Considering Intervention Intensity in Habitat Restoration Planning: an Application to Pacific Salmon

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ABSTRACT:
Habitat restoration is a key strategy for recovering imperiled species, and planning habitat restoration activities cost effectively can help advance recovery objectives. This article examines the incorporation of the amount of restoration undertaken in a given location and time, termed intervention intensity, to inform cost-effective habitat restoration planning. A return on investment framework is developed for incorporating habitat restoration interventions with return on investment analysis. The framework is then applied in the context of planning habitat restoration for Pacific salmon recovery as a case study. Results showed that no single intervention type or location dominated, and several returns to scale relationships emerged across the candidate interventions. Scenarios that considered interventions across multiple intensities outperformed single-intensity scenarios in terms of total benefits and cost effectiveness. . These findings highlight the usefulness of exploratory return on investment analysis for prioritizing habitat restoration interventions, and underscore the importance of systematically considering how much restoration to undertake, in addition to what to do and where.

Keywords: Return on investment analysis; Habitat restoration; Species recovery; Returns to scale; Ecological thresholds

Introduction

Ecological restoration encompasses a broad range of interventions (Noss et al., 2009) and motivations (Wiens and Hobbs, 2015). This article deals with habitat restoration -- a context that involves planning and enacting restoration interventions to meet the habitat needs of focal species and ultimately enhancing the conservation status of those species (Miller and Hobbs, 2007). Approaches to habitat restoration planning are diverse and do not necessarily incorporate economic considerations. However, a growing body of evidence supports the idea that incorporating cost data into conservation prioritization and planning decisions can improve the conservation outcomes (e.g., Ando and Langpap, 2018; Babcock et al., 1997; Murdoch et al., 2007; Naidoo et al., 2006; Newbold and Siikamäki, 2009,). Return on investment (ROI) analysis is an economic paradigm suitable for informing habitat restoration planning that involves quantitatively estimating the costs, benefits, and risks of restoration alternatives to prioritize approaches that maximize restoration benefits subject to a budget constraint[1] (Boyd et al., 2015). In this article we develop a framework for explicitly considering the intensity of habitat restoration interventions -- the amount of restoration undertaken in a given location and time -- when conducting ROI analysis, and apply the framework to restoration planning for a simulated population of endangered spring Chinook salmon (*Oncorhynchus tshawytscha*).

Prioritizing habitat restoration falls within a broader class of systematic conservation planning problems concerned with allocating scarce resources to achieve conservation objectives (Margules and Pressey, 2000). The origins of conservation ROI analysis are rooted in spatial prioritization for the design of biodiversity reserves (Newbold and Siikamäki, 2015), but it has since been applied to a variety of contexts including choices over which types of conservation interventions should be undertaken (Rose et al., 2016), the timing of those interventions (Costello and Polasky, 2004; Speir et al., 2015), and weighing tradeoffs among multiple ecological restoration objectives (Wilson et al., 2011).

Two characteristics typical of habitat restoration planning help to distinguish it from other conservation planning contexts. First, the objective of habitat restoration is typically conservation of a particular species or species group, and the specific habitat requirements of the focal species guide

[1] or that minimize the cost of achieving a conservation goal

restoration planning efforts.[2] If restoration plans are not designed to meet the specific habitat needs of a specific species or species group, they are likely to be piecemeal in nature, and potentially less effective at improving the population viability of target species (Miller and Hobbs, 2007). The emphasis of habitat restoration on recovering focal species is also a function of the institutions behind habitat restoration efforts; for example, habitat restoration planning for the Endangered Species Act in the United States and for similar policies elsewhere (e.g. the Species at Risk Act in Canada) is typically done on a species-by-species basis. Several prior studies identify cost effective habitat restoration plans for focal species using species-specific models of habitat requirements and empirical cost estimates (Goldstein et al., 2008; Newbold and Siikamäki, 2009; Rose et al., 2016). Second, owing to the multidimensional habitat requirements of focal species, habitat restoration planning often involves consideration of several intervention types, and of the amount of resources to expend on each type, across a set of candidate locations. This expands the scope of the planning problem beyond binary decisions over whether or not to establish a nature preserve, or to undertake a pre-specified intervention in a given location. Prior studies have compared the return on investment of alternative habitat restoration intervention types (Kimball et al., 2015; Rose et al., 2016), and others have defined conservation planning problems as a decision over the type, the location, and the timing of candidate interventions (Wilson et al., 2007).

This study focuses decisions over the amount of a given intervention type to undertake in a specified location and time period, termed intervention intensity, in the context of habitat restoration planning. Specifically, we define a planning problem where the objective is to maximize the

[2] We note that many habitat restoration projects yield co-benefits through ecosystem service provision and through conservation of ecosystems and habitats that support other species. The distinction here is whether those co-benefits are considered in restoration planning objectives.

expected future size of a focal population by allocating resources to a set of habitat restoration interventions which differ by intervention type, spatial location, and intensity.

