Technical Supplement: Determining Net Ecological Benefit

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1. INTRODUCTION

A. CONTEXT

The 2018 law (Engrossed Substitute Senate Bill (ESSB) 6091, codified as RCW 90.94, required the Department of Ecology (Ecology) to determine that a **Net Ecological Benefit (NEB)** will result when adopting and approving:

- Watershed plan updates, as required under RCW 90.94.020.
- Watershed restoration and enhancement plans under RCW 90.94.030.
- Water resource mitigation pilot projects under RCW 90.94.090.

Interim guidance (Ecology, June 2018) was developed to inform and evaluate plans that are completed within the following twelve months, or later if there is prior agreement with Ecology, and for pilot projects being conducted under RCW 90.94.090. To assist the agency in their development of a final guidance, Ecology developed a consultation with an academic research team affiliated with the Washington Water Research Center at Washington State University to support the technical aspects of the interim guidance. This report is the product of the academic team. The final NEB guidance will be used to evaluate the remaining plans submitted to Ecology later in 2019 through 2021.

Under RCW 90.94.020 and RCW 90.94.030, the completed plans must, at a minimum, recommend actions to offset the potential consumptive impacts of new permit-exempt domestic water uses to instream flows. Before plans are adopted, Ecology must determine that actions identified in a plan, after accounting for new projected domestic uses of water within a water resource inventory area (WRIA) over the next twenty years, will result in a NEB to instream resources within that WRIA.

RCW 90.94.090 authorizes Ecology to issue permit decisions for a series of water resource mitigation pilot projects. Those pilot project proposal evaluations involve issuance of municipal water right permits rather than permit exempt wells. Therefore, the content of this report will focus on planning and evaluations conducted under RCW 90.94.020 and RCW 90.94.030 only.

B. THE NEB DETERMINATION

In essence, the NEB process under RCW 90.94 is a ***transaction***; plans will be evaluated to see if, given a forecast environmental impact from consumptive water withdrawals, there are sufficient forecast offsets from management actions, to meet or exceed those water withdrawals. Specifically in the Interim Guidance Ecology defines NEB as:

> "*A **Net Ecological Benefit** determination means anticipated benefits to instream resources from actions designed to restore streamflow will offset and exceed the projected impacts to instream resources from new water use.*"

Thus, the transaction will amount to a comparison in the quantity and quality of anticipated instream resources prior to water withdrawals and following the deployment and maturation of offset projects. To evaluate this transaction we need to be clear regarding what instream resources are relevant, and how to structure the assessment of the transaction.

Ecology defines instream resources as:

> *"Ecology interprets "**instream resources**" in the context of this provision of ESSB 6091 to include the instream resources and values protected under RCW 90.22.010 and RCW 90.54.020(3)(a), with an emphasis on measures to support the recovery of threatened and endangered salmonids."*

The references to existing rules add the following:

> From RCW 90.22.010: *The department of ecology may establish minimum water flows or levels for streams, lakes or other public waters for the purposes of protecting fish, game, birds or other wildlife resources, or recreational or aesthetic values of said public waters whenever it appears to be in the public interest to establish the same*.

> From RCW 90.54.020(3)(a): *Perennial rivers and streams of the state shall be retained with base flows necessary to provide for preservation of wildlife, fish, scenic, aesthetic and other environmental values, and navigational values. Lakes and ponds shall be retained substantially in their natural condition. Withdrawals of water which would conflict therewith shall be authorized only in those situations where it is clear that overriding considerations of the public interest will be served*.

While the available language potentially encompasses a diversity of ecosystem goods and services (e.g. "recreational or aesthetic values"), the evaluation of instream resources will focus on endangered salmonids.

This is emphasized in the proposed rule for Chapter 173-566 WAC – Streamflow Restoration Funding, which will establish process and criteria for funding projects under Chapter 90.94 RCW which includes the following definition:

> *"Instream resources" for the purposes of this chapter means fish and related aquatic resources.*

In Washington State, consideration of instream flow generally focuses on salmon and trout to a significant extent, as well as on other instream values. Salmon and trout are the most evident native fish in most Washington freshwaters and have high cultural, economic, and recreational importance, as well as being important ecologically (food for other wildlife, transporters inland of marine-derived nutrients that fertilize riparian vegetation (Ben-David et al. 1998; Helfield and Naiman 2001; Naiman et al. 2002; Shaff 2005), and as geomorphic modifiers (Kondolf and Wolman 1993; Macdonald et al. 2010). Based on this focus, this technical supplement will likewise focus on fish and fish habitat aspects of evaluating instream resources.

C. PURPOSE OF THIS DOCUMENT

This document primarily serves as **technical supplement** to the Department of Ecology's final NEB guidance, and does not supersede information provided in Ecology's final guidance. Ecology intends to distribute final NEB guidance that will inform diverse issues related to the development of proposals, and may include objectives for and descriptions of the planning process, requirements for proposals and process for proposal evaluation. This document in contrast, will not address requirements, and is only intended to provide technical support for the ecological assessments that are part of the NEB process. In particular, while this document describes what information content ***may*** be included in the proposals, it does not require specific information be present in all proposals, and indeed, does not define adequacy nor standards for what will be deemed sufficient by and for the Department of Ecology.

It is also a critical distinction that this document is intended to inform and support proposals and planning in response to RCW 90.94, rather than the implementation process for those plans. This distinction has a number of implications. Clearly each subbasin presents unique sets of opportunities and constraints in terms of managing instream resources that need to be examined by each planning group. Each planning group will know its basin best and it is impossible for this planning document to anticipate all of those opportunities and constraints ahead of time. Most importantly however, the role of monitoring and adaptive management will differ between planning and implementation. In the planning process scientific information, presumably collected with effective monitoring, can inform the models, forecasts and assessments in an NEB determination. However, once plans are implemented, additional monitoring may be required to evaluate project effectiveness, validate that performance targets are met, and inform the decision to deploy contingencies in the event that performance targets are not met (Crawford 2007). The incorporation of monitoring into a management loop consisting of: performance target identification, project implementation, monitoring, and management course correction based on monitoring, is termed adaptive management (Walters 1986, 1997) and is likely to be an important component of implementation. Although planning is distinct from implementation, there are both roles for the information produced with monitoring and opportunities for efficiencies in planning that come from consideration of monitoring in the planning process. Therefore, this report includes a brief discussion of monitoring and adaptive management below.

The framework for this document is derived from work produced for the Canadian Science Advisory Secretariat (CSAS) that sought to provide guidance on evaluating offsets from fisheries-related management in relation to any harm development projects may cause to fish and fisheries (Bradford et al. 2014). That report describes an approach to assessing "equivalency" from impacts and offsets, and applied several approaches to evaluating equivalency. The notion of equivalency is similar to NEB, but it derives from statutes which are themselves different in Washington and Canada. Therefore, we do not coopt the technical definition of equivalency in this document. However, Bradford et al (2014) do provide a number of useful insights. Specifically, they recognize the general distinction of In-kind/In-place/In-time offsets on the one hand, to a diversity of approaches to out-of-kind/place offsets on the other[1].

Across the regulatory and research literature on environmental impact mitigation the terms “in-time” and “in-place” are commonly used. However, each regulatory situation may impose unique definitions based on jurisdictional and planning constraints. In Washington State relative to water right permits, the term “in-place” refers to mitigation that is located in the same place as the pumping impacts. However, RCW 90.94 applies to permit-exempt wells and not permitted wells, and relative to offsets for permit-exempt wells in this law does not have in-time or in-place requirements. RCW 90.94 does describe highest priority offset actions as being capable of “replacing the quantity of consumptive water use during the same time as the impact and in the same basin or tributary”, and Ecology has referred to this latter locational constraint as meaning within the same sub-basin. Thus, within the law “Lower priority projects include projects not in the same basin or tributary and projects that replace consumptive water supply impacts only during critical flow periods”. However, beyond this, RCW 90.94 allows great latitude in where offset projects can be located and what the timing of the benefits will be - provided that collectively the plan will achieve a Net Ecological Benefit.

Therefore the spatial domain used in RCW 90.94 is larger than other uses of the term “in-place” -particularly in the research literature broadly, and Bradford et al. (2014) specifically. For the purposes of this document, there are places where the terminology “In-place” is used equivalent to “same basin or tributary” consistent with RCW 90.94, and there are other places where the term “in-place” is used in same sense as the research literature more broadly – with the text noting areas where one or the other situation applies.

Bradford et al. (2014) recognize the diversity of out-of-kind/place/time approaches based on offset goals and modeling frameworks. We have leveraged these insights here and adopted their organization of offset approaches in this document. We use this definition and the conceptual foundations provided by Bradford et al. 2014 within the context of the decades of salmonid research done in the U.S. Pacific Northwest. As such, this discussion outlines pathways forward for developing scientifically defensible plans to estimate 1) the harm new development may inflict on fish, and 2) the efficacy of proposed offset projects towards preventing, reducing or offsetting harm in Washington State.

Given the unique challenges and opportunities present in each watershed in Washington State, and the diversity of approaches to NEB determination described below, this document does not dictate specific actions to be taken. Rather, this document is meant to provide a scientific framework from which planners and regulators can understand the current state of knowledge. We attempt to identify the most scientifically rigorous method in a given area for estimating the

[1] Separate from defining the spatial boundary of “In-Place”, the meaning of In-Kind/In-place is meaningful only if mitigation is contemporaneous or performed over relevant time scales. The ecological responses of habitat to mitigation may operate on different time scales than the impacts of water withdrawals and if those scales are very different, it may be difficult to associate responses with impacts in the same place and of the same response type. Therefore, while In-Kind/In-Place is used here to be consistent with other nomenclature, it should be understood to be In-Kind/In-Place/In-Time unless otherwise noted.

potential harm and potential benefits to endangered salmonids, other fishes and other instream resources despite continued watershed development.

The following sections outline the steps needed to estimate NEB and discuss anticipated issues associated with monitoring and diverse spatial scales, as well as the rationale for making out-of-kind NEB comparisons. These are followed by a review of five approaches for establishing NEB and the merits and limitations of each. These five approaches include: 1) In-kind/In-place Habitat Replacement, 2) Habitat Function Replacement, 3) Habitat Capacity for Single Species Replacement, 4) Fish Abundance Replacement and 5) Fish Production Replacement[2].

The authors of this report do not have detailed knowledge concerning the level of resources, expertise and research sophistication available to each planning unit. Our professional experience suggests that the resources and expertise available to the planning units will vary widely. For example, the Pacific Northwest is one of the most sophisticated natural resource management domains on this planet, with research, monitoring and evaluation expertise built on Endangered Species Act, Northwest Power Planning Act, Northwest Forest Plan and other regulatory framework legacy experiences. Therefore, in some cases the capacity to perform sophisticated NEB determinations may be quite high. At the same time RCW 90.94 addresses a new framework, with new expectations, and some planning units may find they do not currently possess the tools to perform a detailed, demanding NEB determination as described below. In these cases, planners may perceive the information that follows in this report to be somewhat demanding. We feel however, that it is important to describe the approaches to NEB determination that would represent a contemporary and comprehensive approach, with the understanding that some groups may not exploit that comprehensiveness, rather than compose a report that was narrow in approach, based on lower expectations for regional expertise, and fail to provide guidance to planning groups that did possess more capacity.

In the past, when subbasin planning or recovery planning groups and agencies have lacked expertise or access to appropriate monitoring data, detailed research products have been substituted with subjective assessments, often labelled “best professional judgement” or similar. It is likely that in the case of NEB determinations performed under the RCW 90.94 process this will also be the case for some planning groups. Planning units and the Department of Ecology will need to resolve what expectations are appropriate for each planning group, and where the recruitment of additional expertise is justified and available.

D. STEPS IN A NET ECOLOGICAL BENEFITS DETERMINATION

NEB determination is composed of four key parts as defined by the Interim Guidance:

1. **Characterize and quantify potential impacts to instream resources from the projected 20-year new domestic permit-exempt water use at a scale that allows meaningful determinations of whether the proposed offset is in-time and/or in the same subbasin.**
2. **Describe and evaluate individual offset projects.**

[2] Fish Production Replacement refers to the productivity of fish in the habitat. It does not refer to hatchery supplementation of fish.

3. **Explain how the planned projects are linked or coordinated with other existing plans and actions underway to address existing factors impacting instream resources.**
4. **Provide a narrative description and quantitative evaluation (to the extent practical) of the net ecological effect of the plan.**

1) Characterize and quantify potential impacts to instream resources from the projected 20-year new domestic permit-exempt water use at a scale that allows meaningful determinations of whether the proposed offset is in-time and/or in the same subbasin

Planning groups must evaluate “potential impacts to instream resources” as the losses that must be counterbalanced by the proposed offsetting measures. Of particular interest are those spatially- and temporally-dependent changes in stream flow resulting from consumptive use withdrawals that may have impacts on fish and fish habitat. Impacts on fish may not be uniform across space, time or fish life-stage (e.g. decreases in flow can have a positive impact on some life-stages of fish but negative impacts on others, or mitigation actions may take multiple years to take effect—see below). Thus, estimates of *net* impacts should be determined and quantified for each impact type in each phase of a proposed activity across the forecasted 20-year time horizon. This may include determining the extent, duration, and magnitude of the impacts on fish and fish habitat in terms of the reduced fish numbers, area of habitat lost, area of habitat permanently altered and degree of alteration. There are several approaches to making these forecasts (see below); this document outlines the benefits and limitations of these available methods.

2) Describe and evaluate individual offset projects.

All proposed offsetting measures should include details about the design, implementation, and desired outcomes for the NEB determination. The desired outcomes should be determined by the forecasted potential impacts to instream resources in Part 1 of this section. The NEB determination should include clearly defined measures of success that are linked to the desired outcomes of the offset projects, and be expressed as metrics that can be monitored to evaluate effectiveness.

Potential designs for offset project metadata are provided in an accessory appendix (Appendix 1). The metadata dictionary provides examples of metrics for describing the magnitude, location, and extent of each offset project and a rationale for the key information needs listed.

The steps involved in describing and evaluating projects are discussed in Ecology’s Interim Guidance. Once all of the projects have thus been characterized, it is useful to distinguish between those that are “in-time” and “in-same-subbasin” versus those that are “out-of-time and out of-same-subbasin”.RCW 90.94 establishes a hierarchy of priority for actions (projects) aimed at offsetting the impacts of consumptive domestic permit-exempt well use:

- Highest priority are projects that replace consumptive domestic water use impacts during the same time and in the same subbasin as the impacts occur.
- Lower priority are projects that replace consumptive domestic water use impacts elsewhere within the WRIA or only during critical flow periods.

"In-time and In-same subbasin" offsetting refers to situations in which the water used for permit exempt domestic well consumptive use is replaced by the same quantity and quality of water in the same place—where place is defined as the same subbasin. Additional habitat offsetting may potentially be required to account for uncertainty and time lags. The benefits of in-kind offsetting are assumed to accrue to the fish populations affected by the project. In these situations, balancing the losses to fish and fish habitat caused by a project with the benefits that result from offsetting measures can be a straight-forward calculation. The calculation is based on the impacts, water use, and the comparability of offsets both in terms of the metrics used to describe them and the affected fish populations.

With "out-of-same-subbasin" or "out-of-time" offsets, offset projects address factors limiting fish productivity in a given area, but not by replacing what has been lost. Rather, offsets meet or exceed those losses with increased production elsewhere. Out-of-same subbasin/time offsetting measures may include the restoration or creation of habitat types that are different from the habitat type that was lost, or other types of measures. This is sometimes referred to as off-site mitigation. Measuring and comparing losses with offsetting gains can be more complicated in out-of-same subbasin/time offsets, as the transaction relies on a correct understanding of the relationship between habitat alterations in a given location and a fish productivity response. This has been challenging to demonstrate and it is an important limiting assumption (see Locke et al. 2008 and Monitoring and Evaluation section below).

3) Explain how the planned projects are linked or coordinated with other existing plans and actions underway to address existing factors impacting instream resources.

This step is principally an administrative activity, and the Interim Guidance provides additional details and rationale for coordination within other management plans as well as with other ongoing habitat and fish management within their WRIA's and sub- WRIA planning units. Planned offset projects may indeed benefit in terms of greater environmental benefit if they are planned, designed and implemented in coordination with partners. Effective coordination is likely to leverage a larger set of resources and reduce overall cost per unit ecological benefit.

Notwithstanding the desired outcome that more coordination will produce greater environmental benefit, it is also likely that in some cases there will be a state of diminishing returns. For example, if water temperature is a critical concern for salmon in a given WRIA, and 100 out of 130 miles of the riparian corridor have been addressed previously with restoration, then the next 10 miles may not be as effective in increasing fish populations as the first 10 miles of riparian revegetation, nor as effective as 10 miles of riparian revegetation in a different corridor that has been untreated.

In any case, siting offset projects in the context of historical habitat management and coordinating with on-going management will be critical for supporting the forecasts of potential impact and offsets described in each NEB Determination.

Each WRIA-based planning group is likely aware of much of the habitat management actions occurring within their WRIA. However, additional sources of information on planned and implemented projects can be obtained from known data holders on the following list:

Data Holder	Location	Phone	Web url
Columbia Basin Fish and Wildlife Authority - CBFWA	851 SW 6th Ave # 250, Portland, OR 97204	(503) 229-0191	https://www.cbfish.org/
Nisqually Indian Tribe	Nisqually Tribe 4820 She-Nah-Num Drive S.E. Olympia, WA 98513	(360) 456-5221	http://www.nisqually-nsn.gov/index.php/administration/tribal-services/natural-resources/habitat-restoration/
NOAA – Pacific Coast Salmon Recovery Fund	7600 Sand Point Way, Seattle, WA 98115	(503) 230-5419	https://www.westcoast.fisheries.noaa.gov/protected_species/salmon_steelhead/recovery_planning_and_implementation/pacific_coastal_salmon_recovery_fund.html
NOAA Fisheries Community Based Restoration Center	7600 Sand Point Way, Seattle, WA 98115	(360) 902-2603	https://www.fpir.noaa.gov/HCD/hcd_restoration.html
NOAA Restoration Center	7600 Sand Point Way, Seattle, WA 98115	(360) 902-2603	https://www.westcoast.fisheries.noaa.gov/habitat/restoration_on_the_wc.html
NOAA Pacific Northwest Salmon Habitat Project Database	Northwest Fisheries Science Center 2725 Montlake Blvd. East Seattle, WA 98112	(206) 860-3362	https://www.webapps.nwfsc.noaa.gov/apex/f?p=409:13:::::
Nooksack Salmon Enhancement Association	3057 E. Bakerview Road Bellingham, WA 98226	(360) 715-0283	http://www.n-sea.org/contact (also see: https://wdfw.wa.gov/about/volunteer/rfeg/)
Ocean Trust	1000 Padre Blvd. Suite 528 South Padre Island, TX 78597	(703) 434-1444	https://www.oceantrust.org/contact/
South Puget Sound Salmon Enhancement Group	6700 Martin Way East, Suite 112 Olympia, WA 98516	(360) 412-0808	http://spsseg.org/contact-us/
Stillaguamish-Snohomish Fisheries Enhancement Task Force	425.252.6686 PO Box 5006, 2723 Hoyt Ave Everett, WA 98206	(425) 252-6686	http://www.stillysnofish.org/
United States Army Corps of Engineers	PO Box 3755 Seattle, WA 98124-3755	(206) 764-3742	https://www.nws.usace.army.mil/
United States Bureau of Land Management	Bureau of Land Management 333 S.W. 1st. Avenue Portland, OR 97204	(503) 808-6002	https://www.blm.gov/or/programs/fisheries/salmon_habitat_mgmt.htm
United States Fish and Wildlife Service	911 NE 11th Avenue Portland, OR 97232	(509) 548-2985	https://www.fws.gov/pacific/fisheries/HabitatRestorationMain.cfm

United State Forest Service – Regional Ecosystem Office	1220 SW 3rd Avenue Portland, Oregon 97204	(503) 808-2851	https://www.fs.fed.us/r6/reo/nwfp/
Washington State Department of Fish and Wildlife – Habitat Program		(360) 902-2534	https://wdfw.wa.gov/conservation/habitat/
Washington State Salmon Recovery Funding Board (SRFBD)	1111 Washington Street S.E. Olympia, Washington 98501	(360) 902-3000	https://www.rco.wa.gov/boards/srfb.shtml

4) Provide a narrative description and quantitative evaluation (to the extent practical) of the net ecological effect of the plan.

Similar to the forecasts of potential impacts to instream resources identified in Step 1 of this section, planning groups will also need to forecast anticipated net ecological effect of their planned offset projects. Also similar to Step 1, these forecasts can be performed with a variety of approaches, some of which are identified below. Importantly, each approach must address the following technical issues:

a) The forecasted benefits from offset projects need to meet or exceed the potential environmental impacts to stream resources;

b) Recognition that uncertainties exist on several scales

   a. *Uncertainty in offset magnitude*: Given that the magnitude of offset effects are uncertain, the magnitude of total planned offsets may need to be increased in the plans in order to increase likelihood that net offsets exceed impacts of withdrawals;

   b. *Uncertainty in timescale of response*: Recognizing that although not explicitly considered under the streamflow law, there may be time lags between the implementation of a project and when the potential benefits to instream resources may manifest, but the negative impacts of consumptive water withdrawals may occur in the near-term, the magnitude of the offset projects may need to be increased. For example, if the impacts of a set of withdrawals occur over years 1-20, but the benefits from offsets are only manifest in years 10-20, then the magnitudes of the offsets would need to be larger than the impacts at any moment in time for the NEB to net out positive at the end of the planning horizon;

c) Contingency measures for the event that offsets are not reaching performance targets, timelines for evaluating triggers for those contingencies, and management decision process for employing those contingencies should also be identified if the offsetting measures do not meet expectations.

2. ISSUES COMMON TO ALL NEB DETERMINATION APPROACHES

1) Monitoring

As mentioned above, this guidance is intended to address planning rather than implementation needs. However, we can anticipate opportunities and constraints presented by implementation feeding back into the planning process. One of those issues is monitoring and adaptive management. It is appreciated that that monitoring is a key component of management action implementation, but it may be less clear how monitoring fits into planning exercises prior to management action deployment.

Plans that anticipate the realities of implementation, such as uncertainty, risk and management decision making, are most likely to be successful. Monitoring is a critical tool in addressing these realities, and is most effective when incorporated into an adaptive management framework. As mentioned above, there are several scales of uncertainty in forecasting NEB both in terms of impacts of water withdrawals and the impacts of offset projects. The potential that planned projects will not generate a positive NEB is an important risk associated with these uncertainties. Monitoring of offset impacts informs evaluation of progress in meeting plan objectives, and when invested in a framework for making management decisions can determine if plans need to be modified to reach targets. Importantly, monitoring of offsets can determine if performance targets are not being reached and triggers for contingencies are required prior to an unsuccessful project completion. Incorporating monitoring into this loop of performance target identification, project implementation, monitoring, and management course correction based on monitoring, is termed adaptive management (Walters 1986, 1997; Katz et al. 2007). By allowing informed course corrections over the 20 year time horizon of the NEB process, monitoring is a critical to reducing the risk that NEB will be negative.

