populations are ESA-listed and common bycatch stocks which constrain commercial fishing opportunity within the conventional fall fishery.

2.1.4 Field Protocol in 2016

Testing proceeded in the following manner. Three people were needed on site to operate the gear, including two trained WFC employees, a commercial fisherman, or potential volunteers from the region. When all participants were prepared, the trap spiller was deployed to the river bottom by releasing lines and disengaging the electric winch brake. The tunnel door was opened by tightening the harness pulley line, initiating the soak period and enabling the capture of fishes. Observers noted the beginning set time, tidal stage (m), tide height (m), water temperature (ºC; Extech), and presence of marine mammals. The tunnel door remained open to fish passage until the desired soak period ended or the capacity of the spiller had been reached.

Once the soak period had ended (generally 3 – 60 minutes), the tunnel door was closed by releasing the tunnel harness line, preventing further entry or escape. An observer turned on a live-streaming video recorder through the application “Periscope” and noted the end set time, tidal stage, tide height, water temperature, and presence of marine mammals. The spiller bottom was then carefully lifted by an electric winch to concentrate captured fishes toward the spiller door (positioned adjacent to the live-well of the sorting deck) (Figure 2-5). Once the fish were spilled into the live well, all specimens were individually counted, measured (FL), and identified for species type, origin (hatchery/wild), and capture condition (lively, lethargic, bleeding, lively/bleeding, lethargic/bleeding, dead) (WDFW 2009; WDFW 2014). Upon confirmation of resuscitation and documentation of abnormalities and/or injuries, all fish (hatchery and wild) were passively released through the live-well door and additional sets initiated as described. These field methods enabled documentation of capture/release conditions, bycatch, immediate survival, and CPUE in the fall Chinook and coho salmon fishery.

Figure 2-5. A haul of salmon is concentrated toward to spiller door with an electric winch. Photo courtesy of Aaron Jorgenson (WFC 2018).

2.2 Post-Release Survival Study

2.2.1 Trap Design Modifications in 2017

Modifications to the experimental trap design and operations were made in 2017 to increase capture efficiency and reduce physical and physiological damages to captured fishes. The following modifications were made to each component of the trap:

1) Lead and heart nets – WFC staff dove to the river bottom to ensure nets were fully descended to the sediment to minimize escapement points and increase capture of benthic oriented species (e.g. Chinook salmon).

2) Spiller – The mesh size was reduced to 2-1/2’’ (6.50 cm) stretch knotless black nylon material to minimize gilling and wedging of jacks. Furthermore, the shape of the spiller bunt was arced toward the spiller door and curved in the corners to increase the tendency of fish to naturally migrate out the spiller door and into the live-well during lift.

3) Spiller lifting system – 3/8’’ (0.95 cm) stainless steel cables were attached on the inside of each spiller piling to guide deployment and lift of the spiller along the escape rings at each net-piling attachment point (replacing aluminum poles as the guiding mechanism). This effort was made to reduce friction during lifting and lowering of the spiller compartment, increase the speed of lift for more efficient spills and soaks, and ensure the spiller and tunnel were resting flush with the river bed during all periods of deployment to increase capture efficiency.
4) Winch lifting point –The lifting point of the spiller was raised from 9.14 m above the river bed to 11.58 m to improve the ability of fishers to effectively complete sets during the highest tides and spill fish more efficiently.
5) Heart apex – A 1.50 m X 7.62 m panel of 2 ½’’ (6.50 cm) stretch knotless black mesh (referred to as the “fish gate”) was installed at the outlet of the heart to reduce escapement of fish from the heart compartment during lifting of the spiller and to increase buildup of fish within the heart prior to initiation of each succeeding soak period. The “fish gate” could be lifted or lowered along 3/8’’ (0.95 cm) stainless steel cable through a system of line, pulley, and weights.
6) Marine mammal deterrent – A marine mammal “gate” with 8.26 cm diameter rectangular aluminum frame was installed at the entrance to the heart compartment of the trap to prevent entry of seals and sea lions while enabling passage of salmonids for capture (Figure 2-6). This gate consisted of a series of vertical 3.81 cm diameter aluminum bars spaced at 25.4 cm increments along the frame and was constructed with hinges to enable staff to open and close the gate depending on the abundance of marine mammals within the vicinity of the study location.

Figure 2-6. Marine mammal gate deployed at the entrance to the heart of the pound net trap. Photo courtesy of Aaron Jorgenson (WFC 2018).

2.2.2 Field Protocol in 2017

Operation of the trap and field protocol in 2017 mirrored that of 2016 study, but incorporated use of PIT tags to track upriver migrations and quantitatively estimate post-release survival. As in 2016, salmonids and bycatch species captured with the trap were individually counted, measured (FL), and identified for species type, origin (hatchery/wild), and capture condition (WDFW 2014). However, a subsample of Chinook and steelhead were PIT tagged and/or fin-clipped; these fish were placed into a recovery chamber of the live-well with recirculating freshwater (Farrell et al. 2001). Upon confirmation of resuscitation and documentation of abnormalities and/or injuries, all fish (hatchery and wild) were passively released through the live-well door and additional sets initiated as described.

2.2.3 Study Design in 2017

A paired mark-release-recapture methodology was utilized to estimate post-release survival from the experimental pound net trap to upstream detection points (Cormack 1964). Control and treatment groups of randomly sampled Chinook salmon and steelhead trout were sourced at the study location, tagged, and released for detection at upstream dams and hatcheries. During each test fishing day, control and treatment tagging sessions were generally assigned alternately. These methods were employed to reduce potential for violation of model assumptions: 1) the fate of each fish is independent, 2) control and treatment fish have equivalent handling and tagging survival, 3) control and treatment survivors have the same probability of detection after release (e.g. equivalent stock-composition, marine mammal predation, harvest pressures, environmental stressors, and tag loss), 4) all treatment fish have equal survival and recovery probabilities, and 5) all control fish have equal survival and recovery probabilities (WDFW 2014). It must be noted, however, that there were limitations to alternation of control and treatment group assignment depending on light and water clarity, which affected the ability of field staff to effectively dip-net control group fishes.

For the treatment group—represented by individuals lifted by the winch and spilled from the pound net spiller to the live-well—Chinook salmon and steelhead trout were scanned for existing PIT tags with a Biomark 601 reader. If existing PIT tags were detected, codes were recorded directly into a computer database using P4 software (PTAGIS 2017); these fish were then passively released from the live-well chamber. In the absence of an existing PIT tag, Chinook and steelhead were tagged in the peritoneal cavity (as approved by the FDA) with a 12.5 mm 134.2 kHz full duplex PIT tag and an MK-25 Rapid Implant Gun (Figure 2-7) (Biomark, Boise, ID). These fish were then scanned to document the tag number. Additionally, a subset of Chinook and steelhead received non-lethal 2 mm fin clips for genetic analysis to reduce any potential biases from violation of model assumptions. Tissue samples were stored in 97% ethyl alcohol and unique genetic sample numbers were recorded simultaneously with a specimen's PIT tag code utilizing P4 software. With tagging and fin-clipping procedures complete, fish were released from the live-well recovery chamber for upstream detection at PIT tag arrays (WDFW 2014).

Figure 2-7. Biomark 601 reader, MK-25 Rapid Implant Gun, and 12.5 mm 134.2 kHz full duplex PIT tags used for the mark-recapture study. Photo by Adrian Tuohy (WFC 2018).

Similar to previous alternative gear studies, a control source of Chinook salmon and steelhead were passively captured at the project site, tagged, and released for detection upstream. Unlike treatment fish (which experience commercial harvest procedures and make physical contact with the spiller mesh), control fish had no contact with the mesh of the trap and were not lifted or spilled by the trap winch into the live-well. Instead, passively captured and free-swimming fish were dip-netted with a rubberized hand net at the trap site and PIT tagged for release. The capture procedures for this control source were likely less damaging than procedures utilized for previous studies conducted by Ashbrook (2008) and WDFW (2014) (in which fish were trapped at the Bonneville dam AFF, dip-netted, handled, PIT tagged, and trucked downstream to the upstream end of the test fishing location at rkm 225) (Ashbrook 2008; WDFW 2014). Utilizing a less stressful sourcing technique for the control group, the experimental trap was likely at a comparative disadvantage relative to previously tested gears as the upstream detection probability of treatment to control was less prone to be biased high.

PIT tag recovery information was secured through the PIT Tag Information System (PTAGIS). Tag information was attained through upstream interrogations at dam and hatchery arrays and mortalities in fisheries (www.ptagis.org). The Columbia Basin is equipped with over 100 array stations, many of which have detection rates over 99% (WDFW 2014). PTAGIS provides public access to the PIT-tag data which can be electronically retrieved through the internet.

2.2.4 Survival Analysis

The Cormack (1964)-Jolly (1965)-Seber (1965) method was used to estimate post-release survival between four upstream detection points from the initial capture and release site for fall Chinook salmon and summer steelhead trout (Burnham et al. 1987). This method is particular valuable for separating survival from the effect of detection at each detection site and adjusting for the effects of handling and tagging mortality.

Similar to methods utilized by WDFW (2014), post-release survival was estimated between the capture and release site (rkm 70) and upstream detection sites at Bonneville Dam (rkm 233), The Dalles Dam (rkm 308), McNary Dam (rkm 470), and pooled detection sites above McNary Dam (Figure 2-8). This enabled estimation of short-term and long-term post-release survival—analogous to WDFW's (2014) Ricker relative recovery study. However, use of the Cormack-Jolly-Seber method for this study enabled estimation and correction for discrepancies in stock-specific and site-specific differences in detection probabilities, and isolation of the effect of detection from post-release survival. In contrast, the Ricker relative recovery method can only estimate the joint probability of survival and detection (Ricker 1958). Consequently, post-release mortality can only be estimated through the Ricker method assuming control and treatment fish have equal probabilities of detection.

To maintain consistency with previous Columbia River alternative gear evaluations and enable efficient comparison between studies, immediate survival (i) represented survival from capture to release from the gear; short-term survival (τ_1) was measured from release to Bonneville Dam; long-term survival ($\tau_2 * \tau_3$) was measured from Bonneville to McNary Dam; cumulative survival ($i * \tau_1 * \tau_2 * \tau_3$) represented total survival from capture to McNary Dam (Figure

2-8) (WDFW 2014). McNary Dam was selected by WDFW (2014) as the mainstem terminal point, considering the fact that a hypothetical terminal point located too far upstream might give upriver fish a lower probability of recovery relative to fish returning lower in the basin; if any differences in stock composition exist between treatment and control groups given this hypothetical study design, assumptions of the model may be violated, biasing post-release survival estimates (WDFW 2014; WDFW and ODFW 2018).

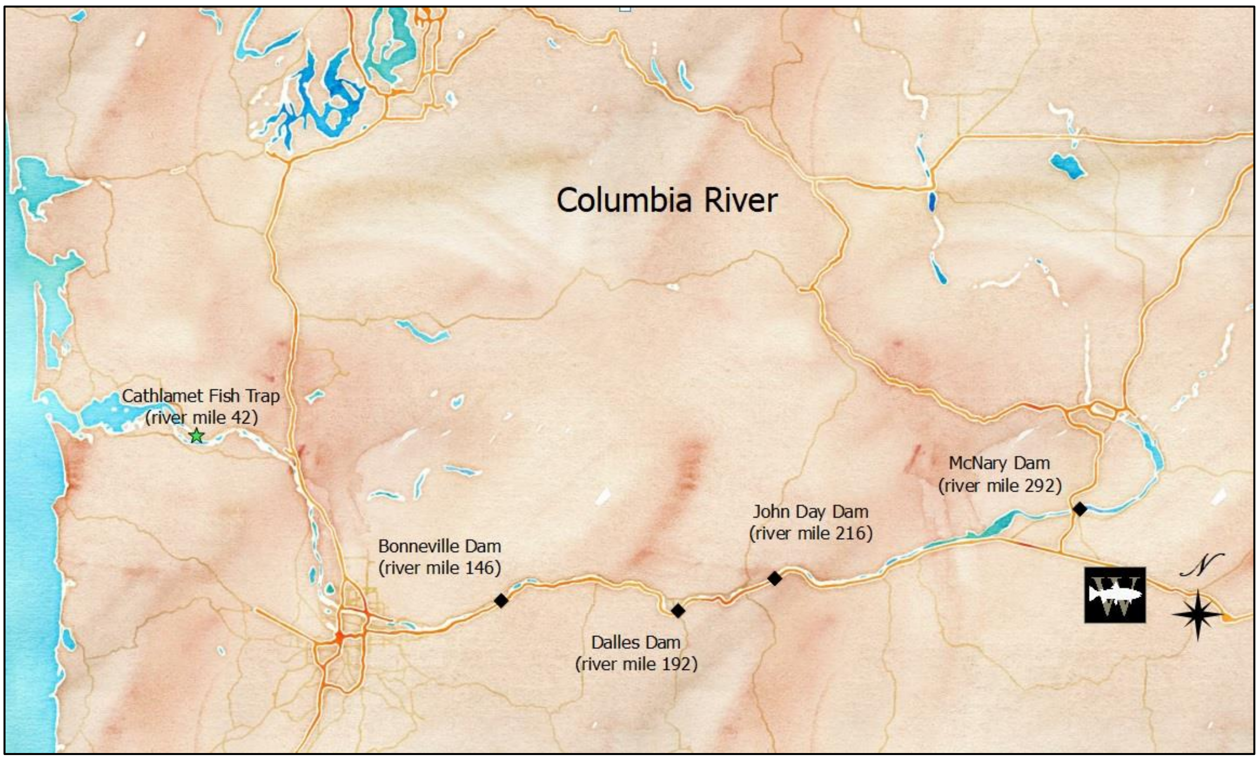

Figure 2-8. Map of the Columbia River and the location of the study site and PIT tag arrays at upstream dam detection points. Image courtesy of Aaron Jorgenson (WFC 2018).

Utilizing the Cormack-Jolly-Seber method to estimate post-release survival at four separate upstream mainstem reaches, potential detection histories for tagged control and treatment group fish (along with model probabilities of occurrence in the paired Cormack–Jolly–Seber model) are described as follows (Table 2-1 and Table 2-2):

Table 2-1. Potential detection histories for control group fish. A “1” denotes detection and “0” nondetection at the four upstream detection locations.

