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The Effectiveness of Large Woody Debris Placement at Improving Freshwater Rearing Habitat and Enhancing Juvenile Salmon (Oncorhynchus spp.) Production

Caroline J. Walls
Western Washington University, cwalls0426@gmail.com

The Effectiveness of Large Woody Debris Placement at Improving Freshwater Rearing Habitat and Enhancing Juvenile Salmon (*Oncorhynchus* spp.) Production

By

Caroline J. Walls

Accepted in Partial Completion
of the Requirements for the Degree
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ADVISORY COMMITTEE

Dr. James M. Helfield, Chair

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Jennifer S. O'Neal, M.S.

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GRADUATE SCHOOL

David L. Patrick, Interim Dean

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Caroline J. Walls

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The Effectiveness of Large Woody Debris Placement at Improving Freshwater Rearing Habitat and Enhancing Juvenile Salmon (*Oncorhynchus* spp.) Production

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Presented to
The Faculty of
Western Washington University

In Partial Fulfillment
Of the Requirements for the Degree
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Abstract

The decline of Pacific salmon (*Oncorhynchus* spp.) is well-documented, and freshwater habitat degradation is a primary contributor. Despite decades of river restoration, salmon populations have not significantly recovered. Large woody debris (LWD) placement is one of the most common forms of restoration. To evaluate the effectiveness of this restoration method, I analyzed long-term monitoring data from 16 LWD placement projects throughout Washington State, implemented between 2004 and 2015. Each project followed a multiple Before-After, Control-Impact study design, which monitored physical habitat and fish populations. I used a series of linear mixed models to evaluate both habitat and fish response. I found that habitat features responded positively, with increases in average residual pool depth, pool area, and habitat complexity. However, fish response varied by species and location. I looked for changes in both abundance and size of juvenile coho (*O. kisutch*), Chinook (*O. tshawytscha*) and steelhead/rainbow trout (*O. mykiss*). The average size of *O. mykiss* increased over time. Coho and coastal Chinook populations were largely unaffected, indicating that these populations are limited by factors unaddressed by LWD placement. Inland Chinook populations increased in abundance immediately, but declined in average size over time, indicating over-crowding at restoration sites due to a lack of high-quality habitat. My results demonstrate that LWD placement is effective at improving freshwater salmon habitat, but these improvements are not generating consistent increases in juvenile salmon abundance or biomass, suggesting that LWD placement does not always address the limiting factors for salmon production. Broader threats to salmon recovery, including declining ocean conditions, climate change, and dams, must also be addressed to improve effectiveness of restoration. My findings also highlight the vital need for comprehensive, long-term monitoring of restoration actions to guide future salmon recovery efforts.

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Introduction

Pacific salmon (*Oncorhynchus* spp.) play important spiritual, cultural, economic, and ecological roles in the Pacific Northwest. They have been described as keystone species due to their importance as a food source for marine and terrestrial vertebrates, and as a vector for nutrients in freshwater and terrestrial food webs (Willson and Halupka 1995, Willson et al. 1998, Lundberg and Moberg 2003, Helfield and Naiman 2006). In the northeastern Pacific Ocean, Chinook salmon (*O. tshawytscha*) provide an essential food source for declining resident killer whales (*Orcinus orca*; Ford et al. 2010). In addition, commercial and recreational fishing of Pacific salmon contribute more than $1 billion to the US economy annually (NOAA 2017a). Thus, the status and health of Pacific salmon are of special concern.

The decline of Pacific salmon has been well-documented, and the causes are conclusively human-derived (NRC 1996, NOAA 2015). Freshwater habitat degradation has long been recognized as a primary driver, although climate change is poised to exacerbate existing challenges with warming temperatures and changing patterns of precipitation and streamflow (Nehlsen et al. 1991, NRC 1996, Mote et al. 2003, Battin et al. 2007, Beechie et al. 2013). As a result, a great deal of effort and funding has gone into freshwater habitat restoration, totaling approximately $2 billion since the year 2000 (NOAA 2017b). With so much at stake, there is great interest in the efficacy of salmon habitat restoration projects.

Of particular concern among habitat managers is the loss of large woody debris (LWD) in freshwater systems. Among other benefits, naturally occurring LWD is positively correlated with pool frequency, pool depth, instream cover, and habitat diversity, which are all vital components of salmon rearing habitat (Trotter 1990, Abbe and Montgomery 1996, Naiman et al. 2002, Beechie et al. 2005, Quinn 2005, Roni et al. 2015). Wood-formed pools are created through scouring processes that occur during high flow events, typically during the winter and spring (Abbe and Montgomery 1996). Historic logging and stream clearing practices have resulted in unnaturally low levels of LWD in Pacific Northwest waterways (Maser and Sedell 1994, Collins, et al. 2002, Wooster and Hilton 2004, Wohl 2014). In an effort to emulate the habitat benefits of natural wood and to bolster declining salmon populations, the placement of LWD has become one of the most popular forms of freshwater habitat restoration (Roni et al. 2002, 2008, 2015, Bernhardt et al. 2005, Katz et al. 2007). More than 2,000 wood placement projects have been implemented since 1980 in the Columbia River Basin alone (Roni et al. 2015). Such projects range from simply placing large wood from the riparian zone into the active stream channel, to the construction of complex engineered log jams (Roni and Beechie 2013).

Stream restoration projects are typically carried out with the expectation that the outcome will be improved habitat, which will then result in increased production of salmon (Roni et al. 2008). However, there is often little follow-up monitoring to determine if this is true. Historically, just 10% of all river restoration projects receive any sort of post-restoration assessment, often because funding for monitoring is difficult to obtain (Bernhardt et al. 2005). Even when monitoring does occur, it is typically insufficient to evaluate fish response. Salmon populations are inherently variable from year to year, so long-term monitoring is especially important. Five to ten years of monitoring is the recommendation for stream restoration projects (Hunt 1976, Kondalf and Micheli 1995), but a 2010 meta-analysis of 211 in-stream restoration projects found that less than 5% of monitoring programs reach that benchmark (Whiteway et al. 2010). Furthermore, most restoration monitoring projects fail to collect any pre-project (i.e., baseline) data, making it extremely difficult to establish causal relationships between restoration actions and outcomes (Bash and Ryan 2002). Multiple years of baseline data are essential to capture inter-annual variations in stream conditions and fish abundance in order to establish an accurate baseline against which to compare post-restoration data.

Given the large number of LWD placement projects in existence, we do at least have a growing body of literature on the resultant physical habitat changes, which are mostly considered beneficial for fish. Roni et al. (2015) summarized the findings of 83 wood placement studies and reported that more than 90% had positive results for at least one habitat metric. However, the authors acknowledged that studies of successful restoration projects are more likely to be published than those of unsuccessful projects (Kondalf and Micheli 1995). Common metrics that improve after LWD placement are habitat complexity, instream cover, pool frequency, and pool depth (Roni et al. 2015). Despite these habitat findings, the effectiveness of LWD placement at improving salmon production remains uncertain (Hunt 1988, Paulson and Fisher 2005, Stewart et al. 2009, Roni et al. 2008 & 2015, Whiteway et al. 2010, Krall et al. 2019).

Physical habitat enhancement itself is typically not the end-goal, but rather a means of increasing the production of the target species. Therefore, it is important to evaluate both physical and biological responses to restoration. Given the rarity of such dual analyses in the published literature (Katz et al. 2007), it is unsurprising that our current understanding of fish response to restoration is inconclusive. Despite the efforts and expenditures directed towards salmon habitat restoration in recent decades, salmon populations have not recovered appreciably. Seventeen distinct population segments, or evolutionarily significant units, of Pacific salmon and steelhead (*O. mykiss*) remain listed as threatened or endangered under the terms of the U.S. Endangered Species Act (NOAA 2015). This is indicative that at least one of the following is occurring: (1) our restoration actions are failing to improve habitat as intended; (2) restoration is improving habitat, but salmon are not responding positively, which would suggest that habitat restoration projects are not addressing the limiting factors for salmon production; or (3) habitat is improving and salmon are responding positively, but those gains are more than offset by additional drivers of salmon decline that are not addressed by our current levels or current focus of habitat restoration.

The research presented here attempts to fill some of the gaps in our understanding of salmon response to freshwater habitat restoration. My research uses data from multiple LWD placement projects across Washington State to evaluate their effectiveness at improving habitat and increasing salmon production. The specific objectives of this work are to assess the immediate and long-term effects of LWD placement on (1) physical habitat and (2) the diversity and abundance of juvenile salmon at restoration sites. In doing so, I hope to promote salmon recovery by informing future restoration and monitoring actions.

Methods

Restoration Projects and Monitoring Design

The data used in my analyses come from 16 LWD placement projects that were implemented in various watersheds throughout Washington State, between 2004 and 2015 (Table 1, Figure 1). Data were collected at project sites through two monitoring programs: the Project Effectiveness program established by Washington State's Salmon Recovery Funding Board (SRFB), and the Action Effectiveness Monitoring program that operated in partnership with the Bonneville Power Administration's Columbia Habitat Monitoring Program (CHaMP). Both programs follow sufficiently similar monitoring protocols to allow the resulting data to be analyzed together (WSRCO 2014, PNWAMP 2015).

The monitoring protocols for both SRFB and CHaMP followed a multiple Before-After Control-Impact experimental design (Stewart-Oaten et al. 1986). The intensity and scope of LWD placement varied among restoration projects, and many included additional restoration actions (Table 1; Bennett et al. 2015, Martin and Buelow 2017, WSRCO 2018). Each project included one or more "treatment" study reach(es) that were located within the area of planned stream restoration. Comparable "control" study reach(es) were selected based on morphological and habitat similarities to the pre-restoration state of their paired treatment study reach(es). Control study reaches were located upstream of the restoration zone, in an area that would remain untreated for the duration of monitoring.

The paired treatment and control study reaches were monitored at varying intervals, both before and after restoration actions were implemented. The pairs were subjected to the same multi-year sampling schedules, though schedules varied among the projects (Table 2). In this analysis, I define the year of restoration as year 0. Survey years before restoration are negative (e.g., years -3, -2, -1), and survey years after restoration are positive (e.g., years 1, 2, 3). If a monitoring survey occurred in year 0, it was completed prior to the implementation of restoration actions. Monitoring occurred from May through November. In order to minimize seasonal differences, the paired treatment and control study reaches were typically sampled during the same week in a given monitoring year, and never more than two weeks apart. Within a single monitoring year, the same technician crew monitored all study reaches of a given project.

Habitat Data Collection and Response Variables

Study reaches ranged in length from 100 m to 600 m, depending on the average bankfull width (m) of the active stream channel (Table 2). Standard habitat units, including riffles, pools, and glides, were delineated by field technicians prior to conducting topographic surveys. The CHaMP and SRFB monitoring protocols conducted topographic surveys through different means, necessitating that some habitat metrics be calculated differently. Habitat data for projects monitored under the CHaMP protocols were collected with a surveying total station, which allowed managers to generate digital elevation models of the active stream channel and surrounding floodplain. This allowed precise calculation of habitat unit area (m^2) and residual depth (m). Habitat data were collected at SRFB-monitored projects via a longitudinal thalweg profile, which limited width and depth measurements to equally spaced, pre-set intervals, meaning the habitat unit calculations were coarser estimates.

In order to evaluate physical habitat changes, I used three response variables:

i) *Pool : Reach Ratio:* The ratio of pool area (m^2) to study reach area (m^2), calculated as the summed area of all of the pools within a study reach, divided by the total area of the study reach.

ii) *Mean Residual Pool Depth (RPD)*: The mean RPD (m) of all habitat units in a study reach that were identified as pools by technicians in the field. RPD was calculated as the difference between the maximum pool depth and the minimum tail-out depth of each pool (Lisle 1987).

iii) *Habitat Diversity:* A modified version of Shannon's diversity index (Shannon and Weaver 1949; formula 1),

$$(1)\ \ Shannon's\ Diversity\ Index_{\,i,j} = -\sum_{i=1}^{n} p_i Ln(p_i),$$

in which n is the total number of different habitat unit types in a study reach, and p_i is the proportional area occupied by each habitat unit type (i).

<u>Fish Data Collection and Response Variables</u>

Fish data were primarily collected via snorkel surveys under both protocols. Snorkelers recorded the species, number, and size (estimated to the nearest 10 mm) of each fish observed within a study reach. On rare occasions, if water quality was too poor to conduct a snorkel survey, backpack electrofishing was used to collect the same information. In order to maintain consistency, if electrofishing was necessary, it would be used at all study reaches within the project during the same monitoring year. Three projects in my study used passive integrated transponder (PIT) tags to track fish response. Because the resulting data were not comparable to the snorkel survey data, they were left out of the fish analyses (Figure 2).

