Econometric Models for Estimating Costs of Salmon Habitat Restoration in the Northwest USA

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Abstract:

Substantial public and private resources are dedicated to restoring aquatic and riparian habitats. These efforts support conservation of fish and wildlife and provide other ecosystem services such as erosion and flood control. Resources are generally not sufficient to undertake all beneficial projects, but information and tools to prioritize habitat restoration based on cost effectiveness are often unavailable due to a lack of cost information. To address this gap, we present statistically estimated models of some of the most common activities used to restore salmon habitat using data from projects completed over the last two decades in Oregon and Washington. We use a hedonic modeling approach that enables us to evaluate how different restoration activities and site characteristics affect the total cost of projects. These models can contribute to strategic planning by providing a means to easily estimate costs of alternative habitat restoration scenarios that may contain hundreds of individual restoration activities spread over large areas. While our study is focused on salmon habitat restoration, the models, and our methodology, have broader applicability to river and riparian habitat restoration since these habitat actions can provide other benefits such as flood and erosion control. Our models demonstrate that most habitat restoration activities exhibit economies of scale that should be considered when designing restoration projects. In addition, our analysis indicates that the costs of most of the modeled restoration activities have been increasing much faster than general inflation, an important consideration when budgeting for restoration activities that will be completed over many years.

Keywords: salmon recovery, salmon habitat, economics, cost-effectiveness, return on investment, economies of scale, cost trends, hedonic model

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1. Introduction

Substantial public and private resources are dedicated to restoring fish and wildlife habitat and related ecosystem services. BenDor et al. (2015) estimated the ecological restoration industry in the United States produced $10 billion in economic activity in 2015. Available resources are still insufficient to undertake all beneficial restoration projects requiring prioritization of the limited funds available. There have long been calls for more consideration of costs and cost-effectiveness in prioritizing land conservation and habitat restoration spending to enable more efficient use of resources (Sutherland et al., 2009; Cook et al., 2017). Studies that have evaluated cost-effectiveness demonstrate that conservation outcomes for a given level of spending on conservation can vary substantially based on type, location, and design of habitat interventions, creating the potential for efficiency gains from considering costs when prioritizing interventions (e.g., Ando and Langpap, 2018; Babcock et al., 1997; Naidoo et al., 2006). A number of studies further show that conservation efforts can exhibit nonconstant returns to scale in conservation outcomes (Newbold and Siikamaki, 2009; Van Deynze et al. 2022) and economies of scale in conservation costs (Armsworth et al., 2011; Armsworth et al., 2018; Kim et al., 2014; Cho et al., 2017), suggesting the scale of interventions is a key consideration for return on investment (ROI).

Despite the potential to increase benefits from limited habitat restoration budgets, evaluation of cost-effectiveness and ROI remains rare in practice, in part due to a lack of cost information (Armsworth, 2014; Pienkowski et al., 2020). In the absence of tools to evaluate ROI, more ad hoc approaches for prioritization are often used. For example, salmon habitat restoration alternatives are commonly prioritized using simple heuristics, expert opinion, or quasi-quantitative scoring systems that generally don't factor in cost or ROI (Beechie et al., 2008; Burch et al., 2024). Scientific and political uncertainty and entrenched special interests can also combine to undermine the effective prioritization of salmon recovery resources (Wu et al., 2003).

In this study we develop models that can provide cost information to support cost-effective planning for river habitat restoration targeted at conservation and recovery of salmon and steelhead populations, many of which have been listed as threatened or endangered under the Endangered Species Act (ESA). The decline of wild salmon populations and their failure to recover relate to a number of causes, but a primary cause for most listed populations is loss and degradation of habitat (Ford et al. 2025). Dams without fish passage have resulted in losses of large areas of historical salmon habitat across the Northwest (Kiffney et al., 2025), but even in areas not blocked by dams, thousands of poorly designed culverts have left large amounts of potentially valuable habitat inaccessible to salmon and steelhead (Sheer and Steel, 2006). Many streams that are accessible have degraded habitat as a result of channelization and loss of flood plains, and a lack of riparian vegetation along streams that can lead to higher stream temperatures and mortality (Bond et al., 2019; Kovach et al., 2019; Crozier et al., 2021).

A vast array of projects has been undertaken over the last few decades to restore riparian and instream salmon habitat and reopen habitat blocked by culverts and dams. These restoration activities are ongoing to satisfy legal obligations associated with ESA and treaties with Northwest Indian tribes (Hickey, 2018). Over $8 billion has been spent over the past four decades on restoration efforts in the Columbia River Basin by federal and state agencies (Jaeger and Scheuerell, 2023). Spending on habitat restoration and protection by the Bonneville Power Administration, which has a responsibility to mitigate the effects of the Columbia River hydropower system on Salmon, averaged over $100 million per year between 2015 and 2024 (Northwest Power and Conservation Council, 2025). Spending on habitat restoration outside the Columbia Basin to support coastal and Puget Sound salmon runs is also substantial. In particular, spending on reopening habitat closed off by barriers to fish passage has been increasing in recent years. The Washington State legislature in 2025 approved an additional $1.1 billion for court ordered mitigation of blocked fish passage by state owned culverts bringing the programs two-decade total to $5.2 billion (Reicher, 2025b). Washington Department of Transportation had requested an additional $5 billion to address state-owned culverts (Reicher, 2025a). In addition, there are ongoing efforts (and grant programs) to remove or replace county, municipal, and privately owned culverts blocking fish passage and restore degraded habitat.

Biophysical models, and particularly life cycle models (e.g., Beechie et al., 2021; Jorgenson et al., 2021), offer a means to objectively evaluate biological benefits for various habitat restoration activities over space. If the cost of those interventions can be estimated, it should be possible to determine ROI and relative cost effectiveness of different restoration activities and strategies (Fonner et al., 2021). Fonner et al. (2021) demonstrate that the ROI of alternative restoration activities can vary widely, even within the same watershed, depending on the type, location, and scale of the restoration activities undertaken. However, for most types of restoration activities, models to estimate costs of potential projects have not been available. Consequently, planners must rely on rules of thumb for estimating costs (e.g., Evergreen Consulting 2003), or use detailed engineering estimates which are themselves costly to produce, and probably not feasible to produce when large numbers of potential projects are being compared. Hildner and Thomson (2007) attempted to estimate restoration cost models for a variety of restoration activities using data from the California Habitat Restoration Project Database supplemented with project data from contractors, but found data was too sparse and inconsistent to estimate multivariate models. They were only able to produce means and ranges of costs for different activities. Van Deynze et al. (2022) used data from the Pacific Northwest Salmon Habitat Project database (PNSHP) on projects undertaken in Oregon and Washington between 2001 and 2015 to estimate multivariate cost models for replacing culverts blocking passage to salmon habitat. These models included a number of variables expected to impact costs including hydrological and road characteristics, terrain, and a variety of other variables expected to impact input costs. They provide a means to create rough cost estimates for replacing a huge inventory of culverts known to block fish passage allowing development of a planning tool that can used to determine cost-effective replacement strategies to reconnect blocked habitat on a regional scale (Jardine et al. 2025; https://upstream-cloud-run-jcypjvvn7a-uw.a.run.app/). These fish passage models need to be updated with more recent data, and there is a need for models to estimate costs of other common types of habitat restoration activities used to improve habitat quality.

We present statistically estimated models of the costs of a variety of salmon habitat restoration activities commonly taken in riparian and instream habitats using data from projects undertaken in Oregon and Washington between 2004 and 2024. Our models provide a means to estimate costs for some of the most common salmon habitat restoration activities that are expected to impact future salmon population abundance and productivity including riparian shading, barrier removals to reopen habitat, increasing instream wood abundance, increasing channel complexity, and floodplain restoration and connection. We follow a modeling approach similar to Van Deynze et al. (2022), but we estimate models for riparian planting and instream habitat restoration activities as well as fish passage. Our approach improves upon the fish passage cost models from Van Deynze et al. (2022) by including a time trend as an explanatory variable to account for changes in costs over time that diverge from our general measure of inflation. We also add ten years of more recent data. Our study indicates that most of these restoration actions exhibit economies of scale, suggesting that increasing the size of habitat interventions relative to the median levels observed over the last few decades could increase efficiency. We also show that the costs of most of these habitat restoration activities have increased substantially faster than the rate of inflation. Understanding cost trends is important since habitat restoration activities can sometimes take many years, and a failure to consider how costs will change can leave managers over budget and short on resources to complete planned restoration. Our models are intended to support strategic planning of habitat restoration to support salmon recovery, but these restoration activities also contribute other benefits such as erosion and flood control and conservation of other aquatic fauna, so they have broader applicability for river restoration planning.