History	Probability of Occurrence (Control)	Count
1111	s1*p21*s2*p22*s3*p23* λ	m_{1111}
0111	s1*q21*s2*p22*s3*p23* λ	m_{0111}
1011	s1*p21*s2*q22*s3*p23*λ	m_{1011}
0011	s1*q21*s2*q22*s3*p23*λ	m_{0011}
1101	s1*p21*s2*p22*s3*q23*λ	m_{1101}
0111	s1*q21*s2*p22*s3*q23*λ	m_{0111}
1001	s1*p21*s2*q22*s3*q23*λ	m_{1001}
0001	s1*q21*s2*q22*s3*q23*λ	m_{0001}
1110	s1*p21*s2*p22*s3*p23*(1-λ)	m_{1110}
0110	s1*q21*s2*p22*s3*p23*(1-λ)	m_{0110}
1010	s1*p21*s2*q22*s3*p23*(1-λ)	m_{1010}
0010	s1*q21*s2*q22*s3*p23*(1-λ)	m_{0010}
1100	s1*p21*s2*p22*((1-s3)+(s3*q23)*(1-λ))	m_{1100}
0100	s1*q21*s2*p22*((1-s3)+(s3*q23)*(1-λ))	m_{0100}
1000	s1*p21*((1-s2)+(s2*q22)*((1-s3)+(s3*q23)*(1-λ)))	m_{1000}
0000	(1-s1)+s1*q21*((1-s2)+s2*q22*((1-s3)+s3*q23*(1-λ)))	m_{0000}

Table 2-2. Potential detection histories for treatment group fish. A “1” denotes detection and “0” nondetection at the four upstream detection locations.

History	Probability of Occurrence (Treatment)	Count
1111	(s1*t1)*p11*(s2*t2)*p12*(s3*t3)*p13*(λ*t4)	m_{1111}
0111	(s1*t1)*q11*(s2*t2)*p12*(s3*t3)*p13*(λ*t4)	m_{0111}
1011	(s1*t1)*p11*(s2*t2)*q12*(s3*t3)*p13*(λ*t4)	m_{1011}
0011	(s1*t1)*q11*(s2*t2)*q12*(s3*t3)*p13*(λ*t4)	m_{0011}
1101	(s1*t1)*p11*(s2*t2)*p12*(s3*t3)*q13*(λ*t4)	m_{1101}
0111	(s1*t1)*q11*(s2*t2)*p12*(s3*t3)*q13*(λ*t4)	m_{0111}
1001	(s1*t1)*p11*(s2*t2)*q12*(s3*t3)*q13*(λ*t4)	m_{1001}
0001	(s1*t1)*q11*(s2*t2)*q12*(s3*t3)*q13*(λ*t4)	m_{0001}
1110	(s1*t1)*p11*(s2*t2)*p12*(s3*t3)*p13*(1-(λ*t4))	m_{1110}
0110	(s1*t1)*q11*(s2*t2)*p12*(s3*t3)*p13*(1-(λ*t4))	m_{0110}
1010	(s1*t1)*p11*(s2*t2)*q12*(s3*t3)*p13*(1-(λ*t4))	m_{1010}
0010	(s1*t1)*q11*(s2*t2)*q12*(s3*t3)*p13*(1-(λ*t4))	m_{0010}

1100	(s1*t1)*p11*(s2*t2)*p12*((1-(s3*t3))+(s3*t3*q13)*(1-(λ*t4)))	m_{1100}
0100	(s1*t1)*q11*(s2*t2)*p12*((1-(s3*t3))+(s3*t3*q13)*(1-(λ*t4)))	m_{0100}
1000	(s1*t1)*p11*((1-(s2*t2))+(s2*t2*q12)*((1-(s3*t3))+(s3*t3*q13)*(1-(λ*t4))))	m_{1000}
0000	(1-s1*t1)+s1*t1*q11*((1-s2*t2)+s2*t2*q12*((1-s3*t3)+s3*t3*q13*(1-(λ*t4))))	m_{0000}

The joint likelihood for the tagging study is expressed as a product multinomial:

$$L\left(S_i, p_{ji} \middle| R_{ji}, m_{ji}\right) = \binom{R_c}{m_{ci}} \prod_{i=1}^{16} p_{ci}^{m_{ci}} \cdot \binom{R_t}{m_{ti}} \prod_{i=1}^{16} p_{ti}^{m_{ti}} \qquad (2.1)$$

where

R_c = number of control group fish tagged and released,

m_{ci} = number of tags recovered from control group fish with detection history i,

p_{ci} = probability of detection history i for the control group,

R_t = number of treatment group fish tagged and released,

m_{ti} = number of tags recovered from treatment group fish with detection history i,

p_{ti} = probability of detection history i for the treatment group,

i = detection history.

In tables 2-1 and 2-2, the model parameters are defined as follows:

s_i = survival rate in reach i,

p_{ji} = probability of detection in reach i for the j treatment group,

q_{ji} = probability of non-detection in reach i for the j treatment group ($1-p_{ji}$),

k = number of potential detection events,

τ_1 = treatment effect on survival from release to Bonneville Dam,

τ_2 = treatment effect on survival from Bonneville Dam to The Dalles Dam,

τ_3 = treatment effect on survival from The Dalles Dam to McNary Dam,

τ_4 = treatment effect on survival from McNary Dam to pooled upstream detection points,

λ = joint probability of survival and detection in reach 4 (s4*p4),

i = reach detection history (1 = Bonneville, 2 = The Dalles, 3 = McNary, 4 = above McNary),

j = treatment group (1 = treatment, 2 = control).

In the final reach between McNary Dam and pooled detection points upstream, it must be noted that survival and detection cannot be differentiated. As a result, s4*p4 = λ for control and treatment groups.

Unique detection histories at upstream dams were downloaded from PTAGIS, processed through the R-platform, and uploaded to Program USER (http://www.cbr.washington.edu/analysis/apps/user) to estimate post-release survival, standard error, and the 95% profile likelihood confidence interval (Kalbfleisch and Sprott 1970; Hudson 1971; Lady and Skalski 2009). The most parsimonious model for parameter estimation was selected through the log-likelihood ratio test (LRT) (Kendall and Stuart 1977). Alternatively, variance and standard error for survival estimates can be calculated through the inverse Hessian.

In the situation where the reduced model ($p_{ci} = p_{ti}$) is statistically equivalent to the full model ($p_{ci} \neq p_{ti}$) and detection probabilities are equated between treatment and control groups, the method of moments estimator for the treatment effect on survival within a given reach is equivalent to that of previous alternative gear studies of the lower Columbia River which used the Ricker relative recovery method:

$$\tau = \frac{\left(\frac{m_{ti}}{R_t}\right)}{\left(\frac{m_{ci}}{R_c}\right)} \tag{2.2}$$

In this reduced model form, survival of tagged fish to a common location is estimated by comparing the upstream recovery probability of the treatment group to that of the control group of tagged fish released at the same location. Therefore, selection of the reduced model results in the following comparisons to the work of WDFW (2014):

τ_1 = Short-term survival (from capture and release to Bonneville Dam),

$\tau_2 * \tau_3$ = Long-term survival (from Bonneville Dam to McNary Dam),

$i * \tau_1 * \tau_2 * \tau_3$ = Cumulative survival (from capture and release to McNary Dam).

Despite efforts to mirror WDFW's (2014) study design and selection of short-term and long-term detection points within the river, it must be noted that the capture/release site used for this study differs dramatically from previous post-release survival studies. Tag and release for purse seine, beach seine, and tangle net studies were conducted between rkm 209 and 233 of the lower Columbia River; the experimental trap is at a comparative disadvantage, with tagging operations occurring at rkm 70 (approximately 150 kilometers downstream). Assuming the gear treatment inflicts greater physical damage and physiological stress to captured and released fishes than control group sourcing, this would give more time to enable a pre-spawn mortality event for treatment sourced fishes, biasing post-release survival of the trap lower relative to previously tested gears in the lower Columbia River (Teffer et al. 2017).

2.2.5 Genetic Analysis

To ensure that random sampling of fish and random assignment of treatment and control groups met the model assumption that both control and treatment fish have equivalent recovery probability, 507 Chinook salmon genetic samples (non-lethal 2 mm diameter fin clips) were randomly selected and analyzed (241 control; 266 treatment) with the appropriate set of

Columbia basin-specific SNP markers by the University of Montana's Conservation Genetics Lab (MCGL) and the Idaho Department of Fish and Wildlife (IDF&G) Genetics Lab. Genetic assignment tests were used to assign individuals to population groups below Bonneville Dam and above Bonneville Dam with a threshold 90% probability to ensure high confidence in assignment (Piry et al. 2004; Miller et al. 2018). A contingency table was constructed for the results of the genetic assignment. GLM/log-linear analysis (Poisson, Log-Link) was performed to test the null-hypothesis of independence between control and treatment groups and population group assignment at the $\alpha \leq 0.05$ significance level. With controversy surrounding previous lower Columbia River post-release survival studies due to hypothesized violation of model assumptions (e.g. equivalent stock-composition between control and treatment groups), this precautionary technique enabled evaluation of model assumptions and correction for potential biases.

2.2.6 Determining CPUE

CPUE (defined by the number of fish captured by a gear-type divided by soak length hours and the mean number of active fishing vessels) was calculated for Chinook salmon, coho salmon, and steelhead trout throughout the study period and compared to that of gillnets in the 2017 lower Columbia River non-Indian commercial fall Chinook and coho salmon fishery (ODFW 2017). Making the assumption that 3 individuals are used to fully staff a commercial drift gillnet operation, calculation of CPUE point estimates between gears shares equivalent man-hour labor inputs, enabling coarse comparison of Chinook salmon and coho salmon CPUE between overlapping weeks of operation (adjusted by one day to account for the migration time of fish between Zone 2 at the fish trap site to Zone 4 where the gillnet fleet operated in 2017). Both hatchery and wild-origin Chinook and coho salmon were used in the comparison of CPUE as wild-origin salmon were surprisingly retained in the 2017 Lower Columbia River non-Indian commercial fishery. Total deliveries was used as a proxy for the number of active fishing vessels.

2.2.7 Regression Analysis of CPUE

Multiple linear regression was conducted to determine the covariates that best explain CPUE at the experimental trap. An $\alpha \leq 0.05$ was used for statistical significance. Covariates considered for this analysis included daily returns to Bonneville Dam (5 days after a given test fishing day to account for the mean migration time of Chinook and steelhead from the test site to Bonneville Dam), time of day (day, night, dawn, or dusk), tide height (m), tidal stage (ebb, flood, high-water, or low-water), water temperature (ºC), use of the marine mammal gate (open or closed), and the intercept term (Table 2-3). Dawn and dusk were defined as the time period one hour before and after sunrise and sunset respectively. High-water and low-water were defined as the time period one hour before and after high-slack and low-slack respectively. The most parsimonious model was selected through the backwards-elimination/deletion approach (Burnham and Anderson 1998). Stock-specific CPUE represented the response variable, which was log transformed to account for right skewness of the data and anticipated multiplicative effects. Association of each covariate with the response variable (positive or negative) was determined independently of the regression model on a single-factor basis.

Table 2-3. Descriptors of covariates used in multiple regression to explain stock-specific CPUE.

Covariate	Unit of Measure	Description
Bonneville Dam Counts	Total salmonids	Total number of a species passing Bonneville Dam five days after CPUE measurement.
Mean Tide Height	Meters	Mean tide height throughout the duration of a soak period.
Water Temperature	ºC	Water temperature at the river surface during the soak period.
Tidal Stage	Categorical	Tide stage (ebb, flood, high-water, low-water) at the end of the soak period.
Time of Day	Categorical	Time of day the set was performed (dawn, day, dusk, night).
Marine Mammal Gate Position	Categorical	Position of the marine mammal gate: open (0), closed (1).

Chapter 3: Results

3.1 Pilot Study in 2016

3.1.1 Total Catch and CPUE

The pound net trap was fished for 258.17 hours over 30 days between August 26 and September 29, 2016. During this period, 124 sets were performed. Soak period, defined as the length of time in which the tunnel door remained open to fish passage per set, ranged from 0.45 to 7.53 h with a median of 1.90 h (Mean = 2.08; SD = 1.12). The median length of time between sets in a day was 9 min with a minimum of 5 (mean = 10 mins).

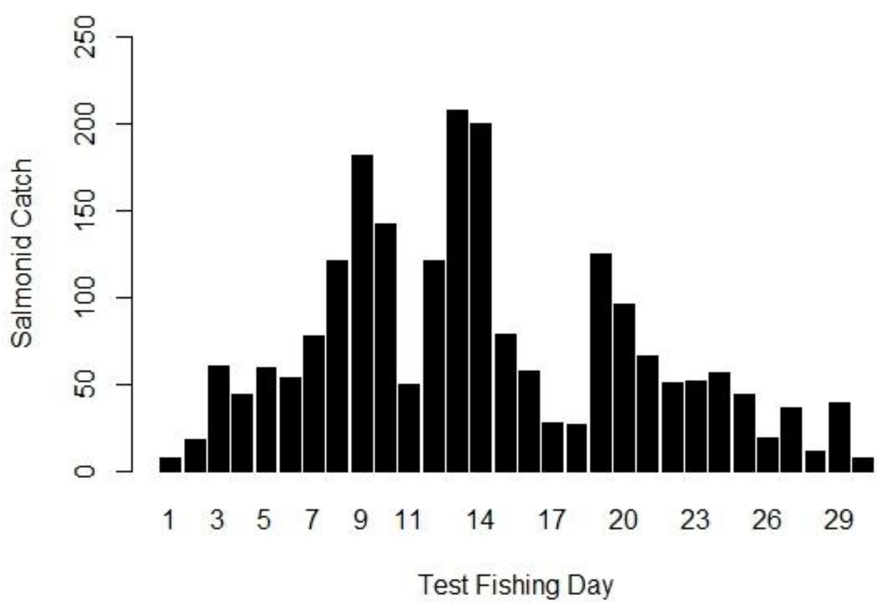

Figure 3-1. Total daily salmonid catch over the 30-day test fishing.

A total of 2,153 salmonids were captured in 2016. Mean daily catch was 71 salmonids with a maximum catch of 208 on September 12 (test fishing day #13) and a minimum catch of 7 on August 26 and September 29 (Figure 3-1). Total catch was composed of 37.0% coho salmon (796 total; 72.5% ad-clipped; 30.3% jack salmon), 24.8% fall Chinook salmon (534 total; 56.1%

ad-clipped; 20.4% jack salmon), 37.9% summer steelhead trout (816 total; 72.3% ad-clipped), 0.2% chum salmon (5 total), sockeye salmon (2 total), American shad (*Alosa sapidissima*) (1 total), and largemouth bass (*Micropterus salmoides*) (1 total).