In order to evaluate the diversity of the salmonid fish community, I considered the abundance and size distribution of all salmonid fishes, excluding individuals that were identified as spawning adults. This included: Chinook salmon, coho salmon (*O. kisutch*), steelhead and rainbow trout (which I will refer to collectively as *O. mykiss*, hereafter), cutthroat trout (*O. clarkii*), bull trout (*Salvelinus confluentus*), brook trout (*S. fontinalis*), and mountain whitefish (*Prosopium williamsoni*). I defined six size class bins for each species: (A) <50 mm, (B) 51-100 mm, (C) 101-150 mm, (D) 151-200 mm, (E) 201-250 mm, and (F) >251 mm (Kiffney et al. 2006). To assess changes in salmonid community diversity at project sites, I used the following response variable:

iv) *Species-Size Class Diversity*: A modified version of Shannon's diversity index (Shannon and Weaver 1949; formula 1),

$$(1)\ Shannon's\ Diversity\ Index_{\ i,j} = -\sum_{i=1}^{n} p_i Ln(p_i),$$

in which $n =$ the number of species-size class bins observed in a study reach, and p_i is the proportional abundance of fish belonging to the i^{th} size-class bin.

In order to evaluate juvenile salmon populations, I focused on three individual target species: coho salmon, Chinook salmon, and *O. mykiss*. I limited my analysis of *O. mykiss* to individuals < 300 mm in total length, because I could be reasonably certain that any individuals larger than 300 mm were resident rainbow trout, based on observations of the upper range of steelhead smolt sizes (Partridge 1985, Peven et al. 1994, Kendall et al. 2014). For each target species, I assessed the following response variables:

v) *Fish $_{Area}$* : Calculated as the number of fish per 100 m^2 of stream area.

vi) *Biomass $_{Area}$* : Calculated as total estimated biomass (g) per 100 m^2 of stream area. Models testing *Biomass $_{Area}$* were designed to evaluate biomass with respect to fish density, by including *Fish $_{Area}$* as an explanatory covariate. Thus, the models can be interpreted as evaluating the average biomass per fish.

vii) *Fish $_{Length}$* : Calculated as the number of fish per 100 m of stream length.

viii) *Biomass $_{Length}$* : Calculated as total estimated biomass (g) per 100 m of stream length. Models testing *Biomass $_{Length}$* were designed to evaluate biomass with respect to fish density, by including *Fish $_{Length}$* as an explanatory covariate. Thus, the models can be interpreted as evaluating the average biomass per fish.

I evaluated fish density and biomass in terms of both study reach area and study reach length because I recognize that stream width may be impacted by restoration actions. By looking at the response variables in terms of both stream area and stream length, I am better able to distinguish if changes in response variables are due to changes in habitat quantity or habitat quality.

To calculate biomass of the target species, I extrapolated weight (g) estimates from the recorded lengths. I obtained length / weight data for each species from the PIT Tag Information System database (PSMFC 2019), which I fit in a least squares regression of the format:

$$(2)\ \log(Weight) \sim a + b\log(Fork\ Length).$$

I then used the results of each regression to generate a predictive weight equation for each species, using the expression

$$(3)\ W = aL^b,$$

in which *W* is the fish weight (g), *L* is the fish length (mm), and *a* and *b* are the parameters of the regression (Crec'hriou et al. 2015; Table 3).

Not every species was present at every project. A project was excluded from a species analysis if that species was absent from all study reaches of that project for all monitoring years (Figure 2 and Table 6).

Data Analysis

I used a linear mixed model approach to evaluate the effects of LWD placement on each response variable. All analyses were carried out in R version 3.6.2 (R Core Team 2019), using the “nlme” package (Pinheiro et al. 2019). Each model followed the structure in Table 4, using the same three fixed effects: (*i*) *Time*, (*ii*) *Treatment*, and (*iii*) *Time Since Treatment*. This model design was intended to capture discontinuous change, in which the treatment of LWD placement can affect both the intercept and slope of the response variables (Singer and Willett 2003). The intercept represents the baseline response variable value at the year of restoration, and the slope represents the change in response variable per year. The fixed effect of *Time* was measured in terms of years since restoration and represents the slope that applies to all projects and all study reaches (i.e., the background rate of change). *Time* was normalized for each project so that the year of restoration was *Time* = 0, while years before were negative and years after were positive. The fixed effect of *Treatment* was defined as a binary condition, for which a study reach was categorized as either having been treated with LWD placement or not. Control study reaches retain a value of *Treatment* = 0 for all monitoring years, whereas treatment study reaches have a value of *Treatment* = 0 for monitoring years before restoration and *Treatment* = 1 for monitoring years after restoration. *Treatment* can be thought of as the static effect of LWD placement, representing the difference in intercept of treatment study reaches, relative to the intercept of control study reaches. *Time Since Treatment* is the interaction of *Time* and *Treatment*, and can be thought of as difference in slope, or rate of change in response variable, that occurs after LWD placement, relative to the background rate of change (i.e. $\text{Slope}_{(Trtmnt\ Study\ Reaches)} = \beta_{Time} + \beta_{Time\ Since\ Trt}$). I did not account for any variations in treatment intensity or additional restoration actions in the fixed effects.

Each model included a continuous autoregressive residual covariance structure, which used individual study reaches as the subject. To account for inherent differences among streams, and to nest paired control / treatment study reaches within their respective projects, each model included random effects for project (Tables 5-6). In all analyses, I tested model assumptions of homogeneity of variance and normality through visual inspection of diagnostic plots. In order to achieve normality, it was often necessary to include explanatory covariates in the models, in addition to the fixed effects (Tables 5-7, Appendix 1-2). I compiled a list of potential explanatory covariates based on ecological knowledge of factors that could influence my response variables, and also based on data availability. I plotted each covariate against the residuals of an "empty" model (i.e., a model including just the random effects for project with no fixed effects). I selected covariates for inclusion in the final model of a response variable if their plots showed a biased distribution of residuals. When appropriate, I used the "weights" statement in the "nlme" package to allow residual variance to vary by different identifying factors (Tables 5-6; Zuur 2009, Pinheiro et al. 2019).

Results and Discussion

Habitat Response

As measured by each of the three habitat response variables, physical habitat improves after LWD placement (Table 8, Figure 3). *Pool : Reach Ratio* exhibits an immediate, static increase after restoration ($\beta_{Treatment} = 0.059$, $P < 0.05$). *Mean RPD* declines immediately ($\beta_{Treatment} = -0.077$, $P < 0.001$), but increases steadily over time ($\beta_{Time\ Since\ Trt} = 0.044$, $P < 0.001$), resulting in a net increase by the third year after restoration. *Habitat Diversity* seems to be increasing over time, regardless of restoration status ($\beta_{Time} = 0.037$, $P < 0.001$). At treatment study reaches, *Habitat Diversity* exhibits an immediate, static increase after restoration ($\beta_{Treatment} = 0.155$, $P < 0.01$). However, this may be an acceleration of the background rate of change, rather than an addition to it, because the net slope for treatment study reaches subsequently flattens (Slope $_{(Trtmnt\ Study\ Reaches)} = \beta_{Time} + \beta_{Time\ Since\ Trt} = 0.037 + (-\ 0.031) \approx 0$).

When taken together, my model results for the three habitat response variables paint a picture of the physical changes achieved from LWD placement projects. The high flows during the first winter and spring after LWD placement generate enough energy to begin scouring pools around the wood. This results in an increase in *Pool : Reach Ratio* and *Habitat Diversity* great enough to be observed at the first post-restoration monitoring event, and both variables then remain constant over time. The young pools are initially shallow, which brings down the average RPD at treatment study reaches that had any deep pool habitat prior to restoration. As time goes on, repeated scour events deepen the new pools, resulting in a net increase in *Mean RPD* after approximately 3 years. When LWD placement projects are proposed, they are often touted for the immediacy with which they improve stream habitat (J. Helfield, *pers. comm.*). While it is clear that LWD placement projects have a much more rapid impact than other forms of restoration (e.g., riparian planting), my results indicate they may take several years to fully realize. This underscores the need for long-term monitoring to properly evaluate restoration effectiveness.

Fish Response

Species-Size Class Diversity

Species-Size Class Diversity exhibits an immediate decline after LWD placement ($\beta_{Treatment} = -0.098$, $P < 0.01$), but then increases over time ($\beta_{Time\ Since\ Trt} = 0.022$, $P < 0.05$; Table 8, Figure 3). This corresponds to a net increase in *Species-Size Class Diversity* after 4-5 years. This closely follows the results for the physical habitat response, particularly *Mean RPD*. Previous research found salmonid diversity largely tracked instream wood volume (Kiffney et al. 2006). This is consistent with my results, which show salmonid diversity tracks the effects of instream wood. The delayed benefits are further evidence of the need for long-term monitoring of restoration actions. Increased *Species-Size Class Diversity* can be considered a desirable outcome of LWD placement, because diverse communities are known to be more resilient to disturbance and environmental challenges (May 1973, McCann 2000, Balvanera et al. 2006, Ives and Carpenter 2007). Diversity is likely to become increasingly important as climate change takes its toll (Battin et al. 2007).

Juvenile Coho Salmon

The models testing coho $Biomass_{Area}$ and $Biomass_{Length}$ do not show any significant fixed effects (Table 9, Figure 4), indicating that LWD placement does not affect juvenile coho size. In contrast, *Time Since Treatment* has a significant positive effect on both $Fish_{Area}$ and $Fish_{Length}$, suggesting that the density of juvenile coho at treatment study reaches increases over time, relative to the density at control study reaches (Table 9, Figure 4). However, the effect of *Time* is also significant in both model results, which means I must consider the beta estimates of both *Time* and *Time Since Treatment* in order to evaluate the net magnitude of change after restoration, which is important for judging the biological significance of the results.

The model for coho $Fish_{Area}$ shows statistically significant results for all three fixed effects (Table 9, Figure 4). Taken together, they suggest a background decline in coho abundance per unit stream area (β_{Time} = - 0.288, P <0.001) and a further, immediate decline in abundance after LWD placement ($\beta_{\ Treatment}$ = - 0.253, P <0.01). Coho abundance then remains stable over time at treatment study reaches (i.e. Slope $_{Trtmnt\ Study\ Reaches}$ = - 0.288 + 0.274 ≈ 0 fish · 100 m^{-2} · yr^{-1}), while continuing to decline at control study reaches. While this indicates that LWD placement might slow the decline in coho density over time, the effect is not great enough to reverse a negative background trend.

In contrast, the model results for coho *Fish* $_{Length}$ (Table 9, Figure 4) indicate that background coho abundance per unit stream length is increasing over time ($\beta_{\ Time}$ = 1.454, P <0.01). There is no immediate effect of LWD placement on coho *Fish* $_{Length}$, but there is a positive effect over time. However, the effect is quite small, resulting in a rate of increase in coho abundance at treatment study reaches that is hardly faster than the background trend (i.e., Slope $_{(Trtmnt\ Study\ Reaches)}$ = 1.454 + 0.022 ≈ 1.476 fish · 100 m^{-1} · yr^{-1}). The beta estimate for *Time Since Treatment* is two degrees of magnitude smaller than the beta estimate for *Time,* which corresponds to a rate of increase that would take approximately 50 years to gain one additional fish per 100 m of stream at treatment study reaches, relative to control study reaches ($\beta_{\ Treatment}$ = 0.022 fish · 100 m^{-1} · yr^{-1}). This rate of increase will not yield biologically significant improvements to coho production on a time scale that is meaningful for salmon recovery.

The conflicting results for the fixed effect of *Time* in the coho *Fish* $_{Area}$ and *Fish* $_{Length}$ models suggest that background coho abundance is declining with respect to stream area, while simultaneously increasing with respect to stream length. This could happen if the project streams were getting wider over time for reasons unrelated to LWD placement, as stream widening could result in more fish per stream length, but less per stream area. However, I found no evidence in the raw topographic data to suggest this is occurring. Thus, I believe these results are more likely a relic of the wide range in survey dates (mid-May to mid-November), which can correspond to dramatically different flow rates and correspondingly different stream widths at the time of the snorkel surveys from year to year. These results call attention to the importance of consistent, long-term baseline and control data in restoration monitoring. Even though I cannot definitively describe background trends for coho abundance in my project streams, the paired design (and the fact that paired treatment and control study reaches were typically surveyed during the same week and under the same flow conditions in each sampling year) allows me to interpret how coho density is responding to LWD placement, relative to background density. My coho $Fish_{Area}$ and $Fish_{Length}$ model results support the conclusion that juvenile coho salmon exhibit little or no biologically significant response to LWD placement, which is counter to previous studies that have reported increases in coho production after freshwater habitat restoration (*see* Roni et al. 2013).