For these reasons, projects that include an explicit effectiveness monitoring plan[3] should garner greater deference in determining project benefits. In addition, the State of Washington and the region more broadly have recognized the critical role of monitoring in validating fish-habitat response models as well as validating the effort and significant investments in habitat management across the Pacific Northwest. The precepts behind monitoring across the region are summarized in the ***Coordinated Habitat Action Effectiveness Monitoring*** guidance from the Pacific Northwest Aquatic Monitoring Partnership (PNAMP), to which Ecology is a partner agency. The following is an excerpt that underscores the role of, and commitment to, effectiveness monitoring:

> *"Habitat action effectiveness monitoring is a critical component of performance tracking and adaptive management needs of the Pacific Coastal Salmon Recovery Fund (PCSRF), the Columbia Basin Fish and Wildlife Program, the 2008 NOAA Federal*

[3] Action effectiveness studies [=effectiveness monitoring] look at "cause and effect" relationships between management actions and improvements to fish survival and/or environmental conditions. In other words, these studies help evaluate whether actions for fish are achieving their biological objectives. https://www.salmonrecovery.gov/Evaluation/ActionEffectiveness.aspx

> *Columbia River Power System (FCRPS) Biological Opinion, several other regional Biological Opinions, and several federal, state and tribal mitigation programs. The current habitat action effectiveness monitoring and assessment strategies being implemented under these Programs requires a combination of project implementation monitoring, project level and watershed scale effectiveness monitoring, along with habitat/fish status and trend monitoring. This information will support a tool box of various habitat and fish population relationships and models that can be used to make assessments and inferences about the effectiveness of various actions. For these strategies to succeed, the components need to be coordinated with compatible and well documented metrics, methods, and designs and balanced across different categories of action types within limited budgeting available for this type of information." PNAMP 2010*[4]

Effectiveness monitoring plans should include:

- Clearly articulated models of the managed system, where 'model' refers to a description of the environmental system that includes hydrologic and ecological process and allows a specific forecast for the effect of the implemented management action in terms that can be monitored with current methods.
- Clearly defined and reportable benchmarks of success and time lines that can be used to determine success in reaching NEB, as well as recognizing when NEB is not being achieved and contingencies must be triggered.
- Methods and designs consistent with effectiveness monitoring guidelines or plans in place across the region (e.g. PNAMP).
- The same level of transparency present in other aspects of the NEB determination.
- Coordination with other status and trends and effectiveness monitoring programs within their planning domain and in adjacent planning domains to ensure interoperability and maximal efficiency with respect to multiple NEB determinations.

Although monitoring has been described here in the context of validating the performance of offsets, it is also likely that monitoring will inform some of the estimates of impacts. However, the degree to which monitoring influences the estimates of impacts from water withdrawals will be highly variable among the approaches to NEB determination described below. This variability makes it difficult to specify the characteristics of monitoring for this objective. However, if monitoring is a component of impact assessment as well as offset forecasts, that monitoring should be designed to generate data that is interoperable between those activities.

2) Accuracy, Precision & Transparency

[4] Crawford, B., J. O'Neal, M. Newsom and J. Geiselman. (2010). Coordinated Habitat Action Effectiveness Monitoring. PNAMP Available at https://www.pnamp.org/document/3039 (accessed Nov. 5, 2018)

The RCW 90.94 interim guidance indicates that plans should characterize relevant uncertainties in their estimates of impacts and offsets in each NEB determination. This direction is specific:

> "*Uncertainty of benefits should be identified and quantified to the extent possible*."

How then to characterize uncertainty? Every model is a simplification of a true and complicated process. As such, it is important to understand both the sources and magnitudes of uncertainties that are part of any model prediction. Uncertainty in this context can actually arise from several aspects of the analyses used to forecast impacts and offsets.

In general, there are two primary elements to uncertainty that one needs to consider: accuracy and precision (fig 1). ***Precision is how close predictions or observations are to each other***. Models that make predictions that are tightly clustered together are said to have high precision, and this is expressed statistically as low "variance". ***Accuracy on the other hand, is how close a prediction is to the truth***. Models that make predictions that are consistently close to the true values are said to have high accuracy, and this is expressed statistically as low "bias". It is important to keep these sources of uncertainty distinct for several reasons. They are statistically different, they affect interpretations differently, and they have different origins such that minimizing one or the other requires different alterations to our methods. For example, there are times when increasing the amount of data can increase precision, but it usually has no effect on accuracy. As a consequence, there are times when no amount of more data will get an answer with more accuracy. In addition, it must be recognized that they are independent in that either one or both can be high or low at the same time. This is illustrated in figure 1 below.

An ideal model has both high accuracy and high precision, but in practice this rarely happens. Especially with models, there are good reasons for the presence of trade-offs among the two. Relatively simple models may have fairly high precision, but low accuracy, especially when applied to novel data. Increasing the complexity of models can increase accuracy, but at the cost of decreased precision.

Several features of the interaction between precision and accuracy can be illustrated with the following simple example of fitting a curve to estimate an underlying process. Consider a simple relationship, or process, in the environment (gray line in each panel of fig 2a-c), that we observe at a sample of X values (indicated by black dots). The values of the dots are determined by the process and a little bit of uncertainty, or error in the data. So the black dots are the data in hand, and the process is what we are trying to model, and in each panel of the figure they are the same. Now if we fit the data with a simple linear model with 2 parameters (the slope and intercept) we get the straight red line in figure 2a. The points deviate from the red line quite a lot (low precision), but the line represents the underlying process fairly accurately; the overall trend is going down with increasing X over this range of X values. In the far right panel the process is estimated with a much more complex model that has 7 parameters (6 polynomial exponents and an intercept). Here the red line goes through all the points, and so is very precise (indicated by the high value of r^2), but the red line is not a very accurate description of the process given all the diversions and "waviness". In the middle, we fit the data with a polynomial with 4 parameters.

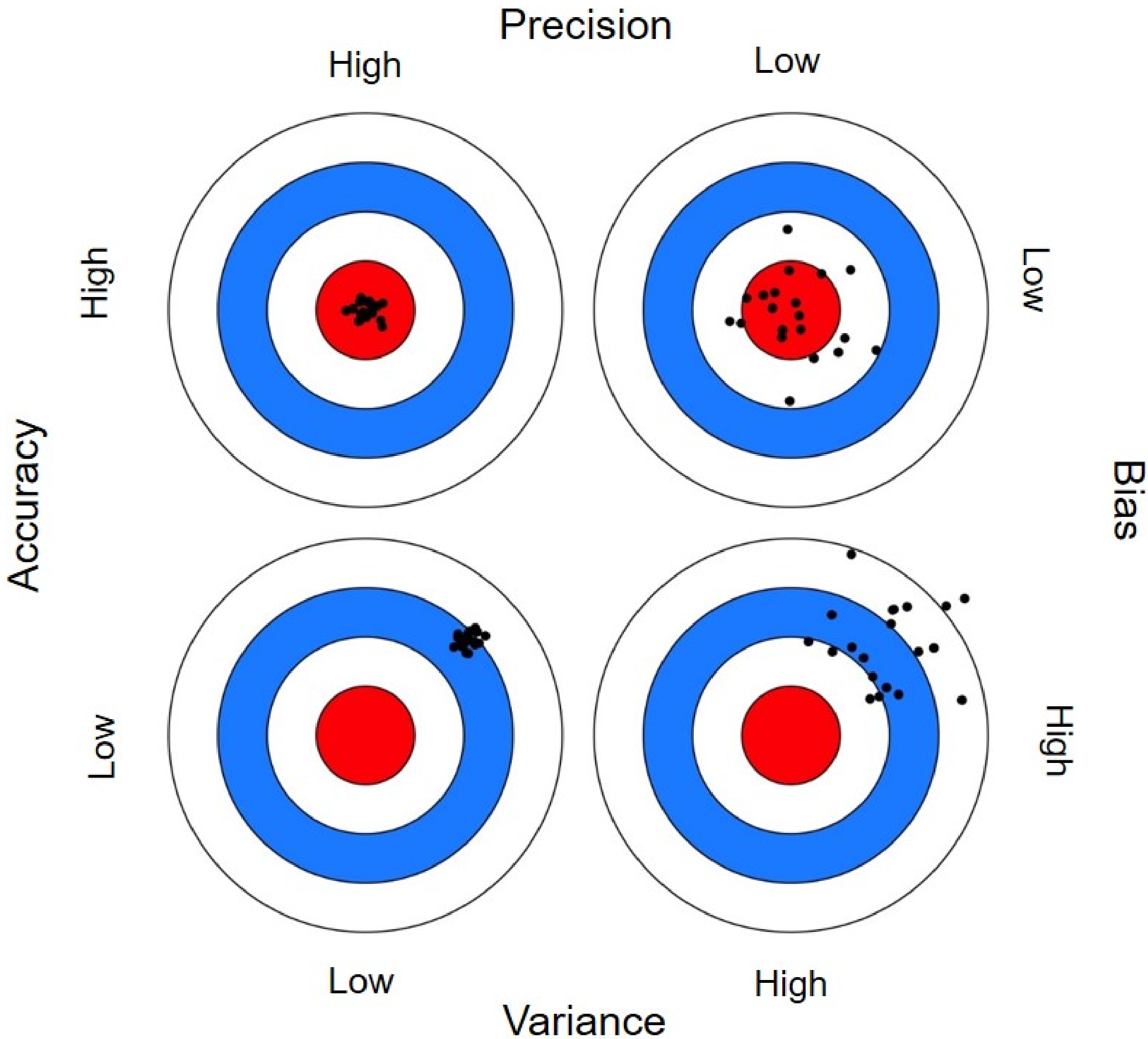

Figure 1 Diagrammatic representation of accuracy and precision. The distribution of holes in the targets is an expression of the statistical properties of accuracy and precision in order to convey 1) that they are independent properties in that either may be high or low regardless of the other, and 2) that they have different impacts on how model forecasts are viewed. In this framework, the bull's eye on the targets represent what is happening in reality, and the bullet holes represent the model descriptions of that reality. In the best of all outcomes, models would be accurate and precise and the bullet holes would all be clustered closely at the center of the target (model=reality). The fact that precision and accuracy are separate and independent means that models can be poor reflections of reality in multiple ways.

This example of a trade-off illustrates a general principle. As model complexity increases, precision generally increases, but accuracy may decrease. The interpretation is that the more complex models are better at approximating all the data, but they perform more poorly at estimating the underlying process. In between is an optimum where the Total Error arising from both imprecision and inaccuracy are at a minimum. This trade-off is graphically summarized in figure 2d. A second implication of the trade-off in figure 2d is that while there is a minimum Total Error, it never goes to zero. So our models will never be free from uncertainty; the question is can we develop useful models and can we make choices among models that get us as close to the minimum as

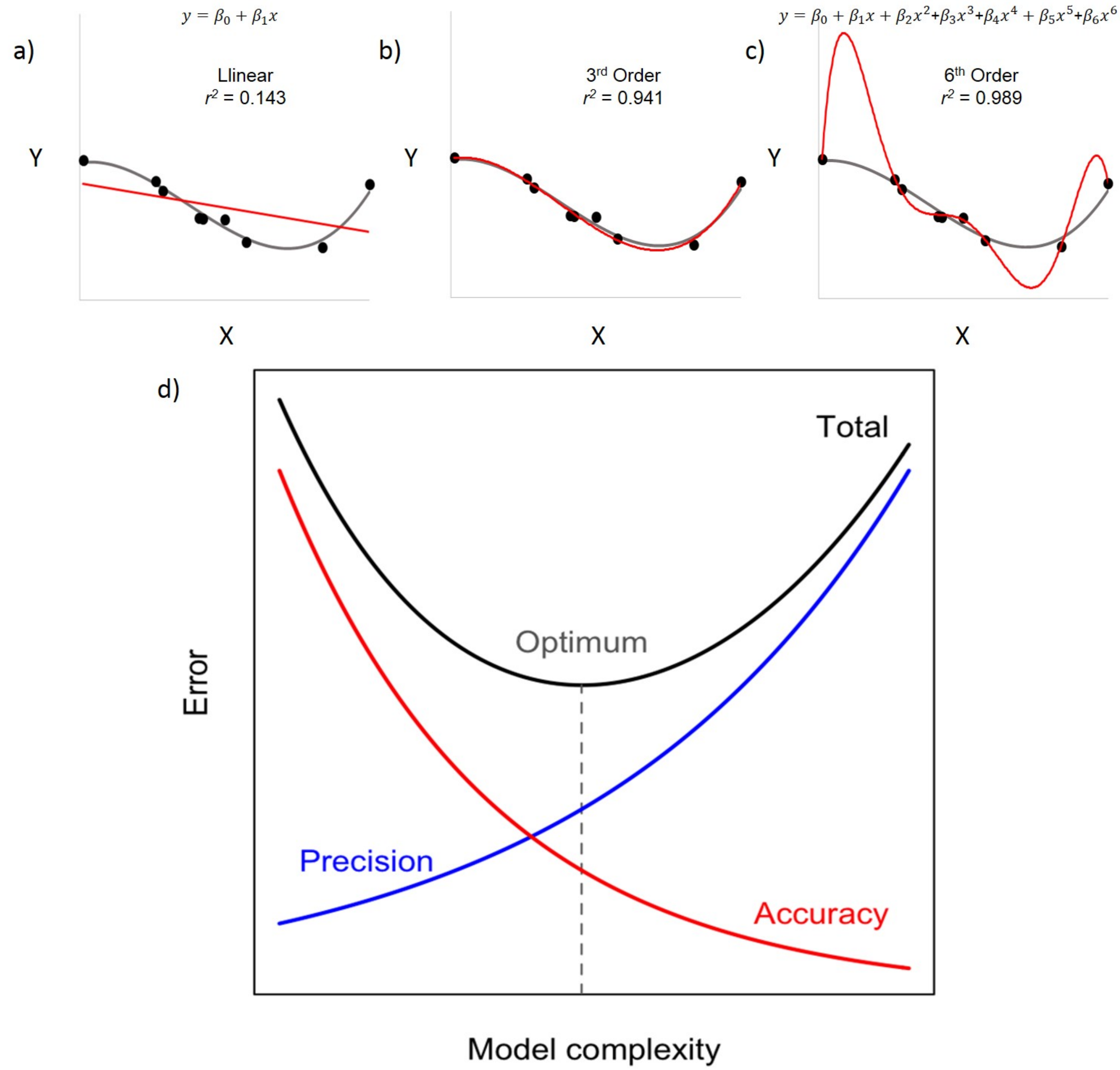

Figure 2 Illustrations of relationship between model complexity and total uncertainty or error. A) A simple process (gray line) observed at various values of X (black dots), and fit with a linear regression model (red line). The form of this most simple model is above the curve. B) The same process and observations as in A, but fit with a 3rd order polynomial as a model of intermediate complexity. C) The same process and observations as in A and B, but fit with a 6rd order polynomial as a model of high complexity. The form of this most complex model is above the curve. D) The generic trade-off between errors due to loss of accuracy and increased precision as model complexity increases (total error is the sum of the other two components). The dashed vertical line indicates the optimum level of model complexity.

possible. In practice, the optimum is often wide—a range of complexity will give similar total error—and spending large efforts to acutely optimize model complexity is not a useful expense of effort. However, especially in the context of fish-habitat association modeling and salmon recovery very complex models are in use with potentially profound inaccuracies. Therefore, planners and reviewers need to be conscious of the choices they make with respect to approaches to NEB determination, and then be clear on how they report those choices.

In the context of NEB determination, this trade-off has an additional important implication. As stated above, ideally we would like both precision and accuracy. In many research contexts however, one is likely interested in describing the data in hand and so precision is very important. In this case, one may tolerate a more complex model to achieve a better “fit” to the data. In contrast, in the present context NEB determinations need to project impacts and offsets out into the future and so one must prioritize a better understanding of the underlying process. The most complex models in use can make very precise forecasts (i.e. 165,500 Coho salmon from a given sub basin), and this is often seductive in a management context where future ecological status is at stake. But those same models can make wildly inaccurate forecasts because of that same complexity. In fig. 2c for example, the model fits the data well, but the 6th order polynomial deviates wildly from the process; would it be prudent to use the red line to make a forecast of where the gray line is going to be at some far outlying value of X? Likely not.

This discussion of precision and accuracy has focused on the implications of model complexity at the level of making choices among NEB determination approaches (see below). Once one has decided on a specific approach to NEB determination, it is possible to then continue to evaluate the complexity of models and tactics to increase precision and accuracy that are related to data quality, sampling design and statistical estimation techniques. All of which are potentially important, but which are also likely to vary quite widely from WRIA to WRIA. That level of model selection and optimization will involve location-specific detail that exceeds the scope of this guidance. However, it is likely that at small scales the benefits of an intensive optimization process will be marginal, and the decisions regarding specific choice of impact and offset forecast will be made based on the data, expertise and other resources at hand to make the determination.

These uncertainties are important, and the design choices that are made have interacting implications for total uncertainty in the impact and offset forecasts, but at the same time planners will need to make judgments about what they need and what they can do to develop their NEB determinations. This reality makes it important to be transparent with respect to what choices are made and how. The interim guidance states:

> *“… plans will provide a **transparent**, structured evaluation to be used in Ecology’s NEB analysis to determine whether the requirement in ESSB 6091 has been met. If the planning group concludes that the planned projects recommended in the plan will achieve NEB, the plan should include a clear explanation and justification for that conclusion.”* (emphasis added)

In this context, transparent means that all methods and assumptions are reported. This would include descriptions, sources and magnitude of bias and uncertainties that affect the impact and offset forecasts. At a minimum, this would include the uncertainties that arise from data, model choice and estimation methods. As noted above and various places in this guidance however, some of the choices made on adopting one approach or another involve complex trade-offs between technical issues, but also practical constraints. Therefore, a transparent description of the methods and approaches taken should also identify where choices were made and what constraints may have been in place to guide those decisions. As demonstrated in figure 2d, it may not be possible to reduce uncertainty to zero, therefore it is critical to document what efforts were taken to address uncertainty throughout the NEB determination process.

3) Ecological Context, Scale and Critical flow periods

The interim guidance defines high and low priority projects, and instructs planners that viability of proposed projects will be evaluated in an ecological context. Ecological context in this case refers to the scales, environmental conditions and scope of biodiversity relevant to the fish affected by consumptive water withdrawals. Specifically, the guidance includes:

> *"Where highest priority projects are not feasible, ESSB 6091 authorizes plans to include lower priority projects—those that do not occur in the same subbasin or tributary (but are within the same WRIA) or only replace water during critical flow periods. To determine the viability of a lower priority water offset project, planning groups will need to determine critical flow periods. The critical flow period determinations should consider fish presence and distribution, and the historic hydrograph (synthesized hydrograph if necessary)."*

Ecological context matters to NEB determinations in a number of ways. Location and setting will be important for both high and low priority projects. Location and scale are important both to correctly account for the ecological system being managed, but also because the approaches to NEB determination described below all have dependencies on data and data aggregation that are affected by location, timing and scale. For high priority projects (i.e. In-the same subbasin and In-time), a clear description of the extent of water use and offset is needed to evaluate the offset equivalence, and determine NEB. The guidance also refers to different opportunities for offsets at scales from the tributary to the subbasin and WRIA scales. Therefore, planners need to be clear about the spatial extent of their projects and impacts if their plans are to be evaluated appropriately. For low priority projects, location and scale will be similarly important as environmental conditions subject to proposed offsets are heterogeneously distributed across the landscape. Accounting for habitat capacity or fish production equivalence and substitution will require habitat descriptions of similarly high detail to capture that heterogeneity.

In addition to the scale dependencies of ecological data, ecological context can also affect the scale of ecological process that impacts instream flow. For example, a specific reduction in stream flow (e.g. 0.75 cubic feet per second, cfs) is likely to have a larger impact on a smaller tributary than a larger river. Alternatively, a given withdrawal may have a larger impact on habitat (i.e. the environmental correlates of stream flow) if taken higher in the watershed than closer to the confluence of the tributary to a large stream. Yet the effect of a withdrawal can also be diminished as other tributaries or groundwater are added downstream.

In addressing ecological process, planners may need to aggregate withdrawals along tributaries, weighted by the tributary stream flow, catchment size, geomorphology and adjacent withdrawals. It is hard to predict how complex such schemes are likely to get across the diversity of stream networks within the geographic range of Washington WRIA's, but planners will need to incorporate at least the basic watershed characteristics listed above in their plans.

Figure 3 is a map of the state of Washington that identifies the watersheds covered under this planning process. These watersheds cover a wide swath of the state and the ecological conditions are distinct among them. These ecological differences will affect the impacts of consumptive water withdrawals from permit exempt wells.

As mentioned above, the magnitude of stream flow changes are anticipated to vary widely from WRIA to WRIA, with some obvious and others almost imperceptible. If one is evaluating withdrawal impacts on a large tributary the effect of a fraction of a cfs change is likely to be very small and perhaps technically challenging to demonstrate. WRIA's dominated by rain inputs in the western portion of the state (e.g. WRIAs 12,14, 22 & 23), may commonly experience localized areas of low flow in the late summer and fall. Here a small cfs reduction could have large impacts seasonally. On the other hand, the Little Spokane and Colville subbasins (WRIAs 55 & 59), while having regulated flows do not support listed salmonids, and so the determination of NEB will likely be made on a basis other than anadromous fish impacts (see RCW 90.22.010 and 90.54.020). This is an additional component to ecological context, and it will create both challenges and opportunities for NEB determinations.

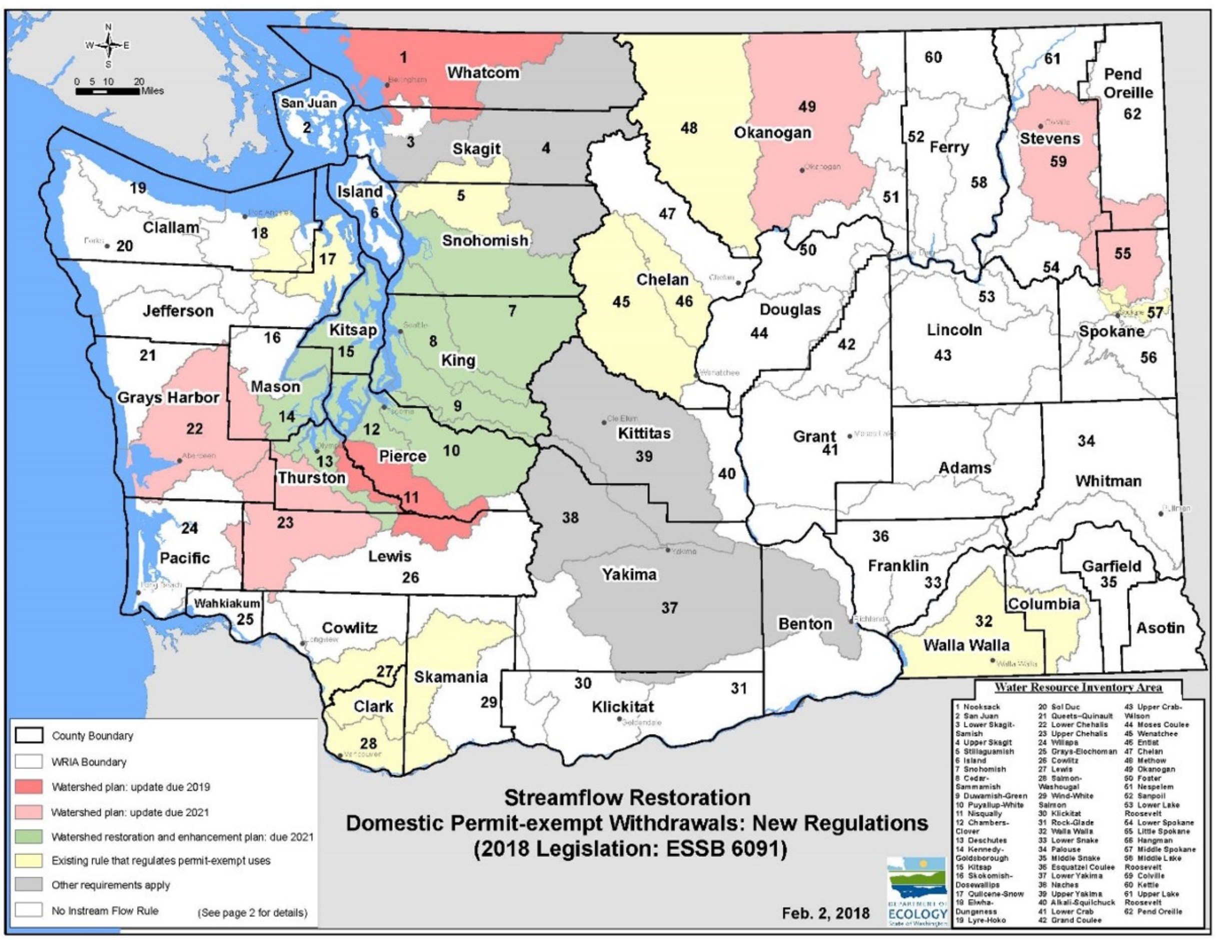

Figure 3 Planning Watersheds under RCW 90.94. The WRIAs submitting plans under section 202 are in red and pink based on timing. WRIAs submitting plans under section 203 are in green.