In practice, the definition of intensity for a given intervention type depends on how location and timing are specified in the planning problem, and on the nature of the intervention type. In the watershed restoration context, for example, locations may be defined as an area, such as subwatersheds, or a distance, such as stream miles. The timing of allocations can be specified at some interval (e.g. annually) or as a one-off decision. With the location and timing specified, defining intensity for a given intervention type depends on if and how managers control the amount of an intervention type undertaken in a given location and period. One key consideration is whether interventions are passive in nature, such as restricting use of an area, or active, such as planting vegetation (Noss et al., 2009). For active interventions, intensity is measured as the amount of restoration (e.g. trees planted) per specified location and period. For passive interventions, intensity may translate, for instance, to the stringency of use restrictions, or other habitat protections, enacted in a given location, during a given period. In cases when managers cannot control the amount of an intervention taken in a given time and area, (e.g. binary decisions whether or not to enact a specified intervention in a given area) a single intensity level may be appropriate. Prior studies that consider intervention intensity include Withey et al. (2012), who addressed the problem of how much land preservation (intensity for a single intervention type) to undertake in each U.S. county (the specified location) simultaneously (timing) in order to maximize the number of species protected with a given budget.

In the context of habitat restoration, returns to scale refers to the proportionate increase in conservation benefits resulting from an increase in restoration effort. Increasing restoration effort corresponds to increased expenditure of the resources being allocated, and can be defined in a variety of ways. It can, for example, refer to increasing the intensity of restoration interventions, expanding the aggregate size of the locations where habitat is restored, lengthening the duration or frequency of a given intervention, or increasing the overall budget being allocated. Restoration exhibits increasing returns to scale when increases in restoration effort yield a greater than proportional increase in the benefits of restoration. Non-linearities, including thresholds, in the relationships between habitat factors and restoration benefits can cause certain restoration interventions to exhibit increasing or decreasing returns to scale (Goldstein et al., 2008; Lamberson et al., 1992; Wu and Skelton-Groth, 2002). Habitat thresholds may be related to the spatial extent of habitat or the condition of features in the landscape. In either case, thresholds may, for example, dictate that a minimum level of restoration effort is required before species recovery benefits materialize or that marginal benefits go to zero after a certain habitat state is attained. Moreover, interventions can exhibit nonlinearities in the relationships between restoration effort and restoration costs driven by economies (diseconomies) of scale (Armsworth, 2014; Cho et al., 2017). For instance, larger habitat restoration efforts may have lower average costs if startup costs are spread over a greater amount of restoration effort or if materials can be purchased more cheaply at larger quantities. Thus, inefficiencies may arise if nonlinearities in the effectiveness and costs of interventions are not considered (Wu and Boggess, 1999). Moreover, ecological thresholds are important to consider when societal values promote equitable distribution of restoration effort across areas, resulting in the potential for threshold habitat levels not being attained (Wu et al., 2003).

Intervention intensity is a topic paid less attention in the conservation planning literature compared to location, type, and timing considerations. We develop a restoration planning framework that incorporates habitat restoration intensity and then apply the framework to a case study of habitat restoration planning for ESA-listed Chinook salmon. Our analysis employs paired ecological models that account for the location-specific habitat requirements of the focal population to estimate the population change associated with each candidate intervention. In particular, we define the planning problem as deciding how to allocate finite resources to a set of restoration interventions that differ by intervention type, spatial area, timing, and intensity. The novel contribution of this study is an explicit focus on intervention intensity considerations in conservation planning and presentation of a case study demonstrating that incorporating intensity into planning decisions can increase the ROI of conservation investments. Additionally, this study illuminates characteristics and techniques typical of the habitat restoration planning context, including a focus on the outcomes of particular species, the use of ecological models that account for the specific habitat requirements of focal species, and the choice among several alternative restoration types. As is described in the following section, a need exists for the development of quantitative habitat restoration planning models to help inform the ongoing habitat restoration planning problem facing salmon recovery managers in the Pacific Northwest of the United States. Life cycle simulation models provide an opportunity for evaluating the expected responses associated with several types of restoration interventions that may have impacts across multiple life stages, and over time and space (Sharma et al., 2005), and a secondary contribution of this research is the development of a life cycle based habitat restoration planning model for a population of imperiled Pacific salmon that incorporates multiple intervention types and empirical estimates of the costs of those interventions.

Habitat Restoration Planning for Pacific Salmon Recovery

Habitat restoration is a central feature of recovery plans for imperiled Pacific salmon (NMFS, 2013a) but planning habitat restoration for salmon comes with inherent challenges (Barnas et al., 2015; Beechie et al., 2008). Restoration actions can be costly and, in the US Pacific Northwest alone, hundreds of millions of dollars are spent annually on interventions to restore and protect coastal and freshwater salmonid habitats (e.g. NPCC, 2017; Roni et al., 2017; SRFB, 2016). This focus on habitat restoration has created an ongoing cycle of funding allocation decisions by salmon recovery managers, who must evaluate, coordinate, and enact restoration interventions on short timelines, with limited data. Planning habitat restoration to recover imperiled salmon is further complicated by their complex life histories that are spread across multiple habitats and economic sectors (Bellanger et al., 2021). Evidence that some prior allocations of habitat restoration effort were not well-aligned with the recovery needs of salmon is unsurprising given the difficulties associated with salmon habitat restoration planning (Barnas et al., 2015).

Multiple approaches for prioritizing salmon habitat restoration are already utilized by recovery managers, but use of economic information and predictive ecological models is currently limited in those approaches. Specifically, salmon habitat restoration alternatives are commonly prioritized using simple heuristics, expert opinion, or quasi-quantitative scoring systems, depending on the objective and the level of information available (Beechie et al., 2008). Salmon recovery scientific advisory bodies and researchers increasingly support using predictive ecological models and economic information to increase the cost effectiveness of allocation decisions (ISAB, 2018). This support, along with the inherent challenges with salmon habitat restoration planning and its ongoing importance in recovery efforts, underscore the need for development of practical ROI prioritization methods.

