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ARTICLE

Review of Tools for Identifying, Planning, and Implementing Habitat Restoration for Pacific Salmon and Steelhead

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Abstract

A key challenge in watershed restoration is identifying the appropriate assessments, data, and analyses needed to identify disrupted natural processes, lost and degraded habitats, and limiting factors to ultimately identify and design successful restoration projects. This has proven particularly challenging for large restoration programs focused on recovery of threatened and endangered salmon and trout where numerous tools, models, and other assessments have been developed to assist with habitat restoration at the watershed, reach, and project scale. Unfortunately, it is often unclear which step in the restoration process these various assessment tools will actually address. To assist with identifying the appropriate assessment tool (e.g., model, data collection, analysis, and survey), we reviewed major categories of watershed restoration assessment tools to determine their goals, inputs, outputs, and their utility in helping plan, prioritize, and implement restoration actions. The major categories of assessment tools reviewed were: (1) life cycle and fish–habitat models, (2) watershed assessment methods and techniques, (3) reach assessments, (4) prioritization tools, and (5) common monitoring methods to identify, prioritize, and plan river and watershed restoration projects. We specifically indicated whether these assessment tools directly or indirectly assisted with the key steps in the restoration process that are required to develop successful restoration plans and projects. These steps involve assessing watershed conditions, identifying limiting habitats and life stages, identifying problems and restoration actions, selecting restoration techniques, prioritizing restoration actions, or designing actual restoration projects. It is important to recognize that no single assessment tool will address all the steps in the restoration process. Selecting appropriate assessment tools requires a clear understanding of the goals of the restoration program and which step in the restoration process will be addressed by a particular tool. We provide recommendations for how restoration practitioners and managers can use our review to help select the appropriate assessment tools needed for their watershed.

The federal listing of several species of Pacific salmon *Oncorhynchus* spp. as threatened or endangered since 1999 has led to large efforts to recover these species and their distinct population segments (evolutionary significant units) (NOAA 2015). Recovery efforts have generally focused on four major factors that have contributed to the decline of Pacific salmon and steelhead *O. mykiss*, often called the four Hs: hatcheries, harvest, hydropower, and

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habitat (Gore and Doerr 2000). While the first three largely focus on modifying policies or infrastructure, habitat restoration has been ubiquitous throughout watersheds used by Pacific salmonids. Several hundred million dollars are spent annually in the Pacific Northwest and California to restore watersheds and recover listed salmonids. For example, the Pacific Coastal Salmon Recovery Fund has funded more than US$1 billion in habitat restoration projects since its initiation in 2000 (NOAA 2015). The Bonneville Power Administration's Fish and Wildlife Program also funds more than $100 million in habitat restoration annually. During the past 15 years more than an estimated 20,000 habitat restoration actions have been implemented in the Columbia River basin alone (NOAA 2016). These efforts include riparian restoration (planting, removal of livestock, invasive species control), instream restoration (placing of logs, boulders, and instream structures), floodplain restoration and reconnection (levee removal or set back, side-channel and off-channel reconnection, channel reconstruction, and floodplain reconnection), and barrier removal (removal of impassable culverts and dams, or installation of fish passage structures). While both the large number and the breadth of the efforts suggest that entire watersheds have been restored, most projects are relatively small in scale, typically treating less than 1 km of stream or 1 ha of habitat, and less than 10% of the habitat in any one watershed is restored (Roni et al. 2010).

Identifying habitat restoration actions that can be implemented and that will result in the recovery of not only habitat conditions but also salmon populations has proven to be extremely challenging. Ideally this would include assessing habitat loss, degradation, impaired watershed processes including the delivery of wood, water, and sediment, and habitats limiting the production and survival of species of interest (Beechie et al. 2013b). Conducting such detailed and comprehensive assessments has proven challenging. This is in part due to the vast array of assessment methods available and the fact that few watersheds have conducted the comprehensive data collection, analyses, and modeling needed and suggested by different watershed assessment manuals to properly identify restoration actions (e.g., REO 1995; Cramer 2012; Beechie et al. 2013b). For example, Beechie et al. (2013b) synthesized watershed assessment approaches and identified more than a dozen approaches for assessing just riparian conditions. The Washington Forest Practices Board Watershed Assessment Manual includes 20 assessment modules, each with multiple assessment methods requiring specific data collection, analysis, and modeling techniques (WFPB 2011). The lack of comprehensive and detailed watershed assessments has also been due in part to the large amounts of effort, time, planning, cost, and interdisciplinary expertise needed to not only conduct and complete the assessment but also to collect the associated data and conduct the analyses. Furthermore, stipulations on current restoration funding has put pressure on practitioners to propose new restoration projects every year and to complete restoration within 6 months to a year, leaving little time or funding to conduct the comprehensive assessments described above.

While an assessment can help identify reaches in need of restoration, no "cook book" exists for translating the assessment data into a list of prioritized reaches and identifying and designing reach-specific restoration actions; all these tasks require a certain level of professional opinion or expertise. This has led to a plethora of tools, models, and monitoring programs designed to help identify restoration actions to recover Pacific salmon and their habitat. The aim of many of these tools is to provide an efficient approach that will bypass the traditional data-intensive approach needed to identify habitat restoration actions that will address limiting habitats and life stages, restore watershed processes and habitats, and ultimately lead to salmon recovery. Almost all of these tools, whether they are a model, a monitoring program, or research, are promoted as being able to help identify restoration actions. For example, life cycle models that identify which life stage has the lowest survival or is most impaired can provide useful information for salmon restoration (e.g., Kareiva et al. 2000; Bartz et al. 2006). Several tools have been developed to help prioritize or sequence restoration projects (e.g., Lichatowich et al. 1995; Beechie and Bolton 1999; Roni et al. 2002; BPA 2015) but are sometimes assumed to be useful for other aspects of restoration planning. Thus, restoration practitioners are often confused about which tools or assessments are needed and which are most appropriate for their watershed. Consequently, some practitioners jump from one tool to the next in an effort to use the latest tool to help them quickly identify and prioritize restoration actions. There are many steps in the restoration process including assessment, identification of problems and actions, selection of techniques, prioritization of actions, and project design and implementation (Roni and Beechie 2013); no single tool can meet all these needs. Furthermore, it is not always clear which part of the restoration process is addressed by each of the available assessments tools.

To address these key uncertainties, the goals of this review are to (1) provide a clear review of the goals and outputs of the various tools for identifying and prioritizing habitat restoration actions for Pacific salmon recovery, and (2) help restoration practitioners, fisheries scientists, and natural resource managers understand which tools will be most useful to them at each step of the restoration process. The intent of this review is not to critique each tool or technique, nor to cover every possible tool in detail. Rather, we provide an overview and a framework for fisheries and restoration practitioners to make more informed choices about the tools that will be most useful at each step of the restoration process in their specific watershed. However, where appropriate, we provide recommendations regarding tools that have proven to be particularly useful for different steps in the restoration process.

METHODS

We reviewed major categories of watershed restoration assessment tools, specifically (1) life cycle and fish–habitat models, (2) watershed assessment methods and techniques, (3) reach assessments, (4) prioritization tools, and (5) common monitoring methods to identify, prioritize, and plan river and watershed restoration projects. We collectively refer to these as restoration assessment and planning "tools." For each category of tools, we first provide an overview, including a brief description of goals, approach, and general inputs and outputs. This is followed by a discussion of the utility of the different tools in assessment and prioritization, and which step in the restoration process they best address. We then discuss considerations for selecting appropriate tools at each step in the restoration process. For steps in the restoration process, we use the steps for developing a comprehensive restoration program as outlined by Roni and Beechie (2013) (Figure 1).