In several places, the interim guidance refers to critical flows as an acknowledgement that the magnitude of stream flows can change significantly during different seasons. In general, stream flows are highest in the winter when precipitation is highest and are lowest in the late summer when precipitation is low or zero and flows are supplied by snow melt, groundwater, or reservoir release. These seasonal patterns are often expressed in the characteristic hydrograph for the stream under study. Periods of low flow are likely to be critical periods for fish. However, it is an oversimplification to suggest that critical flows are times of low stream flow. Anadromous fish life histories are diverse, and involve complex patterns where different species use different habitat at different life stages for different ecological objectives. This makes it difficult to identify a single time and circumstance that is uniquely “critical”.

While the literature on differential habitat use by different species is voluminous and summarized in detail elsewhere (Groot and Margolis 1991), highlighting some specific patterns of fish-habitat relationships can help give context to how withdrawal decisions and fish production may be linked.

Salmon and trout spawn in gravel, burying their eggs below the surface of the gravel, where the eggs stay for a number of weeks or months (depending on temperature) as they develop. Flow generally changes during this incubation period while they develop. If adults spawn in a deeper, mid-channel area because they spawned while flow was unusually low and a flood occurs during incubation, many eggs could be lost to flood scour, resulting in lower production (Tripp and Poulin 1985; Thorne and Ames 1987; DeVries 1997, 2000; Lapointe et al. 2000; Ames and Beecher 2001). Conversely, flow reduction during incubation could result in no water going through the egg pocket in the gravel, a risk that is greatest if incubation occurs during declining flows or if spawning occurred when flows were particularly high (Hawke 1978; Becker et al. 1982, 1983; Reiser and White 1983; Reiser 1990; Connor and Pflug 2004). Given that the volume of groundwater withdrawal by new permit-exempt domestic wells are anticipated to be relatively small, impacts to fish spawning in small streams (e.g. cutthroat trout, coho salmon, some chum salmon) are the most likely to be negatively impacted.

Riffles, the shallowest areas along the length of streams, can be sufficiently shallow to hinder or even halt migration when stream flow is at its lowest (Locke et al. 2008; Grantham 2013). When flow reduction at riffles coincides with upstream spawning migration of salmon and trout, adults can be blocked from reaching spawning areas (Thompson 1972; Smith 1973; Locke et al. 2008; Warren et al. 2015; Holmes et al. 2016), and mortality can be increased through exposure to predation, energy depletion, and injury and infection. Pink salmon, summer chum salmon, fall Chinook salmon, and bull trout may all migrate upstream during late summer and early fall when flows can be lowest.

Lowering stream flow can also impact young fish prior to out migration. Stream flow reduction can impact rearing fish by (1) reducing suitable habitat area and volume, (2) reducing overall system productivity and food transport, and (3) reducing water quality. Coho salmon and cutthroat trout that rear in small streams through summer low flows can be adversely impacted by flow reductions (Brown and Hartman 1988; Beecher et al. 2010; Vadas Jr et al. 2016). Steelhead and Chinook salmon rear in somewhat larger streams, but they are also sensitive to flow reduction. As habitat area and volume are reduced, fish crowding may result in density-dependent reduction in growth and condition, leading to lower survival (Harvey and Nakamoto 1996; Bailey et al. 2010). When flow reduction coincides with higher temperature, water quality (including dissolved oxygen) can also be adversely affected by flow reduction (Elliott 2000). This is particularly true in riparian wetlands, with large surface areas and shallow depths, but which provide important rearing habitat for coho salmon (Brown and Hartman 1988; Swales and Levings 1989; Henning et al. 2006; Jeffres et al. 2008; Rosenfeld et al. 2008; Katz et al. 2017).

Here the major associations between species, life history stage, stream order and usage are summarized in the following table (sources listed above, summarized in Groot and Margolis, 1991):

Species & lifestage	**Small streams**	**Medium streams**	**Large streams**	**Very large streams**	**Largest streams**
Pink salmon adult migration		Early fall	Early fall	Early fall	
Pink salmon spawning & onset of incubation		Early fall	Early fall	Early fall	
Pink salmon incubation		Fall & winter	Fall & winter	Fall & winter	
Pink salmon fry emergence & seaward migration		Early spring	Early spring	Early spring	
Chum salmon adult migration	Late fall (fall chum salmon)	Early (summer chum salmon – Hood Canal, eastern Straits) & late fall(fall chum salmon)	Early (summer chum)& late fall (fall chum)	Late fall (fall chum)	Late fall (fall chum)
Chum salmon spawning & onset of incubation	Late fall (fall chum)	Early (summer chum) & late fall (fall chum)	Early (summer chum) & late fall (fall chum)	Late fall (fall chum)	Late fall (fall chum)
Chum salmon incubation	Winter	Fall (summer chum) & winter (both)	Fall (summer chum) & winter (both)	Winter (fall chum)	Winter (fall chum)
Chum salmon fry emergence & seaward migration	Spring	Spring	Spring	Spring	Spring
Sockeye salmon adult migration	Fall	Fall	Fall	Fall	Fall
Sockeye salmon spawning & onset of incubation	Fall	Fall	Fall	Fall	

Sockeye salmon incubation	Winter	Winter	Winter	Winter	
Sockeye salmon fry emergence & lakeward migration	Spring	Spring	Spring	Spring	
Sockeye salmon rearing in lake	Year-round	Year-round	Year-round	Year-round	
Sockeye salmon smolt migration from lake to sea	Spring	Spring	Spring	Spring	Spring
Coho salmon adult migration	Late fall	Late fall	Late fall	Late fall	Fall
Coho salmon spawning & onset of incubation	Late fall	Late fall			
Coho salmon incubation	Winter	Winter			
Coho salmon fry emergence	Spring	Spring			
Coho salmon rearing	Year-round	Year-round	Year-round	Year-round	Year-round
Chinook salmon adult migration		Spring, summer, early fall	Spring, summer, early fall	Spring, summer, early fall	Spring, summer, early fall
Chinook salmon spawning & onset of incubation		Summer, early fall	Summer, early fall	Summer, early fall	Summer, early fall
Chinook salmon rearing		Year-round	Year-round	Year-round	Year-round
Steelhead adult migration		Spring, summer, fall, winter; spring	Spring, summer, fall, winter; spring	Spring, summer, fall, winter; spring	Spring, summer
Steelhead spawning & incubation		Spring	Spring	Spring	
Steelhead rearing		Year-round	Year-round	Year-round	Year-round
Cutthroat trout adult migration	Winter; spring	Winter; spring	Winter; spring	Winter; spring	Winter; spring

Cutthroat trout spawning & incubation	Spring				
Cutthroat trout rearing	Year-round	Year-round	Year-round	Year-round	Year-round
Bull trout adult migration		Summer, fall	Summer, fall	Summer, fall	Summer, fall
Bull trout spawning & onset of incubation		Fall	Fall		
Bull trout incubation		Winter	Winter		
Bull trout rearing	Year-round	Year-round	Year-round	Year-round	Year-round
Bull trout downstream migration for migratory fish		Spring	Spring	Spring	Spring

4) Basis for comparison for out-of-kind offsets for NEB determination

Offset projects that can be demonstrated to provide benefits in the same subbasin and time with available stream flow (i.e. “in-same subbasin/in-time”) make NEB determination conceptually simple.

When out-of-kind/time/place offsets are proposed, a comparison of impact and offset will necessarily entail implicit or explicit use of relative values, or weights, to complete the NEB determination. Here we illustrate concepts for evaluating these more complex comparisons of impact on instream resources drawn from the field of resource economics. Examples of questions that might arise when perfect water-for-water replacement is not possible include:

1. Does a (set of) project(s) that provides 0.5 cfs in May in one sub-basin compensate for 0.5 cfs loss in May in an adjoining sub-basin; or in August in another sub-basin?[5]
2. Does a set of projects that augments one species’ habitat (such as an ESA-listed steelhead trout) in one basin offset losses of habitat for another species’ habitat (such as a non-listed coho salmon)? How is this tradeoff considered as a part of calculating NEB?

The answer to question 1 likely depends, in part, on the relative importance in each particular watershed of stream flow across time and space, and how instream resources are affected by changes in stream flow. In this case it is likely that the functional effect of stream flow changes on instream resources (including fish) will be treated as similar in character. Question 2

[5] The magnitudes here are only for comparison purposes, and may not necessarily reflect values seen in each planning domain.

however, necessarily requires assessing the *relative value* of changes in the target populations. The basic elements of such a comparison are described next.

Imagine the following simple scenario from question 2 (Fig 4). Projected increases in permit-exempt wells around the Yellow River are expected to lead to lower summer flows, measurable at point A, reducing spawning habitat for Coho salmon, a non-listed and harvested population

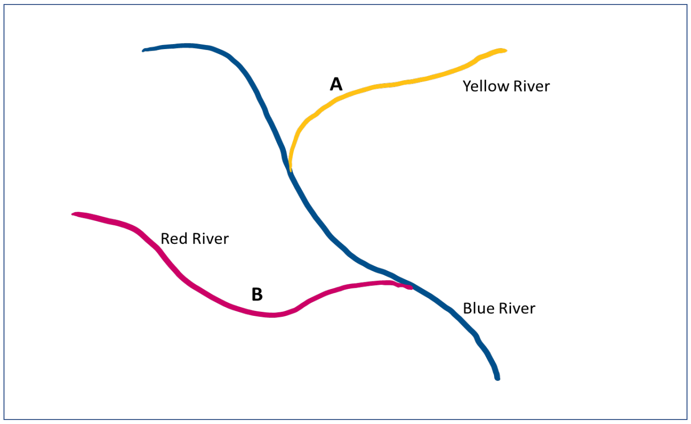

Figure 4: Hypothetical river basin

(suppose this is the only environmental impact). Suppose also that there is no feasible way to replace "water for water" during the critical summer flows in the Yellow River basin. The proposal instead is to provide mitigation along the Red River that increases habitat capacity for the population of ESA-listed steelhead trout in that tributary (again suppose no additional ecological benefits). Do the projected gains in steelhead at point B "outweigh" the lost coho at point A, and provide a "net ecological benefit"?

A little mathematical structure will be used to clarify concepts. Define a generic ecological endpoint condition modeled at location x as $Q_x (S_x, T_x, P_x)$. The endpoint condition Q could be the abundance of steelhead or coho, and is a function of the species affected (S_x, where S_A=coho and S_B=steelhead), the timing of habitat changes (T_x, where T_A could be critical summer flows and T_B could be year-round), and the place (P_x, where P_A is point A in Yellow River and P_B is point B in Red River). One could also think of the time dimension on a longer time-scale: the decline in coho will happen in the next 3 years, but the steelhead populations may not reach full capacity for 30 years.

An NEB determination will assess *changes* in these endpoint conditions, or ΔQ_x. In our example, what is the (positive) gain in steelhead (ΔQ_B), and what is the loss in coho (ΔQ_A, a negative amount) from a watershed perspective, given both losses in streamflow from groundwater use and the offsetting effect of mitigation projects? Because ΔQ_B and ΔQ_A vary in the three dimensions (timing, place, and species) even in this simplistic example, the answer to these questions is complicated and generally uncertain, and is the focus of the majority of this technical document.

Comparing net impacts to steelhead with net impacts to Coho is like comparing apples and oranges. Is the gain in one species due to plan implementation sufficient to offset the loss due to exempt-well-induced streamflow reductions in another species? To answer this question, we need more information. In addition to changes in resources, (ΔQ_B and ΔQ_A), we also need to place a relative value, or weight, on each species to decide whether the gains are "sufficient" to compensate for the losses.

To make this determination we add one more level of mathematical structure. Define $V(\Delta Q_B)$ as the value that affected households in the region place on the gain in steelhead. What is the minimum acceptable increase in steelhead abundance ("willingness to accept") due to the proposed plan necessary to mitigate for the failure to mitigate losses to coho yield in the Yellow river $V(\Delta Q_A)$? A simple economic decision rule might be that if $V(\Delta Q_B) > -(V(\Delta Q_A))$, the combined growth in exempt wells and mitigation projects should be acceptable if this metric is deemed appropriate for NEB determination. The simplest possible representation of such a comparison is provided in Textbox 1. It is not unreasonable to think of placing a "price" or value on each unit (each fish), and multiply the change in each by that price. These "prices" need not reflect market prices, but instead reflect what people are willing to give up (or accept) for an increase or decrease in Q_A and Q_B (even in the absence of a relevant market). Therefore, this is not inherently or necessarily a process of "monetizing" the fish, rather it is a way of formally representing tradeoffs. Below we provide some more context and guidance for how to estimate and compare environmental impacts in an economic framework.

In general, environmental economists have well-developed tools to answer questions of relative value such as these, though the information required to answer them is often difficult and costly to acquire. The most relevant for this application is a "stated preference" approach to surveying the relevant constituent members of the public. A prominent recent example was a survey to assess the damages (at $17.2B) caused by the 2010 Deepwater Horizon oil spill[6]. The Deepwater study team spent over 3 years at great expense developing, testing and refining their survey.

Less costly approaches to valuation include "benefits transfer" approaches in which researchers attempt to find studies already undertaken in a different place with *a similar environmental context and a package of changes* as similar as possible to ΔQ_B and

> Text box 1: A course workflow for value comparisons.
>
> 1) Estimate the physical changes due to streamflow changes and mitigation projects (ΔQ_A and ΔQ_B).
> 2) Estimate the value of individual units of Q_A and Q_B. Call these P_A and P_B.
> 3) The value of changes in Q_A and Q_B could be estimated as
>
> $$V(\Delta Q_A)=P_A \times \Delta Q_A$$
>
> and
>
> $$V(\Delta Q_B)=P_B \times \Delta Q_B$$

[6] Bishop, B.R.C., K.J. Boyle, R.T. Carson, D. Chapman, W. Michael, B. Kanninen, R.J. Kopp, J.A. Krosnick, J. List, R. Paterson, S. Presser, V.K. Smith, R. Tourangeau, M. Welsh, J.M. Wooldridge, M. Debell, C. Donovan, M. Konopka, and N. Scherer. 2017. "Putting a value on injuries to natural assets: The BP oil spill." *Science* 356(6335).

ΔQ_A. The benefits transfer would also include an adjustment for factors that we expect would shift overall willingness-to-pay, like differences in income or general environmental attitudes between the study site and our Blue River Basin site. There are a number of environmental consulting firms that can provide input for such benefit transfer studies; Earth Economics in Tacoma specializes in part on this type of analysis. ECONorthwest is another regional consulting firm with expertise in this area. The Natural Capital project at Stanford University is another useful source for relevant primary studies[7]. Table A1 in Appendix 1 provides a complete (to our knowledge) list of studies estimating the value the public places on changes in anadromous fish populations.

However, the valuation task at hand in our example is not as simple as laid out above. The comparison $V(\Delta Q_B) > -(V(\Delta Q_A))$ was simplified for illustration, but each ΔQ_A depends on *timing (T), place (P) and species (S).* For example, Mansfield et al (2012) asked survey respondents whether they would be willing to make hypothetical, annual payments over 20 years through federal taxes to increase wild salmon populations in the Klamath River Basin from 30 – 150% based on change in extinction risks (low, moderate, high, very high). In this example, timing (20 years), place (Klamath) and species (wild salmon) are all held constant; the only varying attribute is the change in quantity/abundance. In our example, this would be like answering the question of changing coho returns only at point A, at the same point in time. This complexity manifests in several ways; in each case the trade-offs can be identified, but we are unaware of any existing studies examining willingness to pay or consistency in perception and weighting for such complex tradeoff, or explicit trade-offs of different species. One response to this complexity would be to incorporate a stated preference study into the proposals submitted under RCW 90.94, and tailor the scenarios given to respondents to precisely target the package of changes in our decision problem. Done correctly, this would provide a defensible information set to guide an economic decision rule.

To supplement this conceptual summary, we have included an annotated bibliography in Appendix 1 of selected relevant journal articles that may be of use as context for NEB determination.

3. APPROACHES TO NEB DETERMINATION

There are a variety of approaches to NEB determination, with different strengths and weaknesses, demands for data, assumptions and key uncertainties (Table 1). In addition, different constraints on how impacts of consumptive water withdrawals and offset benefits are forecasted will also vary based on the approach taken. For instance, in some cases it may be possible for impacts to be assessed empirically, but forecasting benefits from offset actions over the 20 year planning horizon will most likely rely on calculated projections. Therefore, planners should choose their approach to NEB determination based on data availability and planning goals. The following is a suite of approaches to NEB determination that planners may pursue. Each approach includes a discussion describing the data needs and methods, the assumptions

[7] https://naturalcapitalproject.stanford.edu/

required/used, and the sources of uncertainty. These approaches to determining NEB resulting from planned offset to consumptive water use from permit exempt wells were derived from Bradford et al. (2014), and include:

- In-kind/In-place Habitat Replacement
- Habitat Function Replacement
- Habitat Capacity for Single Species Replacement
- Fish Abundance Replacement
- Fish Production Replacement

The most appropriate approach for any planning unit will depend on the different needs, opportunities, and constraints in each situation. Deciding the right approach requires evaluating what technical or ecological data and expertise are available, but also practical in terms of the size of ecological impacts, and the benefits and the values attributed to those impacts and benefits in each case. Given the uniqueness of each planning unit, it is impossible to set out a single technical NEB determination "recipe" that will work in all cases, nor is it the role of a technical team to decide for planners what approach they must, or even should take given the constraints confronting each planning group.

At the same time, the technical team is conscious of the need for some guidance in this regard, at least to the extent of considerations of how such a decision may be made. In responding to this, this technical report has developed the following decision tree to help planners identify the most appropriate approach to NEB determinations. This decision tree is based on commonly encountered constraints, such as the types and richness of available data, presence of monitoring programs, and the complexity of analysis demanded in each approach. In the decision tree below (fig. 5), the end points are in rectangles; one starts with the need to perform an NEB determination, and arrives at one of the approaches outlined here.

Using the decision tree, planners would evaluate decisions within each diamond. For example, in the first diamond at the top, if the answer is "YES" that water-for-water replacement is possible, then there is no need to perform more complicated modeling exercises to estimate the impacts on instream resources in the execution of the NEB determination. Planners would implement the In-the-same –subbasin/In-kind habitat replacement approach to NEB determination. However, if In-the-same –subbasin/In-kind habitat replacement is not possible (i.e., the answer is "NO"), and one lacks specific information about the fish species of interest in the relevant ecological context (e.g. species and life stage of fish, see above), then one is limited to making the NEB determinations in terms of the habitat as the instream resources being offset for consumptive water use, and extending this to fish contingent on the availability of reliable habitat-fish associations (habitat function or habitat capacity for single species replacement). This can be in the form of modelled or experimentally derived relationships (e.g., Habitat Function Replacement or Habitat Capacity for Single Species Replacement). On the other hand, if fish population data are available, then one can perform the NEB determination on the basis of either fish abundance or productivity replacement depending on the degree to which correlated habitat data are available to inform these approaches. As pointed out in more

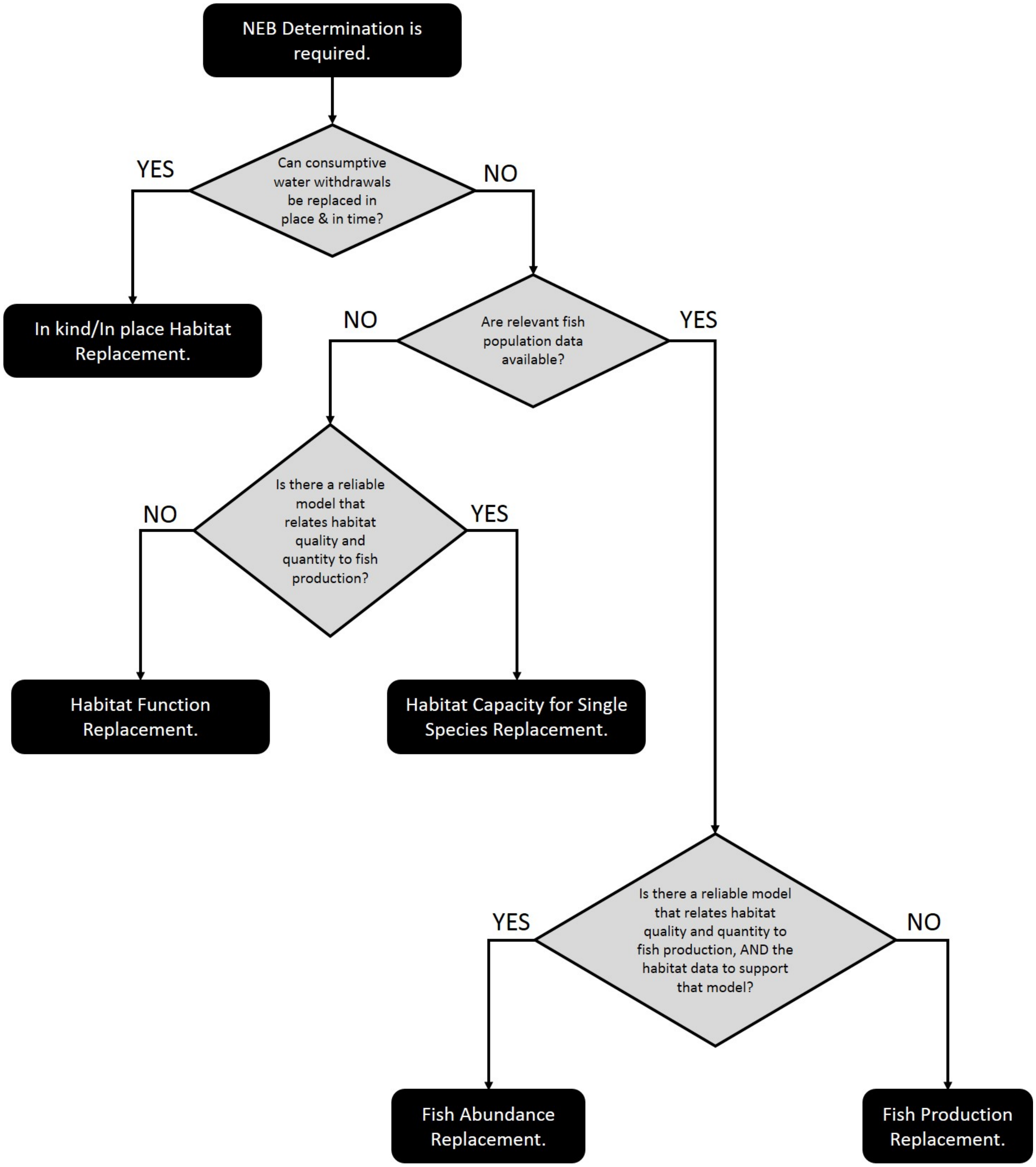

Figure 5 Decision Tree for approaches to NEB determination. Starting and ending points are black rectangles with white text, and decisions are made in diamonds on the basis of answers to the contained questions. Users should keep in mind these are sufficient, but not necessary criteria for making the decision of NEB determination approach. Other considerations may include available funding, time and expertise (see text).

detail for each approach below, if one has long-term time series data (i.e. multiple generations of fish) on fish and habitat at a population scale, then one might be inclined to adopt the fish production replacement approach, such as modeling spawner-recruit relationships, or run reconstructions with habitat metrics as cofactors influencing the production process. However, if fish and habitat data were more limited in time, but at high spatial resolution, then one might be inclined to adopt a fish abundance replacement approach, such as the Ecosystem Diagnosis and Treatment model (EDT).

To use this decision tree effectively, the user should follow the tree to choose an appropriate approach to NEB determination, and then find the description below for that approach. For each tier, there is a secondary work flow chart that indicates the conceptual steps to performing each respective approach. Conceptual steps include where data of different types enter the assessment process, what estimation is performed in each assessment, and where various kinds of outputs exit the assessment process.