Mean salmonid CPUE after 258.17 hours of total fishing effort was 8.34/h (Figure 3-2). Daily CPUE for all salmonids peaked at 21.1 salmonids/h on September 12 (SD = 5.4). For coho salmon, daily CPUE peaked at 8.8/h on September 7 (mean = 3.0, SD = 2.2). Chinook salmon daily CPUE peaked at 5.2/h on September 13 (mean = 2.0, SD = 1.7). Steelhead daily CPUE peaked at 9.7/h on September 12 (mean = 3.1, SD = 2.3).

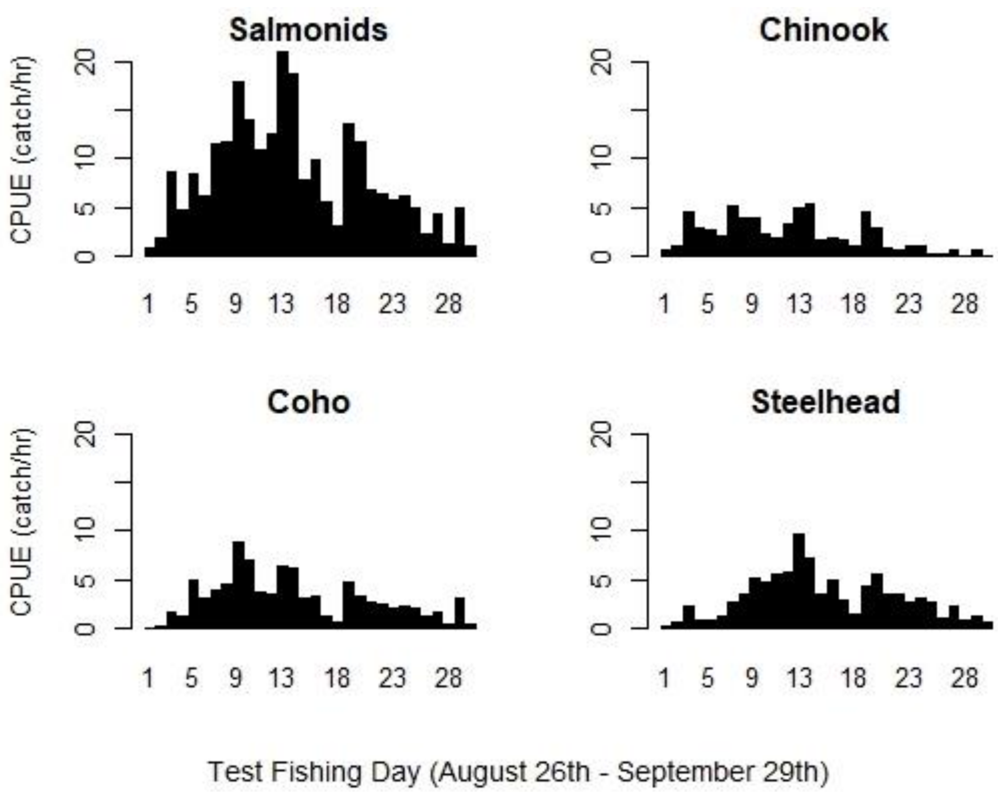

Figure 3-2. Daily CPUE (catch/h) of all salmonids, Chinook salmon, coho salmon, and steelhead trout in 2016.

3.1.2 Immediate Survival

In 2016, 2,144 salmonids (99.58% of catch) were released in a vigorous and lively condition from the trap (Table 3-1). A total of 9 coho salmon jacks were killed (7 of hatchery origin; 2 wild), for an immediate mortality rate of 0.42%. Most jack mortalities occurred from wedging in the spiller mesh; the remainder resulted from wedging in the downstream panel of the heart. From these results, immediate survival for all ages of Chinook salmon and steelhead trout was 100%. Adult coho salmon immediate survival was 100%; combined immediate survival for all ages of the species was 98.87%.

Table 3-1. Stock-specific immediate mortality during the 2016 study period.

Species	Total Captured	Mortalities (Adults)	Mortalities (Jacks)	% Immediate Mortality
Chinook	534	0	0	0.00%
Coho	796	0	9	1.13%
Steelhead	816	0	0	0.00%

3.1.3 Marine Mammal Encounters

The presence of marine mammals was noted at various times throughout the study but was not systematically quantified. Harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and Steller sea lions (*Eumetopias jubatus*) were sighted from August through September. The relative abundance of marine mammals increased in late September. In response, a marine mammal barrier (a 7 ft. aluminum bar positioned vertically) was installed on the tunnel orifice which successfully deterred entry of all mammals sighted at the trap while enabling passage of fish into the spiller compartment. Prior to installation of the marine mammal barrier, two harbor seals entered the spiller and were released unharmed by lowering the spiller mesh. Although encounters with marine mammals were minimal in 2016, improved deterrence was recognized as a necessary modification for following years of research.

3.1.4 Identified Trap Modification Needs Following the 2016 Study

Various potential modifications to engineering and deployment were identified by WFC staff to improve capture efficiency, reduce wedging of jacks, and deter marine mammals in future years of research and/or test fishing. Major issues in 2016 included the following:

1) Lead and heart nets were not fully descended to the river bottom during portions of the study, likely reducing capture of benthic oriented species (e.g. Chinook salmon);
2) Friction was substantial during lifting and lowering of the spiller compartment, reducing the speed of lift and preventing the spiller and tunnel from sitting flush with the river bed;
3) During high tides, the lifting point of the spiller proved too low, reducing the ability of fishers to spill fish during high-tides;
4) The 3-1/8'' mesh size of the upper spiller compartment was too large, enabling alarmed jacks during the lifting process to become wedged or gilled immediately prior to sorting;
5) The shape of the spiller bunt was too flat, allowing fish to get trapped in inaccessible locations (e.g. corners of the spiller compartment) during lift;
6) A marine mammal deterrent was needed to prevent entry of seals and sea lions to the heart compartment.

3.2 Post-Release Survival Study in 2017

3.2.1 Total Catch and CPUE

The experimental trap was fished for 290.52 h over 33 d between August 26 and September 27, 2017. During this period, 381 sets were performed with a median soak length of 36 minutes (min = 6 min; max = 336 min; mean = 46 min; SD = 36 min). The median time between the conclusion of a treatment soak and re-deployment was approximately 3 minutes.

A total of 7,129 salmonids were captured and released. Mean daily catch was 215 salmonids with a maximum catch of 506 on September 8 and a minimum of 4 on September 27 (Figure 3-3). Total catch was composed of 49.1% coho salmon (3501 total; 52.4% ad-clipped; 16.4% jack salmon), 37.4% Chinook salmon (2670 total; 47.9% ad-clipped; 16.3% jack salmon), 12.9% summer steelhead trout (921 total; 80.9% ad-clipped; 10.5% B-run (> 78cm)), 0.4%

resident/residualized *Oncorhynchus mykiss* (29 total; 77.8% ad-clipped), and 0.1% *Oncorhynchus spp.* (8 total) (Figure 3-4). From these salmonid results, the ratio of steelhead bycatch (wild and hatchery-origin) to targeted hatchery-origin Chinook and coho salmon was approximately 1:3. The ratio of total wild-origin to hatchery-origin salmonids captured was also approximately 1:3. In addition to salmonid catch, American shad (3 total), largemouth bass (1 total), common carp (*Cyprinus carpio*) (1 total), and peamouth (*Mylocheilus caurinus*) (1 total) were captured and released throughout the study period.

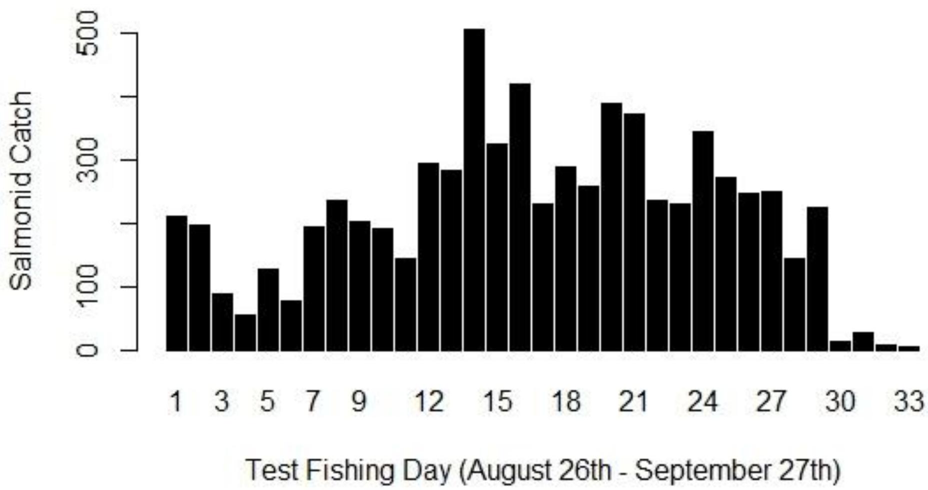

Figure 3-3. Total 2017 catch of Chinook, coho, and steelhead throughout the test fishing period.

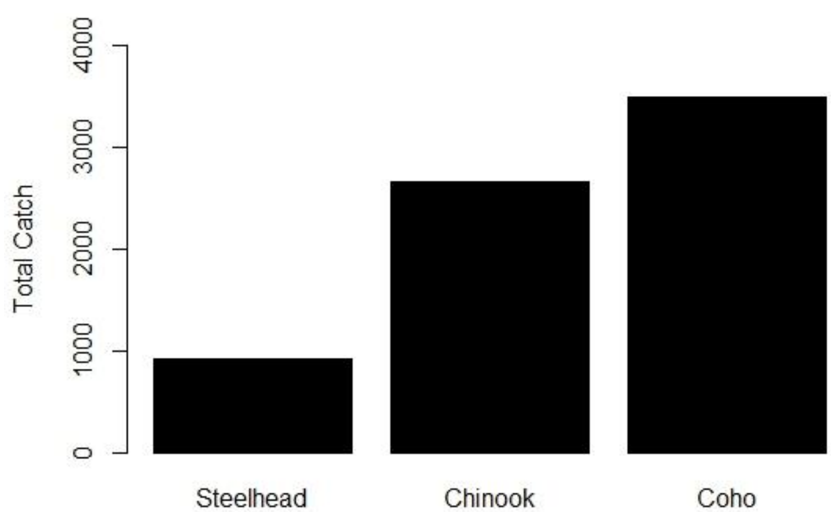

Figure 3-4. Salmonid catch by species from August 26 through September 29, 2017.

Mean salmonid CPUE after 290.52 h of total fishing effort was 24.54/h (Figure 3-5). Daily CPUE for all salmonids peaked at 75.5 salmonids/h on September 9 (mean = 25.2, SD = 16.8). For coho salmon, daily CPUE peaked at 39.3/h on September 7 (mean = 12.5, SD = 10.5). Chinook salmon daily CPUE peaked at 30.9/h on September 8 (mean = 9.4, SD = 7.0). Steelhead daily CPUE peaked at 9.4/h on September 14 (mean = 3.3, SD = 2.5). Analyzing 381 unique sets performed throughout the study period, Chinook CPUE per set ranged from 0 to 225 chinook/h (median = 5.90, mean = 15.48, SD = 27.67). Coho CPUE per set ranged from 0 to 290 coho/h (median = 7.20, mean = 21.27, SD = 37.75). Steelhead CPUE per set ranged from 0 to 110 steelhead/h (median = 1.85, mean = 4.92, SD = 9.74).

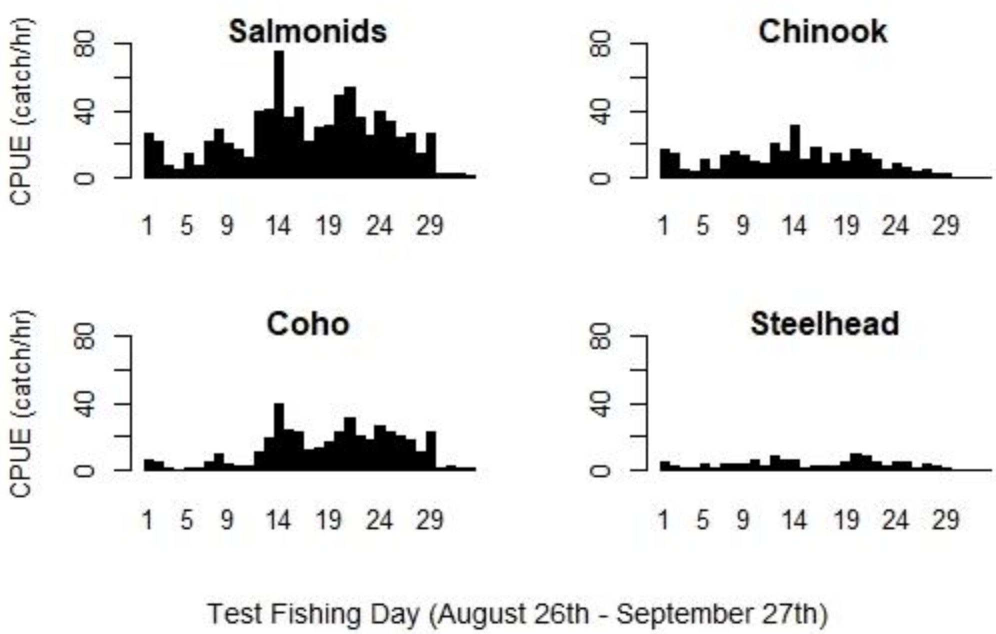

Figure 3-5. Daily CPUE (catch/h) of all salmonids, Chinook salmon, coho salmon, and steelhead trout from August 26 through September 27, 2017.

CPUE of Chinook and coho salmon were analyzed during two periods in which the 2017 August and early-fall lower Columbia River non-Indian commercial Chinook and coho salmon fisheries took place (ODFW 2017). Dates were adjusted by one day to account for the migration of fish from the fish trap site at Zone 2 to the location of the gillnet fleet in Zone 4. Table 3-2 and 3-3 summarize the results during this period for the experimental trap and the commercial gillnet fleet. CPUE in this case represents total catch of a stock divided by the mean number of deliveries (a proxy for the number of fishing vessels) and total hours of operation. Equivalent labor inputs are assumed between gear-types. Mean CPUE for the experimental trap was 5.50 and 6.61 for Chinook and coho salmon respectively. Mean CPUE for the average gillnetter was 3.02 for Chinook salmon and 0.18 for Coho salmon. During these overlapping periods of operation, the trap outperformed the average gillnetter by a factor of 1.82 for Chinook salmon and 35.98 for coho salmon (Tables 3-2 and 3-3).

Table 3-2. Catch results for the experimental trap during weeks in which the lower Columbia River non-Indian commercial gillnet fleet operated in 2017. CPUE represents daily stock-specific catch divided by the number of hours fished in a day.