2. Data and Methods

2.1 Data

We use data on salmon habitat restoration projects in Washington and Oregon completed over the 2004-2024 period to estimate habitat restoration cost models. Data is drawn from two different sources: (1) the Oregon Watershed Restoration Inventory which inventories data on habitat restoration projects for the state of Oregon and is managed by the Oregon Watershed Enhancement Board (OWEB); and (2) the Washington State Recreation and Conservation Office Project Information System (PRISM) database which inventories data on habitat restoration projects in Washington state. Projects in these databases include projects funded by federal, state and private funds as well as in-kind contributions, but projects in the OWEB and PRISM datasets do not overlap. They each only contain projects in the respective states of Oregon and Washington.

The data sets include thousands of habitat restoration projects implemented over the last few decades, including different types of habitat restoration activities that vary in size, scope, intensity, and location. We confine our analysis to a subset of these activities for which there are sufficient numbers of projects with well-defined characteristics and cost data. We also focused on restoration activities being evaluated with salmon life cycle models, which our models are designed to support. These restoration activities are grouped under three basic “activity types”: (1) riparian restoration, which includes riparian planting, fencing, vegetation management, and invasive plant control; (2) instream habitat restoration, which includes placement of instream structures to add complexity, bank stabilization, and channel alteration; and (3) replacement or removal of fish passage blockages (mainly culverts). This grouping reflects activity-type definitions and a data-reporting framework used by NOAA’s Pacific Coastal Salmon Recovery Fund (PCSRF) to track federal grants, which was also adopted by both OWEB and PRISM.

Both datasets include consistently reported quantitative measures of specific restoration activities, project dates, and site locations that can be used to associate additional data on factors that influence project costs. The OWEB and PRISM datasets have similar, but not identical, metrics to describe restoration activities for projects. The primary metrics that define the type, size, and scope of activities in both datasets generally follow the definitions used by PCSRF. Both data sets include the reported costs of projects broken down by activity type (e.g., riparian, instream, fish passage), but costs are not ascribed to more specific activities when there are multiple activities within a particular activity type. For example, a project might include riparian planting, fencing, and vegetation management but only the combined cost of these activities is reported. For this reason, we model costs at the level of activity type using a hedonic framework described in the following section. The costs we model do not include administrative, design or permitting costs. While design and administrative costs can be substantial, they are not reported in the OWEB data, and it is unclear how accurately and completely reported they are in the PRISM data, so we limit our analysis to direct habitat restoration costs.

We identified several variables related to characteristics and location of the worksite that are hypothesized to affect restoration costs (Barnard et al., 2013; Evergreen Consultants, 2003; Washington Department of Fish and Wildlife 2019; Van Deynze et al., 2022) and include these as explanatory variables in our models. Van Deynze et al. (2022) discuss how these variables are expected to influence costs of replacing culverts, but many of these variables are also expected to impact costs of other types of restoration, though potentially in different ways. Hydrologic variables such as bankfull width and channel slope, and road characteristics, can influence the size and strength, and consequently cost, of a bridge replacing a culvert. Hydrologic factors can also influence the costs of other restoration activities due to the complexity and accessibility of the worksite and the materials and methods required. Bankfull width and slope may interact to create important stream characteristics. Van Deynze et al. (2022) found and interactive of these variables was a significant predictor of culvert replacement cost. For example, riparian planting along steep banks or instream habitat restoration in fast-moving streams may be more costly. Landcover types can similarly impact the difficulty of undertaking work. Proximity to paved roads and to urban areas are included as proxies for the availability and cost of labor, equipment, and materials. The influence of these variables may be complex making it difficult to evaluate a priori whether and how some of these variables might impact restoration costs. For example, proximity to urban areas may reduce the costs of transporting materials, but wages and materials costs may be higher in urbanized watersheds as well. All of the models include a variety of site characteristics based on the worksite locations as additional explanatory variables. Some data on site characteristics is drawn directly from the OWEB and PRISM data, but we also draw on other data sources to associate site characteristics based on the worksite coordinates. These sources include the US Environmental Protection Agency (EPA) NHDPlus national database, the U.S. Geological Survey (USGS)'s National Land Cover Database (NLCD), the NAVTEQ/HERE road data, and the U.S. Census. More details on the site characteristics and sources and how they are matched to restoration sites can be found in the supporting materials Table S1.

There is substantial variation in the total costs of projects for each project type within and across watershed basins (Figure 1). While projects are distributed across both Washington and Oregon, projects are more concentrated in the Western part of both states and are relatively sparse in some Eastern watersheds. There are more observations for each activity type in Oregon than in Washington. Projects for all activity types are spread broadly over the 2004-2024 period, but Oregon observations for fish passage and riparian projects are more concentrated between 2004-2014 than more recent years while observations for instream projects are more evenly distributed (Figure 2). For project in Washington observations, particularly for fish passage, are more concentrated after 2013.

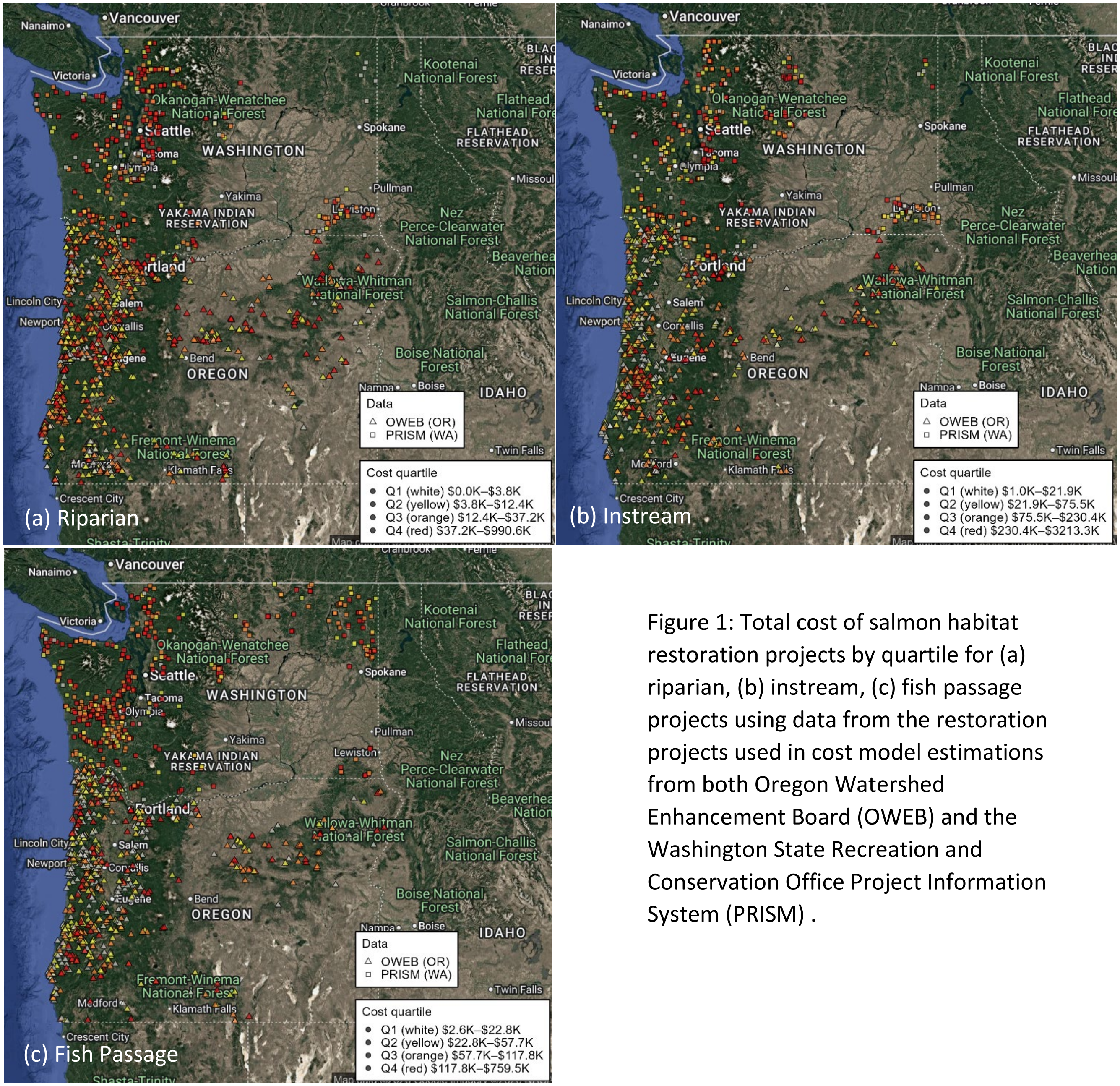

Figure 1: Total cost of salmon habitat restoration projects by quartile for (a) riparian, (b) instream, (c) fish passage projects using data from the restoration projects used in cost model estimations from both Oregon Watershed Enhancement Board (OWEB) and the Washington State Recreation and Conservation Office Project Information System (PRISM) .