Despite their rather limited use in practice, several studies have employed predictive ecological models to evaluate the ROI of salmon recovery interventions (Ettinger et al., 2021; Fullerton et al., 2010; Halsing and Moore, 2008; Newbold and Siikamäki, 2009; Null and Lund, 2012; O'hanley and Tomberlin, 2005; Ogston et al., 2015; Paulsen and Wernstedt, 1995; Speir et al., 2015; Watanabe et al., 2005). These studies have taken a variety of approaches (Supplement S1), but none have utilized coupled life cycle modeling and habitat assessment techniques. Life cycle models can account for salmon survival at each of their diverse life stages based on habitat characteristics of the locations where each life stage occurs. Locations identified to be population bottlenecks due to habitat characteristics that reduce survival through particular life stages can become targets for restoration (Honea et al. 2009). As empirical studies of ecological responses to on-the-ground restoration continue, modeling advancements in predicting salmon population responses to restoration interventions, and life cycle models in particular, will likely present an opportunity for expanding the deployment of ROI for salmon habitat restoration planning (e.g. Anderson et al., 2019; Pess et al., 2012; Polivka and Claeson, 2020). To that end, this research presents a tractable planning model capable of informing prioritization of habitat restoration for Pacific salmon that utilizes empirical cost data and ecological models that account for salmon habitat needs across their complex life histories.

Ecological thresholds are a topic of particular interest in Pacific salmon management (Munsch et al., 2020), and several studies have investigated the returns to scale of salmon habitat restoration. Newbold and Siikamäki (2009) found evidence of decreasing returns to scale for population viability from expanding the spatial extent of subwatershed protections to reduce non-point source pollution: even if selected at random, the first subwatersheds selected for protection tended to yield larger recovery benefits per unit area than subsequent protections. Fullerton et al. (2010) found evidence of increasing returns to scale in restoration intensity when benefits were specified in terms of population outcomes but constant returns to scale when benefits were defined as habitat-based metrics. A related finding from other studies was that using habitat-based benefit metrics to prioritize restoration interventions may create allocation inefficiencies (Newbold and

Siikamäki, 2009; Watanabe et al., 2005), underscoring the importance of developing models capable of predicting the relative impacts of restoration alternatives on recovery objectives.

Salmon habitat is frequently assessed in terms of limiting factors which can include temperature, sediment, or other habitat features (Smith, 2005). Decreasing returns to scale are expected for restoration interventions that modify habitat features (e.g. water temperature) beyond states where they are limiting (Kondolf et al., 2008). Increasing returns to scale in habitat restoration (Fullerton et al., 2010) imply that increases in restoration effort lead to proportionally larger increases in recovery benefits. Ecologically, these effects could relate to the presence of cumulative (threshold) effects, which have been observed for salmonid habitat (Li et al., 1994) and have implications for restoration prioritization (Wu et al., 2000). With regard to sequencing, Null and Lund (2012) found that the optimal amount and distribution of riparian restoration depended on whether or not other large restoration interventions were enacted first.

Methods

This section develops a framework for incorporating intensity into habitat restoration planning and then describes the methods used to apply the framework to a case study of habitat restoration planning for Pacific Salmon. We express the status of the focal species at the end of the planning horizon, $STATUS_T$, as a function of the set of habitat restoration interventions that are enacted, y, where the individual elements of y are given by y_{ijkt}, and are distinguished according to their type (i), location (j), intensity (k), and timing (t) .

$$STATUS_T = f(y) \tag{1}$$

The elements of y are selected from x, the set of candidate habitat restoration interventions. Across the planning horizon, the funds available for restoration in each period are given by the budget set $\{b_0, b_1, \dots, b_T\}$. Likewise, the costs of individual restoration interventions are given by c_{ijkt}. When budgets must be spent in the current period, the budget constraint imposes that restoration alternatives must satisfy: $\sum_{i=1}^{I}\sum_{j=1}^{J}\sum_{k=1}^{K} c_{ijkt} \leq b_t \forall t$.

Three assumptions were imposed to focus the analysis on the relationship between species outcomes and the intensity levels of individual interventions. We defined a one-timestep habitat restoration problem by setting the planning horizon (T) equal to one (Possingham et al., 2009), defined species recovery as the sole objective considered for habitat restoration, and ignored potential interdependencies among specified intervention alternatives in the selection process. These assumptions facilitate definition of a tractable routine for identifying portfolios of habitat restoration interventions that generate the largest possible expected increase in the focal population under a specified budget (2). The distinguishing feature of this framework is the incorporation of intervention intensity, which can be specified by the analyst to systematically probe the landscape for cost effective habitat restoration opportunities across a variety of intervention types.

$$\sum_{i=1}^{I}\sum_{j=1}^{J}\sum_{k=1}^{K} a_{ijk} STATUS_T(x_{ijk}) \tag{2}$$

Subject to:

$$i)\ \sum_{i=1}^{I}\sum_{j=1}^{J}\sum_{k=1}^{K} a_{ijk} C_{ijk} \leq b$$

$$ii)\ \sum_{k=1}^{K} a_{ijk} \leq 1\ \forall\ i,j$$

$$iii) a_{ijk} \in \{0,1\}$$

We applied this framework to salmon habitat restoration planning in a watershed within the Columbia River, the scene of a significant and ongoing habitat restoration effort to promote recovery of endangered salmon (NMFS, 2013b; UCSRB, 2007). The analysis was facilitated by linked ecological models that form a well-defined ecological production function, translating changes on the landscape associated with restoration interventions into predicted responses in the focal population of adult spring Chinook salmon. The first of the two ecological models established statistical relationships between landscape characteristics and habitat conditions for specific life stages of salmon in each subwatershed region (Jorgensen et al., 2009; Table 1). The landscape characteristics consisted of measures of landcover, including forest vegetation, impervious surfaces, roads, and geologic and geographic features such as elevation, gradient, subwatershed drainage area, and alluvial potential, as well as indicators of climate such as mean annual precipitation. Through this landscape characteristics-fish habitat linkage, changes in landscape characteristics occurring as a consequence of habitat restoration were translated into changes in fish habitat conditions for each subwatershed. The second ecological model uses habitat conditions produced by the first model along with estimates of habitat capacity for each subwatershed to estimate the number of spawning adult salmon returning to each subwatershed (Honea et al., 2009; Table 1). This spatially explicit life cycle model estimates the number of spawners returning to the watershed each year based on empirically derived relationships between habitat condition and fish survival through each life history stage—egg, fry, overwinter, smolt, ocean adult, upstream adult and spawner—depending on where the fish are at each stage and the habitat condition at those locations (Honea et al., 2009).

We specified a set of habitat restoration interventions that varied by type, intensity and location—but that all directly modified subwatershed features in one of the linked models (Table 1). We then simulated the restoration interventions and population responses that varied by type and recovery across 22 subwatersheds defined by 12-digit hydrologic unit codes (HUC6s; Seaber et al., 1987). The ecological models are based on a well-studied watershed (Wenatchee River, Washington State) with an endangered population of spring Chinook salmon to provide a realistic simulation. However, the analysis in this paper is meant as an example of what is possible through combining these types of models with return-on-investment analysis and is not designed at this time to be prescriptive.

Table 1. The coupled ecological models used for estimating the benefits of candidate habitat restoration interventions. The Jorgensen et al. (2009) modeling established linkages between landscape attributes (inputs) and life stage-specific fish habitat characteristics (outputs), and the Honea et al. (2009) life cycle model used the outputs from Jorgensen et al. (2009), and estimates of fish capacity, as inputs to estimate fish responses. Elements in italics indicate fixed inputs to the modeling that were unchanged by any of the management actions evaluated in our study.

Landscape-habitat modeling (Jorgensen et al. 2009)		Habitat-fish life cycle model (Honea et al. 2009)	
Inputs:	**Outputs:**	**Inputs:**	**Outputs:**
Subwatershed level[a] -*Elevation*[1] -*Gradient*[1] -*Area*[1] -*Precipitation*[2-3] -*Alluvium*[4] -*Impervious surface*[5] -Riparian forest cover[5] -Total forest cover[5] -Road density[6]	Subwatershed level -Prespawn temperature -Egg incubation temperature -Summer rearing temperature -Fine sediment -Cobble embeddedness	Subwatershed level -Outputs from Jorgensen et al. -Fry capacity[7] -Spawner capacity[7] System level -*Wild harvest*[8] -*Hatchery harvest*[9] -*Hatchery releases*[9] -*Mainstem Columbia migration up & downstream (i.e., Hydropower survival)*[10,11,12] -*Ocean conditions*[10,13]	Population level -Wild spawning adults (summation across all subwatersheds)

a = Inputs in italics are not affected by habitat restoration alternatives
Data sources: 1. USGS (1999); 2. Daly et al. (1994); 3. Daly and Taylor (2000); 4. WDGER (2005); 5. P. Murphy, unpublished data, U.S.D.A. Forest Service, Okanogan - Wenatchee National Forests Headquarters, 215 Melody Lane, Wenatchee, WA 98801; 6. US Census (2000); 7. ICTRT (2007); 8. Parties to US vs Oregon (2005) ; 9. Cooper (2006); 10. McClure et al. (2008); 11. Grant Grant-PUD (2003); 12. Skalski et al. (2005); 13. Cooney et al. (2002).

The candidate intervention types we specified were identified based on past restoration activity in the watershed and the costs are averages based on costs reported for interventions completed in the Wenatchee watershed from 2006-2011 (Table 2). Each intervention type modified landscape features (riparian forest cover, RFR; road density, RDD; upland forest cover, UFR) that influence fish habitat quality or habitat capacity (side channel reconnection, SCH; removal of culverts, a feature that blocked access to habitat, CLV) in the focal population.[3] Based on recent

[3] See supplementary material (S2) for additional details

habitat restoration interventions in the basin, we assumed that removal of each culvert increased habitat area by 1,000 m^2 and raised the carrying capacity in optimal habitat conditions by five additional potential spawning adults (Honea et al., 2009). Culvert removal alternatives were specified based on historical subwatershed habitat capacity rather than known habitat blockages, underscoring our exploratory approach intended to identify potential restoration opportunities rather than to evaluate vetted restoration proposals.