Date	Vessels	Effort (Hours)	Chinook Total	Chinook CPUE	Coho Total	Coho CPUE
26-Aug	1	12.85	128	9.96	46	3.58
27-Aug	1	13.62	129	9.47	47	3.45
28-Aug	1	13.35	52	3.90	17	1.27
29-Aug	1	12.72	40	3.15	3	0.24
30-Aug	1	12.80	90	7.03	11	0.86
31-Aug	1	13.25	49	3.70	15	1.13
16-Sep	1	13.28	67	5.04	137	10.31
17-Sep	1	12.78	40	3.13	171	13.38
18-Sep	1	13.08	75	5.73	231	17.66
19-Sep	1	12.78	48	3.75	185	14.47

Table 3-3. Catch results for the lower Columbia River non-Indian commercial gillnet fleet. CPUE represents daily stock-specific catch divided by the estimated number of fishing vessels and the number of hours fished in a day.

Date	Estimated Vessels	Effort (Hours)	Chinook Total	Chinook CPUE	Coho Total	Coho CPUE
8/27-8/28	122	9	5544	5.05	129	0.12
8/29-8-30	112	9	1805	1.79	20	0.02
8/31-9/1	96	9	1563	1.81	12	0.01

9/17-9/18	107	10	3651	3.41	404	0.38
9/19-9/20	69	10	1788	2.59	309	0.45

3.2.2 Regression Analysis of CPUE

Multiple linear regression was used to explain variation in species-specific CPUE for the 381 sets performed in 2017. Through the backwards-elimination/deletion approach, only water temperature was determined to be non-significant of all considered covariates explaining Chinook salmon CPUE. The following model was selected for Chinook salmon:

$$\ln(CPUE_{chinook} + 1) = \beta_0 + \beta_1 \text{ (Tidal Stage}_i) + \beta_2 \text{ (Tide Height)} + \beta_3 \text{ (Time of Day}_i) + \beta_4 \text{ (MMG Position}_i) + \beta_5 \text{ (Bonneville Count)} + \varepsilon$$

Modeling through the R-platform, all partial regression coefficients were statistically significant at the $P \leq 0.05$ significance level through last-entry analysis (Table 3-4). The association and significance of each coefficient is described in order of association (positive vs. negative), followed by statistical significance: daily Bonneville Dam count (P ($|t| \geq 5.139$) < 0.001, association = positive), the intercept term (P ($|t| \geq 4.025$) < 0.001), mean tide height (P ($|t| \geq$ 3.099) = 0.002, association = positive), tide stage (flood tide) (P ($|t| \geq -5.780$) < 0.001, association = negative), MMG position (P ($|t| \geq -3.896$) < 0.001, association = negative), and time of day (night) (P ($|t| \geq -2.213$) = 0.028, association = negative). Although all covariates had statistically significant impacts on the response variable and the model was significant at the $P \leq$ 0.05 level (P ($|F_{9,343}| \geq 11.67$) < 0.001), only a small proportion of the total variation in Chinook salmon CPUE was explained through the multiple regression model (R^2 = 0.235).

Table 3-4. Summary of covariates from the multiple regression model used to explain Chinook salmon CPUE, ranked by association and *P*-value for last entry into the model.

Independent Variable	***P*-value**	***t*-value**	**Association**	**Coefficient**	$\widehat{SE}$
Bonneville Dam Count	0.000	5.139	+	4.61e-05	8.96e-06
Intercept Term	0.000	4.025	+	1.213	0.302
Mean Tide Height	0.002	3.099	+	0.124	0.040
Tidal Stage (Flood)	0.000	-5.780	-	-0.861	0.149

Marine Mammal Gate	0.000	-3.896	-	-0.666	0.171
Time of Day (Night)	0.028	-2.213	-	-0.725	0.327

Through the backwards-elimination/deletion approach, only water temperature and marine mammal gate position were determined to be non-significant of all considered covariates explaining coho salmon CPUE. The following model was selected for coho salmon:

$$\ln(\text{CPUE}_{coho} + 1) = \beta_0 + \beta_1 (\text{Tidal Stage}_i) + \beta_2 (\text{Tide Height}) + \beta_3 (\text{Time of Day}_i) + \\ + \beta_4 (\text{Bonneville Count}) + \varepsilon$$

Through last-entry analysis, all partial regression coefficients were statistically significant at the $P \leq 0.05$ significance level with the exception of mean tide height, which was significant at the 0.10 level (Table 3-5). The association and significance of each coefficient is described in order of association (positive vs. negative), followed by statistical significance: daily Bonneville Dam count (P ($|t| \geq 10.423$) < 0.001, association = positive), the intercept term (P ($|t| \geq 3.269$) = 0.001), mean tide height (P ($|t| \geq 1.678$) = 0.094, association = positive), tide stage (flood tide) (P ($|t| \geq -3.131$) = 0.002, association = negative), and time of day (night) (P ($|t| \geq -2.920$) = 0.004, association = negative). Although the majority of these covariates had statistically significant impacts on the response variable and the model was significant at the $P \leq 0.05$ level (P ($|F_{8,372}| \geq 18.71$) < 0.001), only a small proportion of the total variation in coho salmon CPUE was explained through the multiple regression model ($R^2 = 0.287$).

Table 3-5. Summary of covariates from the multiple regression model used to explain coho salmon CPUE, ranked by association and *P*-value for last entry into the model.

Independent Variable	***P*-value**	***t*-value**	**Association**	**Coefficient**	$\widehat{SE}$
Bonneville Dam Count	0.000	10.423	+	5.27e-04	5.06e-05
Intercept Term	0.001	3.269	+	0.884	0.270
Mean Tide Height	0.094	1.678	+	0.065	0.039
Tidal Stage (Flood)	0.002	-3.131	-	-0.449	0.143
Time of Day (Night)	0.004	-2.920	-	-0.917	0.314

Of all considered covariates explaining summer steelhead CPUE, only water temperature was determined to be non-significant. The following model was selected for steelhead trout:

$$\ln(\text{CPUE}_{\text{steelhead}} + 1) = \beta_0 + \beta_1\,(\text{Tidal Stage}_i) + \beta_2\,(\text{Tide Height}) + \beta_3\,(\text{Time of Day}_i) + \beta_4\,(\text{MMG Position}_i) + \beta_5\,(\text{Bonneville Count}) + \varepsilon$$

Through last-entry analysis, all partial regression coefficients were statistically significant at the $P \leq 0.05$ significance level with the exception of the intercept term (Table 3-6). The association and significance of each coefficient is described in order of association (positive vs. negative), followed by statistical significance: daily Bonneville Dam count ($P\ (|t| \geq 5.323) < 0.001$, association = positive), mean tide height ($P\ (|t| \geq 3.941) < 0.001$, association = positive), time of day (day) ($P\ (|t| \geq 2.208) = 0.028$, association = positive), time of day (dusk) ($P\ (|t| \geq 2.277) = 0.023$, association = positive), MMG position ($P\ (|t| \geq -4.181) < 0.001$, association = negative), tide stage (flood tide) ($P\ (|t| \geq -3.505) = 0.001$, association = negative), and time of day (night) ($P\ (|t| \geq -2.822) = 0.001$, association = negative). Although all covariates had statistically significant impacts on the response variable and the model was significant at the $P \leq 0.05$ level ($P\ (|F_{9,349}| \geq 12.46) < 0.001$), only a small proportion of the total variation in steelhead trout CPUE was explained through the multiple regression model ($R^2 = 0.243$).

Table 3-6. Summary of covariates from the multiple regression model used to explain summer steelhead CPUE, ranked by association and *P*-value for last entry into the model.

Independent Variable	*P*-value	*t*-value	Association	Coefficient	$\widehat{SE}$
Bonneville Dam Count	0.000	5.323	+	5.31e-04	9.98e-05
Mean Tide Height	0.002	3.941	+	0.119	0.030
Time of Day (Dusk)	0.023	2.277	+	0.523	0.230
Time of Day (Day)	0.028	2.208	+	0.388	0.176
Marine Mammal Gate	0.000	-4.181	-	-0.521	0.125
Tidal Stage (Flood)	0.001	-3.505	-	-0.403	0.115
Time of Day (Night)	0.001	-2.822	-	-0.711	0.252

3.2.3 Total Tagged Fish and Upstream Detections

A total of 2,848 Chinook salmon and steelhead trout were PIT-tagged throughout the study period. Random sampling and assignment of control and treatment tagging sessions resulted in fairly equal representation of mixed-stock throughout the fishing period for both control and treatment groups (Figures 3-6 and 3-7). In addition, 13 previously tagged fish were recaptured at the trap (most of which were previously tagged at the trap site). However, this small group of previously tagged fish was excluded from the analysis due to the potential difference in handling survival from those that had undergone the standard tagging procedure. Of the tagged fish, 2,066 were Chinook salmon (976 control; 1090 treatment); 782 were steelhead trout (379 control; 403 treatment). Through a PTAGIS database query on January 30, 2018, there were 1,848 detections of unique WFC tag codes from 43 active PIT tag arrays throughout the Columbia River Basin. A total of 35 detections were made downstream of the trap site on the Oregon side at the Columbia River Estuary array. Chinook and steelhead were detected in locations hundreds of kilometers upstream at arrays including the lower Okanagan and lower South Fork Clearwater.

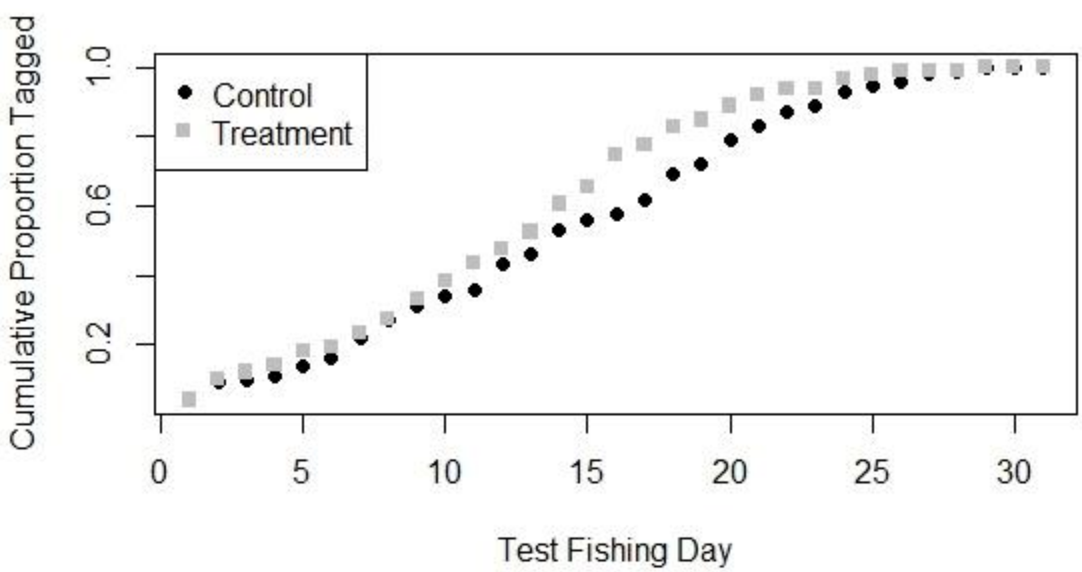

Figure 3-6. Cumulative proportion of tagged Chinook salmon control and treatment groups.

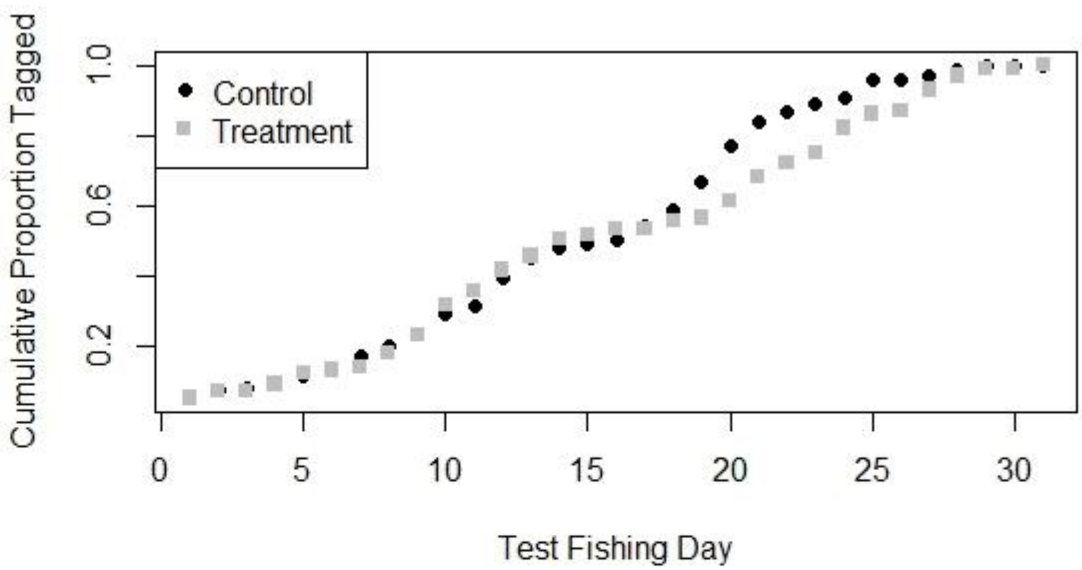

Figure 3-7. Cumulative proportion of tagged steelhead trout control and treatment groups.

3.2.4 Total Fin-clip Samples and Genotyping

Fin-clip samples were obtained from 2,828 Chinook salmon and steelhead trout throughout the study period, representing 99.3% of the tagged population. Of these samples, 2,046 were Chinook salmon (964 control; 1082 treatment); 782 were steelhead trout (380 control; 402 treatment). A random sub-sample of 507 Chinook fin-clip samples were selected from four discrete time periods—separately for control and treatment samples—in proportion to their abundance within each period (Table 3-7). These samples were analyzed with the appropriate set of Columbia basin-specific SNP markers to assign individuals to defined population groups below and above Bonneville Dam.

Table 3-7. Chinook fin-clip samples randomly selected for population group assignment.

Period	Control	Treatment
One (8/26 - 9/2)	65	75
Two (9/3 - 9/10)	74	125
Three (9/11 - 9/18)	85	56
Four (9/19 - 9/27)	17	10
N	**241**	**266**

Of the 507 genetic samples submitted for population group assignment, only 11 samples (6 control, 5 treatment) could not be genotyped with high confidence to reporting groups either above or below Bonneville Dam (Miller et al. 2018). Eliminating these 11 samples from the dataset, 496 were successfully assigned (Table 3-8). Through GLM/log-linear analysis, there was no significant association between control and treatment groups and Columbia Basin population group assignment (P ($\chi_1^2 \geq 0.000$) = 1.000). From these results, it is clear that stock-composition is equivalent between control and treatment groups at the $P \leq 0.05$ significance level.