This decision tree is offered in an effort to be helpful and promote transparency in decision making, but all users must be conscious that the answers to the questions are ***sufficient***, but ***not necessary*** conditions for selecting an NEB determination approach. For example, if one doesn't have any data on fish at the scale of the affected unit of fish (population, Evolutionary Significant Unit = ESU, etc,), then one is not likely to generate a credible fish production replacement-based NEB determination. However, even if one has high quality fish population monitoring data, there may be reasons why a planning group would opt for a habitat function replacement assessment. For example, if the costs of the requisite modeling is perceived as excessive, the analytical expertise is not readily accessible, or the available data inventory of habitat units and their net ecological services is seen as superior in quality to available fish data (see HEA description below), then planners might opt for habitat function replacement in spite of this decision tree.

A. IN-KIND/IN-SAME-SUBBASIN HABITAT REPLACEMENT (AREA/TYPE)

Under RCW 90.94, a disruption or detraction of fish habitat, resulting from reductions in stream flow consequent to consumptive use withdrawals must be balanced by some form of mitigation or redesign to achieve the goal of a positive Net Ecological Benefit. The highest priority (i.e. "most preferred") mechanism, within a range of offset mechanisms, are projects that replace consumptive domestic water use impacts during the same time and in the same subbasin as where the impacts occur. This option is supported by the assumption that keeping impacts and benefits comparable in type, extent and location is most likely to maintain the existing productivity and integrity of the ecosystem; this assumption is broadly relied upon, but is an assumption none the less (Moilanen et al. 2009; McKenney and Kiesecker 2010).

In-kind/In-place offsets are the simplest offset mechanism since the equivalence of habitat for habitat is the most straight-forward. However, establishing that the offsets are indeed In-kind/In-place offsets may become difficult as the area becomes larger and increased habitat diversity makes it difficult to validate that offsets are truly "In-kind". Therefore In-kind/In-place offsets are best suited for smaller habitat units and do not include habitat conversion (e.g., river to reservoir or confined (bank-hardened) to unconfined channels). The biggest advantage of In-kind/In-place offsets is the ease of establishing equivalency as the determination is based on water for water in the same units. This is the most direct comparison and the easiest NEB determination. However, the largest risk in assessing this form of offset is the assumption that the replacement habitat and the associated fisheries productivity will be equivalent to that lost. For example, if the area considered has multiple and diverse habitat types, the benefits to each fish species/life-stage may be different for a given change in stream flow.

a. Data and Methods

The primary metric for in-kind habitat replacement is stream flow (e.g., cfs). Since the units are the same for in-kind/in-place offsets, it is not necessary to determine NEB by assessing fish productivity. Calculation of environmental impact is made by measuring consumptive water use, and establishing offsets. In some cases, particularly in low flow locations where a very small change in stream flow at a critical time can be the difference between passage and no passage for out-migrating juvenile fish, continued stream flow monitoring may be warranted.

b. Assumptions and Implications

In-kind/In-place offsets assumes that habitat and environmental variables (e.g., macrophytes, depth, substrate, nutrients, temperature etc.) will respond to quantities of stream flow equivalently among different locations within a planning unit, and further that responses in habitat variables can be considered surrogates of fish productivity. It also assumes that new habitat generated by the offset will have the same ecological characteristics and associated production values (e.g. primary and secondary production). This assumption may not be supported (Bull et al. 2013), and therefore should be validated with appropriate monitoring.

Because replacing habitat in-kind/in-place is not expected to alter the total habitat available to fish (given that the new location will be in proximity to the lost habitat) it is not anticipated to change fish population dynamics. However, the result of where stream flows are offset may have an impact on that new location’s potential productivity. Specifically, in making an NEB determination it is the marginal increase in productivity at the offset location that must meet or exceed the production lost at the impact locations. The offset locations may have had some prior intrinsic productivity under baseline conditions that would be expected to continue in the absence of the offset, and this should not contribute to the estimate of NEB in the planning scenario.

c. Sources of Uncertainties

Replacing stream flow in-kind and in-place relies on relatively common and well understood measurement methods and consequently have relatively little uncertainty in the measurements themselves. Assuming the nature of in-kind and in-place can be validated, the associated benefits to habitat and productivity are also relatively low uncertainty. However, the validity of these assumptions are more certain for well-studied habitat types (e.g. spawning gravels Fitzsimons 2014). Uncertainty increases for less-well studied habitat types, or especially highly unstable or typically mobile or dynamic habitat types (e.g., gravel bars or pools created and maintained by log-jams that migrate under typical conditions, Abbe et al. 2003; Pess et al. 2012).

B. REPLACING HABITAT FUNCTION

When in-kind/in-time/in-place offsets are not possible, plans will need to determine NEB based upon more complex assessments of equivalence between locations of impact and offset. The first among these offset mechanisms are cases where offsets are based upon replacing the ecological function of certain habitat features with some different combination of features at different locations that would on balance provide the same ecological function. These are called service-to-service equivalency analysis (e.g. NOAA 2000; Lipton et al. 2008). Ideally, these functions relate to fish production, and would include some multivariate description of the habitat, such as habitat structure, cover, or substrate type. Alternatively a description of the habitat might be integrated by measures such as secondary production. Because the NEB determination is made based on the ecological function, the specific habitat provided as offset could be different from the habitat that is impacted, as long as the NEB nets out positive.

Habitats are multifaceted and complex, and a clear mapping of habitat features to fish production can be complex, if it is possible at all (McMillan et al. 2013). Therefore, it is often important to rely on metrics that express some level of integration of habitat features, rather than the features themselves. One example of such integration that expresses ecosystem function is secondary production (i.e., the rate of incorporation of organic matter into body tissue of invertebrate mass per unit time and area (e.g.,Cusson and Bourget 2005). Production, manifesting a rate of energy exchange across trophic levels, is a better indicator of ecological function than standing stock of macroinvertebrate biomass (Benke et al. 1984; Benke 1993). Secondary production can integrate across life stages and generations of invertebrate fauna, and will do so over temporally variable environmental conditions. Thus, secondary production has been suggested as a valid proxy for ecological function. Relationships between secondary production and commercial, recreational, Aboriginal (CRA) fisheries can be determined using productivity-state response curves (Koops et al. 2013).

NEB determination via ecological function replacement can also be complicated by the ecological context. For example, where ecological function that impacts one life stage of fish is replaced with ecological function that impacts a different life stage of fish, those different life stages may not be equally limiting to total population growth. In the case of steelhead, Hall et al. (2016) showed a considerable diversity of life history trajectories that might better accommodate out-of-kind mitigation than a species with a more rigid life history. Given the RCW 90.94's focus on fish, some accounting of fish life history should be part of the assessment of ecological function replacement.

Making an NEB determination based on ecological function is suitable for not only situations with designed offsets to balance the functions lost to impacts, but also situations where alternative functions are preferred in the context of other available habitat in the planning unit. When the intent of offsets is to replace the same ecological function, determining the offset may be as straightforward as the in-kind/in-place. However, if the impacted habitat provides a non-critical ecological function, it may be preferable to design offsets that provide a rate-limiting or rare ecological function.

A specific example of habitat function replacement is Habitat Equivalency Analysis (HEA). HEA is a method developed to determine the compensation for damages to natural resources such as oil discharges, hazardous waste release or physical damage to resources from ship groundings (NOAA 2000). Consequent to statutory requirements, when damage to natural resources occurs, responsible parties are asked to pay damages to cover “compensatory restoration”, where the offsets provided by habitat function at least balance those lost due to the original damage. Thus, the context for its development was principally as a regulatory tool rather than a scientific research tool. Similar to RCW 90.94, restoration plans must determine and quantify injury, develop restoration alternatives that consist of actions that at least match the injury. HEA is a not-In-kind/place approach to evaluate the services provided by the lost habitat, offset habitat and the balance between them. The steps in an HEA determination are:

1. Document and estimate the duration and extent of injury, from the time of injury until the resource recovers to baseline, or possibly to a maximum level below baseline;
2. Document and estimate the services provided by the compensatory project, over the full life of the habitat;
3. Calculate the size of the replacement project for which the total increase in services provided by the replacement project equals the total interim loss of services due to the injury; and
4. Calculate the costs of the replacement project, or specify the performance standards where implementing the compensatory habitat project.

In steps 1 and 2 numerical values for the ecological goods and services provided by each impacted and offset habitat unit must be generated. When aggregated across all the relevant habitat units, injury and offset can be evaluated for net ecological benefit (See fig. 6).

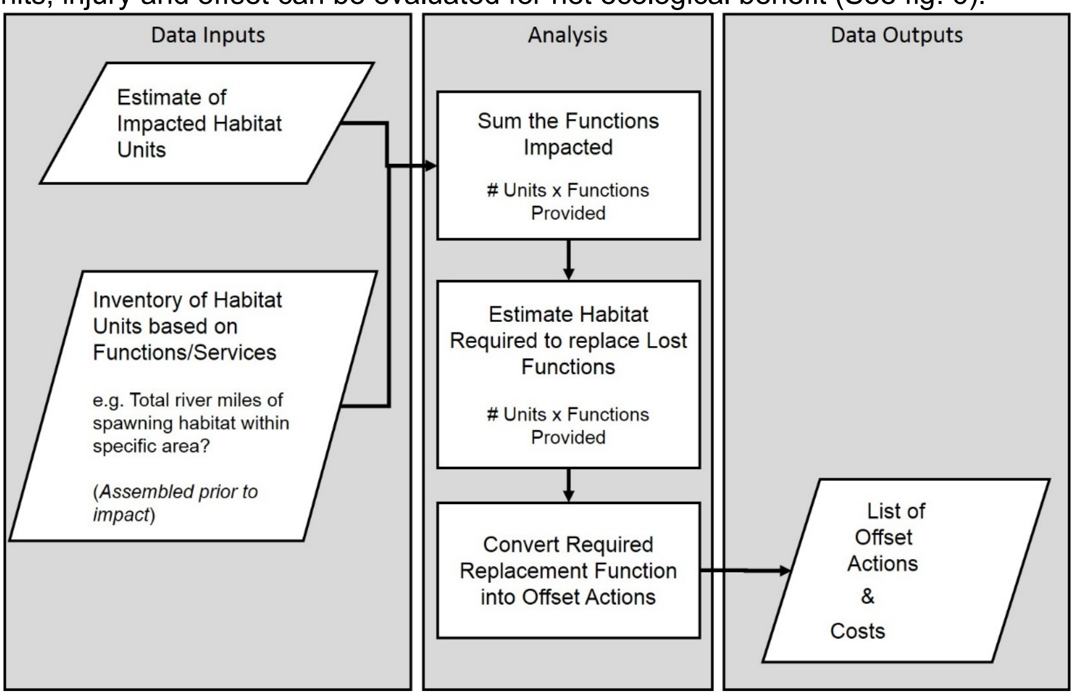

Figure 6 Workflow for Habitat Equivalency Analysis. User data enters the analysis in the form of 1) an inventory of the habitat with the ecological functions or services provided by each habitat unit (e.g. “river miles of Chinook spawning habitat for unit X.”) and 2) an estimate of which habitat units will be impacted by changes in stream flow. The impacts are then added up as number of units times the functions supplied. This number then has to be balanced by the ecological functions or services generated by the habitat offset projects. The inventory of habitat services is prepared ahead of time and can be informed by monitoring data, but is often based on best professional judgement.

HEA has been used in a variety of ecological damage determinations and many lessons have been learned with respect to its strengths and weaknesses (e.g. Dunford et al. 2004; Desvousges et al. 2018). In particular, HEA has a number of critical assumptions that may be difficult to justify: e.g. a design that imposes a preference for offsets to provide the same services that were injured, as well as a constant ratio of habitat services to habitat value, and a constant real value of services and injuries over time (Desvousges et al. 2018). HEA has also been criticized for reducing complex ecological services to a single metric, and for failing to properly account for ecological injuries that continue having incremental or marginal effects over time (Desvousges et al. 2018). In practice, natural resource agencies assemble inventories of their management habitat units and attribute a numerical score for the net services supported by those units. In the absence of targeted monitoring, these scores are assigned based on professional judgement, which can be problematic. Professional judgement by itself is prone to high variability, low and untestable accuracy and hidden bias (e.g. Burgman et al. 2011). Therefore, where habitat function replacement is deployed for NEB determination, significant resources should be applied if possible to pre-impact monitoring to develop testable metrics of the baseline habitat services being replaced (Kennedy and Cheong 2013).

a. Data and Methods

Examples of common indicators associated with habitat function include measures of substrate type and characteristics, densities of riparian or aquatic macrophytes or quantity of large wood. Regionally there are numerous standardized protocols for monitoring and reporting these metrics including:

- CHaMP (http://www.monitoringresources.org/Document/Protocol/Details/2235)
- Washington Dept. of Ecology: (https://ecology.wa.gov/Research-Data/Monitoring-assessment/River-stream-monitoring/Habitat-monitoring/Habitat-monitoring-methods)
- AREMP (https://www.fs.fed.us/pnw/pubs/pnw_gtr625.pdf)
- EPA-EMAP (http://www.epa.gov/emap/html/pubs/docs/groupdocs/surfwatr/field/ewwsm01.html)
- USGS NWQA (http://water.usgs.gov/nawqa/protocols/OFR02-150/OFR02-150.pdf)

Indicators for characterizing secondary production in stream macroinvertebrates include the density and biomass of the entire community of secondary producers (Plotnikoff 1994). The taxonomic level required for this approach can be quite coarse. Methods for sampling secondary consumers are available for many of the same sources of information on metrics for habitat features above.

If the approach is to characterize secondary production it is critically important that methods distinguish between production and biomass. As mentioned, most archived data are reported as biomass, but there are reasons why there would not be a one-to-one mapping of biomass onto productivity (Jenkins 2015). Therefore, NEB determinations should clearly indicate if they are adopting biomass as a proxy for productivity, and if not what model they rely on to convert. Examples include models that relate production to biomass via metabolic energetics, regressions of observed data or reliance on literature benchmarks (Schwinghamer et al. 1986; Tumbiolo and Downing 1994; Wong et al. 2011).

b. Assumptions and Implications

Making a NEB determination on the basis of service-to-service equivalency makes a number of important assumptions. First, it assumes that substitution of equivalent ecological function will result in equivalent fish production. Even if this assumption may work conceptually, salmonid fishes in particular are highly locally adapted (Taylor 1991; Waples 1991, 2006) and the same services provided in a different ecological context may influence fish production differently. In addition, if the substituted ecological functions are different than those impacted, a significant amount of pre-treatment baseline data, benchmarks and models will be required to justify a favorable NEB. In the case of habitat-feature for habitat-feature substitution (equivalent amount and type of structure/cover/substrate) the assumptions and implications are similar for in-kind/in-place above. However, for non-in-kind/in-place mechanisms, there are additional considerations including:

a) Source Data Quality—raw field data are presumed to be accurate, but it is also very site specific, and collecting it over a large domain results in high data density. Therefore, field habitat data are often compiled or aggregated into indexes. Compiled data may do a better job describing a large assessment domain, but may mask the detailed relationships among multivariate data that actually determine fish production. The choice of data type (raw, aggregated, derived, etc.) may be subject to constraints that limit flexibility one way or the other, but planners need to be aware of the character and limitations of source data in this context.
b) Data Interpretation—aggregation can occur in space and time, but also in terms of what ecological feature is being represented. For example, were one to perform a NEB determination on the basis of secondary production, the taxonomic resolution of the consumers can change the interpretation of net biomass. Biomass changes can reflect net energy flow through foodwebs, but animals have food preferences and the details may or may not matter in different ecological contexts.
c) Model structure—relating biomass to productivity will be affected by the structure of the model used and roles that the empirical data, standardized benchmarks and professional knowledge play in the process. Model design choice should reflect a model most similar to the ecological context in the planning unit.

These implications are important, and if NEB determination is to be performed via habitat function substitution, plans should provide details for each of these implications and how they will be addressed to be consistent with the expectation of transparency in the interim guidance.

c. Sources of Uncertainties

There are several sources of critical uncertainty here. As mentioned above, if the offset is designed as ecological function for ecological function, then the uncertainty is dependent on how the service is expressed with a metric or metrics, where the different metrics have different data-related uncertainties. This metric uncertainty is likely to be greater for habitat variables than for stream flow, and will increase rapidly as metrics are built from multivariate habitat features. In addition, the assumption that one will see a given benefit for a given level of service provided generates a model-based uncertainty. When offsets are achieved with services different than those impacted, there are additional uncertainties related to correctly forecasting the effectiveness of projects for those different services. Some locations will be relatively information-rich with respect to the well-researched relationship between habitat features and fish productivity (e.g. Intensively Monitored Watersheds, IMW's). However, many planning units will be information-poor in this regard and NEB determinations will increasingly rely on the scientific literature, expert knowledge, productivity-state curves, and pathways of indicators models. Each of these alternatives can significantly increase uncertainty in the final NEB determination.

C. REPLACING HABITAT CAPACITY FOR SPECIFIC SPECIES

The second NEB approach among the non In-kind/In-place offset mechanisms are cases where offsets are based on applying models of habitat-fish relationships to the amount of available, suitable habitat for selected species. All things being equal, we have good evidence from empirical studies in specific locations that some combinations of habitat features including water depth, water velocity, substrate type and vegetation cover are suitable, and in some cases preferred by specific life-stages and species of fish (Orth and Maughan 1982; Beecher et al. 1993; Thomas and Bovee 1993). If we were to add up all the habitat of the preferred type (including stream flow), we could quantify potential capacity of a given habitat unit within an assessment area. In the present application, quantitative estimates of habitat loss from consumptive use withdrawals could be measured against anticipated amounts of habitat gained from offsets to provide an NEB determination.

Habitat capacity models have been applied to both lake and stream systems. In particular, there is a rich diversity of models that have related stream habitat to fish capacity that include: PHABSIM (Stalnaker et al. 1995), River2D (Katopodis 2003), and MesoHABSIM (Parasiewicz 2001; Parasiewicz and Walker 2007). Many sockeye salmon are lake spawning, and "spawners per hectare" of lake-bottom has also been used as a method to evaluate the environmental benefits of reservoir enhancements (Goodlad et al. 1974; U.S. Department of the Interior Bureau of Reclamation and State of Washington Department of Ecology 2012; ECONorthwest et al. 2012). In either habitat type, the amount of habitat area becomes a comparable currency for impact and offset, but in both cases as well, it is based on the critical assumption that capacity of habitat will be realized by the target species.

PHABSIM and some of its extensions have been used in a diversity of situations to evaluate associations between fish and hydrology (Gallagher and Gard 1999; Parasiewicz and Walker 2007; Beecher et al. 2010; Reiser and Hilgert 2018). PHABSIM (Physical HABitat SIMulation) is part of a family of approaches called the Instream Flow Incremental Methodology (IFIM; Bovee et al. 1998). IFIM is a broad conceptual toolbox that considers a variety of aspects of stream ecology. PHABSIM relates hydraulics to hydrology and to specific components of fish habitat. Other tools in IFIM can integrate PHABSIM results with hydrology over time.

PHABSIM consists of a hydraulic model which is linked with habitat suitability criteria (HSC) to map habitat quality for specified life-stages of target species at different discharges. Several options for hydraulic models are available within PHABSIM. The most common options are step-backwater modeling, depth and velocity regression on transects, and two-dimensional hydraulic modeling based on channel roughness and flow routing. With a hydraulic framework in hand, the amounts of habitat weighted by habitat quality (as indicated by habitat suitability criteria [HSC]) can be integrated across a stream reach at discharges of interest to produce a metric called weighted usable area (WUA). This is accomplished by applying HSCs for each species life-stage of interest to each location within a specific stream network across a range of discharges. This applied weighting is then summed for each discharge and species life-stage to generate a WUA value (e.g., for juvenile Chinook salmon at 300 cfs). The workflow for PHABSIM is illustrated in Fig. 7. Traditional use of PHABSIM incorporates microhabitat (depth, velocity, substrate and/or cover), using HSC for each species life-stage for each microhabitat variable, but consideration of mesohabitat or macrohabitat can be included with appropriate study design. The output allows comparison of the relative habitat value of different discharges for a particular species life-stage.

Experience with PHABSIM has revealed a number of important lessons and constraints to its use. The weightings applied to the area of habitat rely on a representative stream reach (or a critical reach if one is identified) for assessment of impacts of hydrologic changes to fish or other aquatic organisms. Where multiple reaches are expected to be affected by water withdrawal, it may be necessary to model multiple reaches. In either case, there is a critical need to validate that each representative reach where the fish/habitat relationships are developed is truly representative of the locations where the WUA estimates are going to be made. Validation of the HSC's and WUA can be accomplished with field measurements at one or more known discharges. Resources for such validation should be included in plans that perform NEB determinations via habitat capacity replacement.

This approach to NEB determination is flexible in that the WUA can accommodate changes in habitat amount as well as quality. This flexibility is particularly useful in the case where offsets are not located at the impact area or if the offset's ecological result is different from the impact. Of particular importance in this context, there may not be any available water to offset loss of stream flow from consumptive use in the same time and place, and the ecological mechanism of offset has to be of a different kind, such as habitat improvement (McKenney and Kiesecker 2010). The fact that there is a quantitative basis for determining NEB also makes it attractive, but it must be kept in mind that there are good reasons that model predictions may not always be realized in terms of fish numbers. For example, in order for fish-habitat capacity models to be applied generally, the underlying relationship between observed fish preference must reflect a global, or population-wide preference rather than fish making the best of what habitat variability is available (McMillan et al. 2013). Indeed, uncertainty over HSCs has received considerable critique and is discussed elsewhere. Critiques have also challenged if the variables used to construct HSCs are the variables most relevant in the case of changes in stream discharge (Railsback 2016).

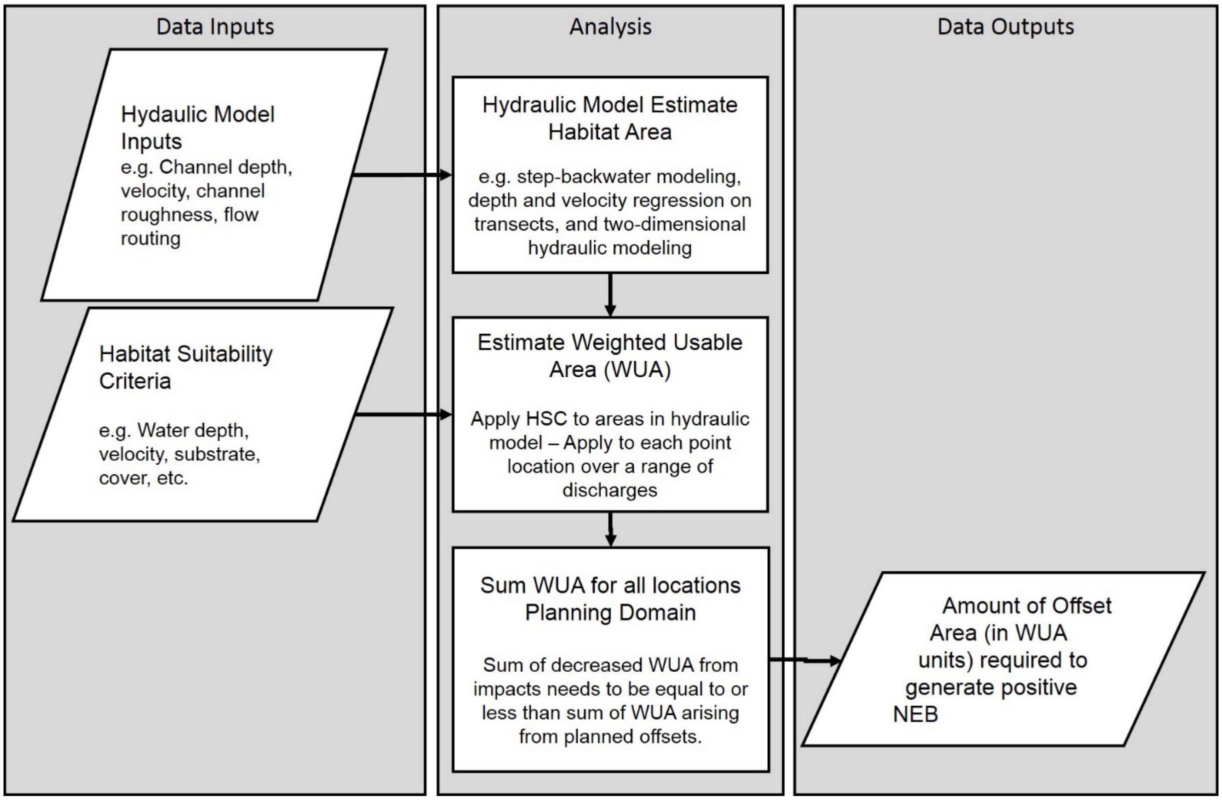

Figure 7 PHABSIM workflow. User data enters the analysis in the form of output from an hydraulic model that provides a framework of a map of stream habitat amount (= area), to which habitat data is applied. User data also enters in the form of Habitat Suitability Criteria (HSC), which are based on a relationship between habitat characteristics and fish preference. HSCs can be derived from other modeling or field monitoring, but should be validated for the planning domain in either case. Modelled areas are weighted by the HSCs and produce an estimate of WUA for each unit across a range of river discharges. This is then summed to get a measure of impact or offset. NEB can be estimated by subtraction.

a. Data and Methods

Habitat capacity has two parts: the suitability or preference of habitat units by fish, and the amount of those units. Measures of suitability or preference are weights that are multiplied by the area of the units producing the WUA metric that is species and life stage specific. Characterizing stream units to characterize their suitability often relies on the lower level, unit-specific data described for the habitat substitution approach above. Examples of these data types include combinations of water depth, water velocity, substrate type and vegetation cover. As described above there are numerous standardized protocols for monitoring and reporting these metrics across the region. Collecting these data is often expensive and labor intensive, but these same regional sources offer data collection protocols that provide the greatest degree of reliability, interoperability and transferability to other locations.