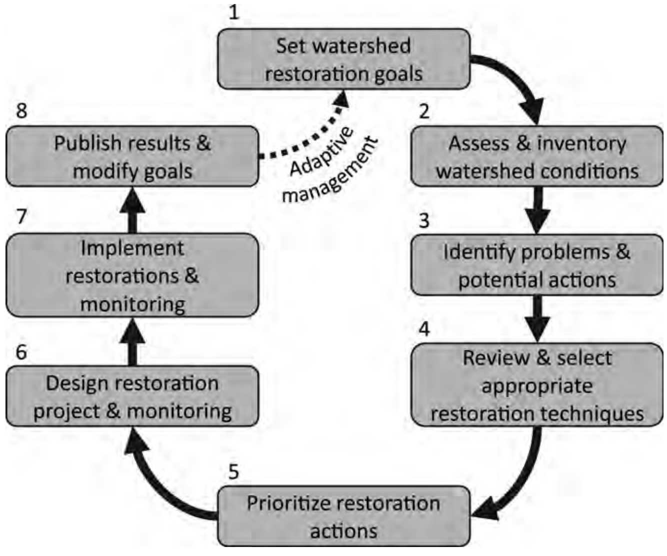

FIGURE 1. Major steps in the restoration process to develop a comprehensive restoration program and successful restoration projects (modified from Roni and Beechie 2013).

OVERVIEW OF ASSESSMENT TOOLS

Life Cycle and Fish–Habitat Models

A variety of general life cycle and fish–habitat models have been developed to assist with salmon management, many of which have been applied to restoration planning. In fact, life cycle modeling is a rapidly growing discipline within fisheries science, and there are more than a dozen life cycle modeling efforts underway to assist in salmon recovery in the Columbia River basin alone (e.g., Zabel et al. 2013). The variety of models developed during the last few decades ranges from relatively simple deterministic models such as limiting-factors models (e.g., Reeves et al. 1989; Beechie et al. 1994) or simple Leslie matrix models (Kareiva et al. 2000), to increasingly complex multistage spawner–recruit models that link life stages to associated habitats and include density-dependent response functions (e.g., Nickelson and Lawson 1998; Greene and Beechie 2004; Scheuerell et al. 2006; Blair et al. 2009; Honea et al. 2009; Zeug et al. 2012; Hendrix et al. 2014). Not surprisingly, the terminology that has evolved to describe these models may be confusing to some restoration practitioners, as there is a wide variety of model types and features that may be beyond their needs for restoration planning. This evolving terminology for model types and attributes includes such terms as stock–recruit, life cycle, Leslie matrix, multistage spawner–recruit, rule-based or scientific model, statistical versus simulation models, deterministic versus stochastic models, individual-based models, and limiting factors. For this review, we do not try to place each model into one of these specific types. Rather, we divide them into four broad categories that are typically encountered in watershed assessments and restoration planning. These are (1) traditional limiting-factors models, (2) life cycle models, (3) the ecosystem diagnosis and treatment (EDT) model, and (4) other models (e.g., intrinsic potential, climate change, and food web models). One key factor for managers and restoration practitioners to keep in mind is that most of these models function primarily as hypothesis-generating tools rather than as tools that generate a putative result before the restoration action has been implemented. Thus, one should not become overly confident on model output and rely more upon field validation and work with researchers to improve the quality of model inputs.

Limiting-factors models.— Limiting-factors models are fish–habitat models that are designed to identify specific limiting habitats and life stages based on fish capacity, and these have been used for many years to assist with restoration planning (e.g., Reeves et al. 1989; Beechie et al. 1994; Nickelson and Lawson 1998). Compared with more complex life cycle models, fish–habitat models use estimates of habitat area and fish densities to estimate capacity at different life stages with fixed survival estimates between life stages to convert life stage-specific capacity to smolt production potential. This life stage-specific, smolt production potential can then be used to determine the habitat type and life stage that is limiting production at a population or watershed scale. The “limiting factor” is typically the life stage-specific habitat type that constrains population size (e.g., summer rearing, winter rearing, spawning), which may also be referred to as a production bottleneck. Typical inputs include data on seasonal area of different habitats, habitat quality, seasonal densities of fish in different habitat types, and estimates of survival from one life stage or season to the next (Reeves et al. 1989; Beechie et al. 1994). Ideally, habitat-specific densities of fish would be collected for the watershed or population in question. However, densities and survival estimates have generally been used from other watersheds, and the unique inputs for each watershed are the seasonal area or availability of different habitat types. Similar to limitations in other types of models, the lack of stream-specific or local data can reduce the accuracy and applicability of model outputs.

Output from limiting-factors models includes the capacity of fish at different life stages, which is converted to smolt production potential for each habitat. This allows comparison of different habitat types to see which ones are limiting total smolt production. For example, a limiting-factors analysis for Coho Salmon *O. kisutch* in the Skagit River basin, Washington, showed that winter rearing habitats were most likely limiting smolt production, and that without restoration of floodplain habitats and beaver ponds, significant increases in Coho Salmon production were unlikely (Beechie et al. 1994, 2003). By contrast, a limiting-factors analysis for Coho Salmon in the North Fork of the Lewis River, Washington, suggested that summer rearing habitat is limiting Coho Salmon production in most subbasins (Figure 2) (Roni and Timm 2016), and that restoration of summer rearing habitats would likely increase population size. Cramer and Ackerman (2009) used a similar approach with additional functions on turbidity, fine sediment, invertebrate drift, alkalinity, and other factors to predict parr and smolt production at reach and watershed scales. While their approach did not specifically identify limiting factors, it can be used to predict changes in capacity based on improvements in habitat. It is important to note that limiting factors predicted from limiting-factors models and other fish–habitat and life cycle models often assume normative conditions, and the limiting factor or habitat may vary from year to year depending upon flow, temperature, and other environmental conditions, which can dramatically influence survival from egg to fry or other life stage-specific survival (Roni et al. 2016).

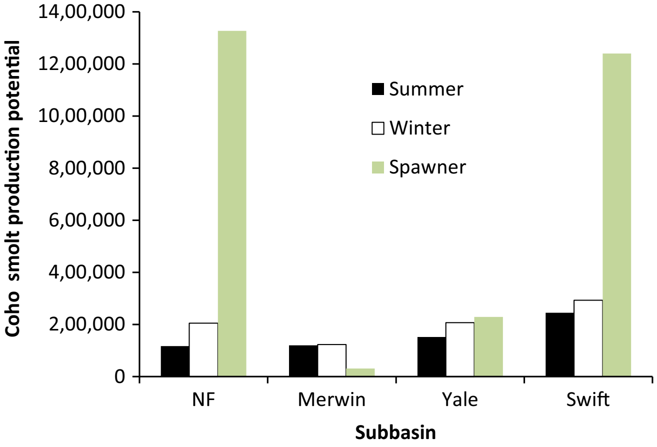

FIGURE 2. Results of Coho Salmon limiting-factors analysis for North Fork of Lewis River, Washington, including smolt production potential by subbasin. Summer habitat is limiting smolt production in North Fork, Yale, and Swift subbasins, while spawning habitat is limiting Coho Salmon production in Merwin subbasin. Input data included seasonal habitat data from within the basin and capacity estimates and survival estimates from other sites in western Washington and Oregon (source: PacifiCorp, unpublished data). NF = North Fork of Lewis River below Merwin Dam. [Color figure can be viewed at afsjournals.org.]