Table 3-8. Contingency table of assigned Columbia Basin population groups for control and treatment Chinook salmon. The observed frequency in each cell is shown, with the frequency expected (in parentheses) if there is no association between control and treatment group and population group assignment.

	Control	**Treatment**	**Frequency**
Below Bonneville Populations	47 *(46.91)*	52 *(52.09)*	99
Above Bonneville Populations	188 *(188.10)*	209 *(208.91)*	397
Frequency	235	261	496

3.2.5 Immediate Survival

Throughout the duration of the study, there were a total of 9 immediate mortalities out of 7,135 fish captured (Table 3-9). Of these mortalities, only 2 were adult fish (1 Chinook; 1 coho) with the remainder being jacks or resident/residualized salmonids (1 Chinook; 4 coho; 2 *O. mykiss*). The two adult mortalities occurred for unknown reasons in the spiller compartment, but were likely caught in a fold of the spiller mesh during lift. Two jack mortalities occurred from wedging in the spiller mesh, with the remainder resulting from wedging in the downstream panel of the heart (typically after noted marine mammal encounters). From these results, immediate mortality of adult Chinook, coho, and steelhead was near zero (e.g. 0.03%).

Table 3-9. Immediate mortalities during the 2017 study period. *O. mykiss** represents resident or residualized hatchery-origin *O. mykiss* < 300 mm.

Species	Total Captured	Mortalities (Adults)	Mortalities (All Life-Stages)	% Immediate Mortality
Chinook	2670	1	2	0.07%
Coho	3501	1	5	0.14%
Steelhead	921	0	0	0.00%
*O. mykiss**	29	n/a	2	6.90%

3.2.6 Analysis of Chinook Salmon Fork-length and Migration Timing

Of 2,066 tagged Chinook salmon (976 control; 1090 treatment), the mean fork length included in the study was 739.30 mm (max = 1,000, min = 500, SD = 85.07). Mean fork length for the control group was 734.00 mm ($\widehat{SE}$ = 2.72) and 744.13 ($\widehat{SE}$ = 2.57) for the treatment group. Analyzed through a two-sample t-test, mean fork length was statistically different between the two groups at the $P \leq 0.05$ significance level (P ($|t_{2067}| \geq 2.71$) = 0.007).

The median arrival date for Chinook salmon was September 12 at Bonneville Dam and September 22 at McNary Dam (Table 3-10). The median travel time between release and Bonneville was 6 d, with a mean of 6.49 d (CI ($6.31 \leq \widehat{T} \leq 6.67$) = 0.95). Mean travel time for the control group was 6.05 d ($\widehat{SE}$ = 0.13) and 6.89 d ($\widehat{SE}$ = 0.13) for the treatment group. Analyzed through a two-sample t-test, the control Chinook salmon group travelled more quickly to Bonneville Dam than the treatment group at the $P \leq 0.05$ significance level (P ($|t_{1189}| \geq 4.627$) < 0.001). The median travel time between release from the gear to McNary Dam was 13 d, with a mean of 14.70 d (CI ($14.19 \leq \widehat{T} \leq 15.22$) = 0.95). Mean travel time for the control group was 14.49 d ($\widehat{SE}$ = 0.38) and 14.90 d ($\widehat{SE}$ = 0.36) for the treatment group. Travel time of control and treatment Chinook salmon did not differ to McNary Dam at the $P \leq 0.05$ significance level (P ($|t_{490}| \geq 0.795$) = 0.427).

Table 3-10. First, last, and median detection date for tagged Chinook salmon as of 1/30/2018.

Detection Site	River km	Number of Tags	Median Detection	First Detection	Last Detection
Bonneville	233	1191	9/13/2017	8/29/2017	10/14/2017
McNary	470	492	9/22/2017	9/5/2017	10/27/2017

3.2.7 Post-release Survival of Chinook Salmon

Retrieving unique capture histories for control and treatment Chinook salmon through PTAGIS, the following cell counts were entered into Program USER to estimate post-release survival (Table 3-11):

Table 3-11. Control and treatment cell counts for all possible capture histories at four mainstem river detection locations. A "1" denotes detection and "0" nondetection at each upstream detection location in order from lowest to highest rkm (Bonneville Dam, The Dalles Dam, McNary Dam, and pooled detection points upstream of McNary Dam). *N* denotes the total number tagged in each group.

History	Control Count	Treatment Count
1111	133	128
0111	1	1
1011	3	0
0011	0	0
1101	0	0
0111	0	0
1001	0	0
0001	0	0
1110	95	127
0110	1	1
1010	0	2
0010	0	0
1100	98	120
0100	1	2
1000	243	242
0000	401	467
N	**976**	**1090**

Given cell counts for each unique capture history (Table 3-11), post-release survival was estimated within four upstream mainstem river reaches through the Cormack-Jolly-Seber method and Program USER (Table 3-12). LRT found no significant difference in PIT tag array detection efficiencies for control and treatment groups at the $P \leq 0.05$ significance level (P ($\chi^2 \geq 0.364$) = 0.540), resulting in selection of the common detection probability (p_i) reduced model. Post-release survival was high from release to Bonneville Dam, at 0.970 (CI ($0.901 \leq \hat{S} \leq 1.000$) = 0.95). The treatment group outperformed the control group between Bonneville Dam and The Dalles Dam, with survival increasing in this reach to 1.060 (CI ($0.965 \leq \hat{S} \leq 1.000$) = 0.95). Post-release survival declined slightly but remained high at 0.968 (CI ($0.877 \leq \hat{S} \leq 1.000$) = 0.95) from The Dalles Dam to McNary Dam—the final detection point used for previous alternative gear studies of the lower Columbia River. Surprisingly, survival declined to 0.847 (CI ($0.719 \leq \hat{S} \leq 0.997$) = 0.95) in the final river reach of this analysis from McNary Dam to pooled upstream detection points. This suggests a potential latent mortality effect. However, it must be noted that the sample size beyond McNary Dam was small ($n_{control}$ = 137, $n_{treatment}$ = 129), resulting in large standard error ($\widehat{SE}$ = 0.070) and a wide confidence interval containing prior reach survival point estimates.

Table 3-12. Post-release survival point-estimates for adult fall Chinook salmon released from the experimental pound net trap, with standard error ($\widehat{SE}$) and the profile likelihood confidence interval estimated through the Cormack-Jolly-Seber method and Program USER.

River Reach	Survival Point Estimate	$\widehat{SE}$	Profile Likelihood 95% Confidence Interval
Gear to Bonneville Dam (τ_1)	0.970	0.036	0.901 – 1.000
Bonneville Dam to The Dalles Dam (τ_2)	1.060	0.051	0.965 – 1.000
The Dalles Dam to McNary Dam (τ_3)	0.968	0.049	0.877 – 1.000
McNary to pooled upstream arrays (τ_4)	0.847	0.070	0.719 – 0.997

Utilizing detection points chosen by WDFW (2014) to estimate short-term, long-term, and cumulative survival of salmon released from purse and beach seines, cumulative survival from the experimental trap to McNary Dam ($i{*}\tau_1{*}\tau_2{*}\tau_3$) was estimated at 0.995 (CI ($0.925 \leq \hat{S} \leq$

1.000) = 0.95) for fall Chinook salmon (Table 3-13). Short-term post-release survival of Chinook salmon from the gear to Bonneville Dam (τ_1) was estimated at 0.970 (CI ($0.901 \leq \hat{S} \leq 1.000$) = 0.95). Long-term post-release survival of Chinook salmon from Bonneville Dam to McNary Dam ($\tau_2 * \tau_3$) was estimated at 1.026 (CI ($0.934 \leq \hat{S} \leq 1.000$) = 0.95). Performing a Z-test on detection probabilities, there was no statistical difference between the control and treatment groups at the $P \leq 0.05$ significance level ($P_{cumulative}$ ($|Z_{cumulative}| \geq 0.059$) = 0.953, P_{short} ($|Z_{short}| \geq 0.808$) = 0.419, P_{long} ($|Z_{short}| \geq 0.370$) = 0.712).

Table 3-13. Fall Chinook post-release survival point estimates and associated profile likelihood 95% confidence intervals from the experimental pound net trap, employing detection points selected by WDFW (2014). *Short-term survival from the gear to Bonneville is a similar metric to that investigated by Vander Haegen (2004) and Ashbrook (2008).

CUMULATIVE: GEAR TO MCNARY				
Treatment	No. Tagged	No. Recaptured	Recapture Prob.	Survival
Control	976	233	0.239	---
Pound Net	1090	259	0.238	0.995 (0.925 - 1.000)
SHORT-TERM: GEAR TO BONNEVILLE*				
Treatment	No. Tagged	No. Recaptured	Recapture Prob.	Survival
Control	976	575	0.589	---
Pound Net	1090	623	0.572	0.970 (0.901 – 1.000)
LONG-TERM: BONNEVILLE TO MCNARY				
Treatment	No. Over BON	No. Recaptured	Recapture Prob.	Survival
Control	575	233	0.405	---
Pound Net	623	259	0.416	1.026 (0.934 – 1.000)

3.2.8 Analysis of Steelhead Trout Fork-length and Migration Timing

Of 792 tagged steelhead trout (383 control; 409 treatment), the mean fork length included in the study was 642.70 mm (max = 1000, min = 500, SD = 82.31). Mean fork length for the control group was 641.51 mm ($\widehat{SE}$ = 4.21) and 643.75 mm ($\widehat{SE}$ = 4.08) for the treatment group. Analyzed through a two-sample t-test (log-transformed to account for right skewness), mean fork length was statistically equivalent between the two groups at the $P \leq 0.05$ significance level (P ($|t_{789}| \geq 0.496$) = 0.620).

The median arrival date for steelhead trout was September 18 at Bonneville Dam and September 30 at McNary Dam (Table 3-14). The median travel time between release and Bonneville was 6 d, with a mean of 8.03 (CI ($7.57 \leq \widehat{T} \leq 8.49$) = 0.95). Mean travel time for the control group was 7.85 d ($\widehat{SE}$ = 0.33) and 8.20 d ($\widehat{SE}$ = 0.33) for the treatment group. Analyzed through a two-sample t-test, travel time of control and treatment steelhead trout from release to Bonneville Dam did not differ at the $P \leq 0.05$ significance level (P ($|t_{622}| \geq 0.741$) = 0.459). The median travel time between release from the gear to McNary Dam was 18 d, with a mean of 21.72 (CI ($20.78 \leq \widehat{T} \leq 22.66$) = 0.95). Mean travel time for the control group was 21.90 d ($\widehat{SE}$ = 0.68) and 21.54 d ($\widehat{SE}$ = 0.68) for the treatment group. Travel time of control and treatment steelhead trout did not differ to McNary Dam at the $P \leq 0.05$ significance level (P ($|t_{529}| \geq -0.375$) = 0.708).

Table 3-14. First, last, and median detection date for tagged steelhead trout as of 1/30/2018.

Detection Site	River Mile	Number of Tags	Median Detection	First Detection	Last Detection
Bonneville	233	624	9/18/2017	8/31/2017	10/26/2017
McNary	470	531	9/30/2017	9/13/2017	12/12/2017

3.2.9 Post-release Survival of Steelhead Trout

Retrieving unique capture histories for control and treatment summer steelhead trout through PTAGIS, the following cell counts were entered into Program USER to estimate post-release survival (Table 3-15):

Table 3-15. Control and treatment cell counts for all possible capture histories at four mainstem river detection locations. A “1” denotes detection and “0” nondetection at each upstream detection location in order from lowest to highest rkm (Bonneville Dam, The Dalles Dam, McNary Dam, and pooled detection points upstream of McNary Dam). *N* denotes the total number tagged in each group.

History	Control Count	Treatment Count
1111	256	255
0111	0	3
1011	0	0
0011	0	0
1101	1	2
0111	0	0
1001	0	0
0001	0	1
1110	10	7
0110	0	0
1010	0	0
0010	0	0
1100	17	22
0100	0	0
1000	24	30
0000	71	83
N	**379**	**403**

Given cell counts for each unique capture history (Table 3-15), post-release survival was estimated for four upstream mainstem river reaches through the Cormack-Jolly-Seber method and Program USER (Table 3-16). LRT found a significant difference in PIT tag array detection efficiencies for control and treatment groups at the $P \leq 0.05$ significance level (P ($\chi 2 \geq 6.874$) = 0.008), resulting in selection of the full model. Post-release survival was high from release to Bonneville Dam, at 0.978 (CI ($0.912 \leq \hat{S} \leq 1.000$) = 0.95). Post-release survival remained high in subsequent reaches between Bonneville Dam and The Dalles Dam and between The Dalles Dam and McNary Dam, increasing to 0.982 (CI ($0.935 \leq \hat{S} \leq 1.000$) = 0.95) and 0.983 (CI ($0.939 \leq \hat{S} \leq 1.000$) = 0.95) respectively. In contrast to Chinook salmon, the treatment group outperformed the control group in the final river reach of this analysis from McNary Dam to pooled upstream detection points, with post-release survival increasing to 1.012 (CI ($0.980 \leq \hat{S} \leq 1.000$) = 0.95).

Table 3-16. Post-release survival point-estimates for adult summer steelhead trout released from the experimental pound net trap, with standard error ($\widehat{SE}$) and the profile likelihood confidence interval estimated through Program USER.

River Reach	Survival Point Estimate	$\widehat{SE}$	Profile Likelihood 95% Confidence Interval
Gear to Bonneville Dam (τ_1)	0.978	0.035	0.912 – 1.000
Bonneville Dam to The Dalles Dam (τ_2)	0.982	0.024	0.935 – 1.000
The Dalles Dam to McNary Dam (τ_3)	0.983	0.022	0.939 – 1.000
McNary to pooled upstream arrays (τ_4)	1.012	0.016	0.980 – 1.000

Utilizing detection points chosen by WDFW (2014) to estimate short-term, long-term, and cumulative survival of steelhead released from purse and beach seines, cumulative survival from the experimental trap to McNary Dam ($i{*}\tau_1{*}\tau_2{*}\tau_3$) was estimated at 0.944 (CI ($0.880 \leq \widehat{S} \leq 1.000$) = 0.95) for summer steelhead trout (Table 3-17). Short-term post-release survival of steelhead from the gear to Bonneville Dam (τ_1) was estimated at 0.977 (CI ($0.911 \leq \widehat{S} \leq 1.000$) = 0.95). Long-term post-release survival of steelhead from Bonneville to McNary Dam ($\tau_2{*}\tau_3$) was estimated at 0.966 (CI ($0.919 \leq \widehat{S} \leq 1.000$) = 0.95). Performing a Z-test on upriver detection probabilities, there was no statistical difference between the control and treatment groups at the 0.05 significance level ($P_{cumulative}$ ($|Z_{cumulative}| \geq 1.187$) = 0.235, P_{short} ($|Z_{short}| \geq 0.654$) = 0.513, P_{long} ($|Z_{short}| \geq 1.036$) = 0.300).