Table 2. Intervention types and associated cost estimates

Intervention type and abbreviation	Subwatershed feature modified	Unit cost estimate [a]	Source [b]
Riparian forest restoration (RFR)	Riparian forest cover	\$8.40/m^2	Average costs from eight interventions conducted from 2006-2011
Road decommissioning (RDD)	Road density	\$9,000/km	Cost estimate for intensive road decommissioning in Okanogan Wenatchee National Forest

Upland forest restoration (UFR)	Total forest cover	$3,000/km^2	Assumed one-time initial expense for mechanical treatments and prescribed burning then annual maintenance costs
Side channel reconnection (SCH)	Side channel area created or reconnected (fry capacity)	$90/m^2	Average cost of 15 interventions enacted in from 2006-2010
Culvert replacement (CLV)	Accessible spawning habitat (spawner capacity)	$150,000/culvert	Average cost of 30 culvert removal interventions conducted 2006-2011

a - Cost estimates in 2011 U.S. Dollars
b - See supplementary material (S2) for details on intervention definitions and cost estimates

We assumed that the marginal costs of restoration effort were constant when expressed in the units specified in the third column of Table 2. Because the landscape-level features affecting salmon populations—riparian forest cover, total forest cover, and road density—are expressed as proportions, the marginal costs of restoring those subwatershed features differed across subwatersheds but were constant within subwatersheds. A set of specified expenditure levels was translated into intensity levels for each candidate type and area combination (hereafter area-type) using the unit costs in Table 2. For our application, the specified expenditure levels were $150K, $600K, and $2.4M, which were used to define up to three discrete intensity levels—low, medium and high—for each area-type combination.[4]

The setup described above—five intervention types, each considered at up to three intensity levels across the 22 subwatersheds—implies that there are 330 potential interventions. To specify the set of candidate interventions considered in our analysis, we reduced this potential set by

[4] The low level corresponds to the assumed cost of removing a single culvert, and the high level roughly equals the total amount spent annually in the basin from 2010-2013 (2010-2013 Upper Columbia Salmon Recovery Board Annual Implementation Reports).

imposing bounds on the intensities considered. Intensity was bound from above so that interventions could not modify subwatershed features beyond the historical conditions estimated in Jorgensen et al. (2009), and bound from below by specifying a minimum expenditure for individual interventions. Imposing these bounds reduced the number of candidate interventions under consideration to 173. The minimum expenditure on any given intervention was set to $50K to reflect startup costs. The highest intensity level specified for a given area-type is either the change corresponding to a $2.4M expenditure based on the specified unit costs, or the change that was expected to return the conditions in the subwatershed to historical levels.

Budget sizes in our application were based on observed budget levels. From 2010 to 2013, the annual budget for restoration in the subbasin ranged from $1.1M to $4.3M, and there was no allowance for carryover of unused budgets (UCSRB, 2011, 2012, 2013, 2014). Using these amounts as guidance, we specified three possible budget levels, $1M, $2.5M, and $5M. The costs of many high-intensity interventions exceed $1M and are thus unattainable at this budget level.

We also specified a set of scenarios that constrained selection to the specified medium intensity interventions to provide a comparison to the multi-intensity scenarios. The medium-intensity-only choice scenarios may reflect prioritization strategies that limit the proportion of budget spent on any individual intervention.

We solved the problem defined in (2) and identified portfolios for each of the specified scenarios using the 'lpSolve' package (Berkelaar, 2020) for linear integer programming in R 4.0.3 (R Team, 2020). As a robustness check of the procedure's assumption of no interdependencies, the selected portfolios were fed through the modeling apparatus to estimate "portfolio models"—which do accommodate interdependencies—to investigate the substitutability or complementarity of selected actions.

A sensitivity analysis of the ecological model sensitivities to habitat intervention inputs was carried out to better understand which interventions have more or less uncertainty in their outcomes (Supplement S3). For example, interventions have the potential to exhibit a range of outcomes that may be consistent with what is known about a particular intervention type, or may point to further on-the-ground research and monitoring to better understand a specific type of intervention or a particular intervention-location combination. If, for instance, there is a big range in model outcomes to manipulating spawning or fry capacity, is this a result of known population dynamics responses to that type of habitat manipulation, or is the large range in the outcomes associated with less precise knowledge of how fish populations respond to this type of habitat action? Resolving answers to questions like this will improve the effectiveness of recovery planners' choices of interventions.

Results

The portfolios of interventions identified using the procedure described in the previous section yield insights into the habitat thresholds limiting salmon survival in the watershed and demonstrate how incorporating intervention intensity into planning can promote cost-effective habitat restoration the presence of such thresholds. Results presented in this section reflect our estimates of the long-run change in the adult spring Chinook salmon population associated with undertaking each of the candidate interventions independently, the costs of those interventions, and their return on investment (Figure 1).

A majority of the potential interventions that were considered were either excluded due to the restrictions placed on intervention intensities (grey tiles, Figure 1), or did not produce additional spawners in the ecological models (tiles with zero values, Figure 1). The large proportion of interventions that were either excluded or produced no biological response imply that naively selected interventions are likely to be ineffective.[5] One intervention type, road decommissioning, produced several of the largest ROI values among the interventions considered. Culvert removal, side channel reconnection, and upland forest restoration interventions were also effective at increasing salmon production for some subwatersheds and intensities. The models predicted that riparian forest restoration, which among other things shades streams from warm sunlight, would be ineffective at increasing salmon production across all areas at the specified intensity levels.

The intervention types we simulated exhibited heterogeneity in ROI across subwatersheds and with respect to intensity in a particular subwatershed (Figure 1). No single action-type or area dominated, and various returns to scale relationships emerged across the candidate interventions. A low-intensity intervention, road decommissioning in area 203, returned the highest estimated ROI, while a high-intensity intervention, road decommissioning in area 304, produced the second-highest estimated ROI (Figure 1, lower right). Likewise, the potential interventions exhibit both increasing and decreasing returns to scale. Road decommissioning in subwatersheds 302, 304, and 502 exhibited increasing returns to scale, shown in the lower right panel of Figure 1 by an increase in ROI from low to high intensity for RDD, while road decommissioning in subwatersheds 203 and 301 exhibited decreasing returns to scale, where ROI for RDD decreases with increasing intensity. These returns-to-scale relationships reflected differences in baseline habitat conditions across subwatersheds as well as non-linearities in the habitat-survivorship curves specified in the biophysical models and in the habitat change associated with a given intensity level and subwatershed.

