Willapa Bay Comprehensive Review of Policy C-3622: Process and Schedule			
Type	**Purpose**	**Date**	**Status**
Public workshop	Public feedback on policy	January 23, 2018	Completed
WBSAG	Proposed process, review public feedback	September 14, 2018	Completed
WBSAG - recreational	Data workshop	October 24, 2018	Completed
WBSAG - commercial	Data workshop	October 25, 2018	Completed
FWC	Proposed process, commissioner feedback	November 2, 2018	Completed
WBSAG	Review of relevant data	Nov/Dec 2018	Completed
FWC - Fish Committee	Briefing on possbile review report structure	December 13, 2018	Completed
FWC	Policy guidance on comprehensive review content, and process and schedule for completion	April 1, 2019	Completed
FWC - Fish Committee	Review draft table of contents for comprehensive review; further review of proposed process and schedule	June 13, 2019	Completed
WBSAG	Feedback on comprehensive review structure and content	August 14, 2019	Proposed
WBSAG	Feedback on comprehensive review structure and content	Sept. TBD, 2019	Proposed
WBSAG	Review of draft final comprehensive review document and consideration of a range of alternatives for policy adjustments	October TBD, 2019	Proposed
FWC - Fish Committee	Briefing on draft final comprehensive review document and a preliminary range of alternatives for policy adjustments	Oct. 17, 2019	Proposed
FWC	Approval of comprehensive review final report and a range of alternatives for policy adjustments to be analyzed by staff	Oct. 18-19, 2019	Proposed
WBSAG	Analysis of range of alternatives for policy adjustments	Nov. 3, 2019 (tentative)	Proposed
WBSAG	Consider recommendations for policy adjustments	Jan. 6, 2020 (tentative)	Proposed
FWC - Fish Committee	Briefing on analysis on the range of alternatives for policy adjustments and any recommendations	Jan. 9, 2020 (tentative)	Proposed
FWC	Consider analysis on the range of alternatives for policy adjustments and select a preliminary preferred alternative for public review	Jan. 10-11, 2020. (tentative)	Proposed
WBSAG	Consider preliminary preferred alternative out for public review	Jan. 24, 2020 (tentative)	Proposed
FWC - Fish Committee	Briefing on further analysis of possible policy adjustments and advisory body/public input; consider recommendation to full Commission	Feb. 13, 2020 (tentative)	Proposed
FWC	Final decision on policy revisions, if any	Feb. 14-15, 2020 (tentative)	Proposed

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Abundance of Adult Fall Chinook Salmon within Willapa Bay as a Function of Policy C-3622

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SUMMARY: We addressed the hypothesis that a harvest strategy mandated by the Washington Fish and Wildlife Commission (FWC) Willapa Bay Salmon Management policy C-3622, which reduced fishing related mortality of fall Chinook salmon within Willapa Bay resulted in detectable increases in estimated redd-based escapement. We used a before-after-impact-control quasi-experimental (BACI) design to control for potential problems with collinearity between typically calculated fisheries parameters such as total runsize, harvest, and escapement. The results of the information theoretic based analysis indicated Chinook escapement estimates to the watersheds of Willapa Bay were not statistically significantly different from pre-policy escapements when runsize was controlled for via the BACI design despite a numerical increase in escapement (4.96%). We outline several commonly accepted mechanisms, which could explain these results including a lack of power to detect patterns (*i.e.* small sample size), bias associated with redd-survey methodology largely based around survey site selection strategy, and compensatory mortality driven by depredation, disease, and dispersal. The routine and unbiased assessment of whether management actions succeeded at achieving objectives provides tools for natural resource managers and decision makers. These data are essential for the development of information based management plans and adaptive management actions.

1. BACKGROUND

Salmon population abundances have declined over the last century across North America where inland and southern populations have been disproportionately influenced (Nehlsen et al. 1991, Thurow et al. 2000). Even though it is challenging to reconstruct historic salmon runs especially for time periods prior to European settlement; research has shown many salmon populations exist in a state of low abundances relative to historic populations. Despite the challenge, several studies have undertaken the effort to calculate an estimate of pre-settlement runsizes using both molecular (Drake et al. 2002, Drake and Naiman 2007) and historic-records (Chapman et al. 1986, Gresh et al. 2000, Meengs and Lackey 2005) based calculations to estimate runsizes. While these studies remain controversial and are based upon unverifiable and incomplete historical data, the results are fairly consistent indicating current pacific salmon returns in the lower United States are 3.0-47.0% of pre-settlement returns (Table 1). Furthermore, salmon have experienced dramatic reductions in their range relative to pre-settlement conditions (Yoshiyama et al. 2001); all of which has prompted managers to respond by reducing harvest pressure, conducting habitat restoration efforts, supplementing populations with hatchery reared fish, and sometimes ultimately listing under the US Endangered Species Act (Williams et al. 1999).

Table 1. Summary of literature review focused on estimating presettlement runsizes for southern east pacific salmon.

Historical Pop Size	Current Pop size	▲Pop Size	Carrying Cap	Method	Species	Citation
1,500,000 - 2,500,000	84,188 - 281,736	(-0.63)-(-0.97)	759,000-1,391,000	Cannery	Coho, Chinook	Meengs, C. C. and R. T. Lackey (2005)
160 - 226 million kg's	11.8 - 13.7 million kg's	(-0.96)-(-0.97)	NA	Cannery	Unidentified	Gresh, T., et al. (2000)
7,500,000 fish	2,500,000 fish	-0.66	NA	Fishing	Coho, Chinook, Sockeye, and Chum	Chapman, D. W. (1986)

Willapa Bay has experienced a long history of anthropogenic influence, which has likely interacted with resident salmon populations. Logging practices, shellfish farming, agriculture, salmon hatchery production, and salmon harvest have interacted with Willapa salmon populations for more than 100 years (Swan 1857, Suzumoto 1992). Yet, there is a paucity of research about the conditions and status of Willapa Bay salmon populations prior to large-scale anthropogenic influence, making it difficult to effectively assess the current status of Willapa Bay's salmon fishery relative to historical reference points. While Willapa's fishery status relative to historic reference points remains unclear, Willapa and its drainages merit significant attention to its conservation status, in part, because the long term economic impact to the coastal communities which are dependent upon fishing would suffer from declines in salmon populations. In response to range-wide and local salmon run declines (Figure 1) coupled with increasing public interest in conserving salmon; WDFW implemented FWC policy C-3622 focused on increasing wild-origin spawner escapement largely through reductions to catch in the commercial fishery.

The successful management of species subject to commercial and recreational exploitation depends upon understanding the interplay between abundance and harvest through time (Rowe and Hutchings 2003, Walters et al. 2019). Many models have been developed to address the calculation of abundance estimates, especially among salmonids whose complex life history interact with the need to partition harvest impacts to the appropriate stocks (Branch and Hilborn 2010, Cunningham et al. 2017, Cunningham et al. 2018). Many of these approaches attempt to reconstruct salmon runs by estimating components such as total catch via commercial and recreational fisheries as well as salmon escapement where escapement is defined as an animal that is geographically and temporally present within the fishery (*e.g.* for Willapa Bay, salmon are within the Willapa Bay terminal area) but does not experience mortality as a function of the fishery.