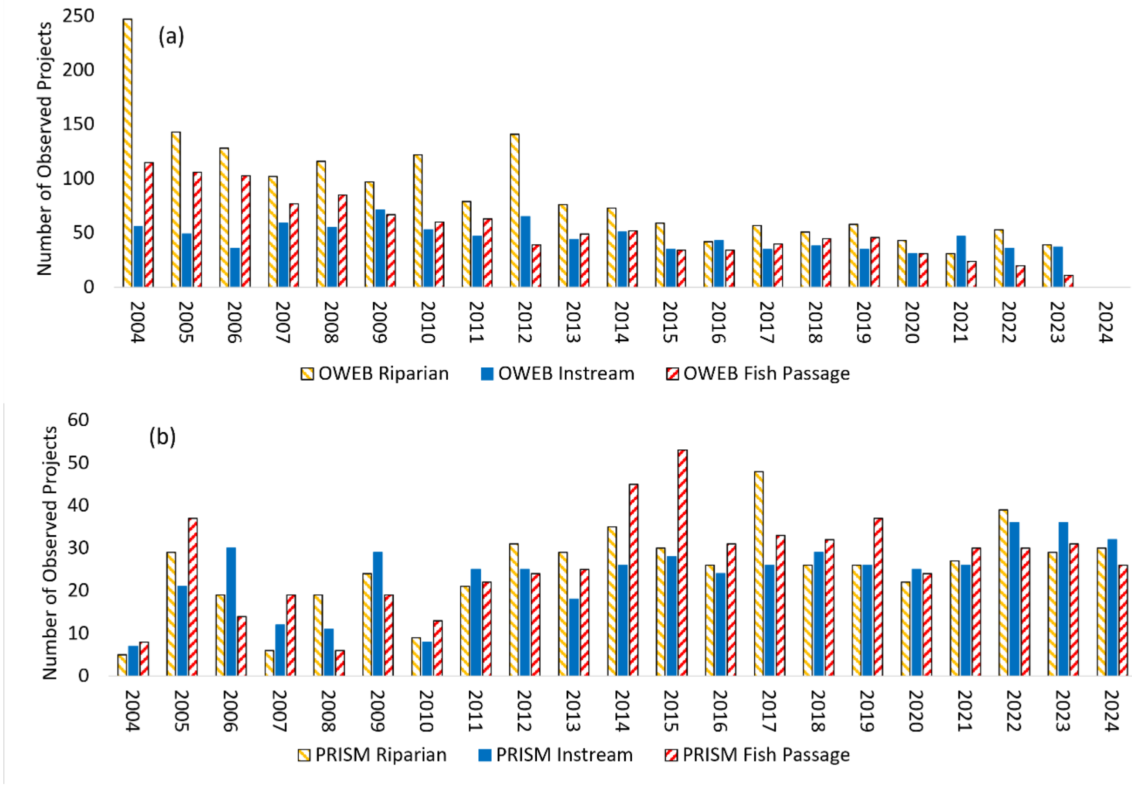

Figure 2: Number of usable restoration action observations by state data set (a) Oregon Watershed Enhancement Board (OWEB) and panel (b) Washington State Recreation and Conservation Office Project Information System (PRISM), and by activity type and completion year within each panel. Note that vertical scales are different for OWEB and PRISM data.

2.2 Methods

For both OWEB and PRISM data sets, cost is only reported for “activity types” (e.g., riparian, instream, and fish passage). These activity type costs can subsume costs of several more specific activities, but we have no way to allocate those costs to specific activities. Consequently, we model total project cost at the activity-type level and use a model specification that can account for multiple activities subsumed in those costs. We considered merging the Oregon and Washington data sets to estimate a single model for each activity type, but this led to increased mean square error in predictions reflecting systematic differences in the relationships among explanatory variables and project costs between the two states. Separate models provide more precise predictions and better insight into cost drivers in the two states. Likelihood ratio tests with models using the combined data also indicated that both intercept and slope coefficients differ by state.

Our cost models are structured similarly to hedonic models which are often used to model prices based on multiple characteristics of a good and to quantify how different characteristics of the good contribute to price (Rosen 1976). Hedonic models date back to Court (1939) who used them to model automobile prices. They are frequently used to model how characteristics of housing impact value (Ball, 1973). Hedonic models have been used to quantify how environmental amenities contribute to property value (e.g., Gillard, 1981; Li and Brown, 1980) including flood risk and mitigation of flood risk by restoring flood plains (Fonner et al. 2023, Kousky and Walls 2014). They have also been used to value the contribution of water rights to agricultural land (e.g., Crouter, 1987; Colby et al., 1993). Hedonic cost functions have been employed to evaluate how different outputs and different shipment characteristic affect shipping costs (Spady and Friedlaender, 1978; Bitzan and Wilson, 2007).

To model costs, we utilize ordinary least squares (OLS) regressions with the natural log of habitat restoration projects costs at the activity type level (adjusted to 2023 dollar values) as the dependent variable (Equation 1). Explanatory variables include metrics describing the primary physical restoration activities (e.g., miles of stream bank planted or number of log structures placed in streams), binary dummy variables indicating ancillary activities (e.g., indicating that a riparian planting project included fencing or vegetation management), site characteristics that are expected to impact costs, and a time trend variable that captures trends in real costs beyond those associated with standard inflation measures. We also include fixed effects for watershed basins (using the 6-digit hydrologic unit code) to control for a variety of unobserved factors that may influence cost. Formally the general model specification is

$$(1)\quad \ln(Y_i) = \alpha + \beta_{primary}\ln(M_i) + \gamma_{ancillary}D_i + \delta_{site\ char}SC_i + \theta T_i + \mu_{basin(i)} + \varepsilon_i$$

where *i* indexes project-worksite, Y_i is project-worksite costs at the activity type level adjusted to 2023 dollars, M_i is the primary activity metric, D_i is a vector of dummy variables indicators for ancillary activities, SC_i is a vector of site characteristics, and T_i is an annual time trend variable. The intercept term α, basin fixed effects μ_i, and the $\beta, \gamma, \delta, and\ \theta$ coefficient vectors are parameters estimated by the model. The error term, ε_i , is assumed to be zero-mean and normally distributed. However, with the log specification this means the error is assumed to be multiplicative on the untransformed costs, as we expect errors to be correlated (increasing) with total cost of projects.

With this mixed log-linear model specification (using the natural log of costs as the dependent variable), the explanatory variables increase or decrease the base project costs multiplicatively (i.e., they scale the total cost as a multiplier as opposed to increasing cost additively). The primary measures of project size are specified as the natural log of the primary activity metrics – e.g., acres planted for riparian planting. With this specification, the primary metrics are assumed to have a constant elasticity (i.e., the percent increase in costs divided by the percent increase in project size remains constant), but that allows marginal costs to change with project size. For the riparian and instream models where the dependent variable is the natural log of total activity type costs, if the coefficient on the primary activity, $\beta_{primary}$, is significantly less than 1, this indicates costs increase at a decreasing rate as the amount of the primary activity is increased, i.e., there are economies of scale, while the opposite is true if the coefficient significantly exceeds 1. For the fish passage models, where the dependent variable is the natural log of cost per crossing treated rather than total cost, the parameter on the number of crossings indicates decreasing marginal costs (i.e., economies of scale) if the coefficient is significantly less than zero and increasing marginal cost (i.e., diseconomies of scale) if the coefficient is greater than zero.

For other ancillary activities that are associated with some projects in the data, but not others, we utilize binary "dummy" variables that indicate whether the additional activities were undertaken. For example, for the OWEB Instream model, the primary activity is the placement of log structures in the stream and the observations only include projects where log structure placement occurred. The estimated coefficients for the ancillary activity dummy variables act as multipliers on predicted costs equal to the exponent of the coefficient. We initially tried estimating log-linear or quadratic models that included absolute quantities for the ancillary activities rather than dummy variables, but, for both the OWEB and PRISM data we found that the fit of the models with binary indicators for ancillary activities being applied was substantially better (higher adjusted R^2) than the models with quantitative measures of the ancillary activities, likely due to the sparseness of data on the ancillary activities.

The site characteristics included in the models include a mix of binary dummy variables (e.g., for land cover types and paved road indicator) and continuous variables (e.g., channel slope, bankfull width, distance to urban areas, etc.). The type of each variable is indicated in figures 3-5 with a "C" in parentheses for continuous variables and a "D" for dummy variables. For the site characteristics modeled with dummy variables, the coefficients act as multipliers on predicted costs equal to the exponent of the coefficient. For the continuous variables, the coefficient together with the value of the variable acts as a multiplier on cost equal to $\exp\,(\delta * SC_i)$.