Table 3-17. Summer steelhead post-release survival point estimates and associated profile likelihood 95% confidence intervals from the experimental pound net trap, employing detection points selected by WDFW (2014). *Short-term survival from the gear to Bonneville is a similar metric to that investigated by Vander Haegen (2004) and Ashbrook (2008).

CUMULATIVE: GEAR TO MCNARY

Treatment	No. Tagged	No. Recaptured	Recapture Prob.	Survival
Control	379	267	0.704	---
Pound Net	403	268	0.665	0.944 (0.880 - 1.000)

SHORT-TERM: GEAR TO BONNEVILLE

Treatment	No. Tagged	No. Recaptured	Recapture Prob.	Survival

Control	379	308	0.813	---
Pound Net	403	320	0.794	0.977 (0.911 - 1.000)

LONG-TERM: BONNEVILLE TO MCNARY

Treatment	No. Over BON	No. Recaptured	Recapture Prob.	Survival
Control	308	267	0.867	---
Pound Net	320	268	0.838	0.966 (0.919 - 1.000)

3.2.10 Marine Mammal Encounters

Of 381 total sets performed, the marine mammal gate was deployed 81 times due to the presence of mammals in the vicinity of the study location. On 11 separate occasions, harbor seals (*Phoca vitulina*) or California sea lions (*Zalophus californianus*) entered the heart of the trap. In most of these situations, marine mammals entered when trap operators were caught off-guard, could not sight the animals, or could not close the marine mammal gate in time. Only in 4 of 11 instances of marine mammal entry was the gate effectively deployed. During these instances, entry was likely achieved through small gaps between the gate frame and the river bottom or the heart mesh lead line and the river bottom when river and tidal currents were strong. With a total of 4 mammal entries during 81 gate closure events, the gate demonstrated a deterrent success rate of 95.1%. In all situations of marine mammal entry, the spiller compartment was lifted and mammals departed within minutes. No physical injury of mammals was observed throughout the duration of the study period. However, fish (species unknown) were taken by marine mammals on 5 separate occasions.

Chapter 4: Discussion

4.1 Relative Performance of the Experimental Fish Trap – Bycatch Impacts

This study has demonstrated the viability of an experimental fish trap as a stock-selective harvest tool, presenting a partial solution to hatchery and bycatch problems within the Columbia Basin and other Pacific Northwest fisheries. Capturing 7,129 salmonids during the 2017 study (including 6,171 coho and Chinook salmon) it is evident that traps can effectively capture commercially viable quantities of fish. Furthermore, when operated with a conservation-minded approach, operators of the gear can successfully release the great majority of non-target salmonids unharmed. Depending on the conservation issues present within a fishery, the fish trap is yet another tool that can be successfully deployed to address bycatch and hatchery management concerns while enabling continuation of commercial fishing (Table 4-1).

Table 4-1. Lower Columbia River cumulative survival estimates from five different gear-types and associated 95% confidence intervals (if available) (WDFW 2014; IFSP 2014; WDFW and ODFW Joint Staff 2018).

Gear	Chinook Survival	Steelhead Survival
Gillnet	0.520	0.555
Tangle Net	0.791	0.764
Beach Seine	0.750 (0.710 – 0.790)	0.920 (0.820 – 1.000)
Purse Seine	0.780 (0.720 – 0.850)	0.980 (0.930 – 1.000)
Fish Trap	0.995 (0.925 – 1.000)	0.944 (0.880 – 1.000)

Mirroring methodology employed by WDFW for consistent comparison between gear-types tested in the lower Columbia River, cumulative survival of Chinook salmon released from the experimental trap represents a statistically significant and dramatic improvement over survival estimates produced from previous alternative and conventional gear studies (Table 4-1). Analyzing cumulative survival from the capture and release site to McNary Dam, the experimental trap outperformed all gears utilized on the lower Columbia River, with survival estimated at 0.995 and the confidence interval exceeding that of all prior studies (CI ($0.925 \leq \widehat{S} \leq 1.000$) = 0.95).

Despite this promising result, the apparent decline in Chinook salmon survival observed above McNary Dam (0.847, CI ($0.719 \leq \widehat{S} \leq 0.997$) = 0.95) in this analysis suggests that further research should be conducted for the fish trap and all other alternative gears to better understand latent mortality effects of commercial fishing on salmonids destined for long-range upriver migration. The sample size utilized in this analysis to determine the point estimate for reach survival above McNary Dam was too small (Table 3-11) and the confidence interval too wide to determine whether survival is statistically better or worse than downstream reaches of the 2017 study or previous alternative gear evaluations. Nevertheless, further investigation in warranted. Although no statistical differences were detected in stock composition between control and treatment groups of Chinook salmon sampled throughout the study period (P ($\chi_1^2 \geq 0.000$) = 1.000) (suggesting assumptions of the model have been met through random sampling and assignment of treatment groups), selection of a terminal point located too far upstream for fall Chinook salmon can magnify small differences in stock composition and bias post-release survival estimates in upriver reaches (WDFW and ODFW 2018). For this reason, WDFW (2014) selected a terminal detection point no further upstream than McNary Dam.

For summer steelhead trout, cumulative survival from the experimental trap to McNary Dam was 0.944 (CI ($0.880 \leq \widehat{S} \leq 1.012$) = 0.95). This point estimate is an improvement over that of the gillnet and beach seine; however, it is less than that of the purse seine at 0.980 (CI ($0.930 \leq \widehat{S} \leq 1.000$) = 0.95). Despite this finding, there is substantial overlap of confidence intervals for the experimental trap and seine point estimates, suggesting that there may be a need for further research to better determine which gear truly has greater steelhead release survival. Interestingly, post-release survival in the reach above McNary Dam (1.012, CI ($0.980 \leq \widehat{S} \leq 1.046$) = 0.95) improved for this species, which would bring cumulative survival to 0.955 (CI ($0.891 \leq \widehat{S} \leq 1.024$) = 0.95). This suggests that the steelhead treatment group released from the trap initially experienced a small degree of acute mortality but recovered in upstream reaches to experience minimal overall latent impact. In contrast with the fall Chinook salmon estimate, the sample size for summer steelhead above McNary Dam was relatively robust (Table 3-15), resulting in a much narrower confidence interval and a more reliable point estimate. Nevertheless, for consistency in comparison to other alternative gear studies, it may be best to consider survival to WDFW's terminal point selection of McNary Dam.

Evaluating the results of this post-release survival study relative to previous work conducted in the lower Columbia River, minor differences in methodology and environmental conditions should be noted. While tangle nets, purse seines, and beach seines were tested between rkm 209 and 233 of the lower Columbia River, tagging operations for the experimental trap occurred approximately 150 kilometers downstream at rkm 70 (Figure 2-8). The median migration time of Chinook salmon was 1.7 d to detection points at Bonneville Dam and 7.8 d to McNary Dam for WDFW's (2014) seine study. Given the experimental trap location in the river, the median migration time of Chinook salmon was 6 d to Bonneville Dam and 13 to McNary Dam. Teffer et al. (2017) demonstrated that premature mortality generally does not occur until 5-10 d after release from a conventional gillnet. Since the gear treatment appears to result in lower survival of captured and released fishes than control group sourcing, this large difference in tagging location and migration timing to upstream detection points provides more time to enable a pre-spawn mortality event for fish trap treatment sourced salmonids from sub-lethal physiological impacts, predation, infection, or disease (Davis 2002; Baker and Schindler 2009; Teffer et al. 2017). The tagging location disadvantage of the trap study has potential to bias post-release survival estimated from the trap lower in comparison to previously tested gears in the Columbia River where the treatment effect of each commercial gear on survival was assessed for a much shorter, and potentially insufficient period of time.

Despite the location disadvantage for the relative post-release survival performance of the 2017-pound net trap, it appears that WDFW (2014) may have designed an experiment that in some ways biased survival of seine captured fishes lower than the control sourced fishes (WDFW 2014). Since control sourced salmon and steelhead were trapped at the Bonneville Dam AFF and trucked downstream for release near the test fishing location (rkm 225), these fish may have had a greater propensity to migrate above Bonneville Dam than treatment sourced fishes (which had not yet displayed a preference for upstream migration above Bonneville). Unfortunately, no genetic samples were collected to test for equivalence in stock-composition between control and treatment groups. Nevertheless, the probability of treatment fish migrating up small tributaries prior to Bonneville Dam seems low given the close proximity of the release site to mainstem detections points at rkm 233.

Although potential violation of model assumptions by WDFW (2014) could have biased detection probability of the treatment group lower than the control group, it could also be argued that the control group sourcing of prior alternative gear studies for tangle nets, purse seines, and beach seines likely inflicted greater damage to control fishes, biasing post-release survival estimates from previously tested alternative gears higher relative to the experimental pound net. For all prior alternative gear studies in the lower Columbia, control group fish were trapped at the Bonneville dam AFF, dip-netted, handled, PIT tagged, and trucked downstream to the upstream end of the test fishing location at rkm 225 to be released into the lower Columbia. This procedure is likely stressful to fishes which are susceptible to the effects of handling and transport (Halvorsen et al. 2009; Harris and Hightower 2011), and they must further undergo a repeated migration through the lower Columbia River and the Bonneville Dam adult ladder or adult fish passage facility for a second time. Ultimately, this results in delayed migration which can potentially impact survival (Keefer et al. 2004; Caudill et al. 2007; Murauskas et al. 2014). Fish that have undergone transport and two experiences of entrapment or ascent of adult ladders at Bonneville Dam may be prone to diminished survival upstream, leading one to hypothesize that prior alternative gear evaluations potentially inflicted greater damage to control group fishes than the 2017-pound net study (biasing post-release survival results high relative to the experimental trap).

Ignoring slight differences in methodology and ways in which one study or another could be prone to hypothesized and untested biases, this study demonstrates that cumulative survival of Chinook salmon released from the experimental pound net trap to McNary Dam is statistically greater than all previously tested gears evaluated to the same upstream detection point; cumulative survival of summer steelhead trout is statistically greater or equivalent to the results of prior studies (Table 4-1). Nevertheless, it must be remembered that the severity of a fishery's bycatch impact is the product of two factors: 1) the bycatch mortality rate inflicted by the gear-type; and 2) the quantity of bycatch encountered. Although this study shows that traps can be operated in a way to reduce bycatch mortality rates of fall Chinook salmon and summer steelhead trout in a commercial salmon fishery, traps also appear more likely to encounter greater quantities of bycatch (as presently defined within the lower Columbia River fall Chinook and coho salmon fishery) relative to tangle nets and purse seines. Test fisheries conducted by WDFW (2016) have enumerated bycatch encountered by purse and beach seines relative to target stocks. The ratio of steelhead bycatch (hatchery and wild-origin) to hatchery-origin Chinook and coho salmon captured was approximately 1:3 for the beach seine and 1:11 for the purse seine (WDFW 2016). Mirroring the performance of the beach seine, the experimental trap exhibited a ratio of nearly 1:3 for steelhead to marked Chinook and coho salmon. This result was expected given the similar means through which fish are captured from the shoreline for these two alternative gears. Despite this finding, the picture looks quite different if managers considered the ratio of wild-origin salmonids to hatchery-origin salmonids by gear-type and reconstitute bycatch in the fishery to prioritize removal of hatchery fish for ESA-salmonid recovery. The ratio of wild-origin to hatchery-origin salmonids captured was approximately 3:2 for purse seines, 2:3 for beach seines, and only 1:3 for the experimental trap (WDFW 2016).

4.2 Relative Performance of the Experimental Fish Trap – CPUE

For commercial implementation of any alternative gear-type, a fishing tool must not only demonstrate potential to achieve conservation objectives but exhibit an ability to meet the economic needs of fishermen and industry. Although this would best be evaluated under real-world commercial fishing conditions (e.g. test fisheries) rather than a research setting, the performance of the fish trap in capturing targeted stocks can be roughly compared to that of the gillnetting fleet operating in the lower Columbia River during the same weekly periods in 2017. Assuming equivalent labor inputs (3 fishers) and utilizing daily retained wild and hatchery-origin Chinook and coho salmon catch data, total daily deliveries (a proxy for the total number of fishers), and the hourly duration of each daily opener fished by gillnetters in the 2017 non-Indian Columbia River mainstem August and late-fall fisheries, the average gillnet vessel captured a mean of 3.12 Chinook and 0.18 coho salmon per man/h (ODFW 2017). In comparison, the experimental trap captured 5.50 Chinook and 6.61 coho salmon per man/h active (from initiation of the first daily soak to completion of the final daily haul) during the same weekly periods of operation. However, it must be noted that the average weight of fish captured at the trap was very likely less than that of conventional gillnets due to use of a smaller mesh size. Also, the minimal extent of overlap between operations of the two gear-types necessitates further investigation of relative CPUE.

Despite the limitations of this coarse CPUE comparison, the experimental trap outperformed the average gillnetter in the lower Columbia fall fishery by a factor of nearly 2 for Chinook salmon and a factor of 35 for coho salmon. Furthermore, this was accomplished with the trap tunnel closed for approximately 30% of each fishing period to enable tagging and other research activities. Based upon this information, I conclude that CPUE of fall Chinook and coho salmon from the experimental trap was likely greater than that of the average gillnetter in the lower Columbia fall fishery in 2017. Given the historical effectiveness of commercial fish traps throughout the Pacific Northwest, this comes as no surprise. Despite this finding, fish traps should be monitored under real-world commercial fishing conditions to determine the feasibility of the technology in the context of full-scale Columbia River commercial fisheries. Total costs, revenue, and profit must be analyzed by fishermen and resource managers over multiple years through emerging, trial, or test fisheries to better understand the long-term economic feasibility of the gear, which, depending on the chosen site location, may involve substantial upfront investment. The 2016-2017 experimental trap cost approximately $100,000 to construct (including pile driving, net construction, installation of the winch, and other required supplies). Although total costs will likely decline over time with standardization of parts and economies of scale, the upfront costs of a trap are presently high and must prove surmountable and recoupable to fishermen or co-ops in order to produce anticipated long-term economic benefits.