Life cycle models.— As noted previously, there has been a rapid expansion in life cycle modeling, particularly for Pacific salmon and trout (Zabel et al. 2013). Most of these are based on a Leslie matrix modeling framework. The goal of these models, and almost all fish–habitat models, has been to translate changes in survival, capacity, or fecundity (demographic rates) at specific life stages into predictions or measures of abundance, survival, or extinction risk. These estimates in turn are typically used to identify areas of low survival that can be addressed by various management actions (e.g., modifications in harvest, hydropower, or habitat). For example, in what is now considered a relatively simple life cycle model of Columbia River spring Chinook Salmon *O. tshawytscha*, Kareiva et al. (2000) identified that improvements in freshwater and estuarine habitat were likely to have more influence on recovery of spring Chinook Salmon than further improvements in the hydropower system (juvenile fish passage improvement). Leslie matrix types of life cycle models track survival and abundance through discrete ages and life stages (e.g., Kareiva et al. 2000; Bartz et al. 2006; Scheuerell et al. 2006). While earlier models were deterministic, not spatially explicit, and relatively simple and did not incorporate habitat or other important predictive factors (e.g., Kareiva et al. 2000), more complex, spatially explicit models have since been developed that include multiple populations, various life histories, density dependence, movement, stochasticity, and climate change (e.g. Green and Beechie 2004; Scheuerell et al. 2006; Zabel et al. 2006, 2013; Crozier et al. 2008; Honea et al. 2009).

While no two models are the same, most recent salmon life cycle models use a series of Beverton–Holt functions linking each life stage to the next, coupled with a series of other functions that modify life stage-specific habitat capacities or survivals (e.g., habitat, fine sediment, temperature, density dependence, harvest) (Beverton and Holt 1957; Moussalli and Hilborn 1986; Sharma et al. 2005; Honea et al. 2009). Basic inputs to the models frequently include habitat-based estimates of survival and capacity for each life stage and a series of functions for other variables to estimate changes in survival or capacity at each life stage due to land use or restoration (e.g., Bartz et al. 2006; Scheuerell et al. 2006). The utility of these models in restoration planning varies depending on the resolution of the model (reach, subwatershed, watershed or population, subpopulation, life history type), but model outputs generally provide information suggesting which life stages, subwatersheds, or habitat types might be limiting salmon abundance, capacity, and survival. This can be used to help prioritize restoration actions. For example, Scheuerell et al. (2006) applied a spatially explicit, multiple life stage Beverton–Holt model to evaluate how a set of habitat variables influenced spring Chinook Salmon in the Snohomish River basin, Washington, under various restoration and habitat degradation scenarios (Table 1). Their modeling effort suggested that reduced rearing capacity has had the largest impact on population size, and restoring main-stem and estuary rearing habitats should be the focus of restoration efforts.

Interpreting outputs in life cycle models can be challenging because many of the driving functions and parameters are difficult to validate. Therefore, it is important to understand whether model outputs (results) are useful as forecasts of population sizes that might be observed in the future, or are more useful as general predictions for comparing relative changes under various scenarios. In most cases, the absolute numbers that emerge from life cycle models contain some uncertainty, and relative comparisons among scenarios are likely more reliable than is relying on absolute numbers to set expectations for restoration outcomes. It is also important to understand the precision and accuracy of the model, whether the results are relative or absolute, and whether the results can be compared with or validated with field observations (Rose et al. 2011). Life cycle models are typically complex, often based on survival functions from other populations, are specific to species and populations, require someone with specific expertise in modeling, can take multiple years to complete, and may not be easily or readily developed by the restoration practitioner (Table 2). Validation of life cycle models continues to be an area in need of additional research.

The ecosystem diagnosis and treatment (EDT) model.— The EDT model is a complex, widely used, habitat-based salmon model that incorporates some elements of limiting-factors and life cycle models (Lichatowich et al. 1995; Mobrand et al. 1997; Blair et al. 2009). This model uses a combination of empirical data, professional opinion, and inputs of more than 40 user-defined attributes composed of tens to hundreds of parameters and a large number of user-defined life history trajectories (up to 10,000 or more), as well as a multistage Beverton–Holt model to estimate capacity, abundance, and reach-specific survival in a watershed under various habitat scenarios (Lichatowich et al. 1995; Blair et al. 2009; Steel et al. 2009; McElhany et al. 2010). Many of the inputs are based on professional opinion and a process developed for working with local experts to set user-defined benchmarks and rules. The model includes additional functions to incorporate the influence of food, age, fecundity, marine survival, harvest, and other factors on reach-level population performance. Earlier versions of the EDT model could not be modified by the practitioner, but the latest version (version 3) allows user modification of many habitat functions and tailoring of the life history trajectories to the model habitat environment. Model outputs include reach-specific estimates of capacity, abundance, life history diversity, and productivity for "template" (reference or historic) and current conditions. A combination of the changes in reach-level abundance (Table 2), diversity, and productivity can then be used to prioritize reaches for restoration (Blair et al. 2009; McElhany et al. 2010). Thus, there are many user-defined rules and inputs to the EDT model, including a wide range of empirical data and professional opinion).

Despite being widely used in more than 100 watersheds in the Pacific Northwest with plausible inputs and results, there has been limited published validation of EDT model outputs at either a population or reach scale (but see

TABLE 1. Estimated percent change in number of spawners following restoration of habitat variables for Snohomish River Chinook Salmon based on model and sensitivity analysis in Scheuerell et al. (2006).

	Location of restoration action				
Habitat change	Headwaters (%)	Lowland (%)	Main stem (%)	Estuary (%)	Entire basin (%)
Increase egg-to-fry survival by 10%	1	1	3		2
Increase fry-to-smolt survival by 10%	1	1	4	2	4
Restore spawner capacity to historical levels	1	1	4		4
Restore juvenile capacity to historical levels	2	2	13	11	23
Restore both spawner and juvenile capacity	3	3	19		41

TABLE 2. Summary of major categories of life cycle and fish–habitat models often used in watershed assessment and restoration planning including description of model inputs and outputs, scale, and level of expertise needed to complete the model or analysis.