Given the observed habitat changes, I would have expected to see a much greater increase in juvenile coho production after restoration. Juvenile coho are known to selectively inhabit deeper pools and pools with a greater abundance of LWD (Bisson et al. 1988, Quinn and Peterson 1996). Such pools are correlated with higher overwinter survival and greater smolt production (Bustard and Narver 1975, Murphy et al. 1986, Nickelson et al 1992*a*, 1992*b*, Sharma and Hilborn 2001). Previous studies have shown increased coho production in response to restoration projects in which artificially-placed LWD generated deep, wood-formed pools (Cederholm et al. 1997, Roni and Quinn 2001). If insufficient deep pool habitat was the initial limiting factor in my study streams, the lackluster coho response is surprising.

It is possible that the availability of deep pools is not the limiting factor for coho production in some of my study streams. Off-channel habitats are important for overwintering survival of juvenile coho, and in some cases may be more important than deep pools in the main channel (Bustard and Narver 1975, Nickelson et al. 1992a, 1992b, Pollock et al. 2004). Coho densities have been found to respond positively to restoration projects that enhance overwintering habitat in the form of reconnected or constructed side channels (Morley et al. 2005, Henning et al. 2006, Roni et al. 2006, Roni et al. 2010). My results are consistent with the recent findings of Anderson et al. (2019), which demonstrate that LWD placement might not enhance coho abundance at sites where other habitat factors are limiting.

The addition of time-varying covariates to the juvenile coho models gives some potential insight into contributing factors that may be limiting juvenile coho response to LWD placement. All four coho models contain three of the same explanatory covariates, which were necessary to include in these models to achieve normality of residuals (Tables 6-7). Those covariates are: (1) *Summer Air Temperature*, which gives the regional average departure from normal air temperature (°C) for the months April-September (NOAA 2019), and serves as a proxy for summer stream temperatures; (2) *Snow Water Equivalent*, which gives the percent of normal snow water equivalent on April 1 for the Pacific Northwest Region (USDA 2019), and serves as a proxy for winter peak flows and summer low flows; and (3) *Snorkel Date*, which is the standardized Julian calendar date of each snorkel survey, and serves as a proxy for the number of growing days that have passed by the time of the survey. These three explanatory covariates are statistically significant in all coho models, with the one exception being *Snow Water Equivalent* in the *Fish* $_{Length}$ model (Table 9). In every model, the magnitudes of the beta estimates for each of these explanatory covariates are greater than those of the significant fixed effects (between ~1.4 – 400 times greater). This indicates that in my study streams, juvenile coho abundance and size are more greatly influenced by climatic variables than by the physical habitat characteristics influenced by LWD placement.

Furthermore, for each of the aforementioned explanatory covariates, the sign of the beta estimate in the *Fish* model results (positive/negative) is opposite to the sign in the *Biomass* model results (Table 9). This suggests that factors correlated with higher juvenile coho density are simultaneously correlated with smaller coho sizes, and vice versa. An inverse relationship between fish density and fish size has been well-documented in systems that exhibit density-dependent growth due to competition and limited food availability (Elliot 1984, Grant and Kramer 1990, Keeley 2001, Imre et al 2004, Grant and Imre 2005, Connor et al. 2013). Therefore, it is possible that I did not observe a greater biological response from coho to LWD placement because the coho populations in my study streams are already close to their carrying capacities, as dictated by the availability of food resources. Further research and restoration actions targeting the autochthonous and allochthonous nutrient sources supporting juvenile coho salmon may be necessary to support coho recovery.

Another possible explanation for the lack of coho response may be the timing of the sampling season (May-November). Previous research evaluating coho production after LWD placement has found the response to be greater in the winter than in the summer (Cederholm et al. 1997, Roni and Quinn 2001). Thus, it is possible that LWD placement has, in fact, improved overwinter survival of coho at the treatment study reaches, but this was not captured with the monitoring schedule. However, given enough time and multiple generations, greater overwinter survival should eventually result in observable increases in juvenile coho abundance throughout the sampling season.

Juvenile Chinook Salmon

The results of my juvenile Chinook models indicate that the species' response to LWD placement projects varies depending on whether the population is coastal or inland (Table 7). There is strong ecological justification to distinguish between coastal and inland populations, due to the different distributions of ocean-type and stream-type Chinook. Ocean-type Chinook, which typically migrate to the oceans during their first three months, are more often found near the coastline in the lower reaches of rivers, whereas stream-type Chinook, which spend one or more year(s) in freshwater before migrating to the ocean, are more likely to be found in smaller tributary streams of major rivers like the Columbia River (Taylor 1990, Healey 1991, Myers et al. 1998, Quinn 2005). Thus, I presume the coastal populations have greater representation of ocean-type Chinook and the inland populations have greater representation of stream-type. The distribution of project locations within my sample made it easy to determine coastal and inland designations. I had a subset of projects located < 200 stream km from the ocean (coastal) and a subset of projects located >500 stream km from the ocean (inland), with nothing in between (Figure 2). All inland projects in my sample that contained Chinook were monitored under CHaMP protocol, while all coastal projects were monitored under SRFB protocol, but I have no reason to believe that this should impact the results, as the two protocols used similar methods for surveying fish. It is highly plausible that inland Chinook would have a more pronounced response to freshwater restoration than coastal Chinook, which is precisely what my results illustrate.

Coastal Chinook exhibit relatively little response to LWD placement (Table 10, Figure 5). The combined results of both $Fish_{Area}$ and $Biomass_{Area}$ suggest that coastal Chinook increase in density per unit area immediately after restoration, but decline over time in both number and biomass. However, these results only appear in the $Fish_{Area}$ and $Biomass_{Area}$ models, indicating that they might be a relic of changes in stream width after LWD placement. The results of *Fish* $_{Length}$ and *Biomass* $_{Length}$ show no significant effects of restoration at coastal Chinook populations.

In contrast, the fixed effects results for inland Chinook are consistent with respect to both study reach area and study reach length (Table 10, Figure 5). The interaction of *Treatment* * *Inland* is statistically significant and positive in both *Fish* $_{Area}$ ($\beta_{Treatment}$ = 3.191, P <0.05) and *Fish* $_{Length}$ ($\beta_{Treatment}$ = 35.585, P <0.01), while the interaction of *Time Since Treatment* * *Inland* has no effect, indicating an immediate increase in abundance, which remains constant over time. This suggests either increased survival at early life stages or increased migration from other habitat to the restoration zones, or both. The consistency of response with respect to both study reach area and study reach length suggests that LWD placement improves both habitat quantity and habitat quality for juvenile inland Chinook.

The models for *Biomass* $_{Area}$ and *Biomass* $_{Length}$ are similarly consistent, both showing a significant positive effect of the interaction of *Treatment* * *Inland* ($\beta_{Treatment,\ Area}$ = 4.390, P <0.05; $\beta_{Treatment,\ Length}$ = 78.167, P <0.05) and a significant negative effect of the interaction of *Time Since Treatment* * *Inland* ($\beta_{Time\ Since\ Trt,\ Area}$ = -3.016, P <0.05; $\beta_{Time\ Since\ Trt,\ Length}$ = -47.803, P <0.05; Table 10, Figure 5). This indicates that the size of inland juvenile Chinook increases immediately after restoration, but then declines steadily over time. The beta estimates for the interaction of *Time Since Treatment* * *Inland* in both *Biomass* models correspond to a net decline in the size of Chinook at restoration sites after just two years. This revelation is yet another example of the value of long-term monitoring, lasting the recommended 5- 10 years (Hunt 1976, Kondalf and Micheli 1995).

The static increase in inland juvenile Chinook abundance at treatment study reaches suggests that LWD placement increases salmon production, but the subsequent decline in biomass over time cannot be ignored. The combined results strongly indicate density-dependent growth after restoration due to resource competition (Murphy et al 1986, Bilby and Bisson 1987 & 1992, Keeley 2001). Slower growth can delay smolting and slow downstream migration, both of which increase the risk of mortality during these later life stages, especially in populations with longer migrations (Giorgi et al. 1997, Quinn 2005, Connor and Tiffan 2012). Furthermore, size is highly correlated with smolt-to-adult survival in salmonids (Ward et al. 1989, Henderson and Cass 1991, Mortensen et al. 2000). Therefore, if left unchecked, the decline in size of inland juvenile Chinook could counteract any productivity gains from increased abundance.

Previous research has found that when high-quality habitat is scarce, juvenile salmonids preferentially aggregate in these areas, even when densities become high enough to inhibit growth (Kahler et al. 2001, Kiffney et al. 2014). My results suggest that such localized "over-crowding" of juvenile Chinook is occurring after LWD placement at inland locations. Therefore, the best way to address this density-dependent growth of inland Chinook might be to greatly increase the number of restoration projects.

Juvenile *O. mykiss*

The model results for *O. mykiss* show that LWD placement has no significant effect on $Fish_{Area}$ or $Fish_{Length}$ (Table 11, Figure 6). However, *Time Since Treatment* has a significant positive effect on both $Biomass_{Area}$ and $Biomass_{Length}$ ($\beta_{Time\ Since\ Trt,\ Area}$ = 10.241, P <0.05; $\beta_{Time\ Since\ Trt,\ Length}$ = 114.721, P <0.05; Table 11, Figure 6). These combined results illustrate that juvenile *O. mykiss* do not increase in abundance after restoration, but their populations are increasingly composed of larger individuals in the years after restoration. This suggests that restoration has been effective at improving freshwater habitat for juvenile *O. mykiss*, because size is positively correlated with survival at every life-stage among salmonids (Ward et al. 1989, Henderson and Cass 1991, Mortensen et al. 2000, Beamish and Mahnken 2001, Quinn 2005). However, it should be noted that it is impossible to distinguish between anadromous and resident individuals from the available data. It is the anadromous expression of most *O. mykiss* populations (i.e., steelhead) that are of primary concern for conservation, but anadromous and resident forms often overlap and interbreed (Christie et al. 2011; Courter et al. 2013; Sloat and Reeves 2014). Enormous effort has gone into trying to determine the influences of genetics, environmental factors, and individual condition on anadromy and residency (*see* Kendall et al. 2015), but much uncertainty remains regarding the underlying patterns and processes. Therefore, it is difficult to speculate to what degree the habitat improvements, and corresponding increase in juvenile *O. mykiss* biomass, improve the production of steelhead, in particular.

Conclusions

LWD placement projects improve freshwater rearing habitat by increasing habitat diversity, pool area, and pool depth. Habitat diversity and pool area improve immediately, while average pool depth initially declines then increases over time, taking approximately three years to exhibit a net increase. Such habitat enhancements are among the desired outcomes for restoration managers targeting salmon production, because deep pools and a diversity of habitat units are known to be vital components of freshwater salmon rearing habitat (Beechie et al. 2005). The response of salmonid species-size class diversity to LWD placement closely follows the response of average residual pool depth, exhibiting an initial decline followed by improvement over time, resulting in a net increase after 4-5 years. This apparent relationship between pool depth and salmonid fish diversity presents further evidence of the significance of deep pool habitat for maintaining healthy salmon communities.

Given our knowledge of what constitutes essential salmon rearing habitat, I would expect improvements in production of coho salmon, Chinook salmon, and *O. mykiss* at LWD placement sites. However, the response of each species was far from uniform. Despite their extended freshwater residency, juvenile coho salmon appear to exhibit very little response to LWD placement. Coho biomass is unaffected, and while coho abundance improves over time, it does so at such a slow rate it cannot be considered biologically significant. My results suggest that coho biomass and abundance are more strongly influenced by climatic factors, such as summer air temperature and snow-pack at the start of spring, and that the coho populations may already be close to their carrying capacities based on available food sources. Likewise, coastal Chinook production is also largely unresponsive to LWD placement, though I presume this is because these population likely have short freshwater residence times, and thus freshwater habitat quality may be less important than ocean conditions. Thus, the habitat improvements attained by LWD placement do not seem to address the limiting factors for juvenile coho or coastal Chinook production.

In contrast, *O. mykiss* and inland Chinook populations respond positively to LWD placement, but with notable caveats. The average size of juvenile *O. mykiss* increases over time after restoration, but I cannot determine to what extent this benefits anadromous as opposed to resident individuals. Inland Chinook exhibit a sustained increase in abundance after LWD placement, suggesting that the habitat improvements do address the limiting factor for juvenile survival. But the average size of juvenile Chinook at inland treatment study reaches declines steadily over time. This suggests that overcrowding at LWD placement sites is resulting in density-dependent limits to growth, likely due to a dearth of high-quality habitat. Given the established positive correlation between size and survival, this finding cannot be ignored. An increase in the spatial extent of freshwater restoration at inland locations may be essential for improving inland Chinook production.

My analyses demonstrate that LWD placement is effective at improving freshwater habitat, but with all things considered, these improvements are not generating consistent increases in juvenile salmon production. This suggests that LWD placement does not always address the limiting factors for salmon production. It is clear that limiting factors vary by species and location, and large-scale threats, such as declining ocean conditions, climate change, and restricted access to freshwater habitat from dams and culverts, undoubtedly impact salmon production in ways that LWD placement alone cannot solve. Moreover, it is possible that competitors and predators of juvenile salmon, including some invasive species, may benefit to an equal or greater extent from the habitat improvements of LWD placement, to a degree that it limits the net gains salmon production. Given the clear habitat improvements and promising responses observed with *O. mykiss* biomass and inland Chinook abundance, LWD placement projects should continue to play a large role in salmon restoration, but should not be relied on as the only action. A broader, more comprehensive strategy for restoration and conservation, which addresses a wider range of limiting factors, is essential in order to ensure the long-term survival of sensitive salmon populations.