Because the project data we used spans two decades, and construction costs have increased substantially over that period (particularly since 2020), we need to account for inflation and other factors that have driven up costs over time in our models. Using producer price indices more closely related to this sector (e.g., a construction wages index) can provide a better way to adjust costs than more general inflation indices (Department of Interior, 2023). Before estimating the model, we adjust costs to 2023 dollars using the producer price index (PPI) for private construction wages private (ECICONWAG) downloaded from the Saint Louis Federal Reserve (https://fred.stlouisfed.org). Projects can last from one to as many as seven years though most projects are less than two years, except for PRISM instream and riparian projects which have a median duration of four years. Since we do not have information on exactly when costs were incurred, the value of the PPI used for adjusting costs is the PPI's value for the middle year (between start and completion) of the project.

Direct adjustment of costs with a national producer price index may still not accurately reflect or fully capture regional cost trends for these specific types of activities. Therefore, we also added a time trend variable to the model as an explanatory variable to capture trends in inflation-adjusted (real) costs. The estimated coefficient on this time trend variable, θ , together with the time trend variable act as a multiplier on the cost estimate equal to $\exp\,(\theta * T_i)$. As the time trend variable increases, so does this multiplier and thus the predicted cost. If the estimated value of θ exceeds zero, this suggests real costs have been increasing over time. This illustrates how costs trends diverged from (mostly exceeded) more general national inflation trends, and provides a means to predict how real restoration costs might change in the future if these trends continue.

For all of the models we used a similar approach to remove outliers before model estimation. We sorted observations by inflation-adjusted cost per unit (e.g., cost per acre of trees planted for riparian planting in 2023 dollars) and removed the observations below the 5^{th} and above the 95^{th} percentile of cost per unit. We feel this is justified as these observations may represent data errors or have cost drivers that differ from more typical observations in unobservable ways, making them less useful for predicting the costs of future projects. In addition, model diagnostics suggested these outliers were leading to non-spherical residual distributions with large residuals for projects with very high or low cost per unit. We evaluated the residuals from the models to assess normality and heteroskedasticity, which can lead to inefficient parameter estimates and incorrect variance estimates (Kennedy, 1992), and we report robust standard errors since heteroskedasticity is present in a few models. Since our cost models include a large number of explanatory variables, there is the potential for multicollinearity, which can inflate standard errors, so we use a variety of tests to evaluate multicollinearity (Kennedy, 1992). Tests are reported in the supplementary materials for normality (Table S15), heteroskedasticity(Table S16), and multicollinearity (Tables S17-24).

3.0 Results

3.1 Summary results

Table 1 indicates the number of observations used in model estimation, the 5th and 95th percentile bounds on cost per unit (2023 U.S. dollars) used to remove outliers for riparian and instream models, the median project cost and size, and the adjusted R^2 for each model. In general, Oregon models using the OWEB data set had higher numbers of observations and better fits. For both states Riparian models had the best fit as measured by adjusted R^2, and fish passage models had the worst fit.

To assess multicollinearity we evaluated condition numbers, correlation matrices of the independent variables, and variance inflation factors (VIFs). Detailed results of this analysis are provided in the supplementary materials (Tables S17-24). For the final models condition numbers ranged from 4.8 to 8.7 for the PRISM models and from 12.8 to 18.7 for the OWEB models, suggesting low overall multicollinearity. VIFs are below 10 for all variables other than the basin fixed effects which are mutually exclusive. There are only a few variables in the models with high cross-correlation coefficients (.7 or higher). For the several of the models the cross-correlation of slope and the interactive variable slope*bankfull-width was high (in the 0.7 to 0.8 range) there is good a priori reason to think that the combination of slope and stream width (in addition to each variable on their own) would be a cost driver (Van Deynze, 2022) and this is borne out by likelihood ratio tests that reject dropping the interactive variable. For the Oregon instream models, the cross-correlation between number of log structures and stream miles treated is 0.71. However, both variables are significant and removing either is rejected by a likelihood ratio test. It is important, however, when using the model to make inferences from these coefficients (e.g., as we do when evaluating economies of scale) to assume these variables will move together. Lastly, the variables in the PRISM fish passage model that indicated if a replacement crossing was a culvert or bridge are highly correlated, but both remain insignificant when the other is dropped. We leave them in the model to demonstrate that there does not appear to be a significant difference in cost between bridge and culvert replacements.

We evaluated residual distributions from all models to test for non-normality and heteroskedasticity. Complete results for these tests and residual plots are provided in the supplementary materials (Table S15, Figure S1-6). Shapiro-Wilk and Anderson-Darling tests cannot reject normality for all but the OWEB riparian model. Breusch-Pagan and White tests for heteroskedasticity produced mixed results for most models, but both suggest heteroskedasticity for the OWEB fish passage models and the PRISM riparian models (Table S16). This appears to be caused by outlier observations with either very high or low costs rather than any correlation with the explanatory variables in the model. While we did remove outliers with cost per unit below the 5th and above the 95th percentile, this did not necessarily remove observations with very high or low total cost. However, residual plots do not suggest any systematic correlation with the fitted predictions from the model. Nevertheless, we report robust standard errors for the models and use these for creating the bounds on the multipliers shown in Figures 3-5.

Table 1: Summary information for models by state data set and activity type

Work Element or Activity Type	Primary Activity Type Metric	Number of Obs.	Median Project Size	Cost Per Unit US (5th Perc)	Cost Per Unit (Median)	Cost Per Unit US$ (95th Perc)	Model Adj. R^2
Oregon Watershed Enhancement Board (OWEB)							
Riparian	Acres of trees planted	1583	1.55	1,028	7,855	72,379	0.77
Instream	Miles of stream	825	0.75	13,937	87,491	1,077,344	0.74
Instream	Structures	825	12	857	5,007	44,161	0.74
Fish Passage	Number of crossings	992	1	3,390	53,218	489,190	0.44
Washington State Recreation and Conservation Office Project Information System (PRISM)							
Riparian	Acres of trees planted	469	5.00	730	8,701	150,010	0.58
Instream	Miles of stream	447	0.40	50,360	606,289	7,841,558	0.53
Instream	Structures	447	16	555	12,581	154,368	0.53
Fish Passage	Number of crossings	504	1	15,200	145,259	917,139	0.26

3.2 Influence of project characteristics, site characteristics on project cost

For all models, the intercept terms were highly significant, but basin fixed effects were mostly not statistically significant (see tables of full model results in the supplementary appendix (Tables S3-S8). This suggests that mean project costs by activity type are an important predictor of total cost, but that there is little systematic variation across watersheds within a state that is not explained by the project characteristics, site characteristics, and price indices included as explanatory variables. The importance of these different characteristics in predicting cost varies across activity types and data sets, as does the overall explanatory power of the models. In the sections below, we describe the implications of parameter estimates for different categories of explanatory variables in terms of whether and by how much they impact project costs.

The cost multipliers presented in Figures 3-5 represent the proportionate changes in total project costs associated with changes in the individual explanatory variables. Table of values and confidence intervals for these multipliers are provided in the supplementary materials (Table S9-S14). For continuous variables (designated with a “C” in parentheses next to the variable names) the multiplier shows the proportionate increase in cost associated with a doubling (100% increase) in the untransformed variable (e.g., doubling acres, not doubling log of acres) from the median level. For dummy variables (designated with a “D”), these figures show the proportionate change in costs when that dummy variable moves from 0 to 1. The filled-in circles in these figures indicate the point estimate of the multiplier, while the whiskers around the circles indicate the 95% confidence interval of the multiplier. If these whiskers intersect with the value of 1, the effect of this variable is not statistically significant at the 95%

confidence level. Details on the derivation of multipliers are provided in the supplementary materials prior to Table S9.

For riparian models (Figure 3), the number of acres of trees planted is an influential predictor of project costs. It is highly significant with a small standard error and has a relatively similar impact on scaling costs for both Washington and Oregon projects. For Oregon, the point estimate of the parameter indicates that a doubling from the median in acres planted increases costs by 64%, while for Washington it increases the total activity-type cost by 59%. Ancillary activities (e.g., fencing of invasive plant control) are specified as dummy variables indicating whether or not they were included in the project. For Oregon, the riparian model suggests that projects that include fencing are predicted to be nearly twice the cost of those that don't on average, while adding other vegetation planting and invasive plant control increase costs by around 16% and 23%, respectively. The multiplier on vegetation management is negative but not significant. For Washington, none of the ancillary activities are statistically significant, though the point estimates of the multipliers for fencing and invasive control are positive. Landscape variables for both Washington and Oregon are mostly insignificant or very small in terms of their impact on costs. Some landcover types are significant for Oregon, suggesting higher costs for wetland, developed land, and forest landcover types relative to the average. All of the significant landcover types could be associated with more difficult access. Developed lands might also require removal/mitigation of built environment or repair of development-based degradation. For Washington, none of the landcover types has a statistically significant impact on riparian planting costs.