The data needs of the NEB determination based on habitat capacity depends on the choice to estimate both parts of the WUA metric. In many cases, the characterization of habitat suitability (e.g. HSC) is taken from the literature or other studies and the quantification of habitat capacity in a specific case amounts to measuring the amount of that habitat type. Indeed, the Departments of Ecology and Fish and Wildlife have collaborated in compiling composite HSC for most WA salmonids. While these data may vary substantially, the range of values may put some bounds on the variability one may encounter in actual implementations.

These methods have been forcefully criticized. In particular, the transportability of HSC's from one location to another is an assumption that has been challenged (Railsback 2016). As a result, the NEB determination may include developing the fish-habitat relationships directly for each stream. In one example where HSC were critically evaluated, a mismatch between HSC and coho salmon response was recognized (Beecher et al. 2010). This led to an updated method of estimating HSC that included estimates of food intake HSC and habitat scale bioenergetics, and resulted in an improved match between fish response & model (Rosenfeld et al. 2016). The performance of the HSC was improved at the cost of additional data, modeling and validation effort. In this event, data needs will include fish density data over a large number of habitat units within the NEB determination area, and the development of quantitative associations between habitat metrics and fish density (e.g. regression, canonical correspondence, etc.). Importantly, HSC will be species and life stage specific, and so fish density data required to evaluate HSC will likewise need to be for the appropriate species and life stages.

b. Assumptions and Implications

Habitat capacity approaches have been criticized for having a number of unrealistic assumptions and for being overly data intensive for the level of precision achieved. In the case of developing study-specific HSC, the demand for both habitat and fish data is intensive and is often viewed as expensive. Among the assumptions that are viewed as problematic are those related to the habitat being oversimplified over a given assessment unit, static in time, independent of species or life stage and equivalent across assessment units (Parasiewicz 2007; Railsback 2016).

These approaches have also been critiqued for issues related to scale of analysis. The hydraulic flow models that are commonly combined with environmental habitat data to evaluate changes in capacity are often developed over different and potentially incompatible scales (Wu and Li 2006). In addition, since one is accumulating habitat units to estimate WUA, the WUA metric may apply over many habitat units, that could each vary greatly. This variance among habitat units may be differentially important to different species and their life-stages, but is missed in most habitat capacity approaches, arguing perhaps for smaller scale assessments. Parasiewicz (2007) however, argues that larger assessment units (10^3-10^5 m^2) are more appropriate for this type of assessment because it is the relevant unit from the management point of view and more fairly represent the concept of "Functional Habitat".

There is also an important assumption that the fish-habitat association represented by HSC reflects a global preference for habitat characteristics, rather than the best of whatever is available. Across a watershed with numerous tributaries and diverse habitat conditions one could sample all the habitat units and correlate the density of fish with the local habitat conditions to develop an HSC rule set. However, in order for fish density in that process to be a true signal of habitat preference, the fish sampled would need to know that the habitat conditions where they are located is more preferable than the possible range of conditions elsewhere across the watershed where the rules were being developed. In reality this is never the case; salmon emerge from gravel and sample a very small subset of habitat conditions in their local surroundings and likely choose the most preferred from the available options. Indeed, when this assumption was tested, the fish made choices within the limits of their local environment and no global rules emerged (McMillan et al. 2013). Over evolutionary time intervals, it is likely globally preferred habitat will have higher net fish productivities, but over that same time the habitat is evolving (e.g. log jam washouts, sediment transport, etc., DeVries et al. 2001; Pess et al. 2012; Fremier et al. 2018). Thus, the pattern of fish density observed at any moment in time is a snap shot of interactions between hydraulic, habitat and biological processes that may or may not permit use of the assumption that HSC are transportable across time and space.

*c. **Sources of Uncertainties***

Habitat capacity models carry large amounts of uncertainty from several sources. Most of the sources of uncertainty are related to the assumptions discussed above, and in particular fish-habitat associations. Uncertainty can be significant with respect to the portability of the fish-habitat relationships to new locations (Freeman et al. 1997; Williams et al. 1999; Railsback 2016), and inappropriate inferences drawn from fish-habitat data (Rose 2000; Minns and Moore 2003; McMillan et al. 2013). There are also potentially large uncertainties from mismatch of scale in the data used to develop WUA measures as hydraulic modeling on one scale married to finer scale habitat data can misrepresent fish preference (Bovee 1986; Railsback 2016). The uncertainties related to habitat preferences being species and life-stage dependent can also be large, especially when preference differences among life stages of fish interacts with seasonal variability in in-stream habitat conditions (Heggenes et al. 1996).

Responding to these uncertainties lies along two lines. On a technical basis, the implementation of NEB determinations should include validations (e.g. Studley et al. 1996; Gallagher and Gard 1999; Beecher et al. 2010), where relationships between flow and fish production are validated in the location where the offsets are implemented. On a policy basis however, the magnitudes of these uncertainties and species dependence may be large, but not addressable. For example, in a watershed with both listed and unlisted salmonids, failure of the assumptions related to the non-listed fish may impact instream resources in the context of fisheries yield, but failure related to the listed fish is a regulatory issue and the Endangered Species Act may impose constraints that one cannot address with a trade-off of habitat functions. It is also possible that the listed fish is in smaller numbers and uncertainties in estimating fish abundance could include zero fish, where the unlisted fish may occur in larger abundance, making mistakes in estimating abundance less costly. Resolving what “large uncertainty” means and how it is handled may represent a policy choice rather than a technical or scientific choice reflecting case-specific, and potentially competing, values.

D. REPLACING FISH ABUNDANCE

In fish abundance replacement, NEB determinations are based on fish abundance in offset areas equaling or exceeding the abundance of fish lost from impact areas. This is one version of a general set of resource-to-resource equivalency analyses (REA, Kim et al. 2017; Holmes and Lipton 2018). This approach expresses NEB in terms of abundance of fish of the same type. For salmonid fishes, this would be fish of the same population group within their Evolutionary Significant Unit (DSP & ESU, Waples 1991, 2006).

Fish abundance replacement approaches to NEB determination are similar to habitat capacity replacement to the extent that it requires similar information on habitat in order to justify the forecast offsets, but more complicated in that it also requires more detailed information on fish abundance. In addition, this more detailed information will be required for both the impacted and offset locations. While it may be possible to acquire some of this more detailed information in pre-treatment monitoring at the impact site, the offsets will usually require forecasts. The challenge is that the forecasts usually must rely on analogs, expert opinion or models that can become quite complex; the dividend is that the detailed information inform the design of a relevant effectiveness monitoring program.

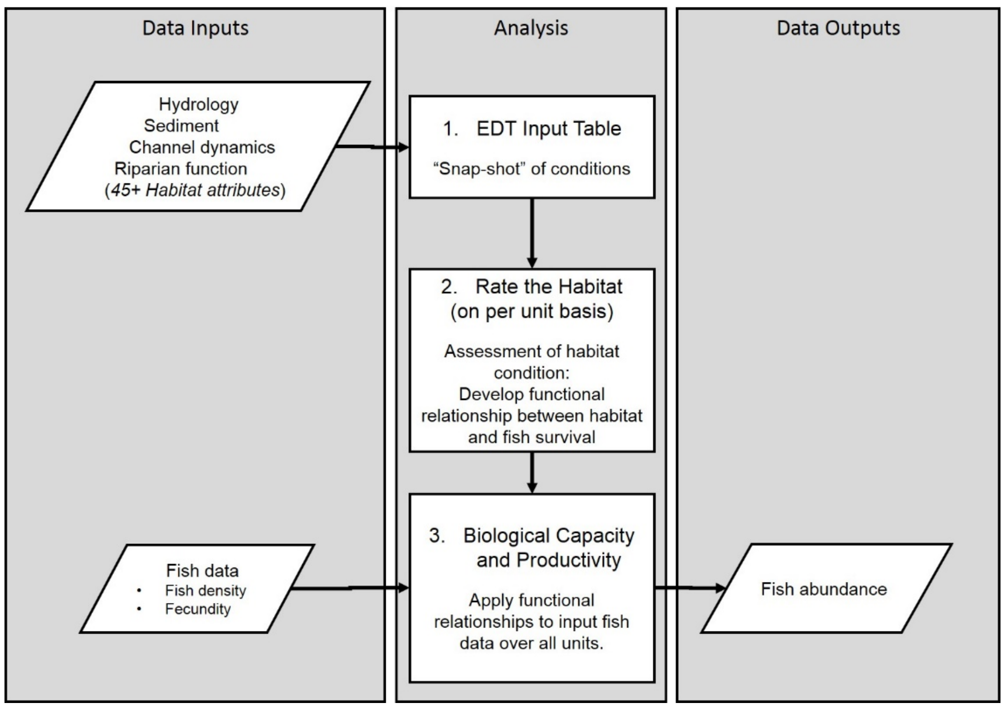

Figure 8 EDT Workflow. User data enters the analysis in the form of habitat data on a per habitat unit basis (e.g. pool) to establish a snap shot of the habitat. In step 2, those data are applied to functional relationships between habitat condition and fish responses. These relationships can be based on empirical monitoring, but are more often based on best professional judgement. In step 3, these functional relationships are applied to additional input data on current or historical fish production and used to generate forecasts of fish abundance. Given the functional relationships and habitat current condition "snap-shot", one can estimate where habitat improvements are likely to generate the greatest fish population response. However, these estimates may not be sufficiently certain to support NEB determination. In addition, much of step 3 is proprietary and requires contracting with the EDT copyright holders.

Quantifying fish abundance becomes increasingly hard as the assessment area gets larger. This is in part due to ecological issues including movement of individuals, but more so due to the logistical aspects of sampling including the expense of sampling a large number of times and the time it takes to conduct monitoring. Therefore, fish numbers-for-fish numbers substitution is better suited to smaller assessment units.

As fish abundance replacement approaches to NEB determination rely on outcomes (numbers of fish) rather than mechanism of achieving those outcomes, they can be quite flexible. Offsets can be achieved through a diversity of habitat alterations, and in quite different ways than at impact locations. Fish abundance replacement is similar to habitat capacity replacement in this flexibility, however it is distinct in that the response metrics are direct measures of fish, rather than relying simply on habitat.

A specific example of habitat function replacement is Ecosystem Diagnosis and Treatment (EDT) model. In principle, EDT works on the premise that each habitat unit has intrinsic qualities that affect the survivorship of the fish that encounter it. These qualities can vary across units, and their functional responses can vary across the life history stages and species of fish; a specific feature of a given pool (e.g. temperature, depth) might positively affect the parr of one species, but negatively affect smolts of a different species. In principle, one could start with a number of eggs in a salmon redd or redds, and then serially apply the survivorships for all units across all life stages of each species and forecast the abundance of fish at the time of the next spawning. This framework is dependent on having detailed, quantitative associations between habitat qualities and fish survival (see fig. 8). This is a strength in that with this information in hand it allows one to evaluate alternative habitat improvement scenarios and prioritize projects; it is a weakness to the extent that all of these relationships are rarely available a priori and EDT often requires significant subjective human input as a substitute. Conceptually however, the idea of net survivorship as a product of many small survivorship steps is a rational approach.

EDT has been used widely across the Pacific Northwest and we have learned a lot about how it works and about a number of limitations. Many of these limitations have been summarized elsewhere (Paine et al. 2001; McElhany et al. 2010) and so here will not be repeated in detail but discussed only as a list of relevant highlights. From a statistical point of view, this approach is a multi-regression with many (many hundreds to thousands, McElhany et al. 2010) parameters used to estimate survival at each point, with the results for all habitat units in the life history applied to the result from the prior habitat unit. This is widely recognized as over-parameterization, and it results in the generation and propagation of errors and generating untestable predictions (Freedman and Freedman 1983; Freedman et al. 1988; Burnham and Anderson 2003; Leinweber 2007). In addition, this approach makes demands on the habitat quality data far in excess of available monitoring data. For those many EDT parameters for which fish and habitat data are lacking, experts are polled for their opinions on what the actual values are likely to be. Thus, much of EDT products result from an “expert-panel” process rather than a data-based, scientific process. As such, many of the uncertainties that exist within the process that might otherwise influence our characterization uncertainty in the ultimate forecasts are subjective, based on opinion rather than data, and ultimately unknowable. Due to its high spatial resolution, EDT does provide very specific forecasts, although its uncertainties mean its accuracy cannot be evaluated. This is an important distinction, that EDT is an expert-panel process does not make its predictions wrong, but it does limit the ability for a scientific review to test its predictions, and is therefore not transparent. That said, the limited literature that attempts to characterize the reliability of EDT forecasts has indicated that it has relatively poor performance and is not useful for forecasting population sizes based on habitat assessment (McElhany et al. 2010).

a. Data and Methods

Fish abundance is expressed and monitored in a variety of ways. The choice of measure has technical implications for how the information is collected and interpreted, and for methods and uncertainties. Common ways that abundance is expressed include total abundance, density, biomass and catch per unit effort (CPUE). Some of these measures are more common in fisheries assessments than in conservation assessments and the choice of method may depend on the motivation for the assessment as well as available technical resources and data. Total abundance is the result of a census or probabilistic sample (Courbois et al. 2008). Total abundance is desirable as a metric as it is a direct answer to the question of abundance; however, it is often expensive or otherwise difficult to obtain in practice.

Density of fish is a measure of abundance within a specific area of habitat and at a specific time. The area is commonly the habitat unit being sampled (e.g. pool area or reach length), and expressed unit of measure is number of individuals per area. Samples that are part of large scale sampling designs often consist of individual density measures, and so density measures are among the most common indexes of fish abundance. It must be kept in mind however, that density of fish, particularly juvenile salmonids can vary greatly habitat unit to habitat unit (e.g. McMillan et al. 2013). Therefore, density measures are often highly location and time dependent and extrapolating from density measures at a limited set of locations can impart large biases to an estimate of abundance (e.g. Courbois et al. 2008).

Fish enumeration is a common activity within fisheries and there are standard methods for enumerating fish. Across the region there are numerous references on methods for enumerating or estimating fish abundance (e.g. Bonar et al. 2000, 2009; Crawford 2007). Additional information on abundance measures is available from Ecology at:

https://ecology.wa.gov/Research-Data/Monitoring-assessment/River-stream-monitoring/Intensively-monitored-watersheds

In addition to estimating some measure of fish abundance, NEB determinations via abundance substitution will require some method to relate or forecast management actions in the watershed plans to future abundances. Therefore, the habitat impacts or baselines, and offset forecasts will need to be characterized, and a method or model that relates offset scenarios with future fish abundance is required. Given the history of habitat management across the Pacific Northwest and the long-term investments in habitat restoration, there is a broad expectation that specific forecasts of fish responses from habitat manipulation are at hand (Roni et al. 2008). However, in practice such expectations for specific habitat types and actions come with large uncertainties (see below).

b. Assumptions and Implications

Given that fish abundance replacement relies on outcomes (numbers of fish) rather than mechanism of achieving those outcomes, the number of assumptions may be less. Nevertheless, there are specific assumptions associated with fish abundance that are similar to assumptions related to habitat capacity replacement. Fish abundance also relies on the assumption that the future ecosystem is similar to the current ecosystem. For example, as water temperatures rise over the next 20 years and centrarchid fishes replace salmonid fishes (Isaak et al. 2015; Rubenson and Olden 2016), the habitat management one would put in place may not be beneficial for centrarchids as they may have been for salmon.

It is also important to remember that the fish are wild, rather than domestic. The relevance to fish-for-fish, or habitat-for-habitat replacement is that human activity can reduce the numbers of fish deterministically (harvest, habitat loss, etc.), but cannot force the production of new wild fish. The foundational assumption of a restoration-based substitution is that by reducing the contribution to mortality from specific sources, such as lost or degraded habitat quality, we will see a consequent increase in the number of wild fish. This may not be unreasonable, but we have to keep in mind that the mechanisms are passive and even if the habitat alteration is successful, there are reasons why that may not correlate with increasing numbers of fish. If current abundances of fish are below the current carrying capacity of the habitat for example, then it suggests something else is limiting population size and increasing habitat capacity further via restoration is unlikely to increase population size. An ecological illustration is the middle fork of the Salmon River; the Frank Church River of No Return wilderness has near-pristine habitat quality and suggesting the need for actions to increase the quality and quantity of habitat is not reasonable. However, Chinook salmon in that area are below carrying capacity and listed as endangered under the ESA.

c. Sources of Uncertainties

Management planning that alters habitat to achieve natural resource responses will have uncertainties. Recent approaches to using the fish abundance metrics to evaluate habitat management and planning (e.g. EDT) have demonstrated a number of critical uncertainties. McElhany et al. (2010) performed an extensive simulation analysis of EDT to evaluate the significance of these issues. That analysis indicated that EDT estimates of fish productivity and habitat capacity were not reliable due to internal parameter uncertainty. However, prioritization of reaches for preservation or restoration based on EDT forecasts were somewhat more robust to given input uncertainties. The interpretation was that EDT may be better as a relative index of where important habitat is, rather than in making specific estimates of fish produced from a given habitat improvement scenario. Like all complex models, EDT outputs are subject to large uncertainties, and therefore it is important to explicitly incorporate the uncertainty and sensitivity analyses into any analyses. Sensitivity analyses should be performed to evaluate the precision of any forecast made with complex models such as EDT (McElhany et al. 2010). As mentioned, much of EDT's products are heavily influenced by subjective, “expert-panel” inputs, rather than data-based, scientific process. Uncertainties introduced with this approach generally have been shown to be highly imprecise, untestable, non-transparent and unreliable (e.g. Burgman et al.

2011). Given the magnitude of these uncertainties, a transparent use of EDT for NEB determination will require an evaluation of the sensitivity of the forecasts to subjective opinion.

E. REPLACING FISH PRODUCTION

Rather than relying on fish abundance, a fish production replacement approach to NEB determination uses population production metrics to evaluate NEB. This amounts to replacing lost fish production in impact areas with equivalent or greater production in offset areas through management actions that are believed to change population growth rate or mortality. As such, this can be viewed as another example of resource-for-resource replacement (Lipton et al. 2008; Clarke and Bradford 2014). Reliance on productivity has a number positive attributes, including direct measures of the productive capacity of a given habitat unit, feeding directly into fisheries-related yield estimates, and in some cases estimation from smaller, less extensive sampling than abundance. However, data needs are often more intensive (one needs more detailed information, albeit from perhaps fewer samples), and population-level assessments must rely on models and methods that are often complicated and technically challenging. Therefore, this approach is likely to be most appropriate when the assessment units are large and heterogeneous.

Estimating fish productivity requires developing a relationship between current abundance of “parent” fish (spawners) and the numbers of “offspring” fish that will return in the future (recruits). In salmon and steelhead trout management, a common approach to estimating productivity are spawner-recruit models. There are a number of familiar formulations of these models that include Ricker, Beverton-Holt, Schaefer or Fox models. Once a particular model is chosen, the parameters must be estimated from the data.

In particular, a Ricker model has been a popular choice due to the ease with which it can be formulated as a linear regression model, such that

$$\log_e(R_t / S_t) = a - b \times S_t + e_t,$$

where R_t is the total number of surviving recruits from brood year t, S_t is the number of spawners in brood year t, a is the intrinsic productivity (i.e., the number of recruits per spawner in the absence of density dependence), b is the per capita strength of density dependence, and e_t is the observation error in brood year t. From a set of R_t and S_t values the log of R_t / S_t is regressed on S_t; the intercept is a and the slope is b in the equation above. The Ricker model is particularly convenient in that the carrying capacity and intrinsic productivity of the population are estimated directly from this regression.

Before beginning with the model fitting, however, R_t must be estimated. For organisms that breed once and all at the same age, the number of recruits is the number of breeding organisms surviving from a prior brood year. In most cases, and in salmonid fishes certainly, the animals that join breeding populations can be from several different years’ production, each with a different survivorship. In these cases we need to evaluate the regression above for recruits that may accumulate over several subsequent years. This is often done with a brood table or run reconstruction, which in turn is derived from combining observed age composition and total spawner counts.

Productivity varies year to year, and so this approach requires multiple years of data to provide reasonable estimates. Indeed, the intrinsic variability in production can, in many cases, require many years of data. It is fortunate that salmon monitoring in the Pacific Northwest has been intensive and ongoing for many years. This is in contrast to many of the fish abundance replacement approaches which rely on a snap shot in time, but extensive data in space. However, if time series data are available it is also possible to evaluate the degree to which co-varying habitat conditions affect the estimates of *a* and *b*, and in so doing develop an estimate of how engineered changes in habitat may alter fish production. This is one approach to forecasting the anticipated positive effects of offset projects for NEB. The workflow for a fish production estimate is illustrated in Fig. 9.