Furthermore, my results highlight the critical need for baseline and long-term monitoring of restoration projects. My key findings would not have been possible without either. It can take three to five years to fully realize some restoration benefits, including increases in average pool depth and salmonid fish diversity. Likewise, long-term monitoring was essential in revealing the biomass declines of inland Chinook at restoration sites. Without pre-restoration and control data, it would have been nearly impossible to confidently identify trends in salmon response to restoration, due to naturally high variability in the populations and seasonal fluctuations of influential climatic factors. Restoration effectiveness cannot progress without such detailed evaluations. Funding agencies must fund, and restoration managers must carry out, comprehensive long-term monitoring if restoration practitioners are going to learn which restoration techniques are most effective. This information is essential for guiding future restoration efforts and supporting salmon recovery.

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Tables and Figures

Table 1: Summary of actions at Large Woody Debris (LWD) placement projects, monitored under protocol of WA's Salmon Recovery Funding Board (SRFB; WSRCO 2018) & Bonneville Power Administration's Columbia Habitat Monitoring Program (CHaMP; PNWAMP 2015, Bennet et al. 2015, Martin and Buelow 2017). Map Code corresponds to project location, shown in Figures 1-2. Project ID is determined by monitoring programs. *Trt*= Treatment study reach, which received LWD placement. *Ctrl* = Control study reach.

Map Code	***Watershed***	***Monitoring Program***	***Project ID***	***Yr of Trt***	***Km Treated***	***# of LWD Struct[1]***	***LWD Struct per km***	***Additional Treatment***
LSV	Little Skookum Valley	SRFB	02-1444	2005	0.48	9	18.75	Streambank Stabilization; Riparian Planting; Non-native plant removal/control
SCWB	Salmon Crk/ Willapa Bay	SRFB	02-1463	2004	2.74	80	29.20	Channel Reconfiguration / Levee Removal
ChC	Chico Creek	SRFB	04-1209IS	2005	0.16	11	68.75	Streambank Stab.; Riparian Planting; Invasive Plant Removal/Control; Channel Reconfig./ Levee Removal
Lnew	Lower Newaukum	SRFB	04-1338	2008	0.24	6	25.00	Riparian Planting; Invasive Plant Removal/ Control; Channel Reconfig./ Levee Removal
Lcol	Lower Columbia	SRFB	04-1448	2005	0.32	9	28.13	Riparian Planting
Uwash	Upper Washougal	SRFB	04-1575	2005	0.8	15	18.75	Streambank Stab; Spawning Gravel; Riparian Planting
Dung	Dungeness	SRFB	04-1589	2005	1.29	7	5.43	N/A
CCL	Cedar Crk / Lewis	SRFB	05-1533	2007	0.42	20	47.62	Riparian Planting; Channel Reconfig. / Levee Removal
SkNook	Skookum / Nooksack	SRFB	07-1803	2009	0.9	3	15.79	Streambank Stabilization; Channel Reconfiguration/ Levee Removal
UTCMC	Upper Trout Crk/ Middle Col	SRFB	02-1515	2005	12.07	44	3.65	Road Abandonment; Upland Vegetation Management
PA3	Tucannon River	CHaMP	PA-3	2014	2.19	48	21.92	N/A
PA14	Tucannon River	CHaMP	PA-14	2014	2.64	88	33.33	Channel Reconfiguration / Levee Removal
PA24	Tucannon River	CHaMP	PA-24	2015	1.59	61	38.36	Channel Reconfiguration / Levee Removal
ACCC	Asotin Creek-Charley Creek	CHaMP	CC-F2 P1BR (Ctrl) CC- F5 P1BR (Ctrl) CC-F3 P1BR (Trt) CC-F3 P2BR (Trt) CC-F4 P2BR (Trt) CC-F4 P3BR (Trt)	2013	4	177	44.25	Riparian Planting; Invasive Plant Removal/Control; Cattle Exclusion; *Supplemental LWD Placement in 2016*
ACNF	Asotin Creek-North Fork	CHaMP	NF-F4 P1BR (Ctrl) NF-F6 P2BR (Ctrl) NF-F1 P1BR (Trt) NF-F1 P2BR (Trt) NF-F2 P1 (Trt) NF-F2 P2 (Trt)	2014	4	121	30.25	Cattle Exclusion; *Supplemental LWD Placement in 2016*
ACSF	Asotin Creek-South Fork	CHaMP	SF-F5 P3BR (Ctrl) SF-F2 P2BR (Ctrl) SF-F3 P2BR (Trt) SF-F3 P3BR (Trt) SF-F4 P1 (Trt) SF-F4 P2 (Trt)	2012	4	146	36.50	Cattle Exclusion; *Supplemental LWD Placement in 2016*

[1] The size of each LWD structure was proportional to the size of the stream in which they were placed.

Table 2: Monitoring schedule and study reach details for LWD restoration projects, monitored under protocol of WA's Salmon Recovery Funding Board (SRFB; WSRCO 2014, 2018) & Bonneville Power Administration's Columbia Habitat Monitoring Program (CHaMP; PNWAMP 2015, Bennet et al. 2015, Martin and Buelow 2017). Map Code corresponds to project location, shown in Figures 1-2. *Trt*= Treatment study reach, which received LWD placement. *Ctrl* = Control study reach, which receives no restoration for the duration of monitoring. Year 0 is the year of LWD placement. If Year 0 is listed, sampling occurred prior to LWD placement. Negative sampling years occurred before restoration; positive years occurred after restoration. Min/max wetted widths refer to the average wetted widths of each study reach during a given monitoring event. Target species is the intended beneficiary of restoration actions, based on what is listed in the planning documents for each restoration project.

Map Code	***Project ID***	***Year 0***	***Habitat Monitoring Yrs [1]***	***Snorkel Survey Yrs [1]***	***Study Reach Length[2] (m)***		***Min/ Max Wetted Width (m)***		***Target Species***
					[Trt]	**[Ctrl]**	**[Trt]**	**[Ctrl]**	
LSV	02-1444	2005	-1, 1, 3, 5, 10	-1, 1, 3, 5	150	90/150	1.18 / 1.50	1.37 / 1.59	Coho
SCWB	02-1463	2004	0, 1, 3, 5, 10	0, 1, 3, 5, 10	180	180	4.52 / 10.43	2.90 / 3.93	Coho
ChC	04-1209IS	2005	0 1, 4, 6, 8	0 1, 4, 6, 8	250	250	5.36 / 8.6	4.97 / 7.06	Chum
Lnew	04-1338	2008	0, 2, 4	0, 2, 4	200	200	5.93 / 9.19	8.37 / 9.70	Chinook
Lcol	04-1448	2005	0, 1, 3, 5, 10	0, 1, 3, 5, 10	320	320	23.67 / 42.05	25.36 / 29.37	Chum
Uwash	04-1575	2005	0, 1, 3, 5, 10	0, 1, 3, 5, 10	500	500	20.42 / 25.87	14.92 / 21.31	Steelhead
Dung	04-1589	2005	0, 1, 3, 5, 7	0, 1, 3, 5, 7	500	500	14.71 / 24.08	15.67 / 19.64	Chinook
CCL	05-1533	2007	-1, 1, 3, 5	-1, 1, 3, 5	300	165/300	13.42 / 15.23	11.90 / 13.45	Chinook
SkNook	07-1803	2009	-1, 1, 3, 5	-1, 1, 3, 5	500	500	28.90 / 30.92	28.28 / 32.84	Chinook
UTCMC	02-1515	2005	-1, 1, 3, 5, 7	-1, 1, 3, 5, 7	150/360	150	4.07 / 12.27	4.07 / 4.87	Steelhead
PA3	PA-3	2014	-3, -2, -1, 0, 1, 2, 3	0, 1, 2, 4	275	295	11.23 / 16.51	8.34/ 9.88	Chin. / SH
PA14	PA-14	2014	-2, -1, 0, 1, 2, 3	0, 1, 2, 3, 4	245	265	9.70 / 10.72	9.55 / 10.45	Chin. / SH
PA24	PA-24	2015	-4, -3, -2, -1, 0, 1, 2	-1, 0, 1, 3	260	275	13.17 / 19.79	10.50 / 11.68	Chin. / SH
ACCC	CC-F2 P1BR (Ctrl)	2013	-2, -1, 0, 1, 2, 3, 4	N/A	165	165	2.78 / 5.08	-	Chinook / Steelhead
	CC- F5 P1BR (Ctrl)		-2, -1, 0, 1, 4		170	170	3.18 / 4.02	-	
	CC-F3 P1BR (Trt)		0, 1, 2, 3, 4		160	160	-	3.18 / 4.43	
	CC-F3 P2BR (Trt)		-1, 0, 1, 2, 4		155	155	-	3.83 / 4.33	
	CC-F4 P2BR (Trt)		0, 1, 3, 4		155	155	-	3.73 / 4.06	
	CC-F4 P3BR (Trt)		0, 1, 2, 3, 4		150	150	-	3.25 / 3.89	
ACNF	NF-F4 P1BR (Ctrl)	2014	-3, -2, -1, 0, 1, 2, 3	N/A	210	210	5.50 / 6.45	-	Chinook / Steelhead
	NF-F6 P2BR (Ctrl)		-3, -2, -1, 0, 1, 2, 3		200	200	6.92 / 9.09	-	
	NF-F1 P1BR (Trt)		-2, -1, 1, 2, 3		205	205	-	5.85 / 6.79	
	NF-F1 P2BR (Trt)		-3, -2, -1, 0, 1, 2, 3		210	210	-	7.62 / 8.51	
	NF-F2 P1 (Trt)		-2, -1, 0, 1, 2, 3		195	195	-	8.17 / 9.03	
	NF-F2 P2 (Trt)		-2, -1, 0, 1, 2, 3		210	210	-	7.85 / 9.49	
ACSF	SF-F5 P3BR (Ctrl)	2012	-1, 0, 1, 2, 3, 4, 5	N/A	175	175	4.76 / 6.13	-	Chinook / Steelhead
	SF-F2 P2BR (Ctrl)		-1, 0, 1, 2, 3, 4		180	180	3.72 / 4.38	-	
	SF-F3 P2BR (Trt)		-1, 0, 1, 3, 4, 5		170	170	-	3.10 / 3.70	
	SF-F3 P3BR (Trt)		0, 1, 2, 3, 4, 5		180	180	-	3.57 / 4.19	
	SF-F4 P1 (Trt)		0, 1, 2, 3, 4, 5		165	165	-	3.59 / 4.03	
	SF-F4 P2 (Trt)		0, 1, 2, 3, 5		160	160	-	4.08 / 4.63	

[1] Monitoring may have occurred at more time points than is listed. The monitoring years listed represent the data that were publicly available at the time of analysis.

[2] Study reach lengths for are the average length from all monitoring years, rounded to the nearest 5 m. If more than one length value is listed, the length changed over time.

Table 3: Parameters and adjusted R^2 values for predictive weight (g) equation for each salmon species (*Oncorhynchus* spp.). $W = aL^b$, in which W is the fish weight (g), L is the estimated fork length (mm) of each fish (Crec'hriou et al. 2015). Parameters a and b were generated by fitting weight and length data from the PIT Tag Information System database (PSMFC 2019) in a least squares regression, following the format $\log(Weight) \sim a + b\log(Fork\ Length)$.

Species	*a*	*b*	Adj. R^2
Chinook	9.48 E-06	3.03	0.979
Coho	1.72 E-05	2.91	0.963
O. mykiss	1.60 E-05	2.91	0.986

Table 4: Model design for testing habitat and fish response variables using linear mixed models. Fixed effects of *Time*, *Treatment*, and *Time Since Treatment* capture discontinuous change in both intercept and slope, as a result of restoration actions (Singer and Willett 2003). Random effects for intercept and slope by *Project* allow the paired *Treatment* and *Control Study Reaches* to be nested within *Project*. Final models were fitted using Restricted Maximum Likelihood (REML) estimation.