For Instream models (Figure 4) the measures of project size are the number of log structures placed and the stream miles treated. For Oregon, a doubling of the number of log structures from the median increases cost by 55% while a doubling of stream miles treated increases cost by 24%. For Washington a doubling of stream miles increases costs by 45% but the multiplier on log structures is not significant. There are a number of observed project characteristics that strongly impact costs of instream habitat restoration in Oregon. In particular, the use of a helicopter on the project doubles costs. The same might be true in Washington but the data does not have an indicator of whether a helicopter was used. Including channel alternation in the projects substantially increases costs, around 92% for Oregon and 77% for Washington based on the point estimate of the multiplier. The use of anchored structures, engineered structures, and bank stabilization also significantly increase predicted cost in the Oregon model. However, for Washington multipliers for ancillary activities for which data were available (other than channel alteration) are not significant. Most of the landscape variables are not significant in the instream cost models, other than bankfull width, which increases costs by 25% for Oregon and 26% for Washington given a 100% increase in bankfull width from the median. When streams are both wider and steeper (i.e., the product of bankfull width and slope is higher) there is a significant increase in cost (see coefficient on Bankfull Width*Slope in the full model results in the appendix). In Washington the percent of private land near the worksite is significantly correlated with lower project costs. Landcover types are not significant predictors of costs for either state except for higher costs associated with developed land in Oregon, which tend to have higher costs, but the uncertainty bounds around this multiplier are wide, suggesting the predictive power of this variable on cost is low.

Fish passage models were estimated with the natural log of cost per crossing treated as the dependent variable rather than total project cost since the great majority of projects treat a single crossing. This specification is consistent with Van Deynze et al. (2022). For both Oregon and Washington models, the coefficients on number of crossings are negative and significant indicating economies of scale (Figure 6).

The estimated multiplier on the number of crossings treated is 0.79 for Oregon and 0.67 for Washington, indicating lower cost per crossing when multiple crossings are treated at the same work site (Figure 5). For both states, simply removing but not replacing a crossing (generally a culvert that blocks upstream passage) costs less than half that of replacing it on average. The data available to the types of treatments to culverts differs by state. For Oregon the model indicates arched culverts or bridges are both 58% more expensive on average than average cost per crossing treatments, while flat culverts average 33% lower than average costs per crossing. For Washington the type of culvert is not specified, only whether a culvert or a bridge was installed, and we cannot identify a statistically significant difference in cost for culverts vs. bridges. For both states the models do not indicate substantial change in cost for increases in bankfull width or channel slope alone but, when streams are both wider and steeper (i.e., the product of bankfull width and slope is higher), there is a significant increase in cost (see coefficient on bankfull width*slope in the full model results in the appendix). We expected the road speed category to be an important indicator of the costs of fish passage projects assuming roads with higher speeds would require more expensive culverts or crossings. The road speed categories run from low numbers for high speeds to higher numbers as road speeds decrease (e.g., category 2 for 65-80 mph and category 8 for road speeds less than 6 mph). For Oregon, the model suggests costs are higher for higher road speeds (lower road speed categories). The multiplier on road speed suggests the opposite for Washington but it is not statistically significant. For Oregon, the model suggests the percent of private land nearby and distance to urban areas are both correlated with lower costs. Oregon fish passage projects in developed landscapes and developed open space were around 50% more costly than others landscape types, while, for Washington, the model indicated lower than average costs for developed areas, cultivated land, forests and wetlands.

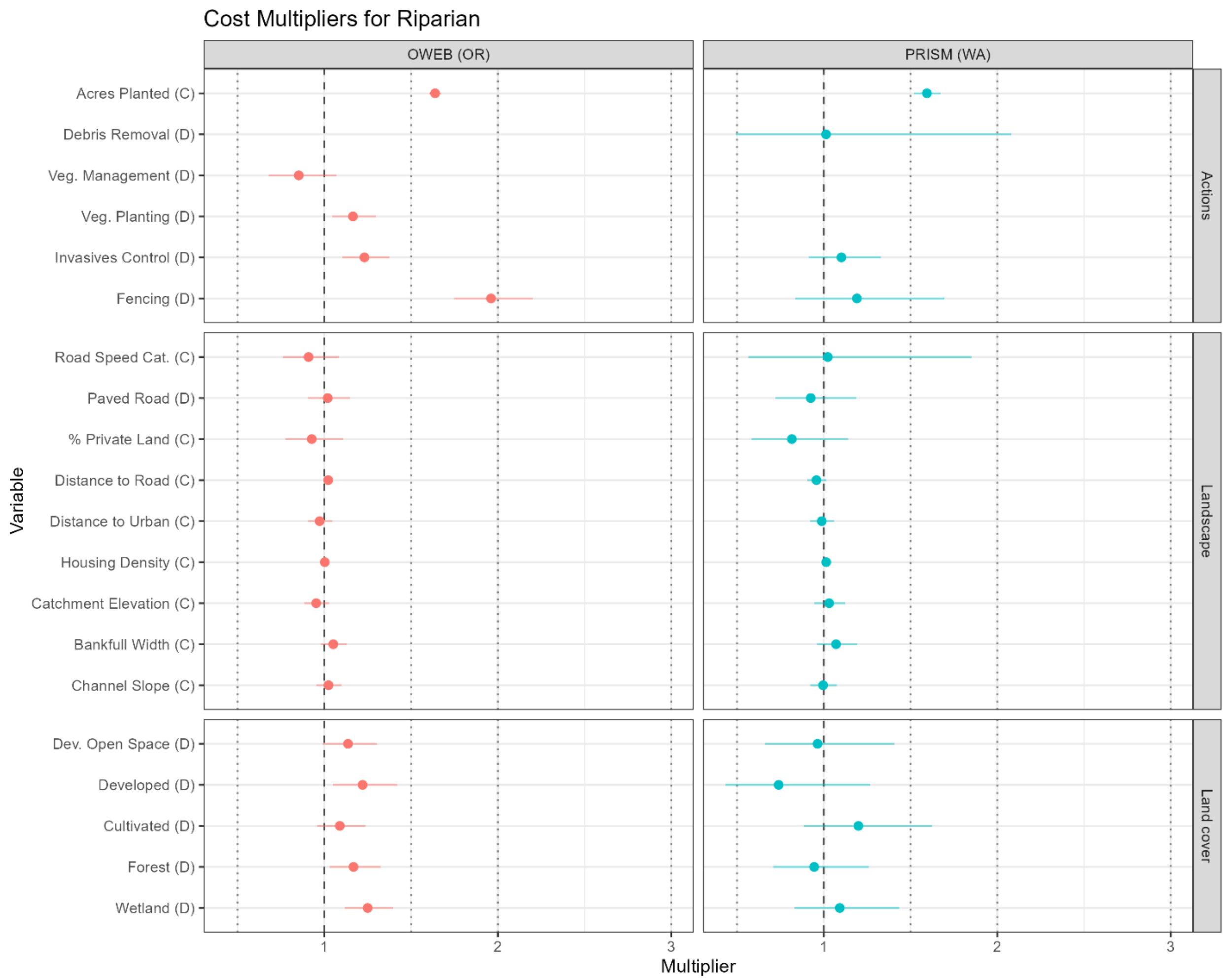

Figure 3. Standardized cost multipliers for hypothesized cost drivers of riparian planting models from Oregon Watershed Enhancement Board (OWEB) and the Washington State Recreation and Conservation Office Project Information System (PRISM). Circles indicate the mean of the cost multiplier for a doubling of the continuous variables or a switch from zero to one for dummy variables, and horizontal bars represent 95 % confidence intervals. Continuous variables are indicated with a C in parentheses next the variable name and binary dummy variables are indicated with a D.