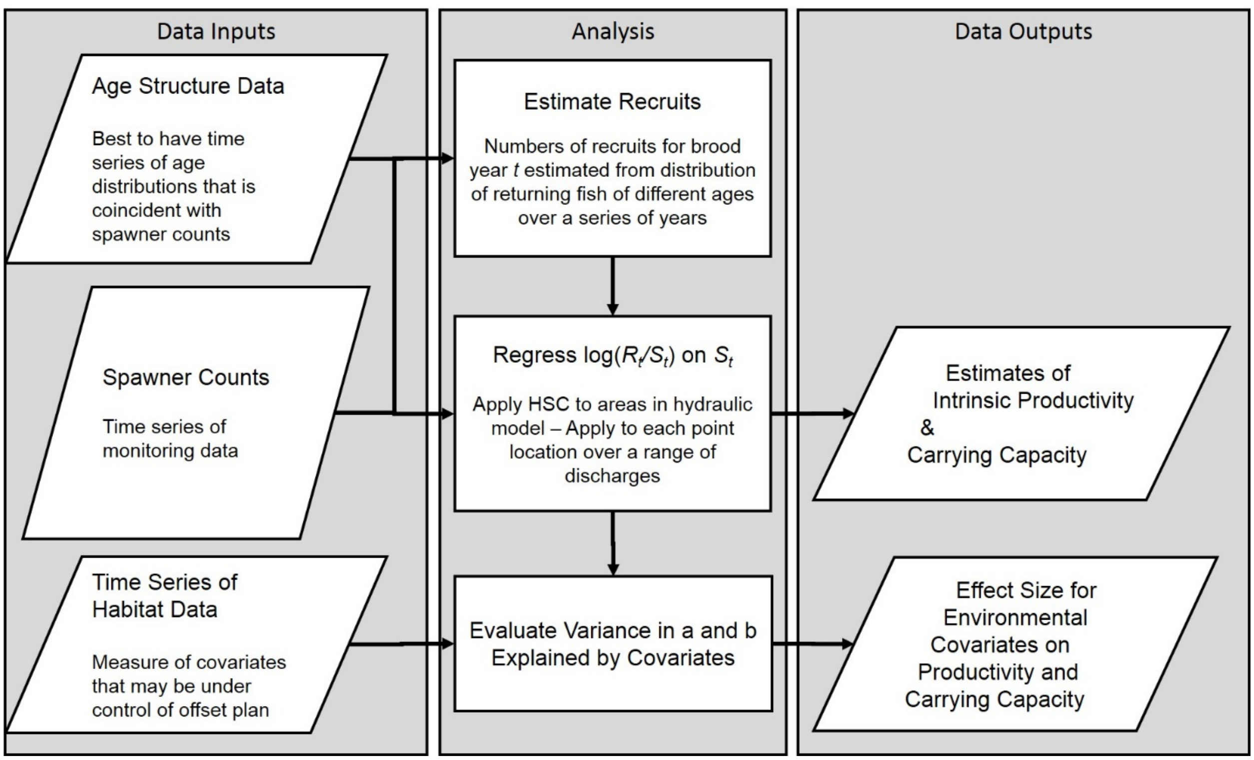

Figure 9 Fish Production replacement workflow. User data enters the analysis in the form time series of spawner counts, age distributions and environmental covariates. Spawner counts from prior years are combined with age structure data to estimate a time series of returning recruits. Production and Carrying capacity estimates for the planned habitat unit are estimated with a linear regression model in the case of a Ricker approach. Slightly different mathematical formulations are used in alternatives to the Ricker model. The outputs from the regression are the principal Data Outputs. If time series data of habitat covariates are available these can be used to explain variation in the productivity and carrying capacity estimates in order to forecast the changes in future production from habitat management actions performed now.

Although convenient, there are a number of inherent challenges when estimating the model parameters in this manner. First, the available raw data are used inefficiently, in that information is lost when summarized into brood tables to calculate demographic rates. Second, the spawner and age data are rarely, if ever, comprehensive or error free due to imperfect detection, misidentification, and non-exhaustive sampling in collecting field data. When not appropriately addressed, these errors in population census may underestimate recruitment (Sanz-Aguilar et al. 2016) or overestimate the strength of density dependence (Knape and de Valpine 2012). Third, failure to acknowledge trade-offs among parameters and the fact that any given type of data (e.g., age structure) may contain information on multiple aspects of population dynamics (e.g., recruitment and survival) can lead to biased parameter estimates.

a. Data and Methods

Fish production rate is the net generation of new biomass in a stock per unit time, whether or not it survives to the end of that time (Ricker 1975). The time unit used to represent the rate can be a variety of units, but in salmon and other fish that have a strong seasonality to their presence in fresh water, the most common time unit is annual. Therefore, the unit of measure for expressing fish production is most often either fish in numbers/year or biomass in kg/year. Consequently, there is an immediate need for data on fish numbers and size. Data on fish abundance and size (i.e. annual estimates of adults and/or biomass) can be collected directly in impacted areas using the methods described above, but estimates of fish abundance for the offset areas will have to be obtained from modeling or other forecasts. The models used in this approach to forecast population responses to habitat management are increasing in use, but remain somewhat rare, and are often complicated. Regardless of the modeling approach, it should include explicit metrics that can be deployed in effectiveness monitoring to allow forecast validation as well as inform the triggering of contingencies for failure to meet forecast goals.

Approaches in use to characterize population productivity in fish include metrics based on and derived from:

- Population structure (e.g. distributions of body size, Productivity: Biomass Ratios),
- Size structure,
- Habitat Productivity Index (HPI) is the product of P:B ratio and seasonal biomass (Randall and Minns 2000, 2002).
- Individual vital rates including growth, survival, fecundity

Methods for implementing these approaches to assessing population productivity are presented in numerous fisheries texts including:

- Hilborn, Ray, and Carl J. Walters, eds. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Springer Science & Business Media, 2013.
- Gulland, John Alan. Fish stock assessment: a manual of basic methods. Vol. 425. New York: Wiley, 1983.
- Pauly, Daniel, and G. R. Morgan, eds. Length-based methods in fisheries research. Vol. 13. WorldFish, 1987.
- Haddon, Malcolm. Modeling and quantitative methods in fisheries. CRC press, 2010.

There are a number of modeling approaches that can be used to investigate the mechanistic relationships between habitat and production. Two of the more common are stock-recruitment models and stage structured habitat supply models.

Stock-recruitment models were originally developed for harvest management and extractive fisheries (e.g. Ricker 1954). Although the techniques have evolved and developed into more general population modeling schemes, the data needs, assumptions and challenges to use remain, as do debates regarding the choice of model form in any given application (e.g. how density dependence is represented in Beverton-Holt vs. Ricker formulations). Stock-recruitment models have high utility in that they incorporate estimates of recruitment, intrinsic growth rates, survival, fecundity and environmental carrying capacity, and they produce estimates of surplus production and sustainability targets for harvest. All of these population properties have wide utility in fisheries management (e.g. Gibson 2006; Parken et al. 2006), although it is less clear that they have similar utility in conservation and habitat-population management scenarios. Most estimation approaches rely on linearizing the stock-recruitment relationship with a log transform of the data, estimation of the parameters and then transform back to linear space where the parameters are reported. While this provides conceptual simplicity and useful outputs (Clark et al. 2009), it results in very large uncertainties in the estimates of recruitment (Ludwig and Walters 1981; Hilborn and Walters 2013), and many of the assumptions are problematic (Walters and Ludwig 1981; Walters 1985, 1987; Kehler et al. 2002; Kope 2006).

Stage-structured models are an alternative that recognizes that fish will encounter different sources of mortality at different times in their life histories. The modeling approach is to take the entire life history of the fish and divide it into a number of stages; the net survivorship is the resulting cumulative probability of survival at each of the steps or life-stage transitions over the lifetime of the fish (Nickelson 1998; Nickelson and Lawson 1998). In salmon, where there is a protracted fresh water period with several recognized developmental stages, it has been possible to construct life cycle models with many survivorship steps. The net survival of fish is calculated as a long series of multiplications of numbers between zero and one (survival probabilities range from 0 to 1), and for the whole life history it can be a very long series. As a consequence, even if survivorship for a specific step is high, or made high by a specific management action, the net survivorship works out close to zero. This is not a surprise when we remember that female fish may lay 3,000 to 7,000 eggs in a salmon redd (Groot and Margolis 1991), but only two fish survive to reproduce if the population is just replacing itself. The other consequence however, is that our sensitivity to detect small changes at specific life history steps is relatively low when we are looking at a population level outcome, such as numbers of returning adult fish

Much of the current paradigm for endangered fish recovery is based on a life-cycle concept. In principle, if we change the mortality at a specific step with a habitat restoration project for example, we could increase the net overall production of fish and put the population on a trajectory to recovery. Unfortunately, this paradigm has an important limitation in that the effect of any change in survival itself is probabilistic. We can’t specify how many fish will survive passing a given dam, or other threat, we can only say what the probability of survival is and if sufficient monitoring data exists, what change in the probability is likely for a given management action. The good news is that while we cannot predict the fate of an actual fish, the probabilistic nature of survival provides a mechanism to estimate our uncertainty in any estimate.

b. Assumptions and Implications

Both life cycle and stock recruitment modeling approaches rely on many of the same data types as the other NEB determination approaches. Therefore, this approach has similar assumptions and constraints as the other methods with respect to data. In addition, the modeling approaches currently in use can become quite complicated. For example, life cycle models in particular can have many steps with different values of survival for each. This is problematic both because the model complexity/bias uncertainty rises and also because there is less monitoring data and empirical studies to support the estimate of survival in a specific habitat and fish life history context. In many of these cases of low empirical data availability, planners have resorted to expert opinion and this introduces additional problems of transparency and model validations (McElhany et al. 2010).

Because production approaches are built on our understanding of a population process, its relevance is greater on the scale over which the survivorship or stock recruitment processes operate. This is usually a large scale, and in the case of endangered anadromous salmon, the scale is the whole population (Bradford et al. 2014). For example, the premise of a habitat restoration project is that by improving habitat-related survivorship we will see an increase in the number of wild fish. This may be a reasonable hypothesis, but we have to keep in mind that the mechanisms are passive—we are not making new fish--and even if the habitat alteration is successful, there are reasons why improved habitat may not correlate with increased fish production. Indeed, in the Columbia River there are a large number of potential sources of mortality occurring outside the basin, with as little as 34 to 64% of mortality occurring in the freshwater life history of anadromous salmonids (Bradford et al. 2014). Thus, improvements in early life survivorship due to actions in the NEB determination may be entirely successful, but out of basin mortality may prevent any of that success from being measurable into the future.

c. Sources of Uncertainties

Uncertainties from this approach to NEB determination arise both from the data and metrics, as well as the modeling approaches chosen to forecast offsets. With respect to data and metrics, the uncertainties are similar to the other approaches to NEB determination that rely on fish abundance and habitat condition measures. Life cycle models have the additional complication that these same data are required for possibly many life stages and habitat conditions.

When any of these approaches to fish production replacement are linked to habitat there are important uncertainties related to ecological context. Empirical measures of habitat affecting life stage survival may be just as site-specific as habitat preference described above. In cases where fish population responses are inferred from localized studies, the lack of transferability of the estimated relationships between habitat and survival may be just as uncertain as habitat capacity modeling.

Perhaps the largest uncertainty is the complex nature of productivity models as forecast tools for the NEB determination. As mentioned above, these models are generally some of the more complex models in fisheries and conservation use, and this complexity exists in models that do not make explicit linkages to habitat metrics as covariates or drivers of population processes. This model complexity imparts large uncertainties to any forecast made that bases fish production from habitat changes. Also important is that the uncertainties in the data that goes into the models and the uncertainties arising from the models themselves may interact in more than a simple additive manner and produce unexpectedly large uncertainties in the forecast results (Caputi 1988). Therefore, if NEB determination is to be made with a fish production replacement approach, large increases in proposed offsets may be needed to increase the likelihood that NEB will be net positive by the end of the planning period. Certainly, monitoring, validation metrics, timelines and triggers for contingencies in the event of failure to reach validation targets will be prudent components for watershed plans developed under RCW 90.94.

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Table 1.

NEB Determination Approach Comparison Summary					
Type of Environmental Offset	In-Kind, In-Place	Out of-Kind, Out of-Place			
	In-Kind Replacement Water for Water	Habitat Function Substitution	Habitat Substitution for Specific Species	Fish Abundance	Fish Production
Example	Water for Water	HEA	PHABSIM	EDT Fish-Flow curves	Life-Cycle Modeling
Basic Information Outputs					
Produces Quantitative Measures of flow? (Will it connect consumptive withdrawals to quantitative changes in instream flow?)	Yes	No	No	No	No
Produces Quantitative Estimates of Habitat Response?	No	Yes	Limited, does estimate hydraulic condition at different flows. However, some professional judgment or input from other model or framework is required to relate hydraulic condition to quantitative habitat responses (e.g. step-backwater modeling, depth and velocity regression on transects, or two-dimensional hydraulic modeling based on channel roughness and flow routing for hydraulic component)	Yes, depending on scale and dimensions and input of habitat variables (cover, mesohabitat, other options)	No
Produces Quantitative Estimates of Fish Population Response?	No	No (associated estimates are from external judgment or model)	Yes, but based upon flow-habitat input	No – requires interpretation of habitat response via model or judgement.	Yes
Estimates responses in other Ecosystem	No	Yes	No	No	No

Goods and Services? (e.g. Recreational, Aesthetic, etc.)					
Qualities of Information Produced					
Data and Methods					
What data are required to perform the assessment?	Streamflow	Impacted habitat estimate; multivariate description of the habitat, such as habitat structure, cover, or substrate type;	Suitability or preference metrics at microhabitat (or meso/marcro habitat) level incl. depth, velocity, substrate, roughness and/or cover) for each species life-stage	Hydrology, sediment, channel dynamics, riparian/habitat function, total fish abundance as count, density, fecundity, biomass, catch per unit effort	Fish age structure (e.g., growth, survival, fecundity), spawner information (e.g. number or biomass per time), habitat productivity, Productivity: Biomass ratios.
Spatial Issues					
Transferability If we have data for one location, can you extend developed inferences to other locations?	No	No	No	No	No
		In all cases, there have been implementations where fish:habitat relationships have been developed in one place and deployed elsewhere. This is done, but it is difficult to support technically, and if done, should be accompanied by extensive validation monitoring.			
Spatial Extent	Stream reach	Arbitrary	Typically a representative reach scale (Can be expand to larger segment-scales if habitat weighting is used. Modeled results have been used to inform watershed-scale decisions for planning and policy purposes.)	Can be made up of very small scale – how close do you want to space measurement points, and at the cost of time and effort	Limited to domain over which the supporting data are relevant. Commonly the distinct population segment (DSP).
Spatial Resolution		Arbitrary (Commonly determined by the spatial resolution of the inventory of habitat units)	Commonly 100 meter reaches	Reach (Although, the resolution is the reach, supporting data at that scale are often unavailable, and supplemented with expert opinion.)	Limited to domain over which the supporting data are relevant. Commonly the distinct population segment (DSP).
Assumptions and Statistical properties of Ecological Benefits Forecast					

Assumptions	That water quantity is a surrogates for habitat metrics and population response	Assumes ecological function is equivalent to fish production	Assumes stationarity in a number of aspects: • that all space are equivalent • static in time, independent of species/life stage, fish have a global knowledge of site suitability	Assumes future ecosystem is similar to the current ecosystem; restoration efforts are directly correlated with wild fish recovery	Ability to accurately model multiple fish life stages; relies on heavily on empirical data or expert opinion
Implications	Any Net benefit from new habitat will have to meet or exceed the lost function in excess of the existing production in the offset location.	Aggregated data may mask detail among data that actually determine fish production; results will depend on structure of the model used and data used.	The pattern of fish density observed at any moment in time is a snap shot of interactions between hydraulic, habitat and biological processes that may or may not permit use of the assumption that HSC are transportable across time and space	Difficult to account for environmental changes associated with a changing climate; restoration activities may not be sufficient to increase wild fish population (other factors may play a role)	Models provide a better understanding for larger scale/process-level estimation, while specific habitat alterations may exact a more local scale response
Uncertainties					
Where do critical uncertainties lie?	Habitat for habitat on a quantity basis is low uncertainty; establishing habitat for habitat on a quality basis is less certain.	Degree of uncertainty is metric-dependent-greater uncertainty for metrics representing complex features (e.g. habitat) and information-poor metrics; reduces complex ecological services to a single metric; does not account for injuries that	Assumptions of fish-habitat associations; portability of fish-habitat relationships; mismatches of scale and/or data;	Estimates of fish productivity and habitat capacity are estimated using other parameters with varying uncertainty; depend heavily on “expert-panel” inputs	Assumptions of fish-habitat associations; portability of fish-habitat relationships; Empirical measures of habitat affecting life stage survival may be just as site-specific as habitat preference; complex models do not make explicit linkages to habitat metrics as covariates or drivers of population processes.

		accumulate over time			
Does it produce a measure of precision of forecast? (What is the uncertainty?)	Lower for well-studied systems when using common/well-understood measurement methods	Can require data aggregation which can mask more detailed/ nuanced relationships	No- hydraulic modeling on one scale married to finer scale habitat data can misrepresent fish preference	Not as currently done, although it could be built in, particularly where hydraulic models are based on regression. Suitability criteria input could also be developed with estimates of precision, but would require extensive work	Estimates may be more robust at the larger/process-based scale
Does it produce a measure of accuracy in the forecast? (How far from the forecast is the truth?)	Higher when modeled as described above	Dependent on data quality	If quantitative habitat:fish models are used, estimates of forecast accuracy are possible. However, as commonly deployed, these approaches rely extensively on expert judgement which does not allow a quantitative assessment of accuracy (see above). As such, resolving what constitutes an estimate of accuracy and how it is handled may represent a policy choice rather than a technical or scientific choice reflecting case-specific, and potentially competing, values.	No, but see above- while EDT does provide specific forecasts, its uncertainties mean its accuracy cannot be evaluated.	Yes, However, uncertainties arising from models being sensitive to data variance can be large. In addition, uncertainties arising from the models themselves may interact in more than a simple additive manner and produce unexpectedly large uncertainties in the forecast results
Is it transparent? (Are all data sources, assumptions and methods documentable and inform estimates of accuracy and precision?)	Yes in principle.	Aggregation and expert opinion can reduce transparency	Yes in principle.	No. As currently implemented, relies extensively on expert opinion, which may be subjective or imprecise.	Yes in principle.

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5. APPENDICIES

Appendix 1: Economic valuation

1) Table A1

Table II: Comparison of Willingness-to-Pay Estimates for Anadromous Salmon in the United States, Post- 1990

Year	Type	Authors	Study location	Sample size	Payment Period	Payment Frequency	Baseline (in 1000s fish	Change (in 1000s fish)	Amount	Payment vehicle	Survey method	Income elasticity
1990	CVM - DC	Hanemann, Loomis, Kanninen	CA, OR, WA	1003	NS	annual	0.1	15	$324	taxes	mult	NR
1991	CVM - OE	Olson, Richards, Scott	WA, OR, ID, MT	1400	one-time	month	2500	2500	$49 - $137	electric bill	phone	NR
1991	CVM - DC	Stevens et al	MA	1000	5 years	annual	NA	complete loss	$13 (Atlantic salmon only)	trust fund	mail	NR
1992	CVM - OE	Duffield and Patterson	MT	796	one-time	lump sum	NS Status Quo	NS	$31	trust fund	mail	NR
1996	MA	Loomis & White	NA	NA	NA	NA	NA	NA	$100	NA	NA	NR
1996	CVM - DC	Loomis	WA & USA	1174	10 years	annual	50	350	$91 - $113	taxes	mail	NR
2001	CVM – R	Layton, Brown, Plummer	WA & OR	1611	20 years	month	2000	3000	$167	utility bill	mail	NR
2003	CVM - DC	Bell, Huppert, Johnson	WA & OR	2209	5 years	annual	64 - 69	64 - 146	$101 - $162 (WA)	taxes	mult	NR
2006	WTP - MC	Montgomery and Helvoigt	OR	5300	NS	month	SQ	NS	$15 - $46 (mode)	utility bill	mail	NR
2007	BT (LBP 1999)	Goodstein and Matson	OR & WA	NA	NA	NA	NA	33 - 66% decrease	$33-$144	NA	NA	0.3
2008	MA	Martin-Lopez et al	NA	NA	NA	NA	NA	NA	$76 - $149	NA	NA	NR
2009	MA	Richardson, L. and Loomis, J.	NA	NA	NA	NA	NA	NA	$92	NA	NA	NR
2009	BT (Loomis 1999)	Helvoigt and Charlton	OR	NA	NA	NA	NA	NA	$33	NA	NA	NR
2009	CVM - CE	Rudd, M.	Canada	2761	20 years	annual	SQ	50 - 200% increase	$86	taxes	online	NR
2012	CVM - CE	Johnston et al	RI	522	NS	annual	SQ	NS	NA	taxes	mail	NR
2012	CVM - CE	Wallmo, K. and Lew, D. K.	US	8476	10 years	annual	SQ	De-list as threatened	$40	taxes	online	NR
2012	CVM - CE	Mansfield et al	OR & CA	3,372	20 years	Annual	SQ	30 – 150% increase	$121 - $213	Taxes	mail	Constant real income

2) Annotated Bibliography

Willingness to Pay

Hanemann, M., Loomis, J., & Kanninen, B. (1991). Statistical efficiency of double-bounded dichotomous choice contingent valuation. *American journal of agricultural economics*, *73*(4), 1255-1263.

The authors conduct a survey of 1,003 residents in Colorado, Oregon, Washington and California to compare willingness-to-pay for various environmental improvement programs in California's San Joaquin Valley. One of those programs is a salmon improvement program. Bid amounts range from $45 - $225 with hypothetical payments made through annual taxes. On average, respondents are willing-to-pay $324 each year to increase fish populations from 100 to 14,900. The authors compare results obtained through both one round and two rounds of "yes/no" questions posed to respondents. The authors concluded that the sequential survey question format provides more efficient estimates of willingness to pay than those obtained through a single question.

Olsen, D., Richards, J., & Scott, R. D. (1991). Existence and sport values for doubling the size of Columbia River basin salmon and steelhead runs. *Rivers*, *2*(1), 44-56

The authors estimate the existence value of doubling Columbia basin salmon runs from 2,500,000 to 5,000,000 fish. A mail survey is used to solicit responses from 1,400 residents of Idaho, Montana, Oregon and Washington, exactly half of which are participants in the commercial fishing industry. Hypothetical payments are made through annual increases in electric bills, and the authors use an open-ended question format. Total economic value is reported as approximately double the non-use values. Non-users of the Columbia basin fishery express an annual willingness-to-pay of $49 per household; for people who may fish at some point in the future, $108; for those who currently participate in the sport or commercial fishing industry, $137.

Stevens, T. H., Echeverria, J., Glass, R. J., Hager, T., & More, T. A. (1991). Measuring the existence value of wildlife: what do CVM estimates really show?. *Land Economics*, *67*(4), 390-400.

The authors conduct a survey of 1000 New England residents soliciting willingness-to-pay to avoid funding cuts for species preservation programs. A sequence of two "yes/no" questions was posed to respondents asking if they would be willing to pay specific amounts between $5 and $150 to a trust fund for the purpose of protecting specific species, including Atlantic salmon. Individuals were told that species would not survive unless the fund was created. Average values for Atlantic salmon were reported as $13 per person. Only 12% of respondents had reported seeing Atlantic salmon. 52%

of respondents did not think their opinion would matter for policy decisions. 64% of respondents expressed a willingness-to-pay of $0.

Duffield, J. W., & Patterson, D. A. (1992). Field testing existence values: comparison of hypothetical and cash transaction values. *Benefits and Costs in Natural Resource Planning, Oregon State University*.

Authors examine the importance of "hypothetical bias" when it comes to valuing instream flows. Hypothetical bias is the tendency of respondents to overstate willingness-to-pay when payments are not actually made. The authors conduct a mail survey of 796 individuals with registered fishing licenses in Montana, approximately half of which are state residents and half non-residents. One version of the survey solicited one-time, actual cash donations to finance a fund for instream flows that would be established through the Montana Nature Conservancy. A second version was identical except that trust fund donations were hypothetical. A final version was similar to the hypothetical version, except that it was delivered through a separate mailing from the University of Montana. Respondents were asked to express Willingness to Pay (WTP) through payment cards in amounts from $10 - $250. Response rates to the actual payment program were only 10%, though the authors did not conduct re-contact respondents for any mailing sent through the Nature Conservancy. Of TNC mailings, the average contributions made by residents to the actual trust program was $31.34 (adjusted to reflect $2012). To the hypothetical program, WTP was reported as $26.25. For non-residents, WTP was reported as $50.04 (actual) and $56.06 (hypothetical).

Loomis, John B. "Measuring the economic benefits of removing dams and restoring the Elwha River: results of a contingent valuation survey." Water Resources Research 32.2 (1996): 441-447.

The author conducts a survey of 1,174 residents in Washington State and other US residents. Respondents are asked to express willingness-to-pay to remove two dams on the Elwha River in order to restore river runs to a natural, pre-dam state. Hypothetical payments were made through annual federal taxes over a period of 10 years and reflect non-use values associated with wild salmon, as opposed to generic salmon populations that include both wild and hatchery fish. Available bid amounts range from $3 to $190 and are solicited through a dichotomous choice, voter referendum format. It is assumed that dam removal would increase salmon populations from 50,000 to 350,000. On a per household basis, the mean annual WTP in Clallam County, WA is $91 mean annual WTP; for rest of Washington State, $113; for the rest of the United States, $105.