	RV	$\sim \beta_0$	$+$ *Time* x β_1	$+$ *Treatment* x β_2	$+$ *Time Since Treatment* x β_3	$+ \mu_{\text{intercept/slope}}$	$+ \epsilon_{\text{residuals}}$
Definition	Response Variable	Intercept	Slope	Has restoration occurred at this study reach? 0 = No; 1 = Yes	Interaction of *Time* and *Treatment*	*Random Effects* intercept and slope for each *Project*	Residuals
Purpose		Value at *Time* = 0.	Slope that applies to all study reaches in all projects.	Allows intercept at treatment study reaches to vary from baseline intercept.	Allows slope at treatment study reaches to vary from baseline slope after restoration occurs.	Allows the intercept and slope for each *Project* to vary from the baseline intercept and slope.	
Interp.		*Time = 0* for the year of restoration at each *Project*	Accounts for background rate of change, regardless of restoration status.	Accounts for immediate, or static, changes in Response Variable due to restoration.	Accounts for changes in Response Variable over time due to restoration.	Nests the treatment and control study reaches within their *Project*	

Table 5: Covariate, Weighted Variance, and Random Effects details for the habitat response variables that were tested using linear mixed effects models. *Habitat Diversity* is based on Shannon's diversity index; *Pool : Reach Ratio* is the summed pool area (m^2) divided by the study reach area (m^2); *Mean RPD* is the average residual pool depth (m). Covariates were added to models when necessary to achieve normality, and were selected based on a combination of visual inspection of residuals and fit statistics. Weighted Variance refers to the identifying factor whose residual variance was allowed to vary, using the "weights" statement in the "nlme" package in R (Pinheiro et al. 2019).

Response Variable	Covariates	Weighted Variance	Random Effects	Projects Included in Model
Habitat Diversity	Stream Width, Annual Regional Air Temp, Annual Regional Precip, Stream Width * Annual Regional Precip	Calendar Year	Intercept by *Project*	All (n = 16)
Pool : Reach Ratio	N / A	Calendar Year	Intercept by *Project* nested in *Monitoring Program*	All (n = 16)
Mean RPD	Stream Width, (Stream Width)^2, Annual Regional Air Temp, Bank Stabilization Treatment	Project	Intercept by *Project* nested in *Monitoring Program*	All (n = 16)

Table 6: Covariate, Weighted Variance, and Random Effects details for the fish response variables that were tested using linear mixed effects models. Explanatory Covariates were added to models when necessary to achieve normality of residuals, based on a combination of visual inspection and fit statistics. Weighted Variance refers to the identifying factor whose residual variance was allowed to vary, using the "weights" statement in the "nlme" package in R (Pinheiro et al. 2019).

Species	Response Variable	Units	Explanatory Covariates	Weighted Variance	Random Effects	Projects Included in Model [1]
Salmonid Community	**Species-Size Class Diversity**	Shannon's Diversity Index	Snorkel Date, SWE, Summer Precipitation	Monitoring Program	Intercept & Slope by *Project*	CCL, ChC, Dung, Lcol, Lnew, LSV, PA14, PA24, PA3, SCWB, SkNook, UTCMC, Uwash (*n* = 13)
Coho	***Fish*** $_{\textit{Area}}$	fish / 100 m^2	Summer Air Temp, Snorkel Date, SWE, Stream Width, Dist. from Ocean (inverse)	Project	Intercept & Slope by *Project*	CCL, ChC, Dung, Lcol, Lnew, LSV, SCWB, SkNook (*n* = 8)
	Biomass $_{\textit{Area}}$	g / 100 m^2	**Coho Density (fish / 100 m^2)**, Summer Air Temp, Snorkel Date, SWE, Snorkel Date * Summer Air Temp			
	Fish $_{\textit{Length}}$	fish / 100 m	Summer Air Temp, Snorkel Date, SWE, Stream Temp during Survey, Dist. from Ocean (inverse)			
	Biomass $_{\textit{Length}}$	g / 100 m	**Coho Density (fish / 100 m)**, Summer Air Temp, Snorkel Date, SWE, Snorkel Date * Summer Air Temp			
Chinook	***Fish*** $_{\textit{Area}}$	fish / 100 m^2	**Inland * *Fixed Effects***, Snorkel Date, Summer Air Temp	Calendar Year	Intercept & Slope by *Project*	CCL, Dung, Lcol, Lnew, SCWB, SkNook, PA14, PA24, PA3 (*n* = 9)
	Biomass $_{\textit{Area}}$	g / 100 m^2	**Inland * *Fixed Effects***, **Chinook Density (fish / 100 m^2)**, Snorkel Date, Summer Air Temp, Inland * Snorkel Date	Project		
	Fish $_{\textit{Length}}$	fish / 100 m	**Inland * *Fixed Effects***, Snorkel Date, Summer Air Temp	Calendar Year		
	Biomass $_{\textit{Length}}$	g / 100 m	**Inland * *Fixed Effects***, **Chinook Density (fish / 100 m)**, Snorkel Date, Summer Air Temp, Inland * Snorkel Date	Project		
O. mykiss	***Fish*** $_{\textit{Area}}$	fish / 100 m^2	Snorkel Date, Dams, SWE	Major River	Intercept & Slope by *Project*	CCL, ChC, Dung, Lcol, Lnew, LSV, PA14, PA24, PA3, SCWB, SkNook, UTCMC, Uwash (*n* = 13)
	Biomass $_{\textit{Area}}$	g / 100 m^2	***O. mykiss* Density (fish / 100 m^2)**, Snorkel Date, Dams, SWE			
	Fish $_{\textit{Length}}$	fish / 100 m	Snorkel Date, Dams, SWE, Stream Width (inverse)			
	Biomass $_{\textit{Length}}$	g / 100 m	***O. mykiss* Density (fish / 100 m)**, Snorkel Date, Dams, SWE			

[1] Project codes: LSV = Little Skookum Valley, SCWB = Salmon Creek / Willapa Bay, ChC = Chico Creek, Lnew = Lower Newaukum, Lcol = Lower Columbia, Uwash = Upper Washougal, Dung = Dungeness, CCL = Cedar Creek / Lewis R., SkNook = Skookum / Nooksack, UTCMC = Upper Trout Creek / Middle Columbia R., PA3/PA14/PA24= Tucannon River Project Areas.

Table 7: Covariates used in one or more linear mixed model(s) used to evaluate physical and biological responses to large woody debris restoration projects. Time-varying covariates vary between monitoring events, time-invariant covariates do not change. Level refers to the subject level at which the covariate varies. *Study Reach* is nested within *Project*, and individual monitoring years for *Projects* are nested within *Calendar Year*. Sources are listed for covariates whose values could not be obtained from the monitoring data or the project planning documents (Bennet et al. 2015, Martin and Buelow 2017, WSRCO 2018).

Covariate	Type	Level	Definition	Source
Annual Regional Air Temperature (depart. from normal)	Time-Varying	*Project*	Average of the regional monthly departures from normal air temperature (°C) for the water year (October -September) of the year of the survey. Regional values for each *Project*, obtained from the nearest NOAA weather station (Appendix 1).	National Centers for Environmental Information (NOAA 2019)
Annual Regional Precipitation (depart. from normal)	Time-Varying	*Project*	Sum of annual water year (October-September) monthly departures from normal precipitation (in) of the year of the survey. Values for each *Project* obtained from the nearest NOAA weather station (Appendix 1).	National Centers for Environmental Information (NOAA 2019)
Bank Stabilization Treatment (binary)	Time-Invariant	*Project*	1 = Bank stabilization was included in the treatment for this *Project*, 0 = Bank stabilization was not included in treatment for this *Project*. [Note: The value applies to both *Treatment/Control Study Reaches* within a *Project*.]	
Dams (binary)	Time-Invariant	*Project*	1= There are 1 or more dams on pathway from ocean to project site, 0= No dams on pathway from ocean to project site	
Distance from Ocean (km)	Time-Invariant	*Project*	Total river/stream distance (km) between the ocean and the Project site	National Hydrography Dataset (USGS 2019)
Habitat Date	Time-Varying	*Study Reach*	Julian calendar date of the habitat survey.	
Inland (binary)	Time-Invariant	*Project*	1= Inland project site (>500 km from ocean), 0= Coastal project site (<200 km from ocean) [Note: For data in Chinook salmon analysis, this is indistinguishable from *Monitoring Program*.]	National Hydrography Dataset (USGS 2019)
Major River (binary)	Time-Invariant	*Project*	1= The project site is on a tributary of a major river (i.e. the Columbia River), 0= The project site is not on the tributary of a major river	
Monitoring Program	Time-Invariant	*Project*	Indicates which monitoring program was responsible for the date collection for each *Project*: Washington State Salmon Recovery Funding Board (SRFB), or Bonneville Power Administration's Columbia Habitat Monitoring Program (CHaMP)	
Snorkel Date (standardized)	Time-Varying	*Study Reach*	Based on Julian calendar date of the snorkel survey. If snorkel survey date was missing, the corresponding habitat survey date was used instead. Julian dates were centered and scaled.	
Snow Water Equiv. (% of normal for April 1)	Time-Varying	*Calendar Year*	Percent of regional normal snow water equivalent for April 1 of the survey year. Normal values based on average of NRCS 1981-2010 values for Pacific Northwest 2-Digit HUC region.	National Water and Climate Center (USDA 2019)
Stream Temp. (°C) at Start of Survey	Time-Varying	*Study Reach*	Temperature (°C) at the start of snorkel survey, centered on the population mean value. [Note: This value is missing from ~15% of site visits, primarily affecting Tucannon River Project Sites]	
Stream Width (m)	Time-Varying	*Study Reach*	Average wetted width (m) of the Study Reach, derived from the habitat survey for a given survey year.	
Summer Air Temp (depart. from normal)	Time-Varying	*Project*	Average of regional monthly departures from normal air temperature (°C) for the months of April-September from the survey year. Regional values for each Project, obtained from the nearest NOAA weather station (Appendix 1).	National Centers for Environmental Information (NOAA 2019)

Table 8: Results for linear mixed models testing physical habitat and salmonid community response to large woody debris placement projects (Tables 4-6). Each restoration project followed a multiple before-after, control-impact study design (Stewart-Oaten et al. 1986; Tables 1-2). Asterisks indicate statistical significance [* *P* <0.05; ** *P* <0.01; *** *P* <0.001].

Response Variable	Fixed Effect	β Est.	P-Value		Covariates	β Est.	P-Value	
Habitat Diversity	*intercept*	1.067	< 0.001	***	*Stream Width (m)*	*0.001*	*0.862*	
	Time	**0.037**	**< 0.001**	***	*Annual Precip.*	*0.388*	*< 0.001*	***
	Treatment	**0.155**	**0.003**	**	*Annual Air Temp*	*- 0.046*	*< 0.001*	***
	Time Since Trt	**- 0.031**	**0.044**	*	*Stream Width * Annual Precip*	*- 0.020*	*0.007*	**
Pool : Reach Ratio	*intercept*	0.235	0.002	**	-			
	Time	0.003	0.471					
	Treatment	**0.059**	**0.013**	*				
	Time Since Trt	0.002	0.762					
Mean RPD	*intercept*	0.142	0.079		*Stream Width (m)*	*0.045*	*< 0.001*	***
	Time	- 0.005	0.227		*(Stream Width)^2*	*-0.001*	*< 0.001*	***
	Treatment	**- 0.077**	**< 0.001**	***	*Annual Air Temp*	*- 0.010*	*0.034*	*
	Time Since Trt	**0.044**	**< 0.001**	***	*Bank Stabilization*	*- 0.083*	*0.022*	*
Species-Size Class Diversity	*intercept*	1.059	0.000	***	*Snorkel Date*	*0.077*	*0.042*	*
	Time	0.005	0.773		*SWE (April 1)*	*0.173*	*0.051*	
	Treatment	**- 0.225**	**0.004**	**	*Summer Precip.*	*0.259*	*< 0.001*	***
	Time Since Trt	**0.051**	**0.014**	*				

Table 9: Results for linear mixed models testing juvenile coho salmon (*Oncorhynchus kisutch*) response to large woody debris placement projects (Tables 4, 6, 7). Each restoration project followed a multiple before-after, control-impact study design (Stewart-Oaten et al. 1986; Tables 1-2). Asterisks indicate statistical significance [***** P <0.05; ****** P <0.01; ******* P <0.001].