Accounting for spatial and temporal patterns of abundance among salmon represents a difficult challenge for managers and researchers alike. A common method of calculating an index of escapement is to estimate the number of animals which have escaped the fishery and other sources of mortality and successfully created visually identifiable nests (Gallagher et al. 2007, Groves et al. 2013), which we term here spawning effort. Once upon the spawning grounds, female salmon sequentially dig a series of nests in gravel substrate where the female deposits eggs externally fertilized by one or more males, covers the eggs with gravel (Neilson and Banford 1983). Because redds are the end result of the breeding process for adult salmon which all experience post copulation mortality, with the exception of steelhead, redd abundance has often been used as an index of animal abundance (Gallagher et al. 2007).

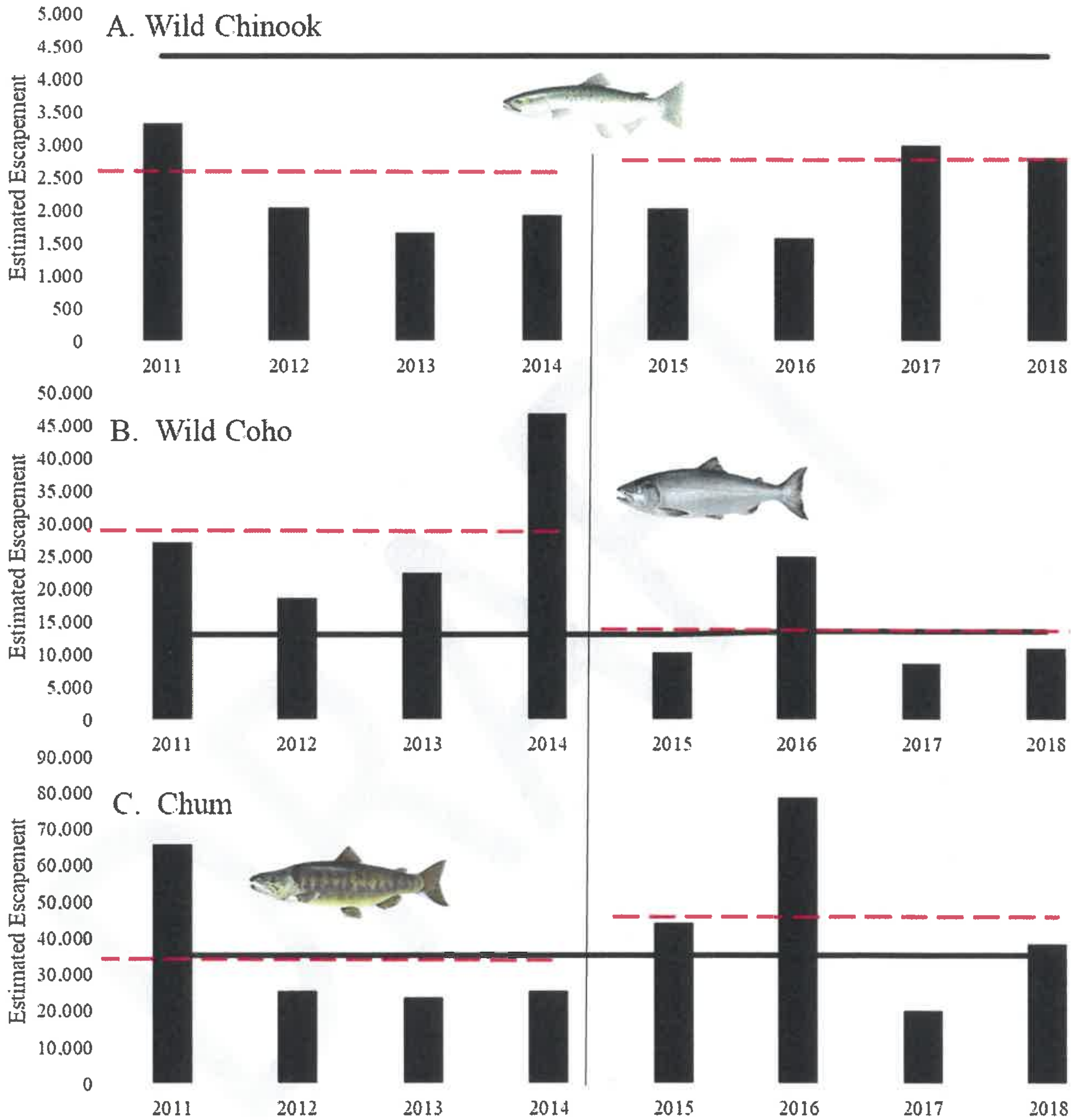

Figure 1. Estimated escapement for wild-origin Chinook and coho. Chum salmon abundance were estimated in the aggregate (wild-origin and hatchery-origin) becasuse hatchery chum lack a visibally identifable external mark (*e.g.* adipose fin clip). The red dotted line represents the mean escapement across the four years prior and post policy adoption. The black line represents the established escapement goal for Willapa Bay.

WDFW collects and maintains large-scale long-term repeatable biological datasets which are intended to detect patterns of relative abundance and calculate estimates of absolute abundance. To calculate estimates of absolute abundance within the terminal area (for application here, both the inland marine waters of Willapa Bay and its freshwater tributaries) fisheries biologists often reconstruct the salmon run using data collected during the fishing season (*i.e.* recreational harvest, commercial harvest, and spawning effort). The most basic form of the run reconstruction paradigm is to calculate estimates of the assumed most important components of salmon runs, that is harvest (*H*) and escapement (*E*) (Equation 1), while ignoring extrinsic sources of mortality (*e.g.* depredation or disease) between terminal area fisheries and the spawning grounds (Starr and Hilborn 1988, Hilborn et al. 1999, Branch and Hilborn 2010, Cunningham et al. 2017).

$$R_g = E_{at} + C_{at} \qquad (1)$$

Where the subscripts *g*, *a*, and *t* indicate Runsize for each population *g* as well as escapement and catch for area *a* at time *t*. This formulation of an estimate of runsize has merit because it is simple, easy to estimate and the data are readily available to the agencies tasked with the management of salmon.

To understand whether reducing fishing pressure on Chinook salmon resulted in a shift in Chinook wild-origin escapement, we used a quasi-experimental before-after-control-impact (BACI) design. Under this design, measurements are taken at the impacted and control sites both before and after the impact occurs (Figure 2) (Underwood 1991, 1994, Schwarz 2015). Using the traditional calculation outlined in equation 1, salmon runsize estimates are a function of escapement and therefore violate the assumption of independence. An alternative method of controlling for runsize is to model multiple management areas relative to Willapa Bay which have experienced either no change in Chinook harvest strategy or a change in harvest strategy based upon recent policy adoptions. An intent of the policy was to reduce fishing impacts to wild-origin Chinook which was quantitatively realized (Figure 3). The expectation is that fish which escape fisheries translate into additional fish counted on the spawning grounds. The BACI design is preferred to the alternative simple before-after comparison because temporal changes may influence the measurements independent of the impact.