We solved the planning problem defined in equation (2) for six different scenarios that varied by the total budget available for allocation, and whether multiple intensity levels were considered

[5] Of note, there is no contemporary production of spring Chinook salmon in subwatersheds 400 and higher.

per area-type combination. The selected portfolios contain interventions of several types and intensities across several subwatersheds (Figure 2), emphasizing the importance of considering interventions that vary along these margins to facilitate cost-effective habitat restoration planning. Road decommissioning in subwatershed 203 is selected in each of the portfolios, but not at high intensity even though that intensity produced the second highest total benefits. The reason is that this intervention exhibits decreasing returns to scale driven by non-linearities embedded in the habitat models, and at high intensity, the ROI declines to the point where it is no longer cost effective.

Examining the multi-intensity results across the budget, ROI peaked at seven additional spawners per $50,000 invested in the $2.5M budget scenario, and then declined to five additional spawners per $50,000 invested in the $5M budget scenario (Table 3). The portfolio model results were similar to the aggregated individual results used in the optimization routine. For example, modeling the interventions selected in the multi-intensity scenarios simultaneously instead of aggregating the modeled returns of individual interventions led to five fewer additional spawners (86) in the $1M scenario, the same increase in spawners (358) in $2.5M budget scenario, and six fewer additional spawners (499) in the $5M budget scenario. Together these results suggest that interactive effects considered in the ecological models were not of great influence in the selected portfolios.

The scenarios constrained to selecting only medium-intensity interventions (second row, Figure 2) provide a comparison against the scenarios that allow for variation in intervention intensity. Under a $1M budget, most of the high-intensity interventions were unattainable, and allowing the interventions of a given area-type to vary by intensity did not lead to a higher return on investment for the selected scenario. However, considering multiple intensities increased ROI by five additional spawning adults in the population per $50000 invested under a $2.5M budget and increased ROI by over three additional spawning adults per \$50000 invested in the \$5M budget scenario.

In the sensitivity analysis, the largest change in spawner number (>+10%) all occurred in an area (304) where *percent fines* was relatively high and constrained survival (Supplement S3). Another relatively large change in spawner number (>-10%) occurred in another area (202) as a result of decreased survival due to an increase in *percent fine sediment* and a decrease in *water temperature* during the incubation period. Variation within the 10% and 90% quantile range in water temperature during the spawning period or the fry rearing periods had no additional impact on survival in any area for any intervention. The lower and upper bounds of the interventions that targeted capacity, CLV and SCH (removing culverts and increasing side-channel habitat, respectively), altered spawner numbers very little (<0.1%) compared to life cycle results from the estimated intervention values.

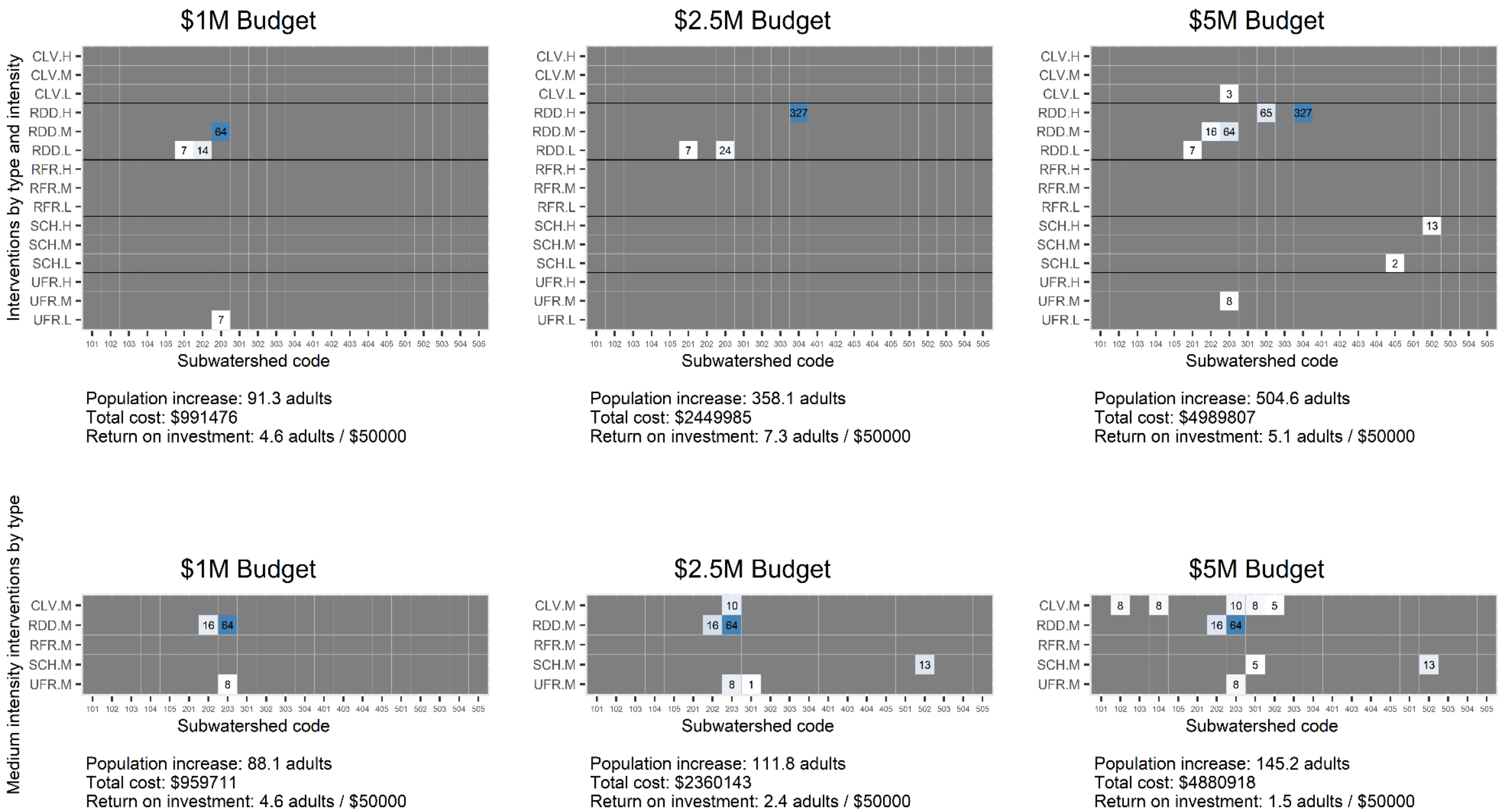
Figure 1. The expected returns, costs, and return on investment associated with the habitat restoration interventions under consideration. Individual interventions are defined by the subwatershed where they are undertaken, the intervention type, and the intensity level. The upper left panel depicts the subwatershed areas under consideration. The code associated with each subwatershed forms the horizontal axis of the remaining panels. The vertical axis of these panels depicts distinct intervention type-intensity combinations. The intervention types considered include culvert removal (CLV), road decommissioning (RDD), riparian forest restoration (RFR), side channel construction (SCH) and upland forest restoration (UFR). The interventions are specified at low (L), medium (M) and high (H) intensity levels. Grayed out boxes represent places where either current conditions equaled historical conditions or potential interventions exceeded historical conditions. Interventions exhibit increasing returns to scale when return on investment (in purple) increases moving from low to high intensity and exhibit decreasing returns to scale when ROI decreases moving from low to high intensity.