Model category (example)	Description (inputs and outputs)	Scale/expertise/notes
Life cycle models (Scheuerell et al. 2006)	Leslie matrix and multistage spawner–recruit models to estimate spawner abundance and survival bottlenecks under various scenarios. Inputs vary but include juvenile and adult abundance and life stage-specific survival. Outputs depend on model	Models are population or watershed specific; requires expertise in ecological modeling
Limiting-factors models (Reeves et al. 1989)	Use habitat area, habitat-specific densities, and life stage-specific survival to determine limiting habitat and life stage	Watershed or subwatershed scale; can be conducted by most fisheries professionals.
EDT (Blair et al. 2009)	Habitat data, professional opinion (40+ habitat attributes), multiple life history trajectories, and a multistage Beverton–Holt model to estimate capacity, abundance, and survival for habitat scenarios	Outputs are at reach scale and can be rolled up for watershed; requires assistance of model developer
Intrinsic potential models (Burnett et al. 2007)	Gradient, confinement, and species habitat preferences to predict suitable reaches	Reach scale; most fisheries professionals with GIS skills can complete these analyses
Climate change models (Beechie et al. 2013a)	Downscaling of predicted regional changes in temperature and precipitation from climate change models to estimate future temperature and flow in order to assist with restoration planning. Often coupled with species temperature preferences	Watershed or reach scale; downscaling climate predictions is beyond capabilities of most fisheries professionals
Food web models (Bellmore et al. 2013)	Use fine-scale fish and macroinvertebrate densities, fish diet, and consumption to determine reach-scale production and consumption of fish and invertebrates to estimate the most productive habitat types for restoration	Reach scale; most fisheries professionals can complete these analyses

Thompson et al. 2009). Sensitivity analysis has shown that the EDT model outputs are very sensitive to user-defined parameters, and small changes in these parameters can lead to large differences in outputs (McElhany et al. 2009; Steel et al. 2009; Mantua et al. 2010). Thus, as with many models, outputs should be seen as hypotheses of how a population functions and need to be tested. While escapement data can be used to test model predictions, this has rarely been done, and as is true for other life cycle models, there is uncertainty associated with the accuracy and precision of model predictions. However, the most common use of the EDT model for restoration planning has been prioritizing reaches for restoration. Sensitivity analyses have indicated that reach priorities that are determined by the EDT model and are largely based on differences in habitat data between template and current conditions, appear to be fairly robust to changes in model parameters (Steel et al. 2009; McElhany et al. 2010). Given that EDT model estimates of current and potential fish production are reach-specific, a potential approach for validating EDT model outputs could be to measure actual fish abundance, growth, and survival in multiple reaches under current conditions and following restoration and compare those to EDT model predictions.

Other models.— A variety of other models have been developed to assist with identifying restoration opportunities including intrinsic potential of stream reaches to produce fish (Burnett et al. 2007) (Table 2), changes in stream temperature and flow with climate change (Beechie et al. 2013a), and food web models that look at differences in productivity (Bellmore et al. 2013). Models that look at the intrinsic potential of habitat, largely based on gradient, valley confinement, and stream size (flow) are used to help determine which reaches should support the highest numbers of a species and are therefore have the best opportunities for restoration. For example, Burnett et al. (2007) modeled the intrinsic potential for juvenile Coho Salmon and steelhead production for the Oregon coast. Flitcroft et al. (2013) coupled intrinsic potential data with existing abundance data to show that when Coho Salmon abundance is high they move into areas of high intrinsic potential and suggested that these areas should be targeted for restoration and protection. Because data input for intrinsic potential is based on readily available information that can be derived from remote sensing (e.g., gradient, confinement, stream size) and information on fish use of different gradient and reach types (Rosenfeld et al. 2000; Burnett et al. 2007), entire watersheds can be mapped fairly easily to identify areas of highest intrinsic potential for different fish species of interest. These outputs can then be used to help identify areas in need of further investigation for changes in habitat condition and opportunities for restoration or protection. The intrinsic potential of a reach could also be considered when prioritizing or selecting reaches or projects for restoration.

Similar to intrinsic potential models, climate change models examining predicted changes in stream flow, stream temperature, and information on fish habitat preferences can be used to predict future suitability of reaches for different salmonid species (Isaak et al. 2010) or vulnerability of species to climate change (Wade et al. 2013). This information can help identify and prioritize basins and stream reaches that need restoration and protection or reaches that under various climate change predictions may become unsuitable for salmon and other fish species. It can also be used to select or prioritize restoration actions that ameliorate the impacts of climate change or increase resilience to climate change (Beechie et al. 2013a). The spatial resolution of model predictions regarding changes in flow and temperature requires downscaling regional climate change models to particular subwatersheds and stream reaches, which has been done for many Pacific Northwest basins (e.g., Mantua et al. 2010; Beechie et al. 2013a). There is, of course, considerable uncertainty regarding the accuracy and precision of predicted changes, particularly at watershed or reach scales (Nover et al. 2016). Considerable analytical ability is needed to convert climate model predictions to changes in stream flow and temperature, although data layers may be available for some watersheds and regions (e.g., https://data.noaa.gov/dataset/stream-flow-and-temperature-maps-effect-of-climate-change-on-salmon-population-vulnerability).

Food web models, which quantify the transfer of energy among trophic levels by analyzing taxa-specific caloric values and fish consumption data, can be used to assess many aspects of salmon recovery including prioritization of specific habitat types for restoration. Some researchers and policy reviews have further suggested that the lack of success in salmon recovery in the Columbia River basin is due in part to the lack of focus on restoring well-functioning food webs (ISAB 2011; Naiman et al. 2012). Food web models for restoration planning have previously focused on nutrient enrichment in lakes and streams (Carpenter et al. 1985; Kohler et al. 2012); Bellmore et al. (2013) and Wall et al. (2015) present some of the few examples in the riverine environment. In their modeling of food webs in different riverine and floodplain habitat types, Bellmore et al. (2013) demonstrated that due to differences in the proportion of prey consumed by salmonid versus nonsalmonid fishes, salmonid carrying capacity was more than 250% higher in side-channel than main-channel habitats, suggesting those areas be the focus of restoration efforts in the Methow River, Washington. Another example is using a model that incorporates net rate of energy intake to estimate steelhead production at the reach level (Wall et al. 2015) or expected changes in steelhead production as a function of wood placement projects (Wall et al. 2017). Typical inputs to food web models include invertebrate and fish production (abundance and biomass) and taxonomic composition, fish diet, and consumption (Wall et al. 2015, 2017). Food web model outputs include estimates of food consumption rates and taxa-specific abundance, biomass, and productivity within and among stream-dwelling invertebrate and vertebrate communities. While the data and analyses required for food web models may be intensive, these data can likely be collected and analyzed by most trained fisheries professionals and restoration practitioners. Moreover, there may be instances, such as in the nearly 20 intensively monitored watersheds in the Pacific Northwest, where constructing food web models with existing data may be possible (Bennett et al. 2016).

Watershed Assessment Tools

There are numerous guides on conducting watershed assessments including ODF (2004), Shilling et al. (2005), Cramer (2012), and most recently Beechie et al. (2013b). Rather than providing a comprehensive review of all possible assessment techniques here, our objective is to review the major categories of assessment tools in order to clarify their goals, inputs, and outputs, and which step or steps they assist with in the restoration process. A comprehensive watershed assessment includes the evaluation of (1) landscape-scale processes (hydrology, sediment supply, and nutrients), and (2) reach-scale processes (riparian, flow, channel dynamics, water quality, habitat alteration, and changes in biota) (Beechie et al. 2013b). In addition, a basic watershed template that includes geology, topography, climate, valley segments, and reach types is needed to adequately support and assist these four basic categories of watershed assessments. The overall goal of all these assessments is to identify disrupted processes, lost or degraded habitat, and changes in biota due to disrupted processes and lost or degraded habitat and ultimately identify restoration opportunities (Beechie et al. 2003a, 2003b, 2013b).

Landscape-Scale Processes and Assessment Tools

A variety of tools are used to examine landscape-scale processes related to hydrology and instream flows, sediment supply and erosion, and nutrients (Table 3). Typically, these include remote sensing to determine where ecological processes have been altered by human activity coupled with field surveys to locate specific areas for restoration of disrupted processes (Beechie et al. 2013b). Estimating the degree of human alteration requires establishing a historical or reference condition, which can be estimated in three main ways: historical maps and data, contemporary reference sites, or modeling. Historical maps are useful for large features, such as wetlands or floodplain channels (Beechie et al. 1994; Collins et al. 2003; Hohensinner et al. 2004). Contemporary reference conditions can be measured at relatively natural sites for features that are too small to appear on historical maps (e.g., pools and riffles) (Beechie et al. 1994, 2001). When historical maps or contemporary reference data are not available, theoretical or empirical models can be used to estimate historical or natural conditions (e.g., Pollock et al. 2004).