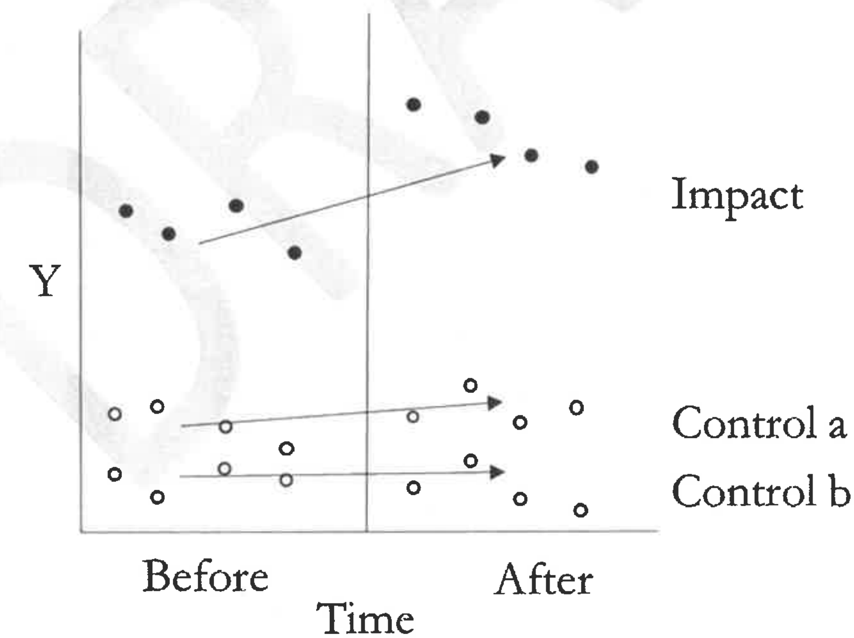

Figure 2. Conceptual model of the BACI design adapted from Schwerz et al. 2015. The key element is that the impact treatment increases more than the control groups effectively controlling for natural time-dependent variation. Alternative versions allow for no change in the impacted site but a shift downward at the control sites due to time dependent variation.

To address the hypothesis that salmon escapement especially wild-origin salmon would increase as a function of a reduction in harvest as mandated by policy C-3622, we assessed redd-based escapement data via a BACI design across five Washington coastal drainages. Conceptually, a reduction in harvest effort in the terminal area would decrease the number of fish impacted and thereby pass more fish to the spawning grounds to be counted by redd surveyors. On the other hand, despite our model where runsize is calculated simplistically as harvest plus escapement, populations may experience patterns of density dependent dispersal or compensatory mortality such that additional fish escaping the fishery would not translate to fish on the surveyed spawning grounds. Furthermore, the power to detect such changes in escapement depends upon the sensitivity of our experimental design and analytical methods; thus understanding whether we should expect to see a pattern in our data given the pattern exists is critical to the interpretation of the dataset and analysis. These data and analyses are intended to inform decision makers about whether and to what degree forgoing harvest opportunities influence wild-origin escapement, the primary metric of conservation.

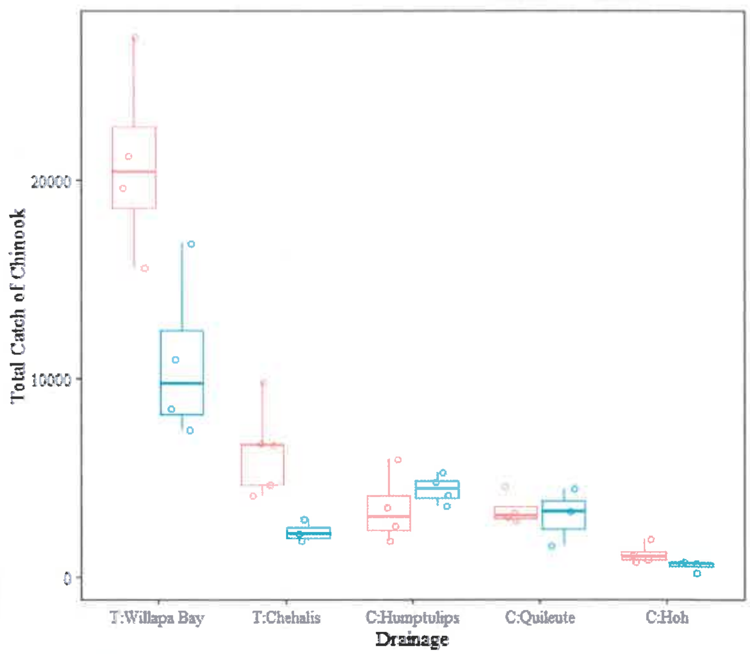

Figure 3. Total catch of Chinook salmon in both the commercial and recreational fisheries for selected major Washington Coastal drainages for four years preceding (red boxes: 2011-2014) and four years after (blue boxes: 2015-2018) policy implementation. Policies for both Willapa and Chehalis significantly impacted Chinook harvest while the control groups experienced little change across the policy and non-policy years.

2. EXPERIMENTAL DESIGN AND ANALYSIS

3.1 Experimental design

We choose to use a before-after-impact-control design to determine whether policy-determined reductions to harvest increased our estimates of spawning effort to avoid issues with autocorrelation between escapement estimates and runsize estimates while also controlling for the possible influence of time dependent effects (Eccles et al. 2003). Quasi-experimental designs such as BACI designs are useful when logistical considerations influence whether impact or control sites are assigned randomly. The analysis of these data relied upon the *a priori* selection of treatment sites as a function of policy development combined with control sites selected based only upon proximity to Willapa Bay and foreknowledge about whether harvest philosophy had changed within the eight year monitoring window

(2011-2018). The five study sites and their assignments were the only sites and assignments considered during the analysis. In other words, study sites and their assignments to either impact or control groups were selected before any exploratory analyses were conducted and were based upon the principal that the policies at Willapa Bay (C-3622) and Grays Harbor (C-3621) would have reduced harvest as intended. The Chehalis and Humptulips drainages are separated into two different groups; impact and control respectively, because despite both drainages being managed under the same policy, the Humptulips River never experienced the estimate thresholds which trigger a policy determined management action and thus experienced the same management paradigm through the eight year monitoring window.

Table 2. Summary of the candidate models explaining wild-origin Chinook escapement. The predicted interaction between the policy status and harvest reduction sites were not among the top performing models ranking 6th. The most parsimonious model is bolded and included the additive model with study site nested within drainage and total catch as covariates.

Candidate Models$_a$	**k**	**AICc**	**▲**	**w**	**Log Liklihood**	**Cum w**
D + H + C	*7*	***695.02***	***0.00***	***1.00***	***-338.70***	***0.95***
C * H + D	9	701.19	6.18	0.05	-338.49	1.00
D	6	709.16	14.14	0.00	-347.31	1.00
D + Y	7	711.97	16.95	0.00	-347.23	1.00
D + P + H	7	712.12	17.10	0.00	-347.31	1.00
P * H + D	8	715.22	20.20	0.00	-347.29	1.00
D * H + D	9	718.00	22.98	0.00	-347.00	1.00
D * Y	11	725.23	30.21	0.00	-346.90	1.00
C	3	753.44	58.42	0.00	-373.38	1.00
P + C	4	755.50	60.48	0.00	-373.16	1.00
H	3	760.53	65.51	0.00	-376.93	1.00
P * H	5	765.05	70.03	0.00	-376.64	1.00
P * H + Y	6	767.69	72.67	0.00	-376.57	1.00
P	3	771.41	76.39	0.00	-382.37	1.00
Y	3	771.71	76.69	0.00	-382.52	1.00
Intercept only	1	804.36	109.34	0.00	-401.13	1.00

Candidate Models$_a$: D = The coastal drainage, Y = Year of the escapement estimate, H = Impact vs control sites, C = Total Catch, P = Policy status based upon year (2011-2014 = Prior to policy, 2015-2018 = Policy years).