Table 3. Individual and combined effects of multi-intensity habitat restoration portfolios

Budget scenario	Intervention type and intensity[a]	Subwatershed code	Estimated population increase	Estimated cost (2011US$)	Estimated ROI (Increase/ $500000)
$1M	RDD.M	203	64	600000	5.3
	RDD.L	202	14	150000	4.7
	RDD.L	201	7	91000	3.8
	UFR.L	203	7	150000	2.3
	Sum of individual interventions		*91*	*991000*	*4.6*
	Portfolio model[b]		*86*	*991000*	*4.3*
$2.5M	RDD.L	203	24	150000	8.0
	RDD.H	304	327	2209000	7.4
	RDD.L	201	7	91000	3.8
	Sum of individual interventions		*358*	*2450000*	*7.3*
	Portfolio model		*358*	*2450000*	*7.3*
$5M	CLV.L	203	3	150000	1.0
	RDD.H	304	327	2209000	7.4
	RDD.M	203	64	600000	5.3
	RDD.M	202	16	179000	4.5
	RDD.L	201	7	91000	3.8
	RDD.H	302	65	915000	3.6
	SCH.L	405	2	65000	1.5
	SCH.H	502	13	600000	1.1
	UFR.M	203	8	181000	2.2
	Sum of individual interventions		*505*	*4990000*	*5.1*
	Portfolio model		*499*	*4990000*	*5.0*

a - Abbreviations: CLV-culvert removal, RDD-Road decommissioning, SCH-side channel reconnection, UFR-upland forest restoration
b - Model with budget-scenario interventions considered simultaneously

Figure 2. Portfolios of interventions selected under each scenario considered. Each potential intervention is represented by a grid cell, and grid cells associated with selected interventions display the expected increase in the adult spawning population, rounded to the nearest integer. Grayed out boxes represent interventions that were not selected. The scenarios in the top row define the set of alternatives to include interventions across three intensities. The scenarios on the bottom row define the set of alternatives to include medium intensity interventions only. A summary of overall performance is provided beneath each scenario plot.



Discussion

The planning framework developed and applied in this paper systematically considers the conservation benefits from interventions of various types undertaken at different intensities in a given area. If planners can estimate costs and benefits for all conceivable intensity levels, then the set of candidate interventions for a particular area-type traces out a curve depicting the relationship between recovery status and restoration intensity. In practice, planners may need to specify discrete intensity levels to maintain tractability in the planning problem. Our application utilized linked ecological models that produce a well-defined ecological production function. To maintain tractability, we specified three discrete intensity levels to probe restoration investment returns along the intensity gradient based on realistic levels of restoration effort and habitat change. Incorporating intervention intensity, in addition to intervention type and location as a selection criterion, facilitates identification of cost-effective interventions in the presence of non-linearities in the benefits and costs of interventions. The application presented in this paper focused on non-linearities in the relationship between salmon survival and habitat restoration intervention effort, known as non-constant returns to scale. However, the presented framework is also appropriate for considering non-linear cost responses associated with economies of scale or non-constant marginal costs. The utility of the presented framework is underscored by the fact that it is not possible to calculate the cost-effective intensity level for a given area-type in isolation, without considering simultaneously the returns associated with other specified interventions and the available budget.