COHO	Fixed Effects	β Est.	P-Value		Covariates	β Est.	P-Value	
Fish Area	*intercept*	8.654	0.009	**	*Summer Air Temp*	0.399	< 0.001	***
	Time	**- 0.288**	**< 0.001**	***	*SWE (April 1)*	1.415	< 0.001	***
	Treatment	**- 0.253**	**< 0.001**	***	*Snorkel Date*	- 0.726	0.001	**
	Time Since Trt	**0.274**	**< 0.001**	***	*Stream Width (m)*	- 0.224	< 0.001	***
					Dist. from Ocean (inverse)	21.32	0.125	
Biomass Area	*intercept*	7.06	0.001	**	*Summer Air Temp*	- 1.784	0.002	**
	Time	- 0.201	0.909		*SWE (April 1)*	- 5.575	0.006	**
	Treatment	- 0.032	0.973		*Snorkel Date*	2.582	< 0.001	***
	Time Since Trt	- 0.032	0.846		*Coho Density (fish / 100 m^2)*	3.603	< 0.001	***
					*Snork. Date * Summer Air Temp*	2.599	0.004	**
Fish Length	*intercept*	45.279	0.237		*Summer Air Temp*	6.205	< 0.001	***
	Time	**1.454**	**0.007**	**	*SWE (April 1)*	18.889	0.096	
	Treatment	-0.613	0.235		*Snorkel Date*	- 8.782	0.009	**
	Time Since Trt	**0.022**	**0.040**	*	*Stream Temp during Survey*	-0.864	< 0.001	***
					Dist. from Ocean (inverse)	87.434	0.474	
Biomass Length	*intercept*	79.667	0.018	*	*Summer Air Temp*	- 26.793	0.022	*
	Time	4.012	0.847		*SWE (April 1)*	- 63.37	0.003	**
	Treatment	1.805	0.765		*Snorkel Date*	17.353	0.011	*
	Time Since Trt	0.295	0.868		*Coho Density (fish / 100 m)*	3.15	< 0.001	***
					*Snork. Date * Summer Air Temp*	26.948	0.036	*

Table 10: Results for linear mixed models testing juvenile Chinook salmon (*Oncorhynchus tshawytscha*) response to LWD placement restoration projects (Tables 4, 6, 7). Each restoration project followed a multiple before-after, control-impact study design (Stewart-Oaten et al. 1986; Tables 1-2). Asterisks indicate statistical significance [* P <0.05; ** P <0.01; *** P <0.001].

CHINOOK	Fixed Effects	β Est.	P-Value		Inland * Fixed Effects	β Est.	P-Value		Covariates	β Est.	P-Value	
Fish Area	*intercept*	0.083	0.857		*Inland*	2.386	0.031	*	*Snorkel Date*	*0.040*	*0.340*	
	Time	0.176	0.486		Time * Inland	0.029	0.951		*Summer Air Temp*	*0.093*	*0.005*	**
	Treatment	**0.038**	**<0.001**	***	**Treatment * Inland**	**3.191**	**0.014**	*				
	Time Since Trt	**-0.013**	**<0.001**	***	Time Since Trt * Inland	- 0.330	0.440					
Biomass Area	*intercept*	- 0.390	0.210		*Inland*	1.504	0.112		*Chinook Dens. (fish / 100 m^2)*	*4.532*	*<0.001*	***
	Time	0.274	0.139		Time * Inland	0.795	0.360		*Snorkel Date*	*0.118*	*0.423*	
	Treatment	0.766	0.075		**Treatment * Inland**	**4.390**	**0.012**	*	*Summer Air Temp*	*0.620*	*<0.001*	***
	Time Since Trt	**- 0.388**	**<0.001**	***	**Time Since Trt * Inland**	**- 3.016**	**0.019**	*	*Snorkel Date * Inland*	*4.116*	*<0.001*	*
Fish Length	*intercept*	3.333	0.369		*Inland*	23.306	0.016	*	*Snorkel Date*	*0.663*	*0.341*	
	Time	1.555	0.560		Time * Inland	- 0.838	0.865		*Summer Air Temp*	*0.714*	*0.157*	
	Treatment	- 0.028	0.735		**Treatment * Inland**	**35.585**	**0.004**	**				
	Time Since Trt	0.009	0.626		Time Since Trt * Inland	- 2.242	0.558					
Biomass Length	*intercept*	-7.324	0.087		*Inland*	20.288	0.149		*Chinook Dens. (fish / 100 m)*	*4.674*	*<0.001*	***
	Time	**4.799**	**0.044**	*	Time * Inland	10.618	0.431		*Snorkel Date*	*-0.895*	*0.742*	
	Treatment	6.456	0.424		**Treatment * Inland**	**78.167**	**0.021**	*	*Summer Air Temp*	*8.368*	*<0.001*	***
	Time Since Trt	- 2.706	0.114		**Time Since Trt * Inland**	**- 47.803**	**0.030**	*	*Snorkel Date * Inland*	*59.603*	*0.002*	**

Table 11: Results for linear mixed effects models testing juvenile *Oncorhynchus mykiss* (< 300 mm) response to large woody debris placement projects (Tables 4, 6, 7). Each restoration project followed a multiple before-after, control-impact study design (Stewart-Oaten et al. 1986; Tables 1-2). Asterisks indicate statistical significance [* *P* <0.05; ** *P* <0.01; *** *P* <0.001].

O. mykiss	*Fixed Effects*	*β Est.*	*P-Value*		*Covariates*	*β Est.*	*P-Value*	
Fish $_{Area}$	*intercept*	12.824	0.001	**	*SWE (April 1)*	*- 6.958*	*0.003*	**
	Time	0.866	0.123		*Dams*	*5.584*	*0.291*	
	Treatment	1.473	0.458		*Snorkel Date*	*0.904*	*0.257*	
	Time Since Trt	- 0.601	0.204					
Biomass $_{Area}$	*intercept*	6.927	0.686		*O. mykiss Dens. (fish / 100 m^2)*	*5.126*	*< 0.001*	***
	Time	- 1.785	0.589		*Dams*	*114.866*	*0.002*	**
	Treatment	- 25.512	0.220		*Snorkel Date*	*16.672*	*0.034*	*
	Time Since Trt	**10.241**	**0.040**	*				
Fish $_{Length}$	*intercept*	191.785	0.001	**	*SWE (April 1)*	*- 85.016*	*0.004*	**
	Time	10.092	0.086		*Dams*	*42.095*	*0.564*	
	Treatment	10.608	0.681		*Snorkel Date*	*19.420*	*0.056*	
	Time Since Trt	- 1.735	0.776		*Stream Width (inverse)*	*- 131.784*	*0.329*	
Biomass $_{Length}$	*intercept*	140.63	0.429		*O. mykiss Dens. (fish / 100 m)*	*4.908*	*< 0.001*	***
	Time	- 38.686	0.163		*Dams*	*1193.828*	*0.001*	**
	Treatment	- 296.909	0.126		*Snorkel Date*	*145.580*	*0.052*	
	Time Since Trt	**114.721**	**0.014**	*				

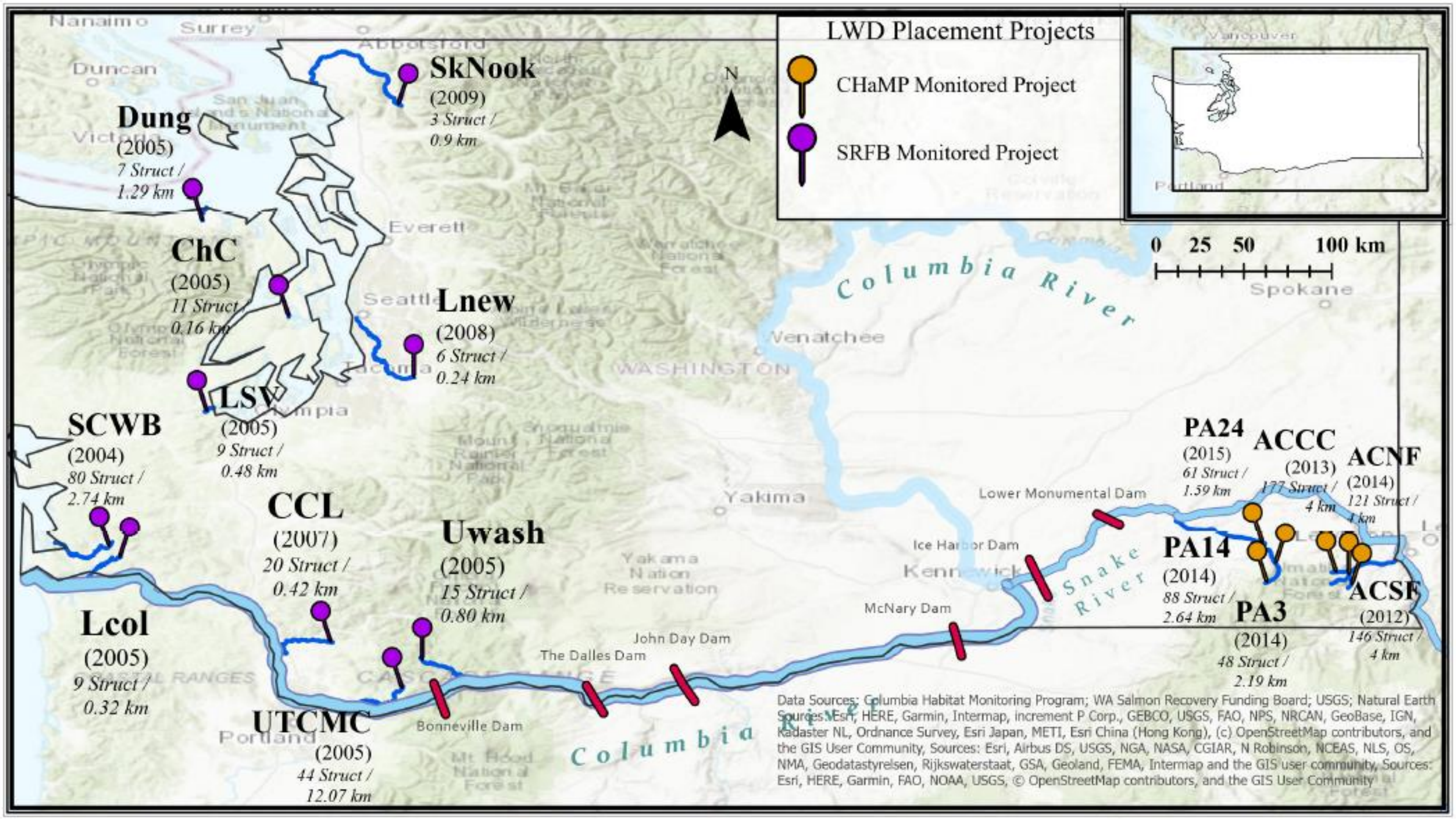


Figure 1: Map showing the approximate locations of large woody debris (LWD) restoration projects, monitored under protocol of Washington State's Salmon Recovery Funding Board (SRFB; WSRCO 2018) & Bonneville Power Administration's Columbia Habitat Monitoring Program (CHaMP; PNWAMP 2015 Bennet et al. 2015, Martin and Buelow 2017). Each point shows the project-specific Map Code (Tables 1-2), the year of restoration, and the number of LWD structures placed over the length of stream (km) receiving actions. LSV = Little Skookum Valley, SCWB = Salmon Creek / Willapa Bay, ChC = Chico Creek, Lnew = Lower Newaukum, Lcol = Lower Columbia, Uwash = Upper Washougal, Dung = Dungeness, CCL = Cedar Creek / Lewis R., SkNook = Skookum / Nooksack, UTCMC = Upper Trout Creek / Middle Columbia R., PA3/PA14/PA24= Tucannon River Project Areas, ACCC = Asotin Creek- Charley Creek, ACNF = Asotin Creek- North Fork, ACSF = Asotin Creek- South Fork.