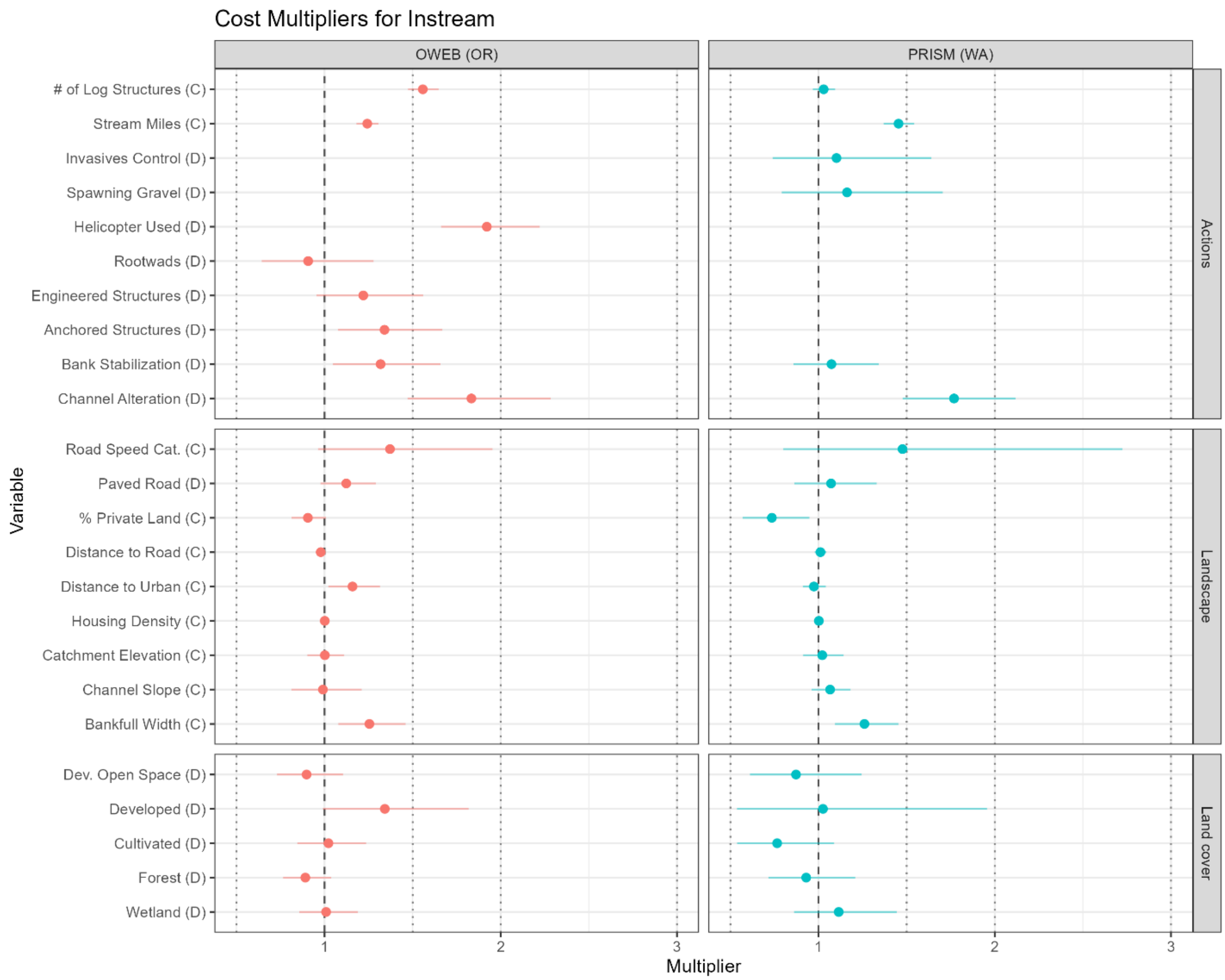

Figure 4. Standardized cost multipliers for hypothesized cost drivers of instream habitat restoration from Oregon Watershed Enhancement Board (OWEB) and the Washington State Recreation and Conservation Office Project Information System (PRISM). Circles indicate the mean of the cost multiplier for a doubling of the continuous variables or a switch from zero to one for dummy variables, and horizontal bars represent 95 % confidence intervals. Continuous variables are indicated with a C in parentheses next the variable name and binary dummy variables are indicated with a D.

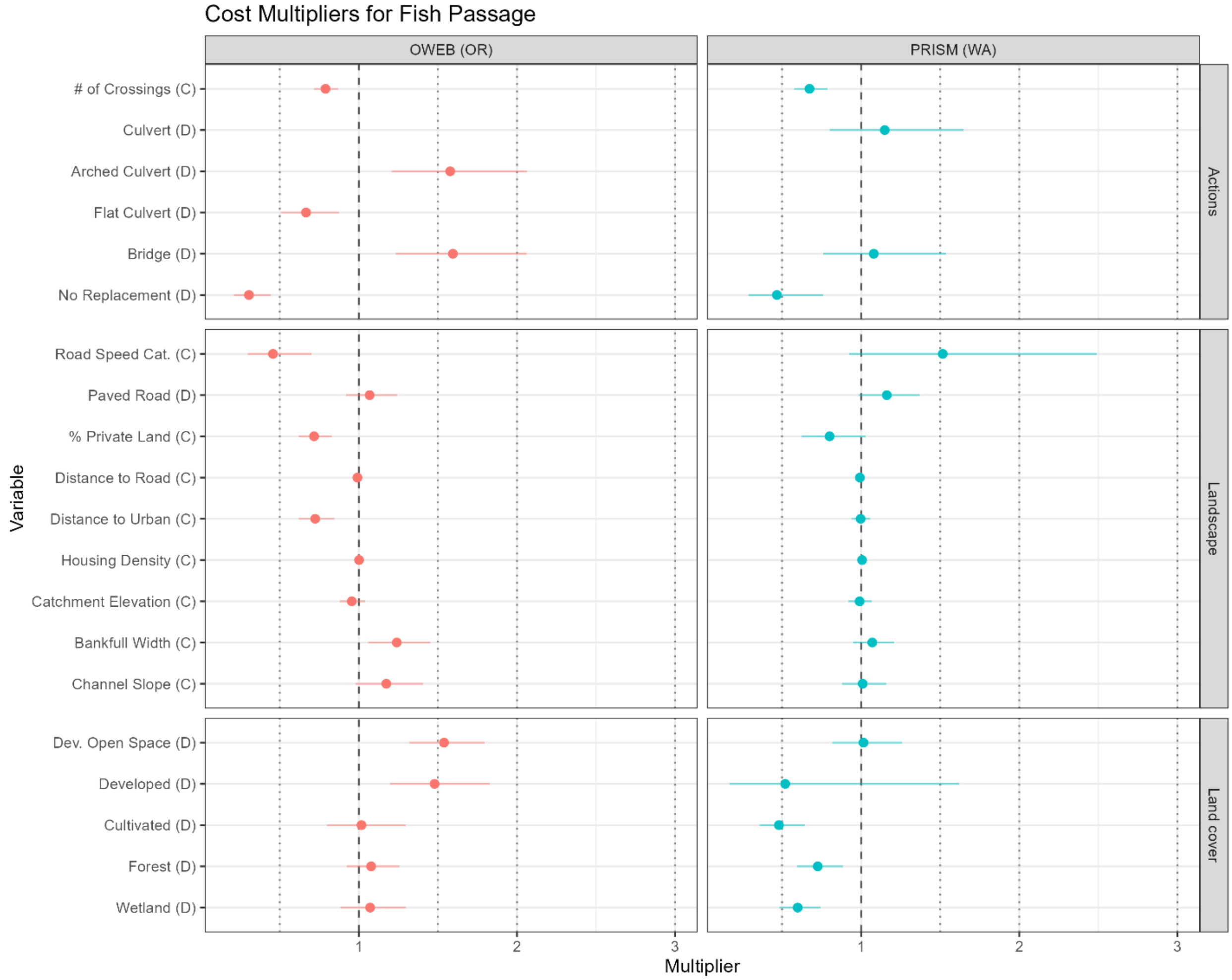

Figure 5. Standardized cost multipliers for hypothesized cost drivers of fish passage projects from Oregon Watershed Enhancement Board (OWEB) and the Washington State Recreation and Conservation Office Project Information System (PRISM). Circles indicate the mean of the cost multiplier for a doubling of the continuous variables or a switch from zero to one for dummy variables, and horizontal bars represent 95 % confidence intervals. Continuous variables are indicated with a C in parentheses next the variable name and binary dummy variables are indicated with a D.

3.3 Economies of scale in the production of habitat

The models can identify economies or diseconomies of scale where the marginal cost per acre, stream mile, or crossing decreases or increases as the quantity of the primary activity increases. The coefficients on acres for both the OWEB and PRISM riparian models are significantly smaller than 1.0 indicating statistically significant economies of scale. The impact of average cost, relative to a project of a single acre (Fig 6a) suggests marginal costs are still declining steeply for OWEB riparian projects at the median project size of 1.55 acres. For the PRISM riparian projects, which have a median size of 5.0 acres, marginal costs are still declining but not steeply suggesting only small average costs reductions for projects larger than the median size.

The model for instream restoration includes both log placements and stream miles treated as primary metrics. Since these variables are highly correlated it may not make sense to evaluate economies of scale based on either coefficient in isolation. In Figure 6b we illustrate how marginal costs increase when both miles treated and log structures are increased proportionately assuming 15 log structures per stream mile. The models suggest that marginal costs per stream mile treated fall by about half in Washington as projects increase from the median size of 0.40 stream miles. In contrast marginal costs in Oregon only decline about 10% before leveling out. Thus, there appears to be more opportunity for costs savings by increasing average instream project size in Washington than in Oregon.