3.2 Analysis

We assessed whether our estimates of local and non-local total catch and estimated redd-based wild escapement (recreational + commercial catch) were best described by the predicted interaction between policy status (*P:* before or after the policy implementation) and harvest reduced treatment group assignment (*H*: impact sites = harvest reduced sites (Willapa Bay and Grays Harbor) or control sites = unchanged harvest (Humptulips, Chehalis, and Hoh)) in two separate analyses. Both total catch and estimated redd-based wild escapement was modeled via general linear models as a function of the coastal drainage (*D*), year of the escapement estimate (*Y*), Impact vs control site (*H*), and policy status based upon year (*P*: 2011-2014 = Prior to policy, 2015-2018 = Policy years). For the total catch analysis, a total of 12 *a priori* models were developed based on hypothesized relationships while we considered 16 *a priori* models to describe the redd-based wild escapement for Chinook salmon.

The most parsimonious model was selected via an information-based approach using Akaike's information criterion developed for small sample sizes (AICc) (Burnham and Anderson 2004, Burnham et al. 2011). Each model was assessed by calculating the AICc difference scores denoted by (▲). All models with a ▲ score < 2.0 were considered supported by the data. A post hoc power analysis (α = 0.05) was also conducted on the wild-origin Chinook model predicted by the BACI design (*P* * *H*) to determine whether the analysis was likely to find a pattern provided it existed (type II error). All data were analyzed in program R (v. 3.2.1 package lme4; 2019).

Table 3. Table depicting the results of the model selection process focused on determining whether total catch of Chinook salmon was determined by the predicted interaction between policy status and treatment status (*i.e.* control drainages or impact drainages). The results of the analysis shows the most parsimonious models both included the predicted interaction between policy status and impact status while also including the nested terms drainage and year.

Candidate Models[a]	k	AICc	▲	w	Log Liklihood	Cum w
P * H + D	***8***	***733.66***	***0.00***	***1.00***	***-356.43***	***0.71***
P * H + D + Y	***9***	***735.42***	***1.76***	***0.41***	***-355.61***	***1.00***
D + Y	7	747.89	14.24	0.00	-365.14	1.00
D + P	7	748.83	15.17	0.00	-365.61	1.00
D + P + H	7	748.83	15.17	0.00	-365.61	1.00
D	6	754.49	20.83	0.00	-369.93	1.00
P * H	5	779.20	45.54	0.00	-383.69	1.00
H	3	779.70	46.05	0.00	-386.51	1.00
P * H + Y	6	780.05	46.39	0.00	-382.71	1.00
Y	3	796.17	62.51	0.00	-394.74	1.00
P	3	796.40	62.75	0.00	-394.86	1.00
Intercept Only	1	818.23	84.57	0.00	-408.06	1.00

Candidate Models[a]: D = The coastal drainage, Y = Year of the escapement estimate, H = Impact vs control sites, P = Policy status based upon year (2011-2014 = Prior to policy, 2015-2018 = Policy years).

3. RESULTS AND DISCUSSION

The primary focus of this section of the document was to understand whether detectable changes in Chinook populations at Willapa Bay were observed as a function of a reduction in fishing pressure associated with the implementation of policy C-3622. Wild-origin Chinook salmon escapement represents a critical metric by which resource managers measure the success or failure of implementing harvest reductions.

The results of the analysis focusing on wild Chinook escapement showed that while escapement estimates increased numerically at Willapa Bay (4.97%; mean pre-policy wild Chinook escapement =

2248, mean post-policy wild Chinook escapement = 2,363), the interaction predicted by the BACI design (the statistical interaction between policy year status and impact-control status) was not supported (Figure 4). In other words, we did not find evidence of increased estimates of spawning effort (WDFW's estimate of escapement) as a function of policy implementation. The results of our information theoretic AICc model selection focusing on wild-origin chinook escapement showed that the most parsimonious model was the simple additive model including drainage (*D*), treatment status (*i.e.* whether or not the site experienced harvest reductions as a function of policy implementation; *H*) and the total number of animals captured in the terminal fishery (*C*). In contrast, the predicted interaction between policy status and impact-control status was among the top six models when drainage was included as a nested term, but performed significantly more poorly than the top performing additive model where the calculated evidence ratio between the top performing model and the predicted BACI model ($P * H + D$) suggested the top model was 24,380 times more likely (Table 2).

It is somewhat perplexing that despite a significant and detectable shift in harvest as a function of policy implementation (Table 3), estimated escapement did not respond as expected via a commensurate shift in the form of added redd counts on the spawning grounds. In other words the fishery reduced the mean number of Chinook salmon harvested by 48.9% (mean yearly difference 9,993 fish unharvested) during policy years, yet the estimated increase in wild-origin spawning effort (pre-policy wild-origin escape) plus estimated hatchery escapement added up to only 22.1% (2,202 additional fish) of the 9,993 fish estimated to have been left unharvested due to the harvest reduction, provided there were no significant shifts in runsize before or after policy implementation. There are several putative explanations for why these conservation measures did not translate to added estimated spawning effort and here we outline several including a lack of power to detect patterns (*i.e.* small sample size), bias associated with redd-survey methodology, and compensatory mortality driven by depredation, disease, and dispersal.

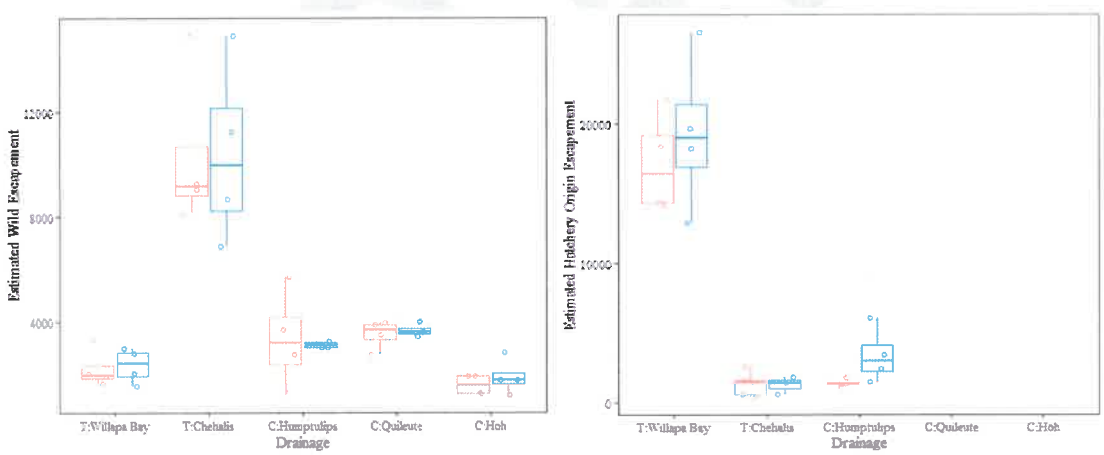

Figure 4. Boxplots of estimated wild- (left panel) and hatchery- (right panel) origin escapement before (red) and after (blue) policy implementation across five Washington State coastal river systems. Each river system is assigned as either a treatment group (T: Willapa Bay and Chehalis River) or as a control group (C: Humptulips, Quileute, and Hoh) based on whether or not the systems experienced significant shifts in harvest strategies as a result of conservation focused policy implementation.