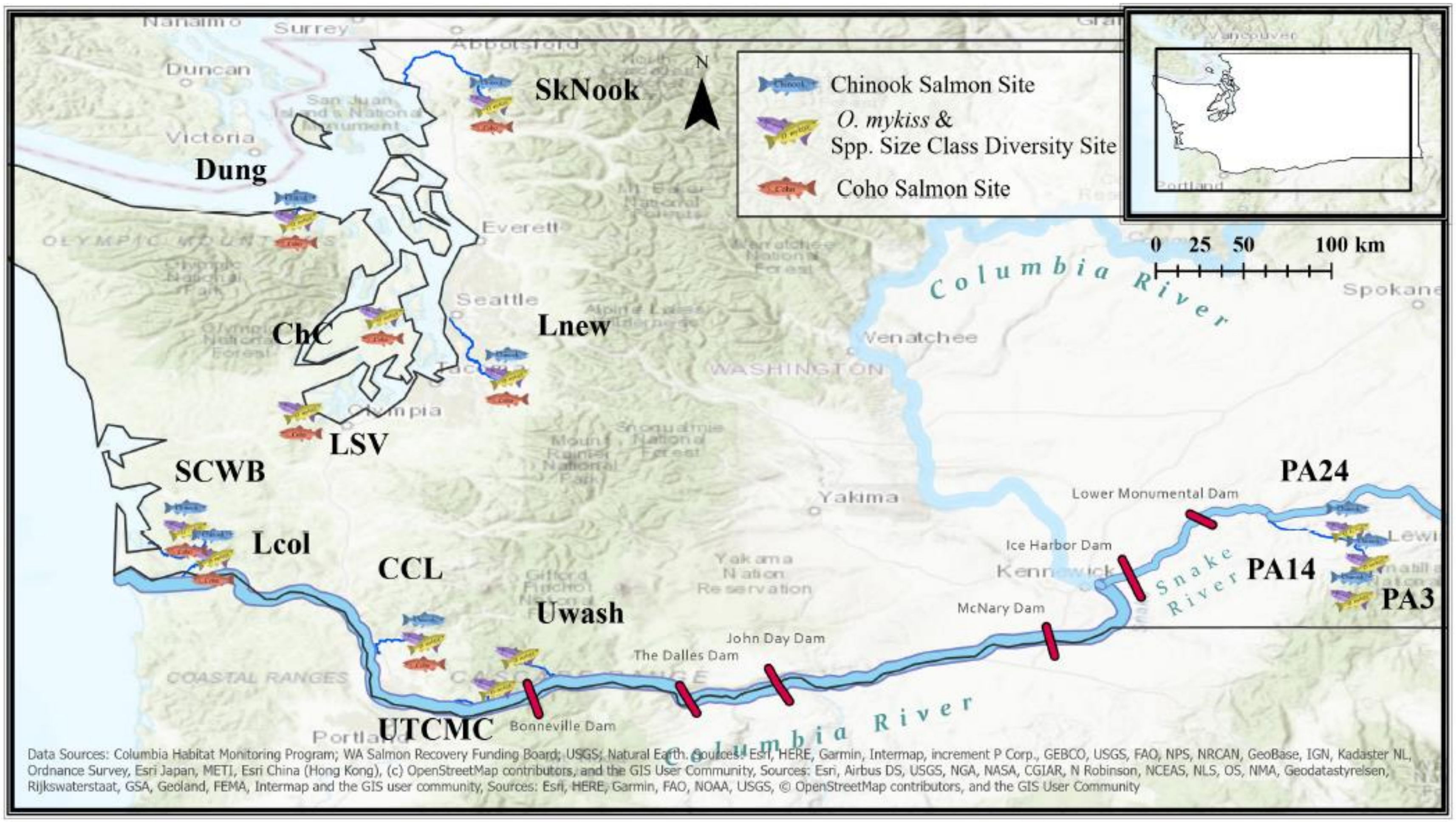


Figure 2: Map showing the approximate locations of large woody debris (LWD) restoration projects, monitored under protocol of Washington State's Salmon Recovery Funding Board (SRFB; WSRCO 2018) & Bonneville Power Administration's Columbia Habitat Monitoring Program (CHaMP; PNWAMP 2015, Martin and Buelow 2017). Each point shows the species analyses in which data from each project was used. Species analyses included: Coho (*Oncorhynchus kisutch*), Chinook (*O. tshawytscha*), *O. mykiss*, and Salmonid Species Size-Class Diversity (Table 6). LSV = Little Skookum Valley, SCWB = Salmon Creek / Willapa Bay, ChC = Chico Creek, Lnew = Lower Newaukum, Lcol = Lower Columbia, Uwash = Upper Washougal, Dung = Dungeness, CCL = Cedar Creek / Lewis R., SkNook = Skookum / Nooksack, UTCMC = Upper Trout Creek / Middle Columbia R., PA3/PA14/PA24= Tucannon River Project Areas.

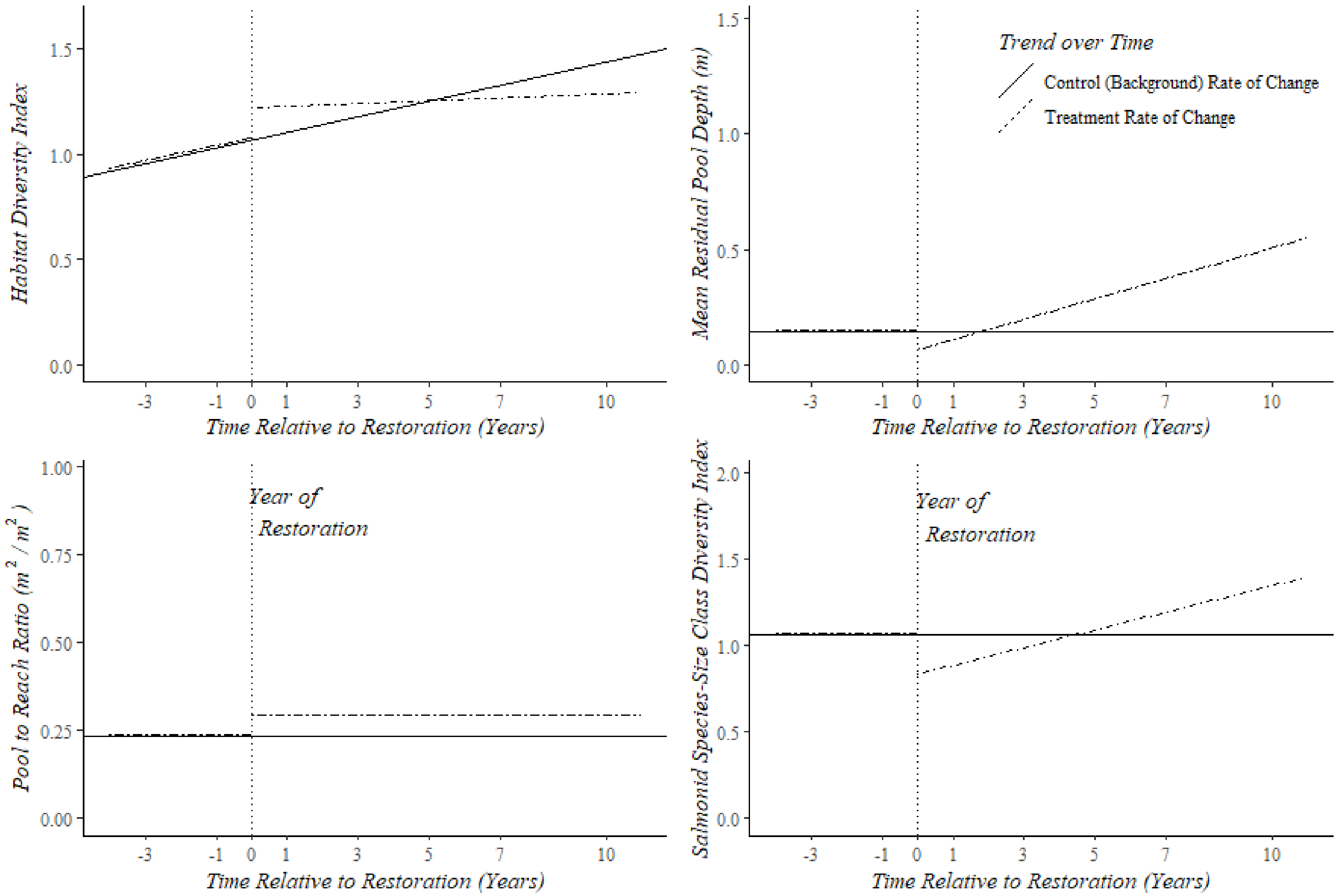


Figure 3: Plots displaying the results for the statistically significant (*p-value* < 0.05) fixed effects from a series of linear mixed models testing the immediate and long-term effects of large woody debris placement in stream restoration. From top left to bottom right: Habitat Diversity Index; Mean Residual Pool Depth (m); Pool to Reach Ratio (m^2 / m^2); and Salmonid Species-Size Class Diversity. See Table 8 for full model results.

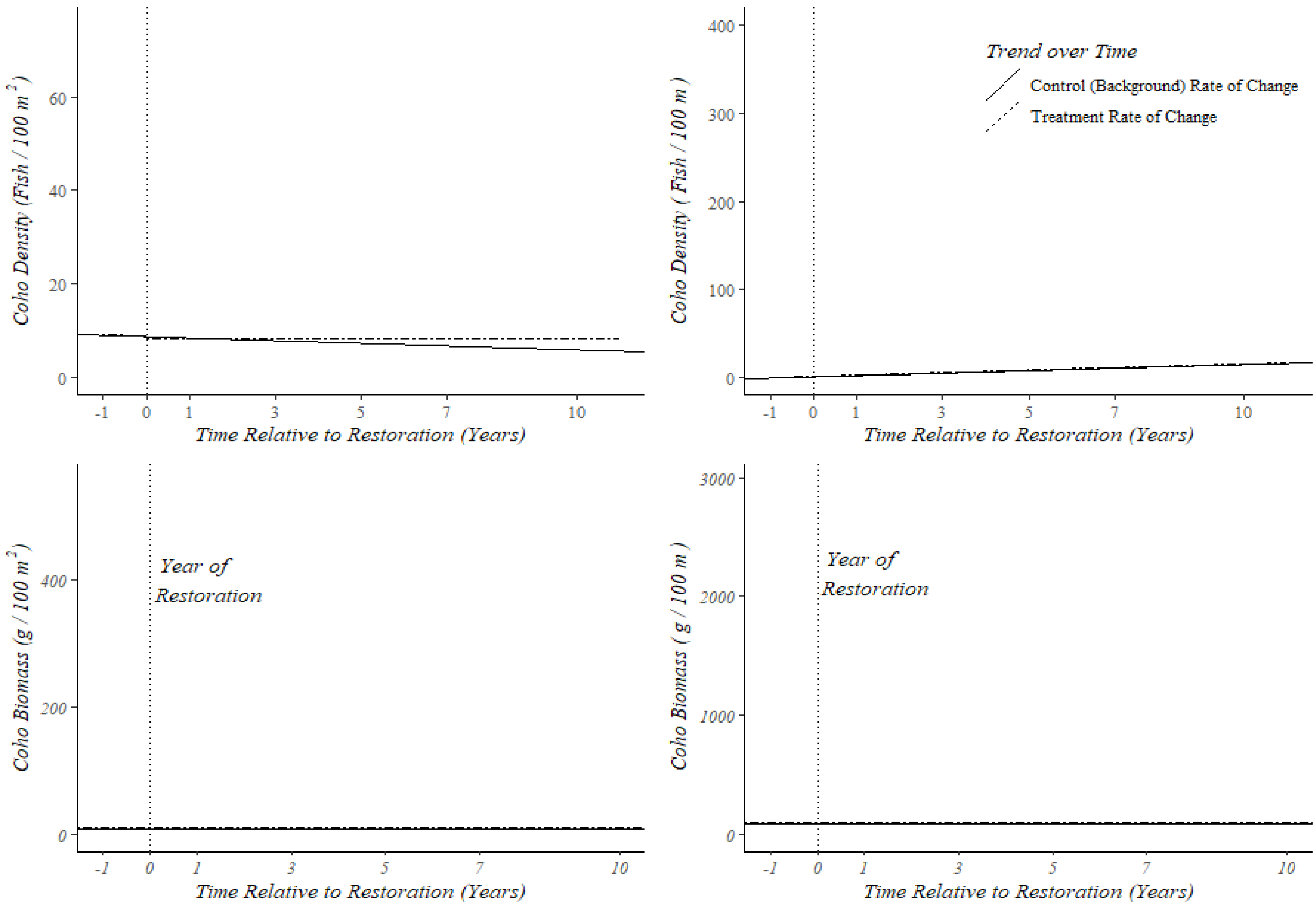
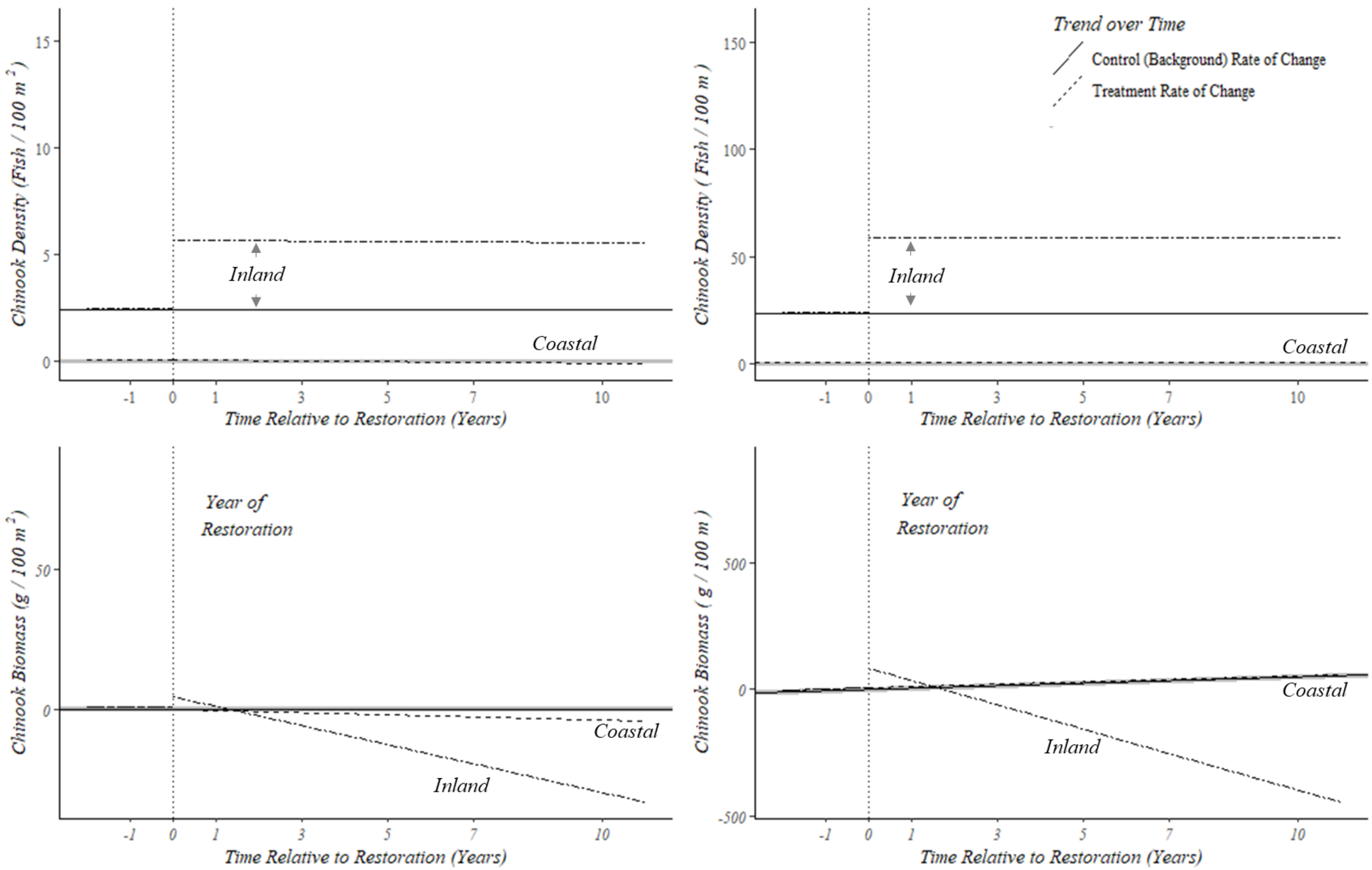