Erosion and sediment supply.— A variety of assessments can be conducted to estimate and evaluate changes in sediment supply to streams from either surface erosion or landslides. These assessments most often focus on

TABLE 3. Examples of common assessment techniques for assessing conditions at the watershed scale including major categories (watershed template, sediment supply, hydrology, nutrients) and their goals, as well as specific types of assessments and their outputs and resolution (scale) of those outputs. SW = subwatershed, R = reach, S = site. Sources: ODF (2004); WFPB (2011); Beechie et al. (2013b).

Types of watershed assessments	Outputs	Resolution
Underlying watershed template		
Overall goal: base maps needed for other assessments, analyses, and monitoring		
Landscape template (geology, topography, climate)	Common physiography, ecoregions, etc.	SW, R
Valley segments	Confinement	R
Reach types	Reach types (current and historical)	R
Sediment supply		
Overall goal: how sediment differs from natural, human impact on sediment supply–budget, restoration opportunities		
Surface erosion (roads)	Current and historical change in sediment supply	R
Surface erosion (agricultural lands, uplands)	Current and historical change in sediment supply	R
Bank erosion	Current and historical change in sediment supply	R, S
Mass wasting (landslides)	Current and historical change in sediment supply, areas at high risk of future landslide	R, S
Road survey	Identification of road segments at risk of failure or in need of restoration	R
Hydrology		
Overall goal: flows different from natural, impact of human uses, restoration opportunities		
Runoff and streamflow	Maps of runoff rates, changes due to land use, etc.	SW, R
Variable infiltration model	Effect of land use and dams on streamflow	SW, R
Water flow balance and simulation model	Effect of land use and dams on streamflow	SW, R
Disturbed soil hydrology and vegetation model	Effect of land use on streamflow	SW, R
Analysis of streamflow data	Correlation of land use to peak flows	SW
Index of hydrologic alteration	Effect of management action on wide range of flow metrics	SW
Altered flows versus historical flows	Level of abstraction; changes in base flows, peak flow, etc.	SW
Nutrients (watershed scale)		
Overall goal: determine where above natural levels, source and fates of nutrients and pollutants, function of land use, and restoration opportunities		
Nutrient budget as function of land use	Effects of land use on nutrients	SW, R
Integrated catchment model	Sources and load of heavy metals (usually mines)	SW
N and P mass–balance models	Nutrient transport, retention, and sources	SW, R

quantifying erosion rates and calculating a sediment budget from aerial photos, maps, or satellite imagery and on estimating total erosion from limited field data collection or modeling (Reid and Dunne 1996; Beechie et al. 2003, 2013a, 2013b) (Table 3). Reid and Dunne (1996) outline a simple and efficient approach for conducting sediment budgets and identifying sources of change in sediment supply to help assess watershed conditions and identify specific areas in need of restoration. The primary output of sediment assessments is the mapped changes in sediment supply due to land use, where map polygons (typically subwatersheds) express an increase in sediment supply (percentage or absolute) over background delivery rates. The change observed from the background is important as natural sediment supply varies with landform, slope, soil type, vegetation, and other factors (Reid and Dunne 1996; Beechie et al. 2013b).

In mountainous areas, a sediment budget typically consists of landslide inventories based on historical aerial photos and estimates of sediment contribution from surface erosion from unpaved forest roads (Reid and Dunne 1996). Identifying the land use associated with landslides is important to quantify the effects of different land uses on sediment supply. Surface erosion from unpaved roads is estimated based on characteristics of the road surface including material type, underlying soil and geology, traffic level, and precipitation. The assessment of mass wasting and surface erosion is often summarized by subbasin to indicate areas that have been most altered from background rates and identifies specific areas in need of additional assessment (Figure 3). This information can be used to focus the mapping of landslide hazard areas and field inventories to locate road segments that are at a high risk of failure or produce large amounts of fine sediment and to identify opportunities for road improvement or removal (Beechie et al. 2003). Assessment of surface erosion from agricultural lands typically incorporates published soil erosion rates from different soil types with varying vegetation cover into surface erosion models to estimate changes in sediment supply (Dunne and Leopold 1978; Beechie et al. 2013b). The outputs can be used to identify and map fields and parcels where erosion and sediment delivery to streams is high and thus there is a need for restoration, protection, or improved land management. These outputs are typically coupled with riparian and other assessments to help understand the causes of increased sediment supply and potential restoration strategies.

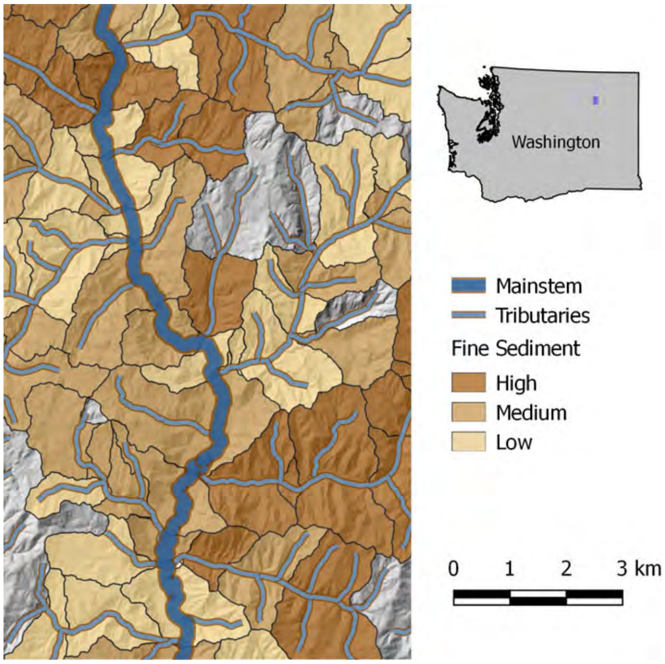

FIGURE 3. Example of outputs from sediment budgeting showing subwatersheds with low, high, and medium levels of fine sediment delivered from roads and other sources in the Sanpoil River basin, Washington (source: courtesy of Confederated Tribes of the Colville Reservation, unpublished data).

Hydrology.— Watershed-scale assessments of runoff and stream flow focus on how land use and dams have altered runoff processes (interception, evapotranspiration, infiltration) or flow timing and volume (Beechie et al. 2013b). A variety of coarse-resolution models have been developed to assess the effects of land uses on runoff processes and stream flow, including the Water Flow and Balance Simulation model and the Variable Infiltration Capacity model (Krause et al. 2007; Cuo et al. 2009) (Table 3). Finer-resolution runoff models such as Hydrology Simulation Program Fortran (HSPF), Hydrologic Engineering Center model (HEC-1; Hydrologic Engineering Center, U.S. Army Corps of Engineers), Distributed Soil Hydrology and Vegetation model (DHSVM), and other finer-resolution models examine similar impacts in smaller watersheds, but are more costly and labor intensive. Finally, the impact of dams can be assessed with the Indicators of Hydrologic Alteration (IHA; Richter et al. 1996; Poff et al. 2010), which are a set of flow metrics that are sensitive to dam operations (e.g., magnitude or timing of peak and low flows, rates of increase or decrease in flow).