Comparing catch results between 2016 and 2017 revealed that minor trap design modifications can dramatically affect capture efficiency. In comparison to the trap's 2016 performance, the modified trap in 2017 increased total salmonid CPUE by a factor of 2.95. This increase in efficiency was achieved with only 79.5% of the August 15 through October 15, 2016 run-size of Chinook, coho, and steelhead (Columbia Basin Research Lab 2017). With fish trap research in its infancy in commercial salmon fisheries, improvements in performance are likely to be largest in the near future from addressing the most pressing and obvious flaws. As testing progresses throughout the years, incremental engineering improvements will likely exhibit diminishing returns to site-specific catch. Regardless, with only two years of fish trap engineering research in the Columbia River, it is evident that efficiency will only continue to increase as lessons are learned and new ideas incorporated into the design and placement of fish traps.

The regression analysis of CPUE from this study lends statistical evidence to inform future years of trap operation in fluvial settings. During the 2017 study, four covariates proved significant in determining CPUE for Chinook salmon, coho salmon, and steelhead trout: adult ladder fish counts at Bonneville Dam, mean tide height during each soak period, tide stage at the completion of the set, and time of day. As expected, time of the season is important for fishing, as explained by the proxy variable Bonneville adult ladder fish counts; the more fish migrating through the river during the fishing season, the more fish are likely to be captured at the trap site. The regression analysis further indicates that catch increases during periods of greater tide-height. This suggests that a trap located at a greater depth could prove more successful in capturing salmon. It also appears that CPUE is impacted by tide stage and time of day, with catch generally decreasing during the flood tide and at night. Nevertheless, the majority of the variation in each stock-specific model could not be explained by the selected covariates, indicating that CPUE at the study site is complicated and results primarily from factors that remain unknown.

While effective in deterring entry of marine mammals to the heart compartment of the trap, results of the CPUE regression analysis also demonstrate that the marine mammal gate, as designed in 2017, reduced catch of Chinook salmon and steelhead trout (Table 3-4; Table 3-6). This result was hypothesized prior to study, as the narrow bars of the gate make entry to the heart compartment more difficult to fish entry. Surprisingly, catch of coho salmon was statistically unaffected from closure of the deterrent device. This is perhaps due to the relatively small size of coho salmon, making closure of the gate to this species less of a perceived barrier. Despite reducing Chinook salmon and steelhead trout CPUE, inclusion of the marine mammal gate proved instrumental in reducing entry of mammals into the heart compartment of the trap relative to 2016 and minimizing potential of fish predation and damage to the gear. In future years, a better system should be developed to quantify encounters with marine mammals to determine if and how animal behavior is affected by operation of the trap. Results should be analyzed within season, between seasons, and between years of operation. While this endeavor may prove challenging given difficulties in sighting marine mammals from above the water-column and inherent detection differences between field observers, there is a need to assess whether marine mammals are being attracted to the gear and the impacts they may have on migrating salmonids near the lead of the trap.

4.3 Recommended Trap Design Modifications and Site Selection Considerations

Future users of traps should consider a number of modifications to further improve efficiency of operations, survival of fishes, and deterrence of marine mammals at a trap site. The following modifications and site selection considerations are recommended based upon experiences during the 2017 study:

1) Inclusion/modification of marine mammal deterrent devices – The marine mammal gate, as engineered in 2017, proved an effective mammal deterrent to the heart compartment, but also reduced CPUE of trap operations. Gaps between deterrent bars could be widened to better enable fish passage. Users should also install a winch system to improve the ease of gate closure. Furthermore, greater weights and heavier lead lines should be applied in between heart compartment pilings to keep the mesh flush with the river bottom (preventing mammal entry) and alternative deterrent methods should be considered and incorporated into trap operations (e.g. visual repellents, noise makers, physical contact, etc.) (NOAA 2015b).
2) Heart compartment mesh alteration – Reduction of the spiller mesh size to 2-1/2'' in 2017 proved effective in reducing wedging of jacks during confinement and lift. The majority of jack mortalities in 2017 were documented in the downstream panel of the heart compartment, which was constructed of 3-1/8'' stretch mesh. Users should consider reducing the mesh size of the heart compartment to 2-1/2'' to further reduce wedging of jacks during trap operation.
3) Addition of a 2^{nd} spiller – During the 2017 study, observers noted the tendency of salmon to congregate in large numbers at the downstream side of the heart compartment during the flood tide. Due to the tendency of salmon to face the current, users should consider installing a 2^{nd} tunnel and spiller at the downstream end of the heart to increase capture of fishes during the flood tide.

4) Addition of a 2^{nd} entrance to the heart – Salmon were observed on multiple occasions in 2017 migrating along the upstream side of the lead during flood tide. Given the location of the trap within the Columbia Estuary, capture efficiency could likely be increased through installation of a 2^{nd} entrance to the heart compartment. Inclusion of this 2^{nd} entrance on the upstream side of the lead would better enable the trap to fish effectively on both tides.
5) Lead extension – The experimental trap tested in 2016-2017 had a lead of approximately 90 m. Historically, trap leads were longer, which would naturally result in increased capture efficiency at a given trap site.
6) Addition of a 2^{nd} heart compartment – Trap engineering evolved substantially from its beginning in the late 19^{th} Century. By the mid-20^{th} Century, most traps included a 2^{nd} heart compartment to increase buildup of captured fishes and reduce the probability of escape. Inclusion of a 2^{nd} heart would likely improve capture efficiency of future operations.
7) Trap placement – From historical accounts, the most effective fish traps were often at depths greater than 6 m. The multiple regression analysis for this study supports these historical anecdotes, as CPUE was positively associated with tide height (greater water depth). Users should consider placement of traps in deeper waters to increase efficiency and capture of benthic oriented species (e.g. Chinook salmon).

4.4 Fisheries Management Applications

This comparison of survival rates, bycatch encounters, and CPUE between alternative gear-types highlights the management applicability of each tool assessed within the lower Columbia River. To date, there is no panacea for resolving the bycatch problem hatcheries inherently enhance in the Pacific Northwest (beyond terminating production hatchery programs altogether). However, each alternative gear-type assessed has potential to address stock-specific management concerns depending on the fishery, designated allocations, and the annual status of returning salmonid runs.

When all salmonid populations are healthy, escapement goals are met, and there are no stocks at risk of failure, conventional gillnets may be utilized sustainably (Gayeski et al. 2018).

This is best exemplified by the Bristol Bay, AK sockeye salmon fishery where salmonid stocks have not faced serious levels of depletion and managers have successfully maintained healthy fisheries for over a century with fishermen utilizing conventional, non-selective harvest tools (Hilborn 2006). Once harvest pressure or other anthropogenic stressors are severe enough to weaken the health of regional salmonid populations, stock-selective harvest practices become increasingly necessary to prevent extirpation. Loss of salmonid life-history and genetic diversity weakens the portfolio effect which maintains the viability and consistency of commercial salmon fisheries (Schindler et al. 2010).

In mixed-stock fisheries where overfished and depleted salmon runs happen to return in relatively robust numbers, application of alternative gears—including tangle nets, purse seines, beach seines, or fish traps—may be appropriate to sustain escapement of wild salmonids of concern. Continuation of the fishery through application of stock-selective harvest tools may prove a more advantageous strategy than fishery closure in the presence of large-scale production hatchery programs (which pose ecological and genetic threats to wild stocks if management cannot remove escaped hatchery-origin fishes). However, during low-return events from depressed or ESA-listed salmonid stocks in hatchery supplemented systems, use of purse seines and fish traps in terminal river or stream locations appear most desirable from a conservation and management perspective. These gears have demonstrated the highest rates of bycatch survival (Table 4-1). Purse seines may be most advantageous for avoidance of shore-oriented species (WDFW 2016). This species avoidance strategy is often sought by Pacific Northwest fisheries management agencies to protect hatchery-origin steelhead runs for recreational interests, which inherently reduces the effectiveness of hatchery fish removal. However, if both protection of bycatch and effective removal of all hatchery-origin salmonids (including hatchery-origin steelhead) is desired to enable commercial fisheries while striving for ESA salmon and steelhead recovery, fish traps appear to be one of the most effective means of doing so. If managers were to consider the ratio of wild-origin salmonids to hatchery-origin salmonids by gear-type to prioritize removal of hatchery-origin fishes, the performance record for beach seines, purse seines, and fish traps is fairly uniform, with purse seines exhibiting a wild to hatchery-origin ratio of approximately 3:2, beach seines at 2:3, and the fish trap at a ratio of 1:3 (WDFW 2016). Although these gears were evaluated during different years with different run sizes and stock-composition, this comparison draws attention to the fact that allocations set by management impact the criteria through which bycatch are defined and the effectiveness of salmonid recovery efforts. Striving for removal of all hatchery-origin salmonids and maximization of wild salmonid survival and fitness, use of the fish traps in terminal river or stream locations appears to be one of the best possible harvest and hatchery management strategies.

4.5 Transition to Stock-selective Commercial Harvest Tools and Benefits

If fishers and resource managers choose to increase efforts to recover threatened wild salmonids and initiate transition to alternative gear-types, fishing capacity reduction and buyout programs will likely be necessary to mitigate existing stakeholders, minimize adverse economic impacts, and ensure fair and equitable allocation of fisheries resources (16 U.S.C. 1851; MSA § 301). Section 312(b) of the MSA outlines fishing capacity reductions, which commonly involve direct purchase of fishing vessels, gear, and/or permits. Carefully crafted buyout or trade programs could work to provide immediate relief to fishermen exiting the fishery or transitioning to alternative gear-types. Financial assistance in transition would be necessary to retool the fishing fleet, reduce overcapacity, increase profitability, and build a firm foundation to sustainable stock-selective fisheries focused around use of alternative gear-types in terminal river or stream locations (Lichatowich et al. 2017; Gayeski et al. 2018). In the absence of such a plan, socio-economic costs would be considerable to the gillnetting community and transition to alternative gears would likely be mired in unnecessary social conflict.

With execution of a well-managed buyout program, partial retooling of the commercial gillnetting fleet to a combination of tangle nets, seines, and fish traps could provide benefit to the Pacific Northwest salmon fishing industry. Presently, commercial gillnetting opportunity is constrained from the onset of the season due in-part to high bycatch mortality rates, which generally exceed 50% (IFSP 2014). Considering such large impacts to ESA-listed stocks, allocation negotiations frequently result in limited openers for the commercial fleet. For example, in the fall of 2017, the lower Columbia non-Indian gillnetting fleet was authorized to fish on only seven occasions as a precautionary measure to protect historic low returns of ESA-listed Snake River steelhead (ODFW 2017). Utilizing stock-selective harvest tools with low bycatch impacts to wild fishes, commercial fishermen would likely see greater allocations of the resource, lengthening the season and increasing profitability. Furthermore, commercial fishing fleets would be less prone to in-season closure from exceeding ESA-take limits. This threat has increasingly become a nuisance to fishers of the lower Columbia which braced for closure in 2017 from the inevitable encounter of just 22 wild-origin B-run steelhead amongst both sport and commercial fisheries (ODFW 2017; Thomas 2017).

While enabling fishermen to fish for longer and more consistently, use of viable stock-selective harvest tools with substantially reduced bycatch impacts could enable more Pacific Salmon fisheries to become certified sustainable in the marketplace, fetching a greater price-point-per-pound (Gayeski et al. 2018). Sustainable market certifiers (including Marine Stewardship Council and Monterrey Bay Aquarium Seafood Watch) brand seafood products in the marketplace that meet specific sustainability criteria. This branding can result in product differentiation to consumers and increased prices received by fishers and processors (Cooper 2004; Kaiser and Edwards-Jones 2006). Concurrently, value-added practices (including bleeding and icing fish on site, and direct marketing of a high quality live-captured product to restaurants and other buyers) could help retooled fisheries increase profitability (Sea Grant 2018). Fishermen that are presently faced with declining salmon returns, greater limitations from ESA-management measures, and diminished prices from increased global competition and introduction of alternate goods (e.g. farm raised salmon) could improve their economic prospects by transitioning to alternative gears and utilizing value-added practices in certified sustainable fisheries. This could result in increased fishing opportunity and prices received for harvested products (Gayeski et al. 2018).

For threatened and endangered wild salmonids of the region, enactment of hatchery reforms and implementation of terminally located stock-selective fisheries could prove essential to survival and recovery (Lichatowich et al. 2017). Since the ESA-listing of salmonids in the Columbia Basin nearly 28 years ago, the most recent status reviews indicate that the same factors that originally caused the decline of wild salmonid populations—including hatcheries, harvest, habitat loss, and dams—continue to impede recovery (NMFS 2016). At present, wild salmonids have declined to 2.5% of historic abundance in the Columbia Basin (Licahtowich et al. 2017). The percentage of hatchery-origin spawners (pHOS) continues to exceed the recommended threshold of 5%, with many spawning populations experiencing pHOS greater than 50% (HSRG

2014; WDFW 2018). From these estimates, it is evident that the situation for wild salmonids is dire and the need for harvest and hatchery reform is urgent.

Although transition from the classic fisheries management paradigm of production hatcheries and conventional harvest will prove challenging, change is necessary to prevent further decline of wild salmonid populations, degradation of the portfolio effect, and curtailment of fishing opportunities (Schindler et al. 2010; Lichatowich et al. 2017; Gayeski et al. 2018). Solutions are at hand to help remedy harvest and hatchery problems in the region. Despite the discomfort that may be caused to stakeholders in the short-term from adoption of an alternative management strategy, the long-term benefits from a well-orchestrated management shift toward selective gears could improve the long-term economic outcome for U.S fishermen and fisheries across the Pacific Northwest (Gayeski et al. 2018). Furthermore, there is potential to benefit wild salmonid escapement and fitness, threatened ecosystems (e.g. the Columbia River Basin, Salish Sea), endangered salmon predators (e.g. Orca Whales, Steller Sea Lions), and future generations who stand to benefit directly through commercial/tribal fisheries or indirectly through recreation and existence value (Hanson et al. 2010; Lichatowich 2013).