Figure 4: Plots displaying the results for the statistically significant (*p-value* < 0.05) fixed effects from a series of linear mixed models testing the immediate and long-term effects of large woody debris placement on juvenile coho salmon (*Oncorhynchus kisutch*). From top left to bottom right: juvenile coho density (fish / 100 m^2); juvenile coho density (fish / 100 m); juvenile coho biomass (g / 100 m^2); and juvenile coho biomass (g / 100 m). Models testing biomass include a covariate that controls for coho density, thus the results can be interpreted as the effects of large woody debris placement on the average weight of each fish. See Table 9 for full model results.



[Figure 5]: Plots displaying the results for the statistically significant (*p-value* < 0.05) fixed effects from a series of linear mixed models testing the immediate and long-term effects of large woody debris placement on juvenile Chinook salmon (*Oncorhynchus tshawytscha*) from both Inland (> 500 km from the ocean) and Coastal (< 200 km from the ocean) restoration locations. From top left to bottom right: juvenile Chinook density (fish / 100 m^2); juvenile Chinook density (fish / 100 m); juvenile Chinook biomass (g / 100 m^2); and juvenile Chinook biomass (g / 100 m). Models testing biomass include a covariate that controls for Chinook density, thus the results can be interpreted as the effects of large woody debris placement on the average weight of each fish. See Table 10 for full model results.

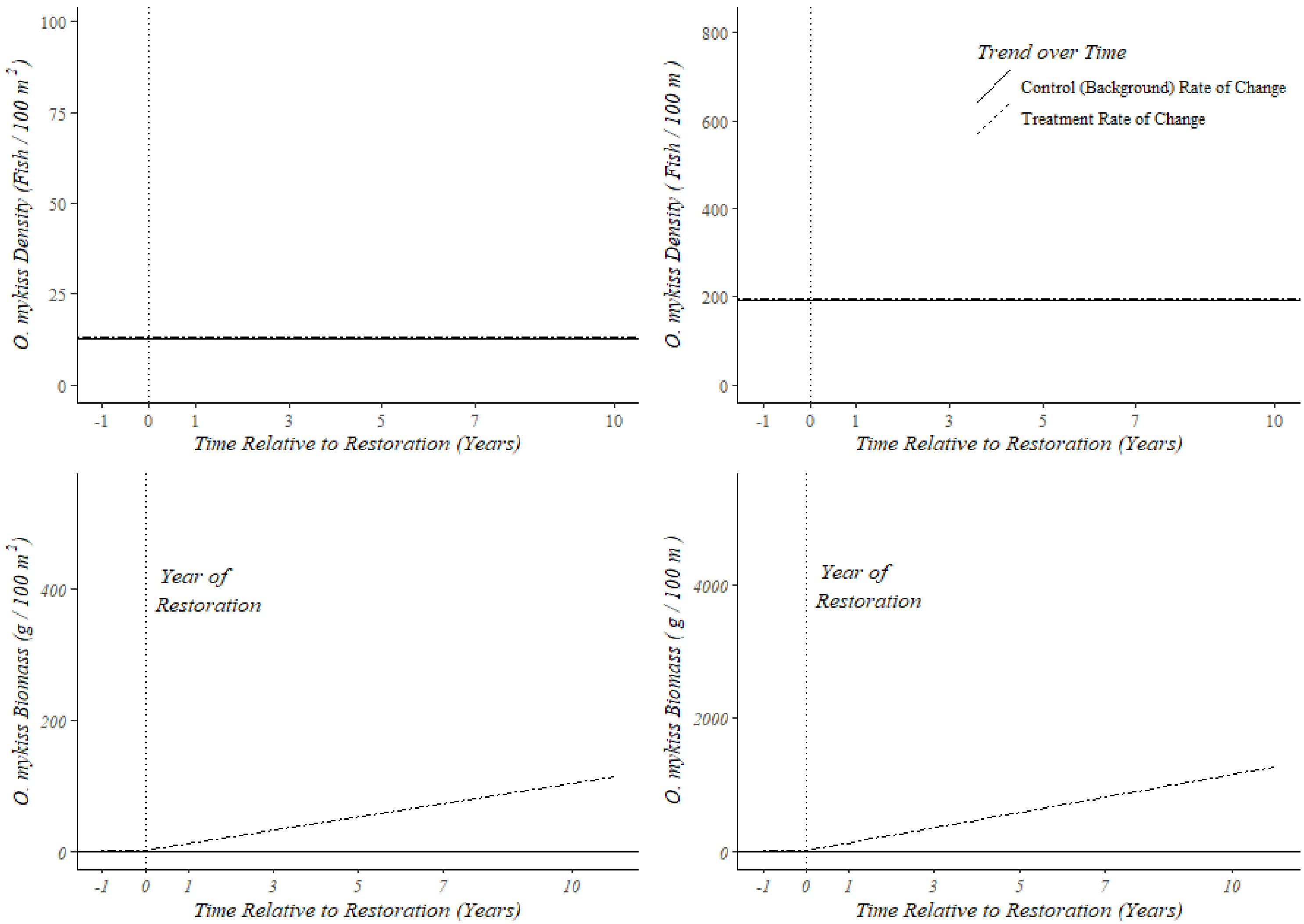


Figure 6: Plots displaying the results for the statistically significant (*p-value* < 0.05) fixed effects from a series of linear mixed models testing the immediate and long-term effects of large woody debris (LWD) placement on juvenile steelhead / rainbow trout (*Oncorhynchus kisutch*). From top left to bottom right: juvenile *O. mykiss* density (fish / 100 m^2); juvenile *O. mykiss* density (fish / 100 m); juvenile *O. mykiss* biomass (g / 100 m^2); and juvenile *O. mykiss* biomass (g / 100 m). Models testing biomass include a covariate that controls for *O. mykiss* density, thus the results can be interpreted as the effects of LWD placement on the average weight of each fish. See Table 11 for full results.

Appendix 1

Weather stations used to determine values for Annual Regional Air Temperature (departure from normal), Annual Regional Precipitation (departure from normal), and Summer Air Temp (departure from normal) for each *Project* (NOAA 2019).

Watershed	***Watershed Code***	***Station Name***	***Station ID***
Little Skookum Valley	LSV	Olympia Airport, WA	WBAN: 24227
Salmon Creek/ Willapa Bay	SCWB	Astoria (Port of), OR	WBAN: 94224
Chico Creek	ChC	Tacoma Narrows Airport, WA	WBAN: 94274
Lower Newaukum	Lnew	Renton Municipal Airport, WA	WBAN: 94248
Lower Columbia	Lcol	Astoria (Port of), OR	WBAN: 94224
Upper Washougal	Uwash	Vancouver Pearson Airport, WA	WBAN: 94298
Dungeness	Dung	Port Angles Fairchild Internat'l Airport, WA	WBAN: 94266
Cedar Creek / Lewis	CCL	Vancouver Pearson Airport, WA	WBAN: 94298
Skookum / Nooksack	SkNook	Bellingham Airport, WA	WBAN: 24217
Upper Trout Creek / Middle Columbia R.	UTCMC	Vancouver Pearson Airport, WA	WBAN: 94298
Tucannon River	PA3 PA14 PA24	Walla Walla Regional Airport, WA	WBAN: 24160
Asotin Creek	ACCC ACNF ACSF	Lewiston Nez Perce Co Airport, ID	WBAN: 24149

Appendix 2

Potential covariates that were considered, but not used in any final linear mixed models used to evaluate physical and biological responses to large woody debris restoration projects. Time-varying covariates vary between monitoring events, time-invariant covariates do not change. Level refers to the subject level at which the covariate varies. Sources are listed for covariates whose values could not be obtained from the monitoring data or project planning documents.

Covariate	Type	Level	Definition	Source
Ave. Depth (m) of Ctrl Study Reach	Time-Varying	*Calendar Year*	The average depth of the control reach for any given year (m). This is was meant as a potential proxy for flow rate.	
Calendar Year	Time-Varying	*Calendar Year*	Calendar year of the year of monitoring. Tested as both numeric and discrete.	
Chinook Spring / Fall Return	Time-Varying	*Calendar Year*	Adult Chinook salmon returns to the Bonneville Dam during the previous year's Fall and Spring runs.	NW Fisheries Science Center (NWFSC 2019)
Coho Prior Ocean Survival	Time-Varying	*Calendar Year*	Estimated smolt to adult survival rate of coho salmon in the Columbia River for the previous year.	NW Fisheries Science Center (NWFSC 2019)
Major River Distance (km)	Time-Invariant	*Project*	Major river (i.e., Columbia River) distance from ocean to tributary streams in which the project is located. Does not include tributary stream distance.	Nat'l Hydrography Dataset (USGS 2019)
Stream Distance (km)	Time-Invariant	*Study Reach*	Small or tributary stream distance from the ocean or a major river (i.e., Columbia River) to the study reach.	Nat'l Hydrography Dataset (USGS 2019)
Target Species (i.e., "Coho as Target Spp", "Chinook as Target Spp", or "SH as Target Spp")	Time-Invariant	*Project*	Binary covariate, indicating whether the target of the restoration actions was either Chinook, coho, or steelhead, depending on which species' response was being testing in the response variable of the model. For instance, if the response variable of the model was a measure of coho salmon production, the value of this covariate would be: 1 = Coho were target of restoration; 0 = Coho were not the target of restoration. Target species were obtained from restoration planning documents. This covariate was applied to both control and treatment study reaches of a project.	
Predator Biomass (g / 100 m; g / 100 m^2); Competitor Biomass (g / 100 m; g / 100 m^2)	Time-Varying	*Study Reach*	Biomass (g / 100 m; g / 100 m^2) of Predators / Competitors of the species being tested in the response variable of the model. Predators were defined as other salmonid fishes observed in the study reach estimated at > 200 mm in length. Competitors were defined as other salmonid fishes observed in the study reach estimated at < 200 mm in length.	
Annual / Winter / Summer Regional Air Temp. (Ave. of Observed Temp. (C))	Time-Varying	*Project*	Average of the observed daily air temperature (C) for the water year of the survey. Annual temperatures were the average from Oct-Sept; Winter temperatures were the average from Oct-March; Summer temperatures were the average from April - Sept. Regional values for each *Project* was obtained from the nearest NOAA weather station (see App. 1 for weather station list).	Nat'l Centers for Environmental Information (NOAA 2019)

Appendix 2 Continued

Covariate	Type	Level	Definition	Source
Winter / Summer Regional Precipitation (depart. from normal)	Time-Varying	*Project*	Sum of monthly departures from normal precipitation (in) of the water year of the survey. Winter values were from October - March; Summer values were from April - September. Values for each *Project* obtained from the nearest NOAA weather station (see Appendix 1 for weather station list)	Nat'l Centers for Environmental Information (NOAA 2019)
Winter Air Temperature (depart. from normal)	Time-Varying	*Project*	Average of regional monthly departures from normal air temperature (C) for the months of October-March of the previous and current survey year. Regional values for each Project obtained from the nearest NOAA weather station (see Appendix 1 for weather station list).	Nat'l Centers for Environmental Information (NOAA 2019)