Simpler approaches for evaluating potential hydrologic change due to land use include quantifying changes in land use indicators that are strongly correlated with peak flow, such as impervious surface area and forest cover (Booth and Jackson 1997; Beschta et al. 2000). These are easier to use than the previously described complex models as they only require assessment of current land cover and provide a good general characterization of which subbasins may have different levels of hydrologic alteration. However, such models do not provide detailed analysis within a basin nor potential effects of mitigation (Beechie et al. 2013b). The inputs to these approaches for assessing changes to runoff and stream flow vary widely depending on the model or method used. For example, the hydrologic models (both coarse and fine resolution) require information on precipitation, topography, soil properties, and vegetative cover, whereas the IHA assessment only requires stream flow data. Notably, all of the hydrologic models provide information to help assess current conditions, levels of impact, and watershed-scale restoration needs. However, they do not necessarily help identify or prioritize site-specific restoration actions. By contrast, the IHA method identifies specific flow alterations by dams and can be used to identify dams in need of flow restoration as well as specific flow restoration recommendations (Grantham et al. 2014). Hydraulic analyses, which focus on in-channel flow characteristics (e.g., channel roughness, depth, velocity, and turbulence), are typically done at a reach scale as part of analyses needed for designing restoration projects.

Nutrients.— Assessments to identify concentrations and sources of nutrients and water quality conditions as they affect salmon restoration are typically conducted under two general scenarios: (1) whether reach-scale water quality or biological assessments have indicated that nutrient loads are higher than expected (Beechie et al. 2013b) or values of water quality variables violate the Clean Water Act or other regulatory water quality standards (Table 3), and (2) in cases of suspected or documented nutrient limitation in the absence of historical marine derived nutrients (MDN) (Ashley and Stockner 2003; Stockner 2003). Similar to a sediment budget, a nutrient budget can be constructed that focuses on the sources of nutrients and pollutants from various land uses, thereby allowing identification of important sources of nutrients and potential restoration opportunities. This has most frequently been conducted in lakes (i.e., see Bennett et al. 1999 for Lake Mendota Wisconsin). In streams, this is often done as part of reach-scale water quality assessment. Determining whether streams are in fact nutrient limited due to reduced MDN—and if nutrient addition may be a suitable restoration approach—requires, at a minimum, an assessment of historical salmon-derived MDN, determining current nutrient status (N and P levels), and an analysis of food web structure (Kiffney et al. 2005).

Reach-Scale Assessment Tools

A variety of tools are used to assess reach-scale processes, which are those processes that directly affect an adjacent reach, including riparian, floodplain, and local fluvial processes (Beechie et al. 2013b).

Riparian processes.— The goals of riparian assessment are to identify where riparian areas have been degraded and where restoration may be needed. Riparian assessments typically use remote sensing data, field data collection, or a combination of the two. Field data collection is required to design specific riparian restoration strategies and often follows a broader assessment done with remote sensing data. Remote sensing data for riparian assessments include a variety of satellite data, aerial photography, and hyperspectral imagery (Klemas 2014). Satellite data are generally used for watershed scale assessment of riparian conditions and provide information on land cover and forest types. Aerial photography is more costly but can provide higher resolution information than satellite data on land cover, land use, riparian condition, and, depending upon the resolution or elevation of the photographs, even species composition (Beechie et al. 2013b; Klemas 2014). Often aerial photography or field surveys are used to assess accuracy of satellite imagery. Field surveys typically consist of visual classification of riparian conditions such as community type and age-class, disturbance, shade, cover, and in some cases bank condition. Typical outputs of riparian assessments include maps and tables documenting reaches or subbasins where riparian function (canopy cover, shade, organic inputs) and condition (age, composition) deviate from natural, historical, or expected conditions, and how land use practices impact these functions and conditions. This information can then be used to identify areas in need of restoration.

Floodplain and fluvial processes.— Assessment of floodplain–channel interactions and fluvial processes in river restoration planning typically includes an assessment of historical and current floodplain conditions including forest age structure, abundance of various habitat types or channel migration rates, and factors such as water and sediment transport, channel migration, and pool–riffle formation. Outputs of these assessments include changes in channel migration rates, changes in forest age structure, or maps of changes in floodplain area and habitat types (e.g., Hohensinner et al. 2004; Beechie et al. 2006; Kloehn et al. 2008; East et al. 2016). These outputs can be used to help characterize reference conditions, identify changes due to land use or dams, and identify restoration opportunities.

Changes in forest age structure, channel migration rates, side channels, and other features are usually assessed using aerial photographs or a combination of historical maps, surveys, and aerial photographs (e.g., Collins and Montgomery 2001; Hohensinner et al. 2004; Kloehn et al. 2008; East et al. 2016). Assessing reach-scale fluvial processes may involve assessing sediment transport or channel migration but often focuses on channel conditions and morphology that indicate disrupted processes or direct modification (Beechie et al. 2013b). This can include field surveys to assess channel type, width : depth ratios, pool depths, sediment size, incision, bank erosion, and bank armoring, although some of these features can be assessed with remote sensing, particularly on larger rivers. Light detection and ranging (LIDAR) techniques can also be used to map current and historical channel patterns, floodplain extent, and isolated habitat (Negishi et al. 2012;

Bizzi et al. 2015), produce digital elevation maps (Zhao et al. 2010), and provide high-resolution, structural, three-dimensional (3D) characterization of floodplain vegetation communities or land cover types in surveyed areas (Van Leeuwen and Nieuwenhuis 2010). Outputs of floodplain and fluvial process assessments include maps and tables that indicate which channel conditions and processes have been altered and degraded and quantify lost or isolated habitats. This information, when coupled with assessments of watershed-scale sediment and hydrologic processes can help target important causal mechanisms to be addressed by restoration (Beechie et al. 2003, 2013b).

Detailed reach assessments and modeling are important components of restoration design as it is critical to understand sediment dynamics, channel migration, and hydraulic processes in a reach to effectively design restoration measures (Skidmore et al. 2013). These include a variety of one-dimensional (1D), two-dimensional (2D), or 3D models to assess changes in velocity, direction, and other flow-field characteristics that can directly and indirectly affect physical, biological, and ecological conditions and functions in rivers and streams. Hydraulic models and analyses in particular are used to model water surface profiles through a reach at different flows, estimate forces acting on the stream bed and on stream banks, and predict microhabitat conditions for aquatic organisms under various restoration designs and flow alternatives. One of the more common models is the U.S. Army Corps of Engineers HEC-RAS model (Hydraulic Engineering Center–River Analysis System), which allows 1D and 2D modeling of flow and sediment transfer capabilities in a reach (USACE 2016). Some reach-scale assessment approaches such as the U.S. Bureau of Reclamation (BOR) reach model include a combination of assessment techniques including analysis of historical photos and maps, topographic and other field surveys, hydraulic modeling to assist with identification, and design of restoration actions within a study reach (BOR 2009, 2010). In addition to these models, habitat suitability index (HSI) models have been developed to use hydraulic model outputs to model the amount of suitable habitat available for different species at different flows. For example, the HSI model developed for use in the Columbia River basin models the amount of suitable habitat for different Pacific salmon and trout species using topographic survey data (North Arrow Research, http://habitat.northarrowresearch.com/index.html) (Figure 4). It provides a useful tool for habitat assessment, and by predicting suitable habitat at different flows, can assist with restoration design.