The results of the power analysis indicated the probability of a type 2 error was low ($\beta = 0.441$), thus the probability of detecting a pattern when it existed was relatively high especially among ecological studies (*power* = 0.669). For example, Jennions and Møller (2003) in a meta-analysis of studies in behavioral ecology estimated the mean power to be 13-16% for a small effect and 40-47% for a medium effect. While the estimated power of our work was higher than typical studies, we did not meet the threshold of 0.8 typically recommended (Cohen 2013). Our power analysis indicated that there was a 44.1% chance that a pattern existed but our data and analysis failed to detect it. Thus, increasing the power of our analysis via additional data collection, increasing the scope of study sites, or increasing the scope of years prior to the treatment may increase the probability of detecting a pattern when it exists.

The validity of statistical analyses depend not only upon sample size, but also on controlling for potential bias via study design. Redd-based escapement estimates, especially those which utilize historical survey sites, represent a mixture of advantages and disadvantages. On the one hand, continuous long term surveys are exceptionally powerful when methods are repeatable and unbiased. On the other hand, science has developed radically over the last century since some of these surveys were first developed. It is often unclear how and why redd survey sites were chosen making it difficult to identify the scope of inference. For example, the scope of inference for sites which were chosen at random across the entire drainage differs from sites which were chosen based upon areas of high spawning density. The spatial and temporal distribution of surveys may have impacted the analysis of the escapement data making it difficult to detect the effects of harvest reductions. For example, some evidence based upon personal communication (Curt Holt, WDFW) indicated historical survey locations within the Willapa Basin were chosen because sites represented some combination of accessibility and perceived fish abundance. Patterns of fish distribution are often patchy because habitat quality relative for the species of interest can vary significantly within a drainage. Absolute abundance can interact with spatial and temporal distribution when fish are sensitive to both the density of animals present in the habitat as well as to the habitat quality such that high quality habitats become less attractive as they become more densely occupied by salmon. Density dependent shifts in spatial and temporal distribution has been documented in some salmon populations (Kokko and Lindstrom 1998, Connor and Pflug 2004, Atlas et al. 2015). Furthermore, Willapa employs a direct extrapolation from redd counts to absolute abundance via the use of two estimates, mean fish per redd and total available habitat. Because we extrapolate the redd data via estimated coefficients calculated at a different time and drainage, the expectation that estimates of catch and escapement via spawning effort would match such that differences in harvest would be detected at the same magnitude on the spawning grounds are likely invalid.

Compensatory adult depredation or natural mortality (*i.e.* adult salmon death due to higher abundances of salmon) could also explain the lack of correlation between the reduction in harvest pressure and the predicted increase in redd-based escapement estimates. Because WDFW's estimates are based only upon catch estimates and spawning effort; unmeasured compensatory mortality could explain the discrepancy between the expected escapement response and harvest reduction. Several studies have shown predators respond quickly to shifts in salmon abundance. For example, Nelson et al 2019 showed seal density best explained the productivity of juvenile Chinook salmon where increasing hatchery releases increased seal density rather than increasing estimates of runsize or estimates of spawning effort. In contrast to Nelson et al's research which requires a shift in predator abundance, optimal foraging theory also provides a mechanism for a compensatory increase in the probability of depredation while predator abundance remains static. For example, a functional predator response to increased salmon availability could increase depredation if predators exhibit sensitivity to nutrient density within their system (Charnov 1976a, b, Pyke et al. 1977). In other words, research has shown predators often respond to changes in prey availability by switching food sources such that the time and effort associated with finding, capturing and processing the prey into calories is balanced by the total nutrition provided by the prey item (Mittelbach 1981). For example, many low energy but easily captured and processed prey items may be more attractive than high-energy difficult-to-find, capture, and process prey items because the energy associated with locating, capturing, and processing may outweigh the benefit of the high calorie meal. An alternative mechanism to density dependent mortality based upon depredation is density dependent pre-spawn mortality or migration failure driven by energetic resources, agonistic competitive behavior, or disease. Little is known about the influence of disease, competitive behavior, or nutrition on Pacific salmon mortality or successful migration. Increases in the probability of transmittable disease or lack of energetic resources could explain why the reduction in fishing effort did not increase our estimates of redd-based escapement estimates.

Anadromous Pacific salmon spend the majority of their lives at sea growing prior to returning to freshwater streams to spawn and ultimately perish frequently leaving their entire biomass within or on the banks of freshwater streams. Nutrients derived from decaying salmon or otherwise consumed salmon are incorporated into the freshwater and riparian biota at various trophic levels (Kline et al. 1993, Wipfli et al. 1998). Pacific salmon migration patterns, therefore, provide a mechanism by which marine derived nutrients supplement the nutrient poor freshwater and terrestrial ecosystems characteristic of the Pacific Northwest. For example, Helfield and Naiman (2001) showed that shrubs and trees in the riparian zone acquired 22-24% of their foliar nitrogen from spawning salmon. It seems clear based upon the data that a reduction in fishing pressure resulted in fewer captured fish within the terminal area of Willapa Bay. If dramatic shifts in runsize does not explain the difference in catch, which our BACI design seems to rule out, the harvest reduction in the fishery at Willapa Bay then is escaping more fish when escapement is defined as runsize minus fishing related mortality. If compensatory mortality accounts for the disconnect between the reduction in harvest and our redd-based escapement estimates, some proportion of the biomass of unharvested salmon was incorporated into the ecosystem likely providing an ecosystem benefit. For example, Wagner and Reynolds (2019) showed that increased salmon biomass was correlated with increasing bird abundance and diversity in the riparian areas. Furthermore, increasing salmon biomass within the freshwater systems have been shown to provide a benefit directly to the next generation of salmon via indirect nutrient supplementation (Wipfli et al. 2003, Heintz et al. 2004). While it is tempting to conclude that because our estimates of spawning effort do not reflect the reduction in harvest, the harvest strategy did not effectively benefit salmon populations. Managers must also be attentive the potential benefits to the ecosystem.

4. CONCLUSIONS:

It is clear based on the harvest data in the terminal areas adult Chinook salmon harvest declined at both treatment sites as a function of policy implementation relative to the control sites. This is not surprising given reducing the fishing pressure as a conservation measure was among the goals of both the Grays Harbor and Willapa Bay policies. The expectation that estimates of escapement via measuring spawning effort would also increase commensurately with the degree of fishing reduction was not detected. We explored several methodological and biological mechanisms for why our analysis failed to detect a shift in spawning. Given the data, we cannot distinguish whether spawning effort increased and our analysis failed to detect the pattern or if compensatory mortality accounted for the lack of expected increased estimated spawning effort or some admixture of the two. Further analyses and targeted data collection should be developed to assess whether reductions in harvest increases escapement.