Bell, K. P., Huppert, D., & Johnson, R. L. 2003. Willingness to pay for local coho salmon enhancement in coastal communities. *Marine Resource Economics*, *18*(1), 15-32.

The authors examine the willingness-to-pay of coastal residents in Oregon and Washington for various coho salmon enhancement programs. The estimates presented are more variable than those of other studies. In total 2,209 respondents were recruited from Grays Harbor, WA; Willapa Bay, WA; Coos Bay, OR; Tillamook Bay, OR; and Yaquina Bay, OR. Questions were posed through a voter referendum format conducted through a combined mail and telephone survey approach. Costs of the programs ranged from \$5 to \$500 in annual tax payments over a period of 5 years. Fish population increases in Washington ranged from 200% to 400%. In Oregon, residents were asked to value fish population increases of sufficient size to de-list Coho as a threatened species under the Endangered Species Act. The authors find wider variation in willingness-to-pay between the Oregon locations than for Washington, and they report a *lower* willingness-to-pay for conservation programs that would result in *higher* increases in coho populations. They also find that participation in the local sport fishing industry is significantly associated with willingness-to-pay, and affiliation with environmental groups is affiliated with greater willingness-to-pay for some, but not all survey locations. For the Washington survey locations, average annual household WTP ranges from \$101 - \$162.

Montgomery, C. A., & Helvoigt, T. L. (2006). Changes in attitudes about importance of and willingness to pay for salmon recovery in Oregon. *Journal of Environmental Management*, *78*(4), 330-340.

Since 1996, the biennial Oregon Population Survey has two questions regarding salmon restoration efforts. First, “as you may know, salmon runs are declining in Oregon. How important do you feel it is to improve salmon runs in Oregon?” Second, “How much per month would you be willing to pay to for water quality and habitat improvement efforts to help improve salmon runs in Oregon?” Oregon residents have become less supportive of salmon recovery efforts from 1996- 2002, and the authors attempt to explain those changes. More than 30% of respondents were willing to pay \$1-\$3 per month in 2002 (the largest category for each survey year). Greater willingness to pay is reported for younger and unmarried respondents, males, American Indians, those with higher levels of education, people living in urban areas or areas that are less economically depressed. Long-term trends are not clear, as “an important portion of the decline in expressed support for salmon recovery and salmon recovery efforts is not explained by [socioeconomic information]” (p.2006, p.338).

Wallmo, K., & Lew, D. K. (2012). Public Willingness to Pay for Recovering and Downlisting Threatened and Endangered Marine Species. *Conservation Biology*, *26*(5), 830-839.

The authors conduct an online survey of 8,476 randomly selected U.S. households to estimate willingness to pay to downlist eight threatened and endangered species. Hypothetical payments would be made annually so that species could be de-listed 50 years in the future. Respondents expressed a willingness to pay of $40.65 ($37.94, $43.19) for Chinook salmon in the Willamette River and $40.49 ($37.91, $42.87) for those in Puget Sound. The survey took the form of a choice experiment. Responses were dropped for individuals who either unsure about their feeling regarding threatened and endangered species or not confident in their answers. The authors caution against using the estimates in benefit transfer applications to value more than three species at once.

Johnston, R. J., Schultz, E. T., Segerson, K., Besedin, E. Y., & Ramachandran, M. (2012). Enhancing the content validity of stated preference valuation: the structure and function of ecological indicators. *Land Economics*, *88*(1), 102-120.

While the public has expressed a willingness to pay to protect fish populations, the authors test whether values are sensitive to alternative ecological indicators of fish population changes. Indicators include the amount of river acres that are made accessible (1), the probability that a restored fish run will still exist in 50 years (2), changes in harvest (3), the amount of wildlife (4), and the overall ecological condition of a watershed as measured through a biological index (5). Public access to enhanced streams is associated with an additional $20 per household per year compared with streams with no public access. One percentage point increases in both biological quality index scores and the number of new acres made accessible to migratory fish is associated with an $0.80 increase in annual, per household values. This is approximately twice the effect of a one percent increases in harvest and one percent increase in the probability of fish survival 50 years later. Results were obtained from a 2008 mail survey of Rhode Island residents and a review of the existing scholarly literature. See also Johnston et al (2005) and Zhau et al (2013).

Mansfield, Carol, Van Houtven, George, Amy Hendershott, Patrick Chen, Jeremy Porter, Vesall Nourani, and Vikram Kilambi. Klamath River Basin Restoration Nonuse Value Survey. RTI International, 2012.

Mansfield and colleagues estimate the total economic value of salmon restoration in the Klamath River Basin of southern Oregon and northern California. The study was commissioned by USBR and asks respondents to express preferences between the status quo and the proposed alternative Klamath River Basin Agreement, a river restoration project that would remove four dams. Respondents are asked to make hypothetical, annual payments over 20 years through federal taxes. The survey asks respondents to value increases in wild salmon populations from 30 – 150% and changes in extinction risks (low, moderate, high, very high). The survey was administered through the mail in June 2011, but respondents had the option to complete the survey online. The sample was stratified into three geographic zones, oversampling residents in the 12-county area closest to the Basin. The overall response rate was 32.8%. The survey included cheap talk and followed up the valuation question by asking respondents how sure they were about their response. The authors estimate annual household WTP for the project in the amount of $121 for Klamath area households; $213 for all other U.S. households. A reduction in extinction risk for coho salmon from "very high" to "high" is associated with an annual WTP of $70 for Klamath-area households, $54 for households in the rest of Oregon and California; and $78 for households in the rest of the U.S.

Meta-Analyses

Loomis, J. B., & White, D. S. (1996). Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics*, *18*(3), 197-206.

The authors conduct a meta-analysis similar to Richardson and Loomis (2009). They use the same variables to explain willingness-to-pay estimates, though they do not control for survey mode. The authors review 20 studies from both the published and non- published literature and report "best" estimates where multiple estimates are reported from a single study. They report an annual household willingness-to-pay (in $1993) for Pacific salmon/ steelhead of $49 - $140 (average $100); for Atlantic salmon $11 - $13 (average $13). Neither survey response rate nor study date was found to have a significant influence on willingness-to-pay estimates in any model estimated. Over 50% of the variation in willingness-to-pay estimates is explained by payment frequency, change in population size, species type, and whether respondents are visitors or residents. They argue that economic values are insensitive to the format of questions posed to respondents.

Johnston, R. J., Besedin, E. Y., Iovanna, R., Miller, C. J., Wardwell, R. F., & Ranson, M. H. (2005). Systematic Variation in Willingness to Pay for Aquatic Resource Improvements and Implications for Benefit Transfer: A Meta-Analysis. *Canadian Journal of Agricultural Economics/Revue canadienne d'agroeconomie*, *53*(2-3), 221-248.

Meta-analysis is used to determine the importance of contextual variables on estimates of total economic value of water quality and habitat improvements that benefit aquatic species. In total the authors use 81 observations from 34 studies conducted from 1973 – 2001, two of which include salmon and steelhead (Olsen et al, 1991; Loomis, 1996). The preferred specification assumes a semi-log form, where all right hand side variables are additive. Multilevel models are used reject random effects that might explain willingness-to-pay estimates through unobservable, study specific characteristics. Willingness-to-pay increases with the number of waterbodies affected and water quality enhancements.

The authors distinguish between valuations for large and small fish population increases (greater or less than 50%). Estimates vary by region and methodological approach, with lower estimates reported in the Pacific Northwest and higher values associated with both mandatory payments and higher response rates. The authors find that most studies express WTP in terms of annual payments of an indefinite duration. Where the duration is reported, a short time horizon is most common (i.e. 3-5 years).

MARTÍN-LÓPEZ, Berta., Montes, C., & Benayas, J. 2008. Economic valuation of biodiversity conservation: the meaning of numbers. *Conservation Biology*, *22*(3), 624-635.

The authors conduct a systematic review of 60 articles valuing different indicators of biodiversity in an attempt to explain variation in the reported estimates. Two studies relate to values for Atlantic salmon (Stevens et al, 1991; Bulte and Kooten, 1999) and two studies relate to values for Pacific salmon/ steelhead (Hanemann et al, 1991; Olsen et al, 1991). In per household terms, the authors report an average annual willingness to pay for Chinook salmon of $76 and for steelhead, $149. Results largely confirm the findings of previous meta-analyses (Loomis and White, 1996; Richardson and Loomis, 1999) that explain variation in willingness to pay estimates from multiple studies.

Richardson, L., & Loomis, J. (2009). The total economic value of threatened, endangered and rare species: an updated meta-analysis. *Ecological Economics*, *68*(5), 1535-1548.

The authors conduct a meta-analysis of 31 estimates of public willingness-to-pay for threatened and endangered species from academic studies published from 1984-2001. Updated from Loomis and White (1996), the study reviews four previous estimates for Pacific Salmon (Olsen et al, 1991; Loomis, 1996; Layton et al, 2001; Bell et al, 2003) and one for Atlantic salmon (Stevens et al, 1991). Average error of the willingness-to-pay model compared with individual study estimates ranges from 34-45%. Willingness-to-pay increases with changes in population size (1), payment frequency (2), dichotomous choice survey formats (3), respondents who are visitors rather than residents (4), more recent study years (5), mammals compared to other species types (6), “charismatic” species (7), phone and in person surveys compared to mail survey modes (8), and lower response rates (9). An indicator of survey quality, higher response rates tend to be associated with lower WTP estimates. LBP is the only study to use a variation of traditional contingent valuation methods, which the authors refer to as conjoint technique. The authors point out that this technique tends to generate higher WTP estimates (Stevens et al, 2001) and "drives a lot of the difference between new and old studies" (2009, p.1542). The annual economic value of salmon/ steelhead ranges from $11 to $158 (average $92) per household in $2012. The authors prefer a double log specification that includes a variable for study year (model 3, p.1545). Using this model, a 1 year increase is associated with an 8% increase in economic value.

Other Benefit-Transfer Studies

Goodstein, Eban, and Laura Matson. "Climate change in the Pacific Northwest: Valuing snowpack loss for agriculture and salmon." *Frontiers in Ecological Economic Theory and Application*. Northampton, MA: Edward Elgar (2007).

The authors review previous estimates of the non-use values associated with anadromous salmon and report a range in annual, per household willingness-to-pay from $33 - $144. They then estimate the total willingness-to-pay for Oregon and Washington residents to avoid a one-third decrease in the size future salmon populations as $398 million. They interpret this amount as the required compensation for the public to be "made whole." As a basis for the estimate, the authors use a modified version of the valuation model presented in Layton, Brown and Plummer (1999).

Helvoight T. and Charlton, D. 2009. The Economic Value of Rogue River Salmon. ECONorthwest. Accessed online 6 December 2013 from http://www.americanrivers.org/assets/pdfs/wild-and-scenic-rivers/RogueSalmonFinalReport0130090e8f.pdf

In a report commissioned by the Save the Wild Rogue Campaign, ECONorthwest analyzed the economic value of salmon and steelhead in Oregon's Wild & Scenic Rogue River. Citing Goodstein and Matson (2007), the authors argue that Washington and Oregon residents are typically willing to pay $30 - $130 per year for salmon recovery programs. Using fish count data from the Oregon Department of Fish and Wildlife and estimates from academic studies, the authors then estimate a range of values associated with salmon use. The per fish economic value of commercial caught salmon ranges from $13 - $68 with sport – caught values ranging up to $900 per fish (Meyer Resources, 1986). To estimate non-use values associated with Rogue River salmon, the authors apply a marginal willingness-to-pay function from Loomis (1999). They calculate the total annual non – use willingness to pay for the Rogue River salmon fishery at $1.5 billion, or $32.67 per person per year. The authors assume a salmon population size of 830,000 based on escapement numbers.

Niemi, C.L., E. G., Buckley, M., Neculae, C., & Reich, S. (2009). An Overview of Potential Economic Costs to Washington of a Business-As-Usual Approach to Climate Change.

The authors provide an overview of potential costs that climate change may impose on Washington State residents under a status quo management scenario. With a total projected cost to Washington State residents of $530 million per year in 2020 and growing to $3 billion per year in 2080, decreases in future salmon populations are one of the three largest climate-related costs out of 18 cost categories considered (the other two, estimated for 2020, are $1.3 billion in annual health-related costs and $220 million in energy costs). The authors use the model presented by Layton, Brown and Plummer (1999) as the basis for salmon-related costs, though they assume that the status quo would result in a 22% reduction in the size of salmon runs by 2090.

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Appendix 2: Restoration Metadata Needs for Assessing impacts of Water Plans under RCW 90.94

A. Implementation tracking information

The Streamflow Restoration Act, RCW 90.94, calls for plans to be developed to evaluate the impacts and responses to consumptive water use, and in particular the in-stream impacts of ground water withdrawals. Those plans will include four key functionalities that follow directly from the Act and the interim guidance. When assembled, these four functions form a workflow that addresses the following questions:

1. What are the specific plans for consumptive withdrawals of water (projected new wells in context of existing withdrawals)?
2. What are the forecasted environmental outcomes of the water use described in part 1?
3. What are the planned mitigation/restoration actions anticipated to produce environmental benefits in response to the environmental outcomes identified in part 2?
4. Do the net environmental benefits in part 3 outweigh the outcomes in part 2, or result in a positive NEB?

As a consequence, RCW 90.94 requests planners to draw ***inferences*** regarding the environmental impacts (in the form of forecasts in parts 2 & 4 above) of water withdrawals and management actions in response (parts 1 & 3). Here we describe the types of ***information*** concerning the actions taken that are needed to draw those inferences (see below for rationale). Those information needs, and their technical specification, are relevant to both the forecasting of impacts and the possible monitoring of action effectiveness. Critically, ***information*** in this context is distinct from ***data***; in any given case, different kinds of data (e.g. latitude/ longitude) could convey the same information need (*where is it?*). This distinction also highlights that the need for information follows from the inferences defined in RCW 90.94, but they are not specified by RCW 90.94 itself.

Regardless of the origin of the information need, this document describes the kinds of relevant metadata for actions proposed or undertaken as part of RCW 90.94 planning or assessment, but it does not provide a specific prescription for minimum requirements regarding planning and monitoring data systems. Recognizing that new calls for data and reporting are often perceived as onerous and demanding, where they exist we provide pointers to existing data systems across the region that can deliver those informational needs, and we provide an example data dictionary in the Appendix below that could be used to address these information needs. However, this document avoids specifying a single data system that constitutes a minimum requirement under RCW 90.94.

B. Common information needs consistent with RCW 90.94

There are common information needs for management actions regardless of whether one is planning future actions, or assessing existing actions. The needs derive from the questions being asked and the assessment techniques deployed, and amount to specific information on the ***who***, ***what kind***, ***how much***, ***where*** and ***when*** of the restoration or mitigation that has occurred or is planned for the near term. The specific technical questions raised in any water management plan will be implementation and location specific. However, there are several overarching classes of questions that are likely to be encountered (Katz et al., 2007):

- Q1. How does a single restoration action alter environmental resources?
- Q2. How does a diverse set of restoration actions implemented within some spatial domain, such as a watershed or subbasin alter environmental resources?
- Q3. How does a given class of similar restoration projects alter environmental resources?

In each case, “alter environmental resources” is contextual and must be defined in a manner relevant to the project and the study (and for the purposes of RCW 90.94 is considered elsewhere). In one case, it might refer to alterations in habitat quantity, while in another it could refer to responses seen in a population of salmon that are impacted by a change in some habitat character. For example, to address question Q2, we would need to know about all restoration projects in a particular basin, including their type, extent and abundance. We also appreciate that actions take time to implement, and their immediate impacts on things like salmon may take several years to be realized due to the complex life cycle of those fish. Thus, we would also need to know projects’ planning and implementation dates. In addition, we would also need to connect this information to a functional model that links the impacts of actions to changes in habitat, and perhaps in turn to changes in the net productivity of fish. Information about the distribution of restoration projects and productivity in adjacent basins would provide contrast and thereby separate the impact of those restoration actions from some other large-scale driver of the system such as climate variability. More explicitly, the *who*, *what when*, *where* and *how much* information should include:

- Spatially explicit data on project location (i.e. the work-site), not the location of the project contract (which has been common for project metadata in the past—see Katz et al., 2007). If the planned actions are to be connected to a model of environmental response and ultimately fish response, project data will need to be linked to spatially explicit environmental data. To identify the relevant habitat data to analyze these projects in a particular reach or stream unit, such as stream gradient, vegetation cover type and so on, the geographic coordinates for the restoration project are needed. There are a number of potential data types to express this information including latitude and longitude or LLID (latitude-longitude identification; https://www.oregon.gov/deq/Data-and-Reports/Pages/default.aspx) and stream mile. However, while larger scale spatial data, such as HUC or County, can be easily generated given a latitude and longitude, the converse is not true – given only a County, one cannot translate that into specific locations for the purposes of supporting these assessments. Fortunately, any spatially explicit coordinate system (e.g. latitude and longitude in decimal degrees or LLID and Stream km), the others can be generated in an automated data system. This recommendation is consistent with the Best Practices for Reporting Location and Time Related Data developed by the Northwest Environmental Data-network (NED 2006).

- Project level data on all implementations—not just projects undertaken as part of RCW 90.94. Characterizing the net impact of diverse restoration actions, and clearly identifying areas that are unimpacted by adjacent restoration actions, require knowledge of all restoration actions in the watershed or relevant spatial domain. In the former case one needs to accurately model or forecast the net magnitude of the treatments, while in the latter, one needs to identify the presence of potentially confounding treatments. Therefore, both the design and analysis of net restoration or mitigation require information about RCW 90.94-specific as well as all the other existing projects, regardless of funding source (e.g., SRFB, CBFWA, TNC). Fortunately, there are publicly-available data systems that provide information on pre-existing and non-RCW 90.94 projects (e.g. the Pacific Coast Salmon Recovery Fund (PCSRF), the Pacific Northwest Salmon Habitat Project Tracking Database (PNSHPTD)), so planners have resources to address this need at hand.

- Measures of magnitude or extent of treatment for each action proposed or implemented. These measures of treatment magnitude are useful in several contexts.
    - To identify the net management effect from a diversity of individual project effects, the level of treatment is critical. One would not compare the effect of 10 fencing projects that excluded cattle from 5 miles of stream

length each, with 10 projects that excluded cattle from ¼ mile of stream length each.

- Many forecasts of environmental effects for the purposes of satisfying RCW 90.94 will amount to comparisons of levels of treatment with levels of environmental response. This is illustrated in the figure below (Fig. A2-1), although the actual statistical comparison may be more sophisticated and complicated (e.g. multivariate and/or non-linear or saturating responses), on a conceptual level the comparison is straight forward. If projects are to be forecast as having a net ecological benefit, one expects to see more recovery (e.g. # of fish) with more treatment (e.g. # of culverts), although there may be reasons this relationship would have an upper or lower limit. Therefore, some measure of treatment extent needs to be incorporated into a water plans in response to RCW 90.94.

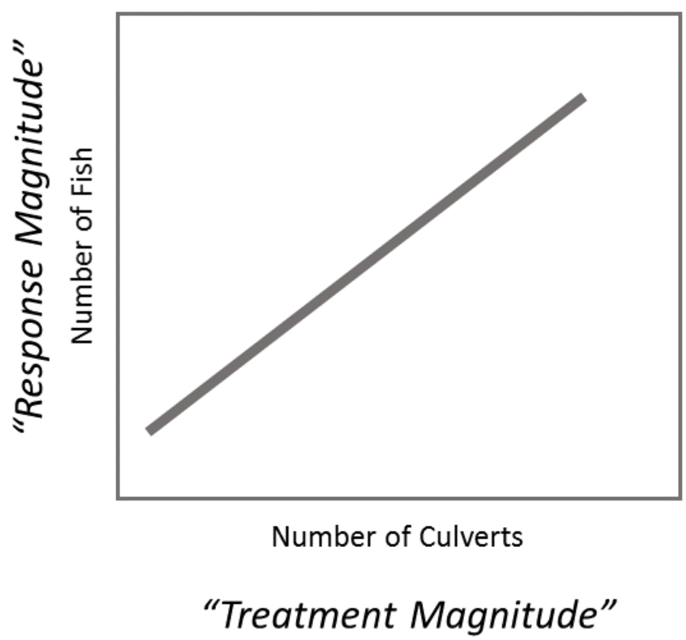

Figure A2-1 Conceptual application of project inventory to assessment of project impact. Conceptually forecasts will amount to estimates of response for a given amount of treatment. Given the lack of simple systems with single restoration types, the actual statistical analysis will require more sophistication.

- Prioritizing project placement – Planning and prioritizing restoration has often occurred at local levels (Beechie et al., 2008). If prior effectiveness monitoring and evaluation efforts are to inform the prioritization of new action implementation at any scale, then some measure of implemented treatments must be available to planners. Historically, project tracking and planning systems in the Pacific Northwest have not included explicit measures of project extent and this has been a significant impediment to regional coordination (Katz et al., 2007; Barnas and Katz, 2010), .

There is a diversity of specific metrics one could employ to express project extent. Indeed, different project types will have different metrics that are relevant to only those projects. For example, change in instream flow is unlikely to be useful to express the extent of a riparian fencing project. Thus, the relevance of a given metric may be case-specific. However, there are existing data management systems that capture and organize information in a manner that is portable across regional planning, funding and monitoring programs (e.g. the Pacific Coast Salmon Recovery Fund (PCSRF), the Pacific Northwest Habitat Restoration Project Tracking Database (PNSHPTD)), and would represent cost-effective data management to satisfy data needs for NEB planning and assessment under RCW 90.94. Appendix A & B are example data definitions from the PNSHPTD data system that is widely deployed across the states of Washington, Oregon and Idaho that provides an example of how these information needs have been interpreted in data structures and metrics. It is intended to be an example of how these data may be defined, but not presented as a requirement.

C. Specific information needs associated with RCW 90.94

There are different sets of management actions that could be undertaken in different parts of the above workflow. Part 1 of the workflow addresses a limited variety of water withdrawals, but the restoration/mitigation actions that are possible in response, and referenced in part 3, include a much larger diversity of mitigation possibilities in terms of type, location and coincidence in time with the consumptive use withdrawal. The actions specific to the context of RCW 90.94 are not covered under other regional guidance, and so are described here. In the interim guidance, these additional actions include:

1. Water right acquisitions (including period of use, instantaneous and annual volume as ac-ft/yr, and source location); and

2. Other projects that provide flow benefits such as:
   - Shallow aquifer recharge;
   - Floodplain restoration/levee removal;

- Floodplain reconnection;
- Switching the source of withdrawal from surface to ground, or other beneficial source of withdrawal change;
- Streamflow augmentation;
- Off-channel storage.

3. In addition, plans may recommend other actions that may or may not be eligible for funding under 90.94 to protect instream resources or offset potential impacts to instream flows such as:
   - Specific conservation requirements for new water users to be adopted by local or state permitting authorities;
   - Requesting rule-making to establish standards for water use quantities that are less than authorized RCW 90.44.050, or more or less than authorized under RCW 90.94;
   - Requesting rule-making to modify fees established under RCW 90.94;
   - Subbasin scale stormwater management strategies to protect or restore hydrologic processes.

This last set of new actions includes in part regulatory decisions (e.g. “conservation requirements”, “rule-making”, etc.) and are therefore outside the scope of this guidance. As such, they will not be covered here.

Current information tracking systems for habitat management actions do not cover all of these project types (e.g. PCSRF or PNSHPTD). Therefore, in meeting the information needs for these actions, some new information will be required. In Table 2 of the appendix below, there are definitions for project types and examples of metrics for water and non-water control projects identified in RCW 90.94 that could satisfy the information needs identified above for projects generally. These are offered as examples of data that, if collected, would be consistent with the conceptual information needs identified in RCW 90.94 and the interim guidance. Indeed, this is true for all of the metrics provided in the appendix; they do not represent a minimum requirement.

D. References

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E. Reportable Metrics

Part 1: Project-Level information Common to All Projects.