Channel and habitat conditions.— There is a suite of reach surveys that can occur at a reach or basin scale to assess in-channel conditions, including mapping or surveying of bank armoring (e.g., levees, revetments, other modifications), surveying of isolated channels and floodplain habitats, and mapping of channel incision, bank erosion, and channel condition (often done as part of a habitat survey; e.g., pattern, width : depth ratio, pool depth and frequency, bar size and height, and grain size) (Collins and Montgomery 2001; Beechie et al. 2008b, 2013b). All of these are usually done to characterize condition and to identify degraded habitat and restoration opportunities. Some of these methods are used in conjunction with

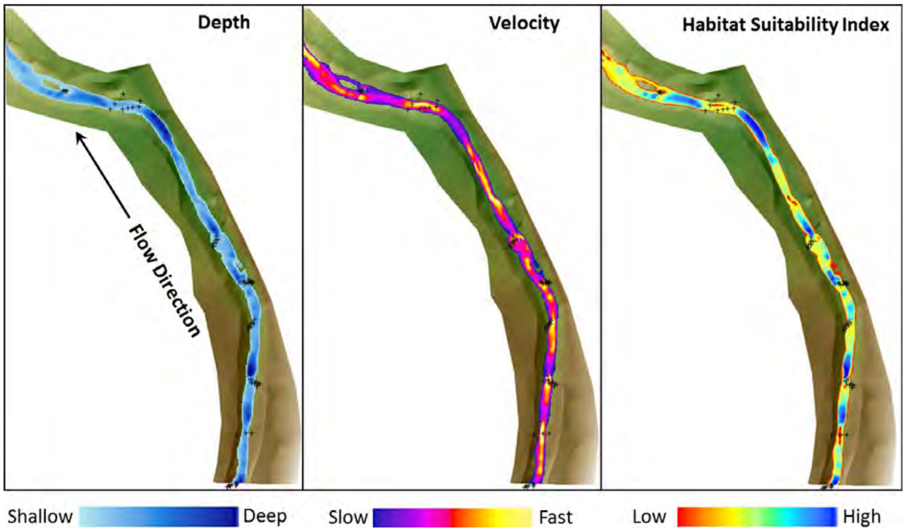

FIGURE 4. Example of habitat suitability index (HSI) modeling for steelhead on Asotin Creek, Washington. The modeling of depth and velocity is based on a Delft3D hydraulic model, which uses discharge, topographic data, and an estimate of channel roughness (D84) and is combined with habitat preferences for steelhead (Maret et al. 2006) to produce a HSI map of areas of highest and lowest suitability for juvenile steelhead during summer. Plus symbols (+) indicate location of large woody debris (source of data and map: courtesy of Andrew Hill, Eco Logical Research).

habitat assessments to quantify instream habitat quality and quantity.

Numerous protocols have been developed to assess wadeable streams in the Pacific Northwest, including the Pacfish/Infish Biological Monitoring Program (PIBO), Columbia Habitat Monitoring Program (CHaMP), Timber Fish and Wildlife (TFW), National Rivers and Streams Assessment (NRSA, formerly EMAP), Oregon Department of Fish and Wildlife (ODFW) Aquatic Inventories Project, and others (Johnson et al. 2001; CHaMP 2016). While all of these approaches have strengths and weaknesses, they all focus on characterizing channel units, cover, wood, sediment size, and in some cases bank conditions. Many of these are not continuous surveys (e.g., CHaMP, PIBO, NRSA/EMAP), but rather use a general random tessellation sampling design or other sampling designs to identify and sample multiple short reaches (100 to 500 m long) throughout a basin (Stevens and Olsen 2004). Some habitat survey methods such as CHaMP include detailed transect or topographic surveys, provide detailed information for short reaches, and collect data on dozens of habitat metrics, but are too costly to apply to entire basins. Intensive survey approaches like these can provide useful information for restoration design in a specific reach. For example, outputs from the topographic survey portion of CHaMP provide a detailed topographic map of the stream channel, which can be useful for designing restoration in that specific reach (CHaMP 2016).

Less intensive and less costly methods that focus on key habitat metrics that are repeatable, relevant to causes of habitat degradation and restoration, known to be important to fish (e.g., pools, large woody debris, cover), and cost effective can be used to census all or portions of a basin (e.g., Hankin and Reeves 1988). Continuous habitat surveys are among the most useful for assessing conditions and identifying restoration opportunities. Most habitat surveys focus on field surveys in wadeable streams or channels less than about 20 m bankfull width. Approaches for larger channels that look at both mainstem (pool, riffle, glide, edge, and backwater habitats) and floodplain channels and habitats (side channel, tributary, pond, oxbow, lake) typically use a combination of remote sensing (aerial photography) and field surveys to quantify habitat (Beechie et al. 2005). From an assessment standpoint, continuous surveys of all or most of the stream reaches of interest in a basin are most useful for identifying degraded habitat and restoration opportunities, in part because they provide consistent information across a basin rather than information from only specific locations (Figure 5) (Beechie et al. 2017).

Another key habitat assessment method is the identification and mapping of natural and anthropogenic barriers to fish migration; these include dams, weirs, road crossings, and other infrastructure (e.g., Beechie et al. 2003). These maps are often completed in conjunction with road surveys as many stream crossings (e.g., culverts, bridges, and fords) create barriers to fish migration. These surveys help identify obvious isolated habitats and restoration opportunities. Mapping barrier features and locations can also inform nonnative species management decisions by limiting the spread of nonnative fishes into habitats containing native fish populations (Muhlfeld et al. 2009, 2012). When coupled with basin-wide habitat surveys or habitat surveys above barriers, these surveys can be used to prioritize restoration actions (Figure 5).

Water quality.— Assessments of water quality variables such as temperature, dissolved oxygen, pH, and fecal coliforms are often performed with point measurements and sometimes done in conjunction with habitat surveys. The availability of inexpensive temperature data loggers allows their placement throughout a watershed to measure temperature continuously year round. However, analyzing and summarizing these data can sometimes be challenging. Online tools such as the NorWest Stream Temperature Database (http://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html) allow input of temperature data and provide tools to map stream temperatures at the reach scale (Figure 6), enabling the evaluation of predicted changes in temperature due to various climate change scenarios. Such approaches can help identify stream reaches that may

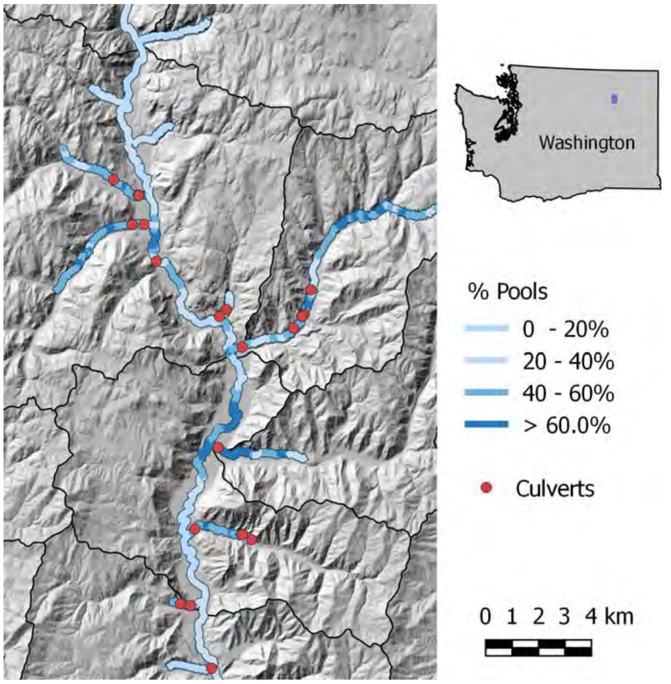

FIGURE 5. Summary of habitat and barrier survey data showing barriers and percentage of pool habitat in reaches of the Sanpoil River basin, Washington (source: courtesy of Confederated Tribes of the Colville Reservation, unpublished data).

have high stream temperature, but need to be coupled with other assessment data to determine the root cause of elevated stream temperatures to identify restoration opportunities.