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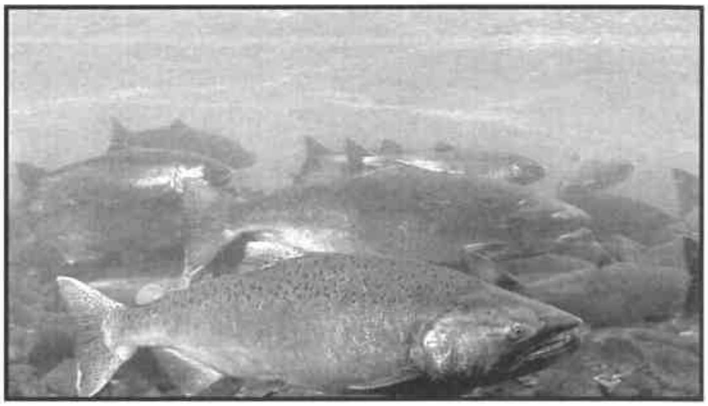
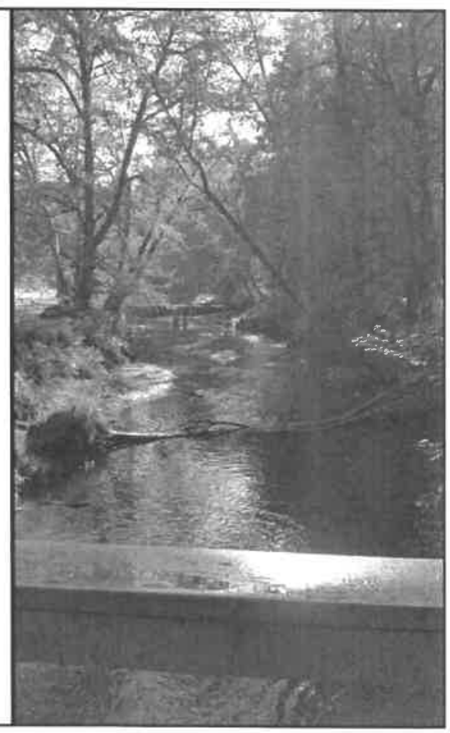
Abundance of Salmon withinWillapa Bay as a Function of Policy C-3622



Damon Peterson, Chad J. Herring, Barbara McClellan, and Lyle F. Jennings
Washington Department of Fish and Wildlife

Pacific salmon are in trouble:

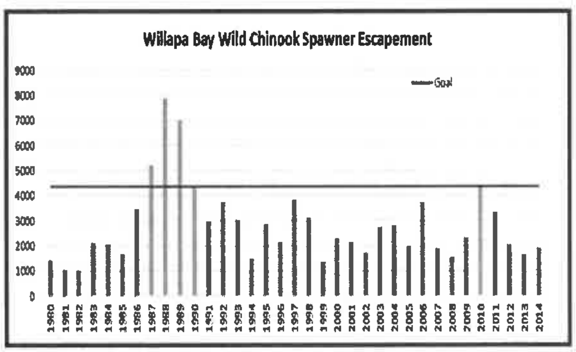
- Reduction in distribution
  - Barriers to upstream movement
    - Complete
      - Dams (hydropower)
      - Channel alteration
      - Shifts in flow
    - Incomplete
      - Abiotic conditions
        - Temperature
      - Predators
      - Flow patterns
- Habitat Degradation
  - Freshwater spawning habitat
  - Freshwater and Juvenile rearing
  - Estuary
  - Ocean
- Exploitation
  - Harvest practices
  - Hatchery practices

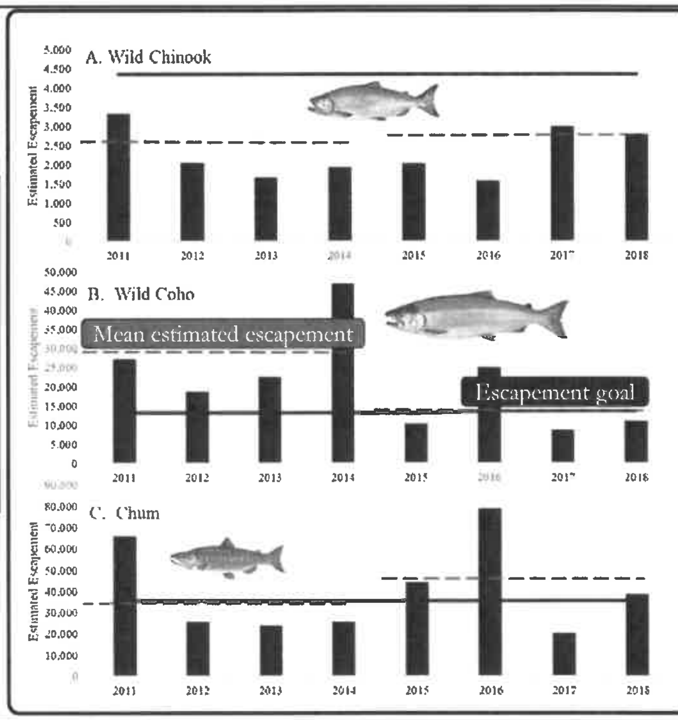


Pacific Salmon populations as a function of presettlement runsizes

Historical Pop Size	Current Pop size	▲Pop Size	Carrying Cap	Method	Species	Citation
1,500,000 - 2,500,000	84,188 - 281,736	(-0.63)-(-0.97)	759,000-1,391,000	Cannery	Coho, Chinook	Meengs, C. C. and R. T. Lackey (2005)
160 - 226 million kg's	11.8 - 13.7 million kg's	(-0.96)-(-0.97)	NA	Cannery	Unidentified	Gresh, T., et al. (2000)
7,500,000 fish	2,500,000 fish	-0.66	NA	Fishing	Coho, Chinook, Sockeye, and Chum	Chapman, D. W. (1986)

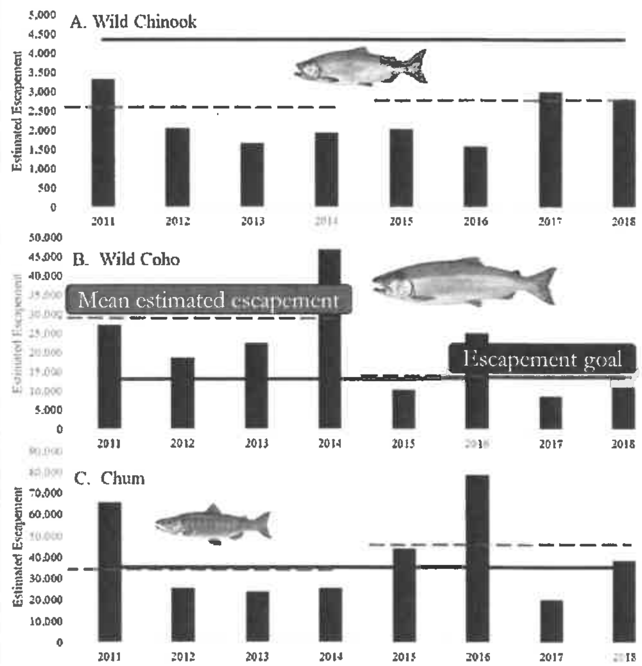
Willapa Bay Salmon Escapement





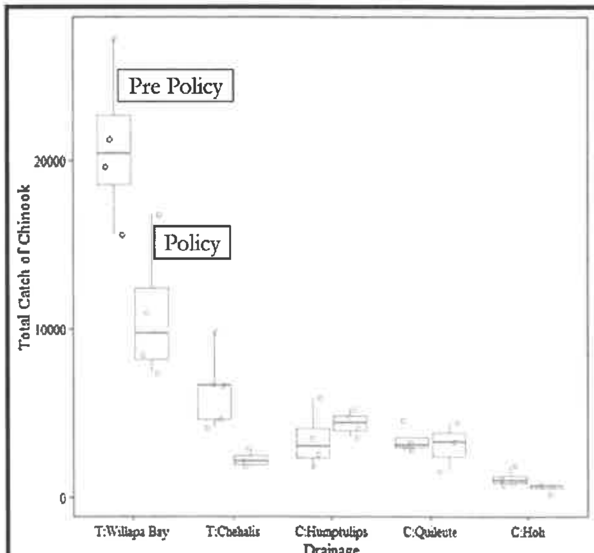
Policy C-3622

- Why was it needed
  - Enhance conservation
  - Frustration with harvest allocation
  - Lack of trust
- Policy goals
  - Achieve restoration of wild salmon
  - Avoid ESA designation
  - Maintain or enhance economic well-being
  - Partition fishing opportunities
  - Enhance transparency, information sharing and technical rigor
  - Restore and maintain public trust and support



What did the Policy C-3622 do to harvest?