Information on the location, timing and contact is needed for all restoration/mitigation actions. Table A2.1 provides examples of project metadata and data definitions that would provide that information. This is one example mechanism to acquire that information in a manner that would be consistent with ESSB 6091 and the interim guidance.

Table	Definition	format (units) for proposed actions and field length	format (units) for completed actions
Project identification number	This is the number given to the project by the State or Tribe	text field	not applicable
Project Grantee	This the official Grantee (State or Tribal group)	Lookup Value	
Primary Subgrantee	The Tribe or State Agency that will assign the project work to be completed (i.e. NWIFC member tribe, CRITFC member tribe, KRITFWC member tribe, OR state agencies, WA state agencies)	Lookup Value	
Project name	This is the name given to the project by the State or Tribe	text field	not applicable
Project start date	The date that the project lead/subgrantee proposes to start the project.	mm/dd/yyyy	not applicable
Project end date	The date that the project's lead/subgrantee contract is completed	mm/dd/yyyy	mm/dd/yyyy
Project description	Short description of the project. The fish stock(s) and or ESUs targeted by the project should be identified as a part of this description.	Narrative, limited to 1000 char. Additional documentation can be attached (e.g. project plans).	Narrative, limited to 1000 char. Additional documentation can be attached (e.g. project plans).
Project Contact(s)	Contact person/people for the project.	Lookup List	
Everything below can be populated automatically once spatial location of worksite is provided.			
State	State that worksite is located in.	Lookup Value	
County	County that worksite is located in.	Lookup Value	
Latitude	The Latitude coordinate value for the worksite. Value should be reported as a positive number from	Number (0-180 Degrees and up to 8 Decimal Places)	
Longitude	The Longitude coordinate value for the worksite. Value should be reported as a negative number from 0 to -180 degrees with up to 8 decimal places.	Number (0-180 Degrees and up to 8 Decimal Places)	
Streamname	The name of the stream where the worksite is located. This name should be taken from the stream data layer provided by StreamNet, so that this name is consistent.	Varchar Text (60 Char.)	
LLID	The LLID of the stream where the worksite is located. An LLID is a stream number method used only in the Northwest region that is based on Latitude/Longitude coordinates of the stream confluences.	Number (25 Char.)	
Begin Ft.	This marks where on a stream network a worksite begins. Begin Ft is a distance measure on a stream network from the confluence.	Number	
End Ft.	This marks where on a stream network a worksite ends. End Ft is a distance measure on a stream network from the confluence.	Number	

Township	A public land surveying unit of 36 sections or 36 square miles. This displays the Township where the worksite is located.	Varchar Text (20 Char.)	
Range	A north-south strip of townships, each six miles square, numbered east and west from a specified meridian in a U.S. public land survey. This displays the Range within a Township that the worksite is located in.	Varchar Text (20 Char.)	
Section	A land unit equal to one square mile (2.59 square kilometers), 640 acres, or 1/36 of a Township. This displays the Section that the worksite is located in.	Varchar Text (20 Char.)	
3rd Field HUC	H.U.C. is an acronym for Hydrologic Unit Codes. Hydrologic unit codes are a way of identifying all of the drainage basins in the United States in a nested arrangement from largest (Regions) to smallest (Cataloging Units). A drainage basin is an area or region.	Lookup Value	
4th Field HUC	H.U.C. is an acronym for Hydrologic Unit Codes. Hydrologic unit codes are a way of identifying all of the drainage basins in the United States in a nested arrangement from largest (Regions) to smallest (Cataloging Units). A drainage basin is an area or region.	Number (25 Char.)	
5th Field HUC	H.U.C. is an acronym for Hydrologic Unit Codes. Hydrologic unit codes are a way of identifying all of the drainage basins in the United States in a nested arrangement from largest (Regions) to smallest (Cataloging Units). A drainage basin is an area or region.	Number (25 Char.)	
Targeted ESU (Evolutionary		Lookup Value	
Geographic area name	On land the Geographic Area Name is defined as the name of the 5th field Hydrologic Unit (HUC). For ocean/estuarine areas not covered by 5th field HUC's the Geographic Area is the name of the water body as shown on NOAA charts or the name of the statistical area. The NWFSC will provide web access to a set of NOAA nautical charts.	text field	not applicable
Geospatial reference/location	This is locational data for each treatment site where the project work is done. Report as a point, line or polygon for all treatment locations. Latitude and longitude from GPS is preferred.	Point, line or polygon. Latitude/ longitude from GPS is preferred Beginning and end points of stream segment can also be provided if available.	Point, line or polygon. Latitude/ longitude from GPS is preferred. Beginning and end points of stream segment can also be provided if available.

PART 2: Project-Level information Common to All Projects

In addition to information on the location, timing and contact for each action, information is needed on what kind of action is taken and how extensive it is. Table A2.2 provides example action metadata and data definitions for project type and extent metrics. In practice, those reporting the data would not report all of these metrics, but rather only those metrics that are specific to the project type undertaken – everything from the top of part 1 and one element from part 2. These metadata definitions are not provided to indicate a minimum standard, but rather to provide examples of what would be consistent with ESSB 6091 not only in terms of information needs as described above, but also in terms of the expectation in ESSB 6091 that where possible, actions undertaken as part of ESSB 6091 will be coordinated and consistent with other state and regional programs.

Table A2.2

Type	Type Definition	Subtype	Subtype Definition	Metric	Metric Definition
Water Projects (*Highest Priority from Funding Guidance*)					
Instream Flow	Projects that maintain and/or increase the flow of water to provide needed habitat conditions. These can include releases of water from dams or impoundments or water conservation projects to reduce stream diversions or extractions.	Water leased or purchased	Purchase of water rights. These water allocations are not withdrawn from the stream.	Annual volume as ac-ft/yr, cfs_a, cfs_i	Water volume proposed for lease or purchase and actually leased or purchased should be reported in (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis.
		Irrigation practice improvement	Installation of a headgate with water gauge that controls water flow into irrigation canals and ditches. Regulates flow on previously unregulated diversions. Also the addition of other water sources (wells etc.) so that water from diversion is less needed or improvement in irrigation systems eg. Replacing open canals with pipes to reduce water loss to evaporation.	cfs_a, cfs_i	The flow of water returned to the stream (not including water that is maintained in the stream) (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis
		Shallow aquifer recharge	Reclaimed water, stormwater collection projects directing water to shallow water aquifer via rock gallery, beaver relocation, beaver dam analogs or direct pumping located near a body of surface water in need of flow and temperature improvements.	cfs_a, cfs_i	The flow of water returned to the stream (not including water that is maintained in the stream). (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis
		Switch to ground water withdrawal	Switching the source of withdrawal from surface to ground, or other beneficial source of withdrawal change	cfs_a, cfs_i	The flow of water returned to the stream (not including water that is maintained in the stream). (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis
		Streamflow augmentation	Reclaimed water, stored water, reduction of surface diversions, or other means that are redirected with an ecologically relevant water quality (temperature & chemistry for affected species), back to a natural channel directly or via an infiltration gallery or shallow aquifer recharge.	cfs_a, cfs_i	The flow of water returned to the stream (not including water that is maintained in the stream). (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis

		Off-channel storage	Off channel storage that diversts high flows from surface waters, either through gravity or pumping into off channel holdings. This project category also includes stormwater projects designed to slow and treat residential or urban runoff stored for later realease to surface waters.	cfsa, cfsi	The flow of water returned to the stream (not including water that is maintained in the stream). (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis

Non-Water Projects (*Lower Priority from Funding Guidance*)

Fish Screening	Projects that result in the installation or improvement of screening systems that prevent Salmonids from passing into areas that do not support salmonid survival, for example into irrigation diversion channels.	Fish Screen Installed	to an unscreened diversion to keep juveniles from being diverted.	#, cfs_a, cfs_i	A total count of screens proposed for installation and actually installed, recognizing that a project may install more than one screen, The flow rate at the screened diversion(s) from the water right. (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis
		Fish Screen Replaced	Replacement, repair or improvement of an existing fish screen	#, cfs_a, cfs_i	A total count of screens proposed for installation and actually installed, recognizing that a project may install more than one screen, The flow rate at the screened diversion(s) from the water right. (CFS to nearest 0.01 CFS), on both an annual and instantaneous basis
Fish Passage	Projects that affect or provide fish migration up and down stream including road crossings (bridges or culverts), barriers (dams or log jams), fishways (ladders, chutes or pools), and weirs (log or rock). Barriers may be complete or partial.	Fish Ladder Improved	Improvement or upgrade of an existing fish ladder	#, target species	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Fish Ladder Installed	Installation of a fish ladder where there was not one previously	#, target species	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Fishways (chutes or pools) Installed	Placement of an engineered way around a barrier (usually a side channel/ or pool) or any by-pass that isn't specified as a fish ladder that is used by salmon migrating upstream; or a chute, used to ease salmon migrating downstream over a dam.	#, target species	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Barriers (dams or log jams)	Removal of a dam other than a push-up or diversion dam; or removal of a naturally formed log or debris jam that created a passage barrier	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Diversion Dam/ push up dam removal	Removal of a push-up dam (earthen dam), or removal of a diversion dam (permanent structure)	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Road Crossings in stream bed	Establishment of engineered passage associated with road placement that may include placement of a bridge.	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Culvert Improvements or	Improve, upgrade or replace an existing culvert	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.

		Culvert Installation	Add a passable culvert where none previously existed.	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Culvert Removal	Removal of culvert (often replaced by a non-blocking structure, bridge etc. or removed because the structure it was associated with was removed, a road etc.)	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.
		Weirs (Incomplete dams)	Placement, modification or removal of an incomplete dam that is a passage barrier to fish	#	There may be more than one fish passage installation per project. Report a count of all blockages that are proposed for removal or improvement and those that are actually removed or improved as part of this project. Latin name of target species.

Instream Habitat	Projects that increase or improve the physical conditions within the stream environment (below the ordinary high water mark of the stream) to support an increased salmonid population.	Streambank Stabilization	The use of rock barbs, log barbs, revetments, gabions etc. to stabilize stream banks	length treated in miles	The number of miles of treatment. Add length treated on both sides when both sides are stabilized. Add one side when one side is treated. (miles to .01 miles)
		Channel Connectivity	Increasing channel connectivity between stream channels, wetlands, and/ or off-channel habitat and floodplain channels. May include increase of historic or new	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Channel reconfiguration	Changes in channel morphology, e.g. pools added/created, meanders added, former channel bed	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Deflectors/ barbs	Placement of triangular structures of rock or logs that extend into the stream to narrow and deepen the channel	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Log weirs	Placement of logs to collect and retain gravel for spawning habitat, to deepen existing resting/jumping pools, to create new pools above and/or below the structure, to trap sediment, aerate the water, or promote deposition of organic debris.	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Off channel habitat	Creation of off-channel habitat consisting of side-channels, backwater areas, alcoves or side-pools, off-channel pools, off- channel ponds, and oxbows.	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Plant Removal/ Control	The removal or control of aquatic non-native plants and noxious weeds growing in the stream channel.	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Rock Weirs	The placement of rocks to collect and retain gravel for spawning habitat, to deepen existing resting/jumping pools; and/or to create new pools, to trap sediment, aerate the water, and to promote deposition of organic debris.	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Spawning Gravel Placement	Addition of spawning gravel to the channel	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Large Woody Debris	Placement of individual logs in the stream that are not part of engineered structures or log jams or other large woody debris not specified as rootwads	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.

		Boulders	Addition of large rocks or boulders to a stream channel	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Rootwads	Placement of a stump with roots attached extending into the stream. Root wads are a type of large woody debris.	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Wood Structure/ Log Jam	Placement of Wood Structure/Log Jam with multiple logs fastened together to form increasing instream habitat	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.
		Beaver Introduction	The introduction or management of beavers to add natural stream complexity (beaver dams, ponds,	# of beavers introduced	# of beavers introduced to increase instream structure/ complexity
Instream-Wetland	Projects designed to protect, create or improve connected wetland areas (that meet the standard for federal delineation) that are known to support salmonid production. For example salmonid populations, especially juveniles, can benefit from access to connected wetland areas where conditions provide food supply, protection from high flows and protection from predators.	Wetland Creation	Creation of wetland area where it did not previously exist	area treated (acres)	Acres of artificial wetland proposed to be created and actually created from an area not formerly a wetland. (Acres to nearest whole acre)
		Wetland Improvement/ Enhancement	Improvements or enhancements to an existing wetland	area treated (acres)	Acres of wetland proposed for treatment and actually treated. (Acre to nearest whole acre)
		Wetland Restoration	Restoration of existing or historic wetland	area treated	Acres of wetland proposed for treatment and actually treated. (Acre to nearest whole acre)
		Wetland Vegetation Planting	Planting of native wetland species in wetland areas.	area treated	Acres of wetland proposed for treatment and actually treated. (Acre to nearest whole acre)
		Wetland Invasive/Noxious Weed Species Removal	Remove or control Non-native species and/or noxious weeds in wetland area	area treated (acres)	The acreage of invasive species proposed for treatment and actuall treated in the wetland project. The proposed project area may only be a portion of an existing wetland such as removing an area of purple loosestrife. (Acres)
Riparian	Projects that change areas (above the ordinary high water mark of the stream and within the flood plain of streams) in order to improve the environmental conditions necessary to sustain Salmonids throughout their life cycle.	Livestock Water Development	Provision of water supply for livestock that is out of the riparian zone. Also called livestock water development or livestock water supply.	# of installations	# of installations, may be more than 1 per project
		Water Gap Development	Provision of a fenced livestock stream crossing	# of installations	# of installations, may be more than 1 per project
		Fencing	Creation of livestock exclusion or other riparian fencing	length of fencing	This refers to meander miles of stream bank proposed for treatment and treated. Report the actual length of proposed treatment, adding lengths of treatment on both sides if treatment was on both sides. (miles to .01 miles)
		Forestry Practices/ Stand Management	Prescribed burnings, stand thinnings, stand conversions, silviculture, vegetation management	area treated (acres)	Total acres proposed and actually treated to nearest whole acre. Examples of treatment include riparian plantings, or protection of riparian zone with a fence.
		Planting	Riparian planting, native plant establishment	Species; area treated (acres	Species Planted (Latin name); Total riparian acres proposed and actually treated to nearest whole acre. Examples of treatment include riparian plantings, or protection of riparian zone with a fence
		Livestock Exclusion	Remove livestock from riparian areas	area treated (acres)	Total riparian acres proposed and actually treated to nearest whole acre. Examples of treatment include riparian plantings, or protection of riparian zone with a fence.
		Conservation Grazing Management	Alteration of agricultural land use practices to reducing grazing pressure for conservation. E.g. Rotate livestock grazing to minimize impact on riparian areas	area treated (acres)	Total riparian acres proposed and actually treated to nearest whole acre. Examples of treatment include riparian plantings, or protection of riparian zone with a fence.

		Weed Control	Removal and/or control of non-native species and noxious weed	Species; area treated (acres)	Invasive species (latin name); the total riparian acres proposed and actually treated to nearest whole acre. Examples of treatment include riparian plantings, or protection of riparian zone with a fence
Sediment Reduction	Projects the diminish sediment transport into streams	Road Reconstruction	Reconstruction and restoration of road in place (not a road relocation) and for a restoration purpose (e.g. road is crumbling into stream and needs to be reinforced). Road reconstruction does not include drainage improvement	miles	Proposed and actual treatments include road(s) decommissioned (closed, obliterated), upgraded, relocated or restored. (miles to .01 miles)
		Road Relocation	Abandonment of existing road in riparian or streambed area with or without rehabilitation and with a new road constructed in a less sensitive area.	Miles	Proposed and actual treatments include road(s) decommissioned (closed, obliterated), upgraded, relocated or restored. (miles to .01 miles)
		Road Stream Crossing Improvements (same as Rocked Ford)	Creation or improvement of a reinforced rock roadbed that crosses the stream without restricting the stream flow. Does not include stream crossing improvements that have a fish passage goal.	miles	Proposed and actual treatments include road(s) decommissioned (closed, obliterated), upgraded, relocated or restored. (miles to .01 miles)
		Road Drainage System Improvements	Placement of structures to contain/ control run-off from roads. Includes surface drainage, peak flow drainage improvements and roadside vegetation	miles	Proposed and actual treatments include road(s) decommissioned (closed, obliterated), upgraded, relocated or restored. (miles to .01 miles)
		Road Obliteration	Road closed with or without rehabilitation. Not a road relocation	miles	Proposed and actual treatments include road(s) decommissioned (closed, obliterated), upgraded, relocated or restored. (miles to .01 miles)
		Erosion Control Structures	Hillside stabilization, grassed waterways wind breaks, planting, conservation land management, and waterbars.	# of erosion structures	# of sediment control installations
		Sediment Control	Sediment basins, sediment ponds and sediment traps.	# of erosion structures	# of sediment control installations
Upland-Agriculture	Upland restoration activities relating to agricultural	Livestock Management	Any upland livestock management including livestock watering schedules and grazing management plans	acres	Total acres proposed for each treatment to nearest whole acre.
		Agriculture Management Best Management Practices	Implementation of best management practices eg low/ no till agriculture	acres	Total acres proposed for each treatment to nearest whole acre.
		Fencing	Placement of exclusion and non-exclusion fencing	miles	Total miles of fencing to nearest 0.01 mile
		Water Development	Irrigation and livestock water development including ditches, wells, ponds, springs etc.	type and #	Type of water development project (ditch, well, pond, etc.) and number of treatments.
Upland- Vegetation	Upland restoration activities relating to vegetation, includes forestry	Planting	Upland plant installation, seeding, and revegetation	area treated	Total acres for each treatment to nearest whole acre.
		Invasive Plant Control	Removal and control of non-native plants and noxious weeds	area treated	Total acres for each treatment to nearest whole acre.
		Vegetation/ Stand Management	Prescribed burns, stand thinning, stand conversion, silviculture, vegetation management, selective thinning, hazard reduction	area treated (acres)	Total acres for each treatment to nearest whole acre.
		Slope Stabilization	Implementation of slope stabilization methods including landslide reparation and terracing.	area treated	Total acres for each treatment to nearest whole acre.
Upland Wetland	Projects designed to protect, create or improve connected wetland areas (that meet the standard for federal delineation)	Wetland Creation	Wetland area created where it did not previously exist	area treated	Acres of artificial wetland created from an area not formerly a wetland. (Acres to nearest whole acre)
		Wetland Improvement/ Enhancement	Changes to an existing wetland	area treated	Acres of wetland actually treated. (Acres to nearest whole acre)

		Wetland Restoration	Restoration of existing or historic wetland	area treated	Acres of wetland actually treated. (Acres to nearest whole acre)
		Wetland Vegetation Planting	The planting of native wetland species in wetland areas.	area treated	Acres of wetland actually treated. (Acres to nearest whole acre)
		Wetland Invasive Species Removal	Removal and/or control of non-native species and/or noxious weeds in a wetland area.	area treated (acres)	The acreage of invasive species actually treated in the wetland project. The proposed project area may only be a portion of an existing wetland such as removing an area of purple loosestrife. (Acres to nearest whole acre)
Water Quality Improvement	Projects that result in an improvement of water quality conditions for example through improved water quality treatment, capture toxic highway runoff, reduction in the use of herbicides, pesticides and fertilizers, and other point sources.	Return Flow Cooling	All projects with a goal of directly reducing or directly limiting increase in water temperature. Most are return flow cooling projects which generally consist of replacing old open return ditches with underground PVC pipe. The primary benefits are t eliminate nutrient and thermal loading, by filtering flows underground where they cool before returning to the river.	water temp	Water temp before and after project completion (if at a point source then avg water temp before at after of point source emission) in degrees Celsius to nearest whole degree.
		Refuse Removal	Removal of garbage in the waterway	lbs of trash collected	Pounds of trash collected from stream and wetland areas to nearest 100 pounds.
		Sewage Clean-up	Clean up of sewage outfall, etc.	Toxin, area treated (acres)	Name of Toxic species, element or material Total acres, wet and/or dry for each cleaned up to nearest whole acre.
		Toxic Clean-up	Clean up/prevention of mine tailings, hebicide, pesticide, toxic sediments, etc.	Toxin, area treated (acres)	Name of Toxic species, element or material Total acres, wet and/or dry for each cleaned up to nearest whole acre.
Outmigrant Survival Improvement (Estuary)	Projects that result in improvement of or increase in the availability of estuarine habitat such as tidal channel restoration, floodplain connectivity, floodgate fish passage or diked land conversion. This habitat is important for salmonid out migration where juvenile Salmonids begin the transition from fresh to salt water environments and where predatory pressures are known to be high. Estuarine habitat is distinct from other wetland habitat in being tidally influenced.	Invasive Species Treated	Control or removal of invasive or exotic estuarine species e.g. Spartina alterniflora	Invasive species, area treated (acres)	Invasive species (latin name); Acres of estuary proposed for treatment and actually treated to nearest whole acre.
		Creation of new estuarine habitat	Creation of an estuarine area where one did not exist previously	area created (acres)	Acres of estuary proposed for treatment and actually treated to nearest whole acre.
		Restoration/Rehabilitation of estuarine habitat	Restoration of existing or historic estuarine habitat	area created (acres)	Acres of estuary proposed for treatment and actually treated to nearest whole acre.
		Removal of existing fill material	Removal of fill that isn't associated with a dike e.g. removal of tideflat fill.	area treated	Acres of estuary proposed for treatment and actually created to nearest whole acre.
		Channel Modification	Deepening or widening existing tidal channel	Type of modification length treate in miles	Type of channel modification and Length of channel modified in miles to nearest 0.01 miles)
		Dike Breaching/ Removal	Removal or breaching of a barrier constructed to contain tidal flooding. Breaching/ removal allows for natural flow/flood regime and potential for off-channel habitat usage.	#; length of treatment (miles)	Number of Dikes breached or removed, total aggregate length of dike reconfigured in miles to .01 miles.
		Tidegate Alteration/ Removal	Removal or changes to tidegate that allows water to flow freely when the tide goes out, but which prevents the water from flowing in the other direction. Changes are generally made to allow fish passage at low and high tide.	#	Number of tide gaits removed or altered
		Dike Reconfiguration	Modification of location or design of an embankment to confine or control water flow.	#, length of treatment (miles)	Number of reconfigurings, total aggregate length of dike reconfigured in miles to .01 miles.
Land Protected, Acquired, or	Projects that involve the acquisition or lease of land or riparian areas.	Streambank Protection	Protection of section of streambank from further degradation or development through purchase, lease, negotiated agreement, statute or other mechanism.	meander miles	This refers to meander miles (to nearest 0.01 mile) of stream bank proposed for protection and actually protected by acquisition, easement or lease. Count miles on both sides of stream if both sides are acquired. Count on one side if only one side is acquired.

Leased		Wetland or Estuarine Area Protection	Protection of wetland or estuarine area from further degradation or development through purchase, lease, negotiated agreement statute or other mechanism.	acres	The acreage reported should be the total acreage proposed for protection and actually protected regardless of whether all of the habitat is applicable to the desired goals for acquisition. (Acres to nearest whole acre)
Nutrient Enrichment	Projects to add marine derived nutrients back into the system	Fertilizer	Nutrients placed in stream to increase nutrient availability	Weight of fertilizer, area treated (acres)	Total of fertilizer delivered (pounds to nearest 100 pounds); Total acres of each treatment to nearest whole acre
		Carcass Analog	Fish meal bricks placed in the stream to increase nutrient availability	Weight of fertilizer, area treated (acres)	Total of fertilizer delivered (pounds to nearest 100 pounds); Total acres of each treatment to nearest whole acre.
		Carcass Placement	Dead salmon added to stream	area treated (acres), weight of carcasses	Total acres of each treatment to nearest whole acre, total weight of salmon carcasses placed in the stream
Project Maintenance	Projects that maintain the functionality of Salmonid Restoration Projects	Site Maintenance	Maintenance of the restoration project site eg.replanting trees that failed to survive	length treated in miles	This refers to meander miles of instream habitat treatments. Count actual stream length treated to nearest 0.01 miles.