4.6 Conclusion

This study represents the first successful attempt to design, construct, and operate a commercial pound net trap for the harvest of salmon in Washington State waters in over 80 years. Furthermore, it is the first ever evaluation of salmonid post-release survival from a commercial-scale salmon trap. During the 2017 study, the modified experimental trap demonstrated an ability to capture commercially viable quantities of Chinook and coho salmon in the lower Columbia River. Carefully designed and operated with a conservation-minded approach, Chinook salmon survival from release to McNary Dam was estimated at 99.5% ($\widehat{SE}$= 0.078), representing a significant and dramatic improvement over the performance of conventional and alternative gears utilized in the Columbia River. Steelhead trout survival to McNary Dam was estimated at 94.4% ($\widehat{SE}$ = 0.046); a result that is statistically greater than the performance of conventional gears and equivalent to the best performing alternative gears. Given these results, consideration of fish traps as stock-selective harvest tools for hatchery-origin salmonids and other healthy stocks in Columbia River commercial fisheries is warranted. The gear could be further applied by fishermen and management in depressed fisheries across the region to reduce detrimental bycatch and hatchery impacts, or as a preventative measure to preserve the status of healthy salmon stocks and fisheries. Further research is needed to determine the economic feasibility of the gear for fishermen and industry. Ecological impacts of the gear must also be investigated during other seasons, in small-scale fluvial settings, open coastal waters, and open-estuarine waters. Nevertheless, the fish trap is a tool that can be successfully deployed in fluvial settings to address bycatch and hatchery management concerns while enabling continuation of commercial fishing.

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APPENDICES

Figure A-1. Researching historical trap blueprints to design the 2017 pound net trap. Photo by Jamie Glasgow (WFC 2018).

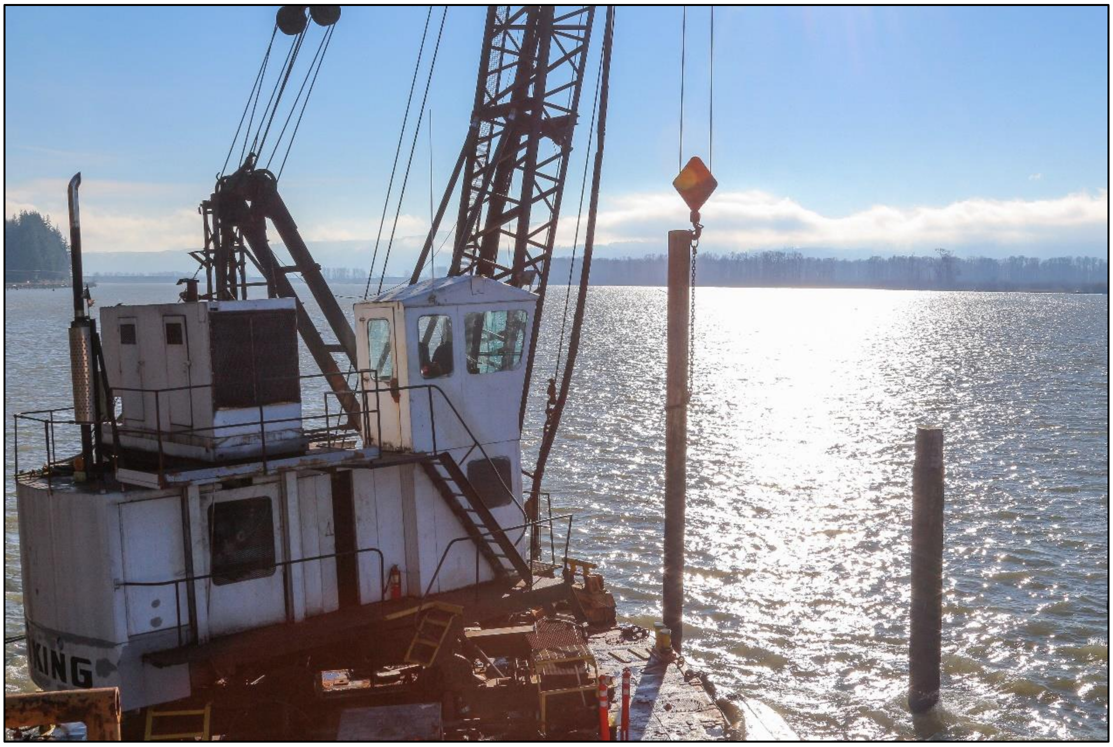

Figure A-2. Pile driving in December 2015. Photo by Kurt Beardslee (WFC 2018).

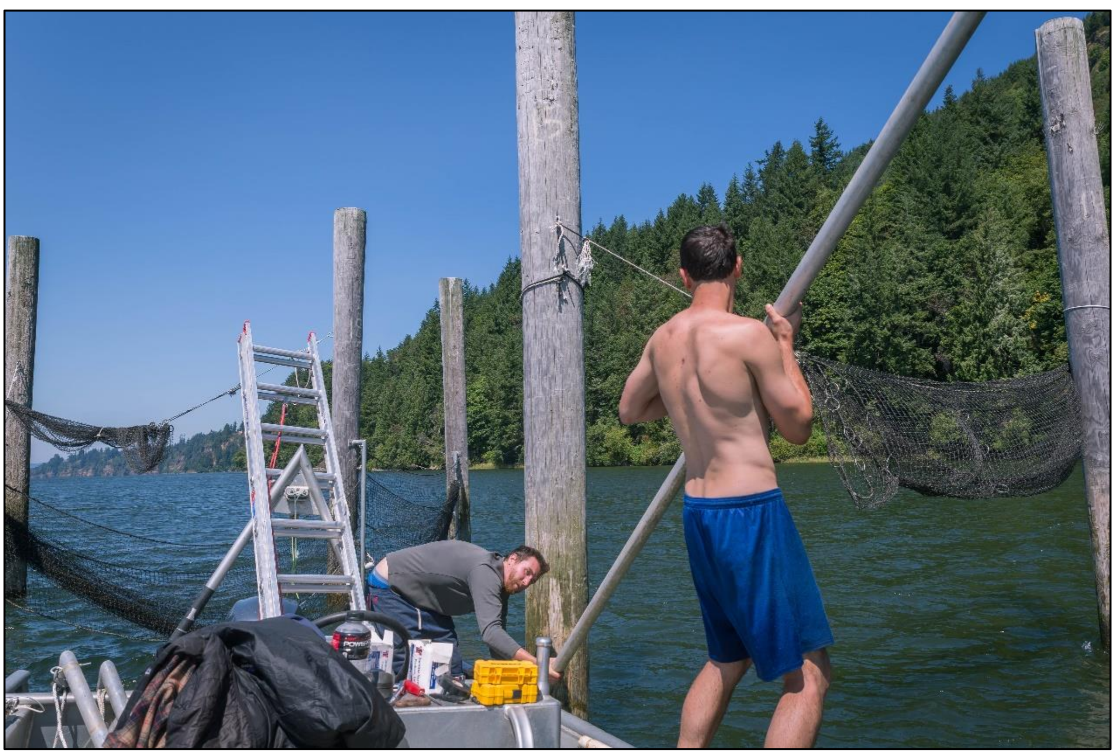

Figure A-3. Constructing the pound net trap in August 2017. Photo by Aaron Jorgenson (WFC 2018).

Figure A-4. Hanging the lead web on the pound net trap in August 2017. Photo by Aaron Jorgenson (WFC 2018).

Figure A-5. Constructing the spiller compartment in August 2017. Photo by Justin Eastman (WFC 2018).

Figure A-6. Modifying and orienting the spiller compartment in August 2017. Photo by Adrian Tuohy (WFC 2018).

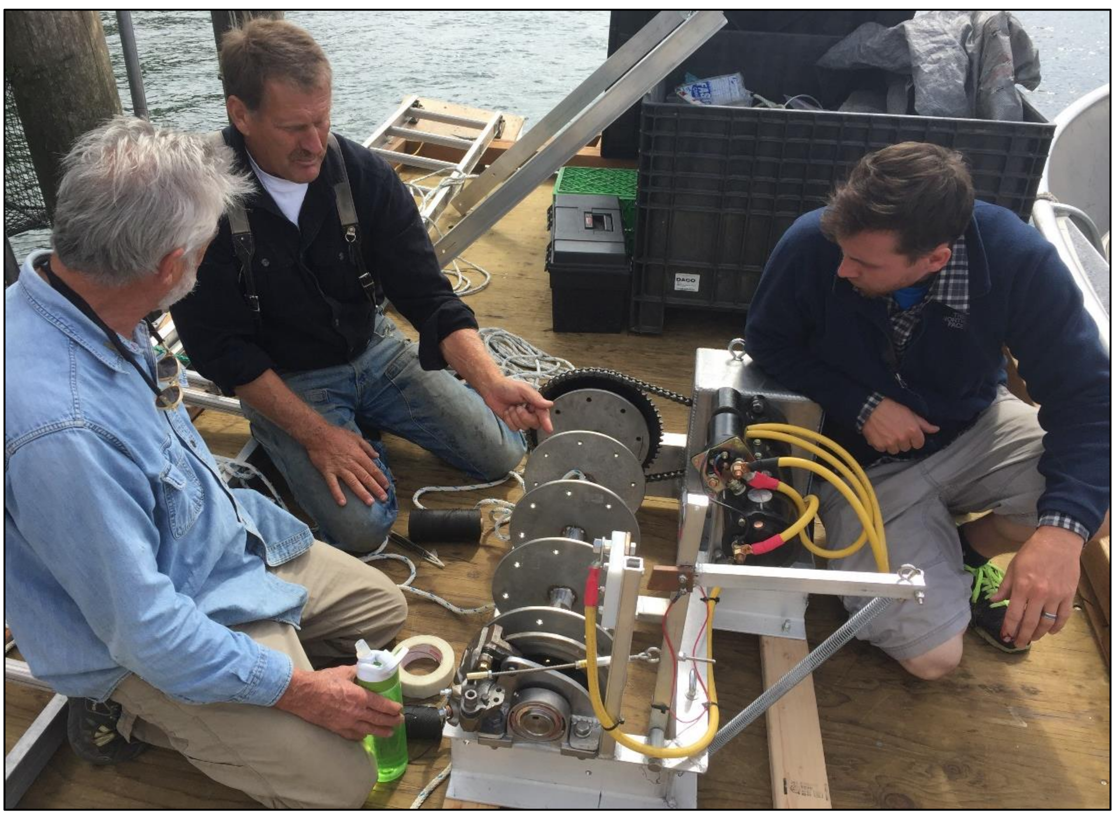

Figure A-7. Installing the solar powered electric winch in August 2017. Photo by Adrian Tuohy (WFC 2018).

Figure A-8. The perforated live-well compartment positioned adjacent to the spiller. This compartment enabled river flows to continuously oxygenate the water for recovering fish. The live well door can be viewed near the top of the photo. Photo by Adrian Tuohy (WFC 2018).

Figure A-9. Field camp for the 2017 study. Photo by Aaron Jorgenson (WFC 2018).

Figure A-10. Completed pound net trap viewed from above. Photo courtesy of Ann Stephenson (WDFW 2017).

Figure A-11. WFC field staff lift the spiller compartment with a solar powered electric winch. Photo by Jamie Glasgow (WFC 2018).

Figure A-12. WFC field staff prepare to spill a small haul of fish through the spiller door. Photo by Aaron Jorgenson (WFC 2018).

Figure A-13. WFC field staff PIT tag an adult Chinook salmon from the live-well. Photo by Jamie Glasgow (WFC 2018).

Figure A-14. A wild Chinook, tagged, fin-clipped, and ready for release upstream. Photo by Aaron Jorgenson (WFC 2018).

Figure A-15. A University intern records PIT tag data through P4 software on the live-well dock. Photo by Aaron Jorgenson (WFC 2018).

Figure A-16. Data entry in between sets from the data booth (positioned on the live-well dock). Photo by Aaron Jorgenson (WFC 2018).

Figure A-17. Lead fisherman Blair Peterson mending mesh in the heart compartment. Photo by Kurt Beardlsee (WFC 2018).

Table A-1. All species and stocks captured throughout the 2017 study period.

Species - Common and Scientific name	Life Stage	Origin	Number Encountered	Disposition of Collection	Genetic Sampling (Y/N)	Comments
Coho Salmon *(Oncorhynchus kisutch)*	adult	hatchery	1817	Release at capture site, alive	N	
Coho Salmon *(Oncorhynchus kisutch)*	adult	hatchery	4	Release at capture site, dead	N	1 adult, 3 jacks
Coho Salmon *(Oncorhynchus kisutch)*	adult	wild	1653	Release at capture site, alive	N	
Coho Salmon *(Oncorhynchus kisutch)*	adult	wild	1	Release at capture site, dead	N	jack
Coho Salmon *(Oncorhynchus kisutch)*	adult	unknown	26	Release at capture site, alive	N	
Chinook Salmon *(Oncorhynchus tshawytscha)*	adult	hatchery	233	Release at capture site, alive	N	
Chinook Salmon *(Oncorhynchus tshawytscha)*	adult	hatchery	2	Release at capture site, dead	N	1 adult, 1 jack
Chinook Salmon *(Oncorhynchus tshawytscha)*	adult	hatchery	1029	Tag and release	Y	2mm caudal fin-clip
Chinook Salmon *(Oncorhynchus tshawytscha)*	adult	wild	335	Release at capture site, alive	N	
Chinook Salmon *(Oncorhynchus tshawytscha)*	adult	wild	1040	Tag and release	Y	2mm caudal fin-clip
Chinook Salmon *(Oncorhynchus tshawytscha)*	adult	unknown	31	Release at capture site, alive	N	
Steelhead Trout *(Oncorhynchus mykiss)*	adult	hatchery	88	Release at capture site, alive	N	
Steelhead Trout *(Oncorhynchus mykiss)*	adult	hatchery	647	Tag and release	Y	2mm caudal fin-clip
Steelhead Trout *(Oncorhynchus mykiss)*	adult	wild	28	release at capture site, alive	N	
Steelhead Trout *(Oncorhynchus mykiss)*	adult	wild	145	Tag and release	Y	2mm caudal fin-clip
Steelhead Trout *(Oncorhynchus mykiss)*	adult	unknown	13	Release at capture site, alive	N	

Resident/Residualized Rainbow Trout *(Oncorhynchus mykiss)*	juvenile	hatchery	19	Release at capture site, alive	N	sub-adult
Resident/Residualized Rainbow Trout *(Oncorhynchus mykiss)*	juvenile	hatchery	2	Release at capture site, dead	N	sub-adult
Resident/Residualized Rainbow Trout *(Oncorhynchus mykiss)*	juvenile	wild	6	Release at capture site, alive	N	sub-adult
Resident/Residualized Rainbow Trout *(Oncorhynchus mykiss)*	juvenile	unknown	2	Release at capture site, alive	N	sub-adult
Unknown Salmonid *(Oncorhynchus sp.)*	unknown	unknown	8	Release at capture site, alive	N	
American Shad *(Alosa sapidissima)*	adult	wild	3	Release at capture site, alive	N	
Largescale Sucker (*Catostomus macrocheilus)*	adult	wild	1	Release at capture site, alive	N	
Peamouth *(Mylocheilus caurinus)*	adult	wild	1	Release at capture site, alive	N	
Largemouth Bass *(Micropterus salmoides)*	adult	wild	1	Release at capture site, alive	N	