- Harvest control rules
  - Chinook
    - Limited wild Chinook impacts at 20% to Willapa and Naselle rivers



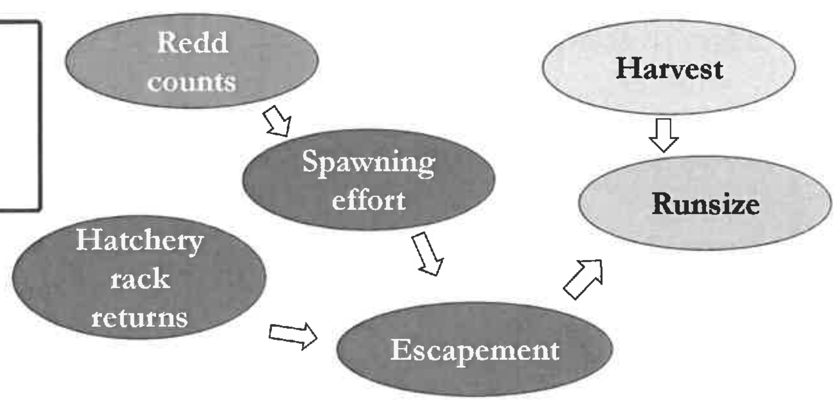
- Problem:
  - Salmon populations in Willapa Bay are thought to be underperforming

Question:
  - Does reducing fishing pressure on Chinook salmon in Willapa Bay result in a measureable conservation benefit?

- Prediction:
  - Reductions in fishing pressure will increase wild-origin redd-based salmon escapement
  - Reductions in fishing pressure will increase hatchery-origin salmon escapement
  - While controlling for changes in runsize

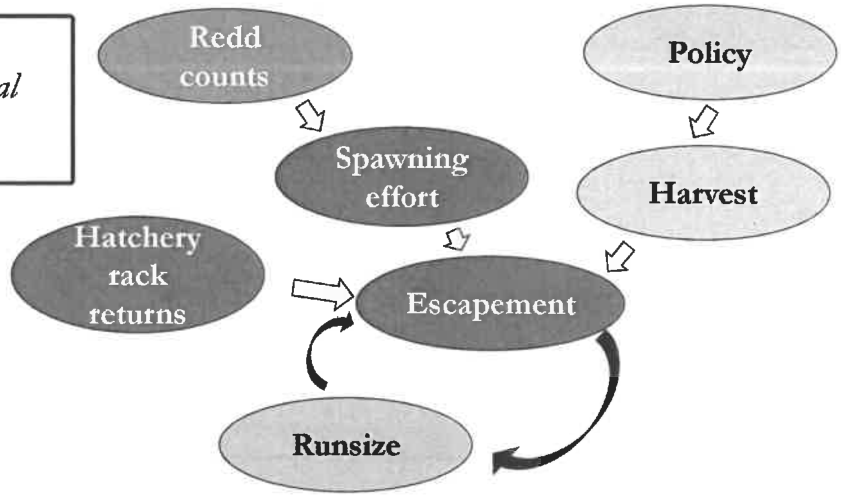
Our calculation of runsize

- Runsize = Harvest + Escapement (*Hilborn et al 2009*)



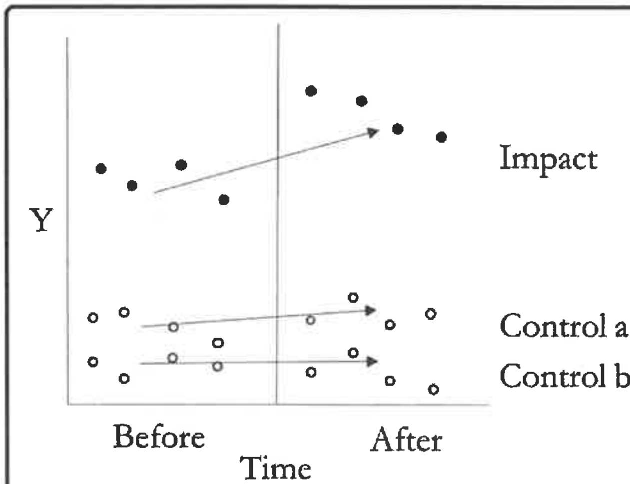
Our methods cannot account for changes in escapement while also controlling for changes in runsize

- Runsize = Harvest + Escapement (*Hilborn et al xx*)



Quasi-experimental Design: Before After Control Impact

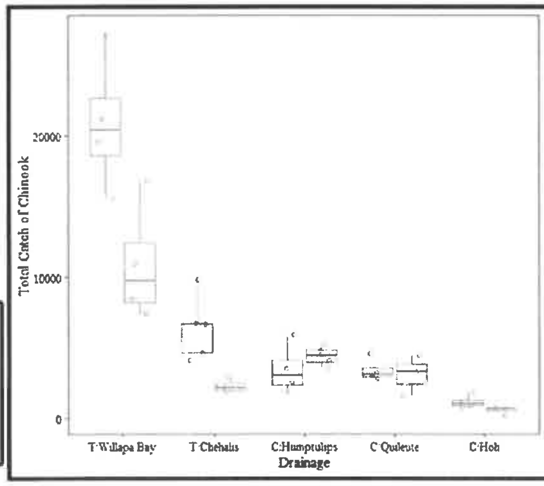
- Controls for natural changes in variation due to time (*Underwood et al 1991, 1992*)
- Multiple sites
- Assign treatment and control sites
  - Impact sites
    - Willapa (C-3622)
    - Chehalis (C-3621)
  - Control
    - Hoh
    - Quileute
    - Humptulips



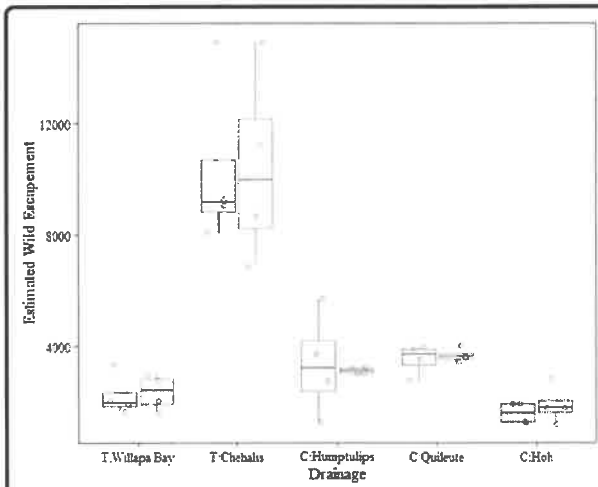
Total catch (Recreational + Commercial) declined as a function of policy implementation