Intensity is particularly important to consider when (1) budgets are larger than the costs of individual candidate interventions, rendering allocation of all resources to a single action-area based on ROI impossible, (2) when allocating resources to single actions results in modifications to habitat conditions well beyond limiting thresholds, and (3) when a certain amount of intensity is required to achieve a habitat threshold. In these cases, algorithms designed to allocate all funds to a single intervention per budget period may not produce cost-effective habitat restoration plans.

If only one intensity level is considered per intervention type and area ($K = 1$) then differences between the portfolio selected using (2) and the portfolio generated by simply selecting interventions with the highest ROI until the budget is exhausted are attributable to budget remainders that can occur when the latter method is employed (Duke et al., 2013). When more than one intensity level is considered ($K > 1$), the optimal portfolio can differ from iteratively selecting interventions with the highest ROI because the benefit-maximizing intensity level for a given area-type is not always the one with the highest ROI.

The habitat restoration planning approach utilized for this study is attractive for several reasons in addition to its accommodation of intervention intensity as a choice dimension. In particular, the framework reflects the features typical of habitat restoration planning, including a species-specific conservation objective and decisions among several intervention types across a set of possible locations. Likewise, the benefits estimated for candidate interventions are derived from ecological models that account for the location-specific habitat needs of the focal population of Chinook salmon, and the estimated unit costs are based on costs reported for past restoration interventions undertaken in the study watershed. Another attractive feature of this approach is that the selection algorithm is computationally tractable and can readily be operationalized with ecological models that are already complete or in development.

The limitations of this study mostly arise from the assumptions imposed to simplify the specified habitat restoration planning problem. In particular, the approach assumes no interdependencies among candidate interventions and does not incorporate time or sequencing considerations. Concerns that substitutability in the effects of selected interventions may result in reduced effectiveness in the selected portfolio are partially eased by the portfolio model results, which indicate that the estimated increase in focal population when the entire portfolio of selected interventions is undertaken in unison is comparable to the summation of all of the individual intervention benefits estimated separately (Table 3). While these results suggest that the interventions in the selected portfolios are not highly interdependent, the specified model does not incorporate information about interdependencies when selecting restoration portfolios. Data and modeling limitations in our case study also precluded an analysis of the timing of interventions, uncertainty in restoration outcomes, and the evolution of habitat threats over time. While our case study did not address these factors, we briefly note their importance below. Intervention timing can be an important consideration, if, for instance, interventions with short-term habitat enhancements are required to mitigate extinction risk while ecosystem processes are restored in the long run. Likewise, managers typically do not know the costs and benefits of interventions or future funding levels with certainty and the nature of this uncertainty can influence planning strategies. Threats can evolve over time as a result of changing climate conditions (Pressey et al., 2007), proliferation of invasive species, or habitat conversion, and the evolution of threats is another potentially important consideration in habitat restoration planning.

In addition to relaxing the assumptions of intervention independence and a one-timestep planning problem, future ROI models of habitat restoration planning may improve model performance through development of more sophisticated models of restoration costs (Armsworth, 2014; Burkhalter et al., 2016; Naidoo et al., 2006), and through exploring alternative definitions of the restoration objectives. Consideration of cost non-linearities with respect to intensity, analogous to the ecological non-linearities considered in the application, is a straightforward methodological extension, as is the incorporation of potential cost interdependencies across interventions. Another worthwhile future direction for habitat restoration ROI models is to consider benefit metrics that incorporate economic non-market values (Duke et al., 2013; Iftekhar et al., 2017) as well as non-economic societal values such as equity and cultural values (Breslow, 2014). Notably, multiple studies have estimated the non-market values of Pacific salmon recovery that could be leveraged to express habitat restoration benefits in terms of public welfare (Anderson and Lee, 2013; Bell et al., 2003; Lewis et al., 2019; Wallmo and Lew, 2012). Moreover, definitions of restoration benefits can also be enhanced by leveraging indigenous and local knowledge of particular ecosystems (Walsh et al., 2020).

Exploratory habitat restoration ROI analyses such as the application presented in this article are typically not specified at a level of data granularity sufficient to characterize conditions at particular candidate restoration sites. Likewise, in practice, these types of exploratory analyses are helpful for identifying habitat restoration opportunities and informing the allocation of restoration resources at the landscape scale, rather than for direct selection of site-specific interventions proposed by project sponsors. Due in part to data limitations and a lack of integration in governance across resources and jurisdictions, development of habitat restoration policies is commonly framed as a multistage process that involves social and political, as well as technical inputs (Baker and Eckerberg, 2013). When habitat restoration planning proceeds in a multi-step process, exploratory ROI analysis can inform finer-scale evaluations in subsequent steps. In particular, when data constraints make it cost prohibitive to develop specific proposals for all feasible interventions (Phillips-Mao, Refsland, & Galatowitsch, 2015) exploratory ROI analysis can inform the solicitation and development of specific proposals, including evaluating intervention feasibility, refining cost estimates, and negotiating or incentivizing participation from landowners. This paper demonstrated the use of exploratory ROI for informing prioritization of interventions by location, type and intensity. Subsequent applications may further explore the appropriate role of exploratory analysis and data collection in linking the stages of planning to facilitate cost-effective conservation. In particular, this could involve assessing whether available data and models are sufficient to prioritize actionable restoration alternatives, and if expected benefits undertaking exploratory analysis or data collection outweigh the costs of these activities. It may also involve integrating ROI methods with other prioritization systems such as expert opinion, as these combined approaches can improve planning outcomes (Langhans et al., 2016). For example, the results of the application may then be integrated expert opinion guidance to further prioritize interventions within subwatersheds.

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