For fish passage projects both OWEB and PRISM models indicate economies of scale as the number of crossings treated by a project increase. Most fish passage projects only treat a single crossing, but our results suggest cost per crossing could be reduced when treating two or three crossings in a project, though this might depend on how near to each other the crossings are which was not specified in the model.

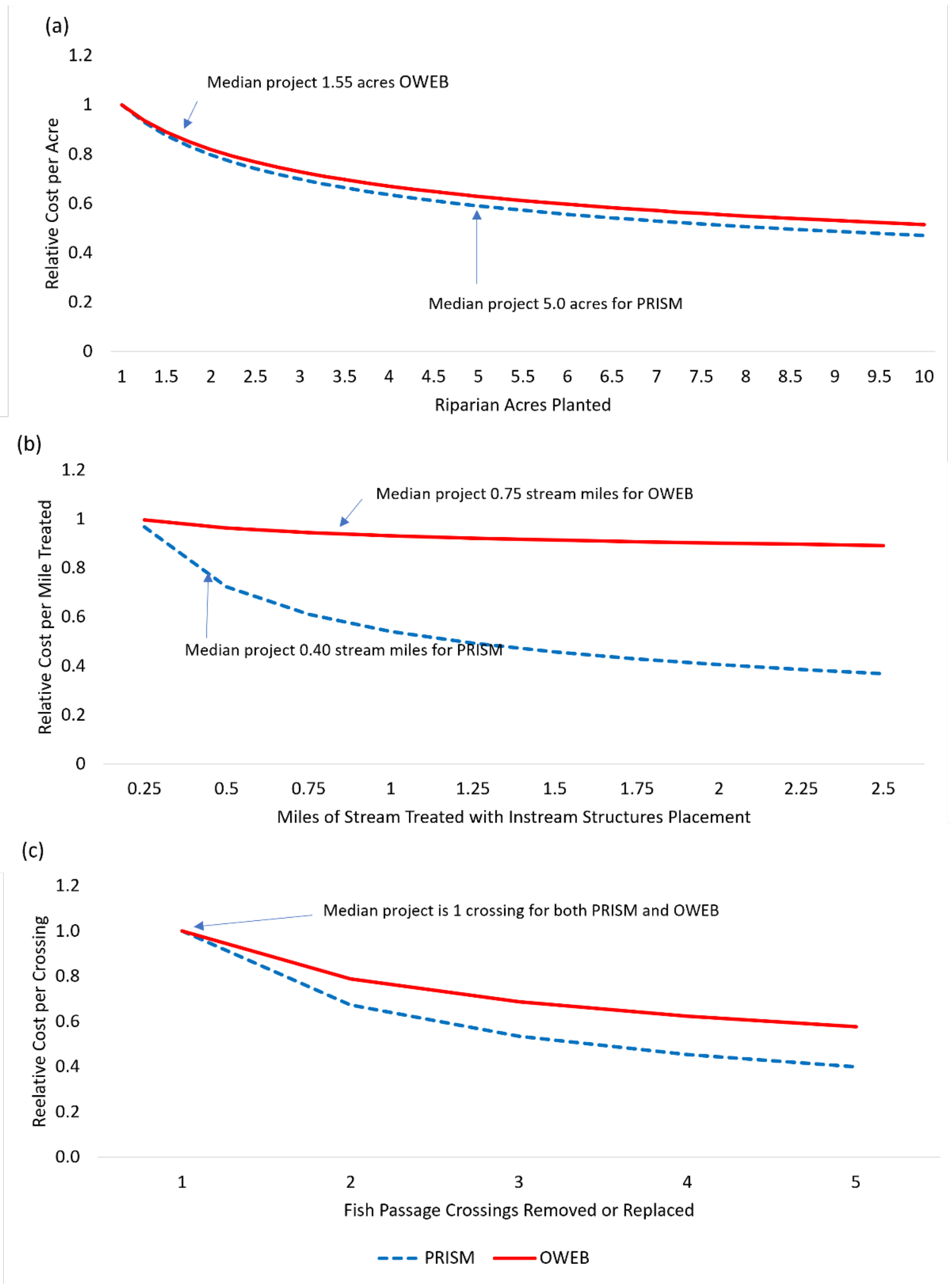

Figure 6. Cost per unit of restoration undertaken relative to the cost of baseline size (e.g., 1 acre of planting, 0.25 mile of stream, or 1 crossing replacement) as the number of units of restoration at a worksite increases based Oregon Watershed Enhancement Board (OWEB) and the Washington State Recreation and Conservation Office Project Information System (PRISM) models for (a) riparian planting panel, (b) instream structure placement, and (c) fish passage panel. For instream projects the relative cost per mile shown above assumes both miles treated and log structures are increased proportionately assuming 15 log structures per stream mile.

3.4 Increases in costs over time relative to indicators of inflation

As noted in the methods section, the PPI for construction wages was used to adjust cost for inflation prior to model estimation, but we also added a time trend variable to the model to account for trends in costs that diverge from the PPI. If observed nominal cost increases mirrored the PPI for construction wages, the coefficient on the time trend would not be statistically different from zero, and the respective trend lines in Figure 7 below would be flat. If they are increasing, and rise above 1.0, this indicates costs increased faster than the PPI. As illustrated in Figure 7, the models suggest that for most activity types, in both states, costs have risen substantially faster than the PPI for construction wages. Increases are greatest for Washington instream projects. The only model that did not suggest cost rising faster than the PPI was the riparian planting in Washington. However, the coefficient on the time trend in the PRISM riparian model was not significantly different from zero, so we cannot say with any certainty that cost trends differ from the PPI. For all the other models the coefficient on the time trend is significantly greater than zero, indicating that those costs have been rising faster than the PPI (see Tables S4-S9 in the supplementary materials).

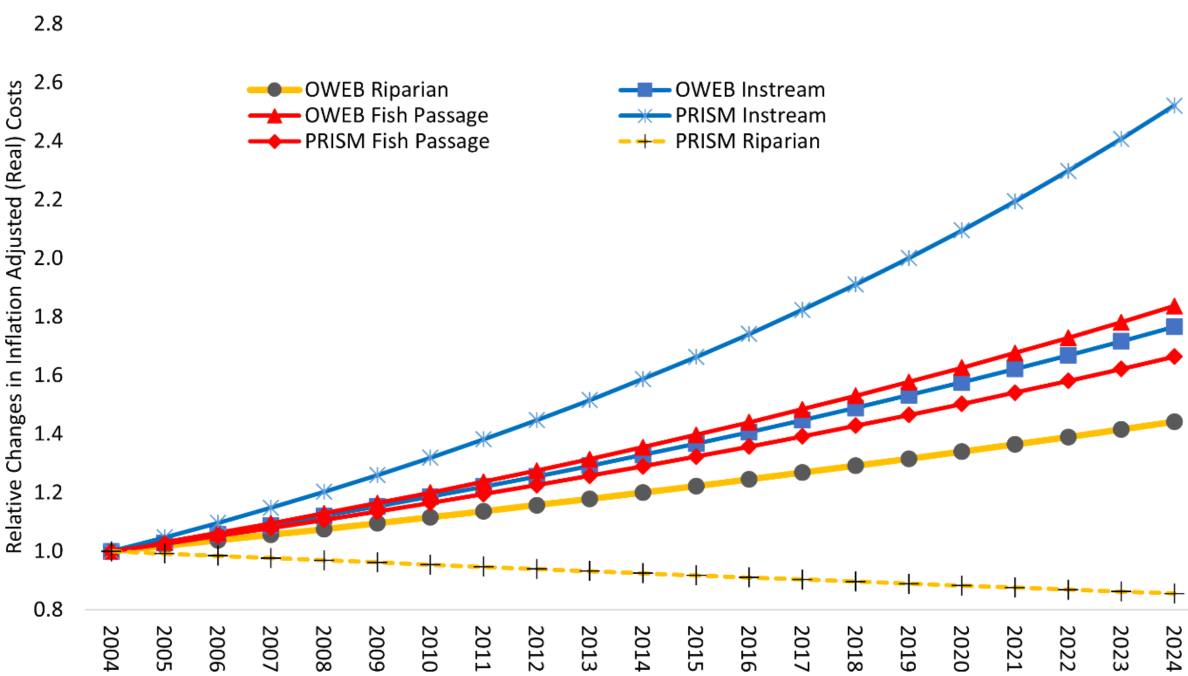

Figure 7. Relative changes in inflation-adjusted (real) costs by restoration type as predicted by Oregon Watershed Enhancement Board (OWEB) and the Washington State Recreation and Conservation Office Project Information System (PRISM) models. Trend lines show the predicted percent change in average real cost relative to 2004.