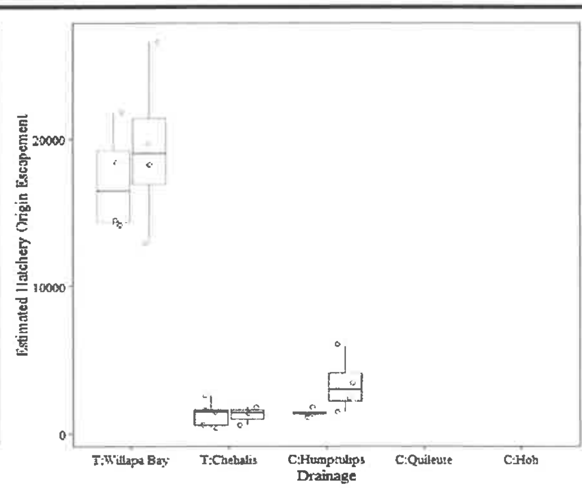
Candidate Models,	k	AICc	▲	w	Log Liklihood	Cum w
P * H + D	***8***	***733.66***	***0.00***	***1.00***	***-356.43***	***0.71***
P * H + D + Y	***9***	***735.42***	***1.76***	***0.41***	***-355.61***	***1.00***
D + Y	7	747.89	14.24	0.00	-365.14	1.00
D + P	7	748.83	15.17	0.00	-365.61	1.00
D + P + H	7	748.83	15.17	0.00	-365.61	1.00
D	6	754.49	20.83	0.00	-369.93	1.00
P * H	5	779.20	45.54	0.00	-383.69	1.00
H	3	779.70	46.05	0.00	-386.51	1.00
P * H + Y	6	780.05	46.39	0.00	-382.71	1.00
Y	3	796.17	62.51	0.00	-394.74	1.00
P	3	796.40	62.75	0.00	-394.86	1.00
Intercept Only	1	818.23		0.00	-408.06	1.00

- D = The coastal drainage,
- Y = Year of the escapement estimate,
- H = Impact vs control sites,
- C = Total Catch,
- P = Policy status based upon year (2011-2014 = Prior to policy, 2015-2018 = Policy years).



Redd-based escapement estimates for wild-origin and hatchery-rack based estimate for hatchery-origin salmon





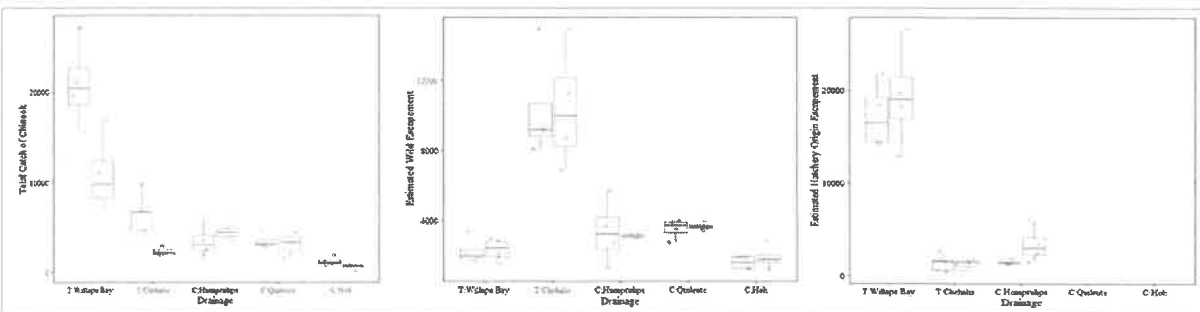
Wild-origin Escapement as a function of the BACI predicted interaction.

Candidate Models$_a$	k	AICc	▲	w	Log Liklihood	Cum w
D + H + C	*7*	***695.02***	***0.00***	***1.00***	***-338.70***	***0.95***
C * H + D	9	701.19	6.18	0.05	-338.49	1.00
D	6	709.16	14.14	0.00	-347.31	1.00
D + Y	7	711.97	16.95	0.00	-347.23	1.00
D + P + H	7	712.12	17.10	0.00	-347.31	1.00
P * H + D	8	715.22	20.20	0.00	-347.29	1.00
D * H + D	9	718.00	22.98	0.00	-347.00	1.00
D * Y	11	725.23	30.21	0.00	-346.90	1.00
C	3	753.44	58.42	0.00	-373.38	1.00
P + C	4	755.50	60.48	0.00	-373.16	1.00
H	3	760.53	65.51	0.00	-376.93	1.00
P * H	5	765.05	70.03	0.00	-376.64	1.00
P * H + Y	6	767.69	72.67	0.00	-376.57	1.00
P	3	771.41	76.39	0.00	-382.37	1.00
Y	3	771.71	76.69	0.00	-382.52	1.00
Intercept only	1	804.36	109.34	0.00	-401.13	1.00

- D = The coastal drainage,
- Y = Year of the escapement estimate,
- H = Impact vs control sites,

C = Total Catch,
P = Policy status based upon year (2011-2014 = Prior to policy, 2015-2018 = Policy years).

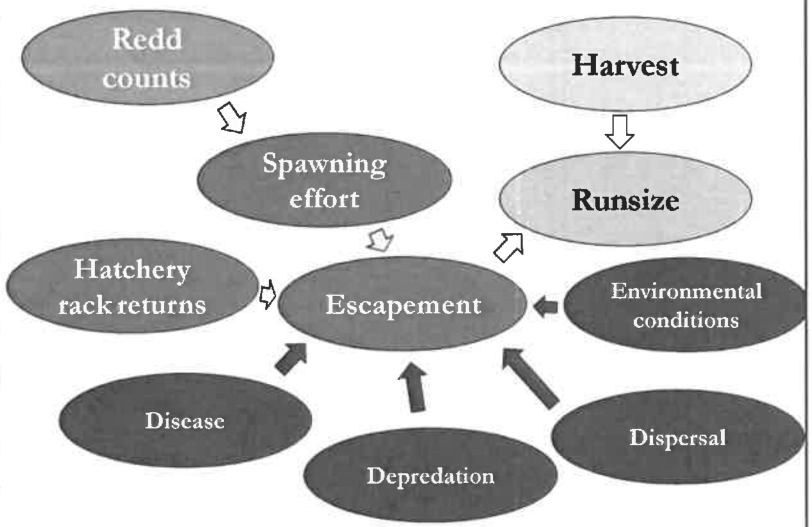
Redd-based escapement estimates for wild-origin and hatchery-rack based estimate for hatchery-origin salmon



Willapa Bay results:
- Harvest was reduced by 48.9%
- Wild Chinook redd-based escapement estimate increased by 4.97%
- Hatchery Chinook rack-based escapement plus hatchery spawning increased by 17.13%
- Total increased escapement was 22.1% increase
- * *given constant runsize.*

This is likely the actual model for determining runsize

- Runsize is more complicated than just spawning effort and harvest
  - Disease
  - Depredation
  - Dispersal
  - Mortality from abiotic conditions
- Problems with redd-based escapement survey methodology
  - Site selection
  - Estimated coefficients for extrapolation



CONCLUSIONS

- We reduced fishing pressure
- We did not find significant evidence that reductions in harvest resulted in increased wild-origin Chinook escapement
- Not finding a pattern does not mean there is no pattern!
  - *i.e.* There may have been a pattern and we failed to detect it
    - Sampling protocol
    - Simplistic assumptions associated with calculations
    - Small sample sizes

Questions?