4.0 Discussion

There are a vast number of potential river restoration projects that could improve salmon habitat and provide other ecosystem services, but resources are limited and only a fraction of potential projects can be undertaken. The efficacy and the cost of different potential projects vary widely. Some very expensive activities may provide little ecological benefit, while other less expensive activities may provide substantial benefits. Prioritizing the projects based on cost-effectiveness is likely to generate more ecological lift per dollar of expenditure. In addition, evaluating projects individually (i.e. ignoring interdependencies in costs and benefits across projects) is unlikely to result in the most effective outcome because projects are often complementary and because of connectivity issues (e.g., restoring habitat upstream of a blockage to fish passage will be ineffective). Effective strategic planning will require evaluating and comparing restoration scenarios that include large numbers of projects spread over large areas so that effects of connectivity, complementarity, and economies of scale can be considered. Developing engineering-level cost estimates for each individual project would be costly and thus infeasible when comparing large scale restoration scenarios capable of making substantial contributions to salmon recovery, but the cost models presented here can provide a quick way to get rough cost estimates for these scenarios.

The cost models developed in this paper can facilitate ROI analysis with salmon life cycle models. Decades of monitoring have produced data on survival and growth rates of different salmon populations at all life stages. This has enabled development of life cycle models that can be used to predict how opening new habitat or improving habitat quality can promote recovery of specific salmon populations and subpopulations. Such models can be very useful for strategic planning of restoration, but life cycle models and other models for estimating the effects of recovery actions on salmon survival need to be paired with cost models that can estimate restoration costs to evaluate ROI. Our cost models were designed to estimate costs of common restoration activities that can be evaluated with life cycle models that are being developed for Northwest salmon populations. The cost models presented here provide a means to estimate costs for large numbers of projects so that the ROI of alternative portfolios of habitat restoration can be compared. These ROI analyses should be able to identify and capitalize on the substantial heterogeneity in cost effectiveness that likely exists across project activity types and locations, as well as inform how project scale can be optimized to reduce costs.

While our cost models have substantial explanatory power, the uncertainty bounds around individual project cost estimates are wide. These models are not a good substitute for engineering cost estimates when evaluating how much a specific project will cost or whether to undertake it. However, when considering portfolios of large numbers of projects of varying sizes, the uncertainty about the costs of the overall portfolio should be reduced (with under and overestimates cancelling out across projects). Thus, we see our cost models as being useful for strategic planning and comparing the ROI of large-scale restoration scenarios, but less so for developing budgets or evaluating grant applications for individual projects. Even when comparing portfolios of projects, care should be taken with predictions if some projects are very large, since error bounds are multiplicative and errors in cost predictions for large projects can be large.

We found that estimating separate models for Oregon and Washington for each activity type improved model predictions relative to models estimated with combined data from both states. This may reflect systematic differences in cost drivers, including materials and the services of companies that undertake habitat restoration activities. It may also reflect differences in the way that projects are designed (e.g., structures per mile of instream treatment or structure types used), the distributions of project scale, the scope of activities and activity types undertaken at the project site, and the way that costs and metrics of activities are reported. For example, OWEB often reports multiple site locations while only a single location is reported in PRISM data. Also, the median project sizes differ substantially across states for both riparian and instream activity types. Given the differences in the models, it is unclear how well they might perform in predicting costs in other states. Data from the BMA's CBFISH data program includes some projects in Idaho and may provide a better alternative for predicting costs in Idaho.

Habitat restoration cost models could likely be improved with additional or more accurate project data. For the most part, site characteristics did not provide a great deal of explanatory power in our models. Some of the uncertainty in our model parameter estimates, or inability to identify significant effects, may be due to inaccuracy in geospatial matching. For many of the Oregon projects there were multiple worksite locations in the data, but the specific activities at different worksites were not identified. Thus, we had to use average or minimum values of site characteristics across the different worksite locations. In other cases, there may be inaccuracy in the location used for matching due to errors in the data or because a single location is specified for a project that took place over a larger area. This is less likely to be important for variables such as distance of urban areas and housing density, where small errors in location should have little effect on the estimate of the variable, but for other variables like channel slope and bankfull width, or road speed category, a small error in worksite locations could create substantial error in the site characteristic measure. More information about the equipment used could also improve the models. Use of helicopters proved to be an important predictor of costs for instream projects in Oregon, but this information was not included in the PRISM project data. It should also be noted that projects in both states were more concentrated in the Western parts of the states. As a consequence, the models are likely more accurate for predicting costs in these areas than in areas where observations were sparse. There may also be cost attribution errors that worsened model fits. Many projects include more than one activity type (e.g., they include both instream and riparian activity types), but the data allocates total project costs across activity types and the attribution of costs across these activity types may be inaccurate or arbitrary. In addition, total project costs often include estimates of in-kind costs, which may be somewhat subjective.

Our cost models clearly demonstrate importance of considering how project scale impacts costs. Fewer larger projects may be more cost-effective, and economies of scale may be underestimated if they also exist for indirect costs that we did not model. The models suggest relatively greater potential cost savings in Oregon from increasing the average size of riparian planting projects than in Washington where projects are larger on average. The opposite is true for instream projects, where greater cost savings appear possible than in Washington. Economies of scope may exist as well if set-up costs at a worksite or equipment that is expensive to move can be shared across activity types. It is also important to consider whether there are increasing returns to scale in terms of the biological impacts of larger-sized projects related to ecological thresholds (Munsch et al., 2020). If so, these could further increase relative benefits from larger projects, but there could also be decreasing returns to scale for some activities and locations, and there may be biological risk reduction benefits to spreading habitat improvement over space. For fish passage projects, connectivity is a key consideration in choosing projects and considering how much habitat is opened up by a project is critical since another untreated barrier further upstream could negate most of the gains of removing a barrier. For all types of projects, it is important to consider other factors that may limit the gains from new and improved habitat, such as pollutants in streams (Feist et al. 2011; 2017). These trade-offs can be evaluated by pairing cost models with life cycle models and networked fish passage models that can evaluate both the costs and biological benefits of alternative portfolios of habitat restoration projects that vary in terms of scale, location and restoration activities.

Our results suggest that costs for most habitat restoration activities have increased faster than general inflation over time. This suggests that making an assumption during strategic planning that costs will rise in proportion to inflation may be risky and lead to insufficient budgets to complete planned restoration. This clearly has been occurring for fish passage mitigation in Washington where cost increases have led to substantial increases in the costs of complying with treaty obligations and necessitated billions of dollars in increased appropriations for fish passage projects relative to initial estimates of funding needs. The inclusion of a time trend variable in our model provides a means to predict how real costs may increase in future years, providing insights into how overall restoration budgets will have to increase to keep pace with increasing costs. However, it is unclear why costs have increased more than national inflation in construction costs and whether these trends will continue. Plausible, and not mutually exclusive, hypothesis include (1) less expensive projects were undertaken first leaving more expensive ones for later years; (2) access to materials such as dead logs and other woody debris may have become more expensive due to having to source it from further away; or (3) demand for restoration services has outpaced supply allowing providers to increase the prices they charge, (4) insufficient competition for projects allowing contractors to charge above market rates. These are, as yet, unverified hypotheses for why costs have increased and there may be other reasons. Understanding the true causes of cost increases is important to improve budget planning and, potentially, to find ways to slow future cost increases. Further research on these questions would be valuable.

We did not include design and permitting costs in our models, but these costs can be substantial. Estimates of design and permitting costs could be useful to complement our models of direct project costs. Data on design (architectural and engineering) costs is available for projects in PRISM data set, but is not in the OWEB data. Even for the PRISM Models, design and permitting costs are at the work site level and are not specified at the level of activity type as we have modeled project costs. We attempted to model design costs at the worksite level as a function of the project costs. While we found that project costs were statistically significant predictors of design costs, the models tended to overpredict observations with low design costs and provide very imprecise predictions in general. We were unable to find a model specification that successfully addressed these issues, so we did not include these models in the paper.

Cost-effectiveness is an important consideration for planning future habitat restoration, given limited resources , but costs and ecological benefits are only two of many factors that are important in prioritization. Equity, feasibility, and cooperation by local landowners and other stakeholders are all important factors for prioritization (King and Fonner 2023; Stanford et a. 2018; Rosenberg et al. 2008; Lubinski et al. 2026), and they may conflict with cost-effectiveness in some cases. In addition, relative cost-effectiveness may be of limited relevance when comparing projects across different watersheds that support different species or ESUs. However, the lack of cost information paired with a good understanding of biological impacts is likely to undermine effective decision making and the recovery of salmon. Continued work on these types of cost models and on integrating them with biophysical evaluations is imperative in fighting the uphill and under resourced battle to recover depleted salmon populations.

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