Where sufficient data are available, the U.S. Environmental Protection Agency (EPA) identifies stream reaches that fall below surface water quality standards for temperature, turbidity, dissolved oxygen, pH, recreational use, fish use, and other water quality standards. The EPA and state departments of ecology or water quality provide detailed lists and maps of these reaches online. Thus, water quality information can be used to help identify impaired reaches in need of restoration. Similar to other assessment tools, water quality data need to be coupled with other assessment data to determine appropriate restoration measures for a reach and to prioritize reaches for restoration.

Longitudinal temperature profiles can be created for entire reaches or river segments with thermal infrared remote sensing by using a drone or aircraft or by dragging a temperature probe along the thalweg of a stream (Torgersen et al. 2001; Vaccaro and Maloy 2006). These surveys have the advantage of covering broad areas, but only provide temperatures for the dates surveyed. By contrast, continuous data loggers provide data only at specific sites but at set time intervals (e.g., every minute, hour, day) for as long as they are deployed (typically months or years). Thus, one approach is not necessarily better than another, but collectively they can provide a detailed assessment of temperatures across reaches within a watershed. When coupled with riparian and sediment surveys, they can be used to identify causes of increased temperatures and restoration opportunities.

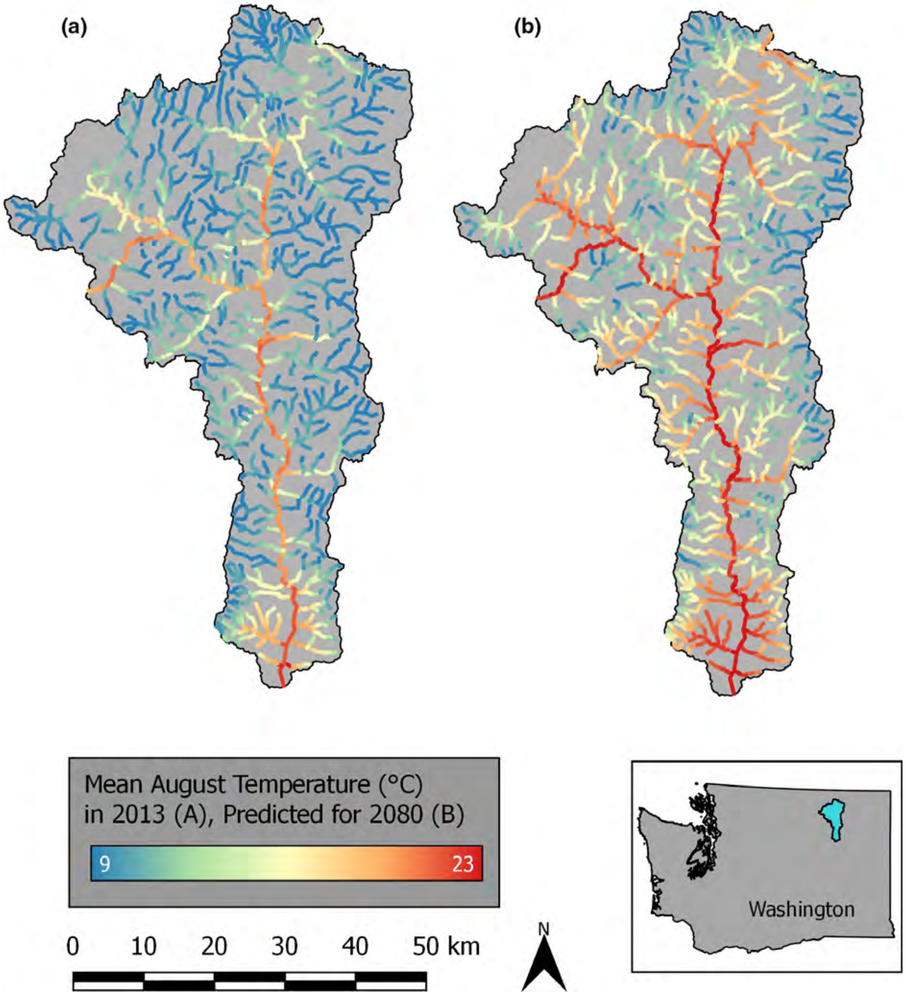

FIGURE 6. Example of **(A)** current (2016) and **(B)** predicted (2080) August stream temperature in Sanpoil River basin, Washington. Stream temperature data and modeled predicted changes in temperature can be downloaded from NorWest Stream Temp at http://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST. Temperatures above 20°C are generally inhospitable for *O. mykiss* and other salmonids native to the basin.

Biological assessments.— Biological assessments typically focus on one of two major approaches: (1) assessing and mapping the distribution and abundance (number or density) of fish or other aquatic species throughout a reach or watershed, or (2) using diversity or multimetric indices to characterize the health of reach or watershed (Beechie et al. 2013b). The first approach often involves common methods including electrofishing, snorkel surveys, spawner surveys, and various trapping methods (e.g., smolt traps, weir counts, minnow traps, fyke netting). Adult, spawner, and juvenile fish surveys are typically done by reach and habitat type, and when coupled with habitat surveys can help identify areas of high and low fish use and assess quality of spawning or rearing habitat. Like many other assessment tools, their utility for this purpose depends upon their spatial and temporal resolution and coverage. Assuming juvenile, adult, or spawner surveys are done throughout the basin or in representative reaches that can be extrapolated to other parts of the basin, they can be extremely useful in identifying opportunities for restoration or protection. In fact, they often provide the necessary data on habitat-specific densities needed to populate limiting factors and life cycle models. When coupled with historical and current habitat data, they can also be used to estimate current, historical, and potential increases in fish production in a reach due to restoration (e.g. Beechie et al. 2003, 2015; Roni et al. 2010). Smolt and adult traps collect useful information for understanding survival and production from a watershed or subwatershed and provide key inputs for life cycle modeling. However, operating traps or weirs is expensive and typically only done at the mouth of a watershed or at or above selected tributary junctions. Thus, unless traps are located in multiple locations in main stems, tributaries, and off-channel areas, traps and weirs are of limited utility for assessing reach conditions and identifying specific restoration opportunities, apart from indicating overall fish production from a watershed or tributary. When multiple traps are run and extensive smolt-trapping data are available for a variety of off-channel restoration or other restoration actions, they can provide useful information for predicting increases in fish due to restoration actions (Roni et al. 2010; Ogston et al. 2015).

Multimetric indices that are correlated with water quality, reach characteristics, or watershed condition, such as the index of biotic integrity (IBI) and benthic index of biotic integrity (B-IBI) in North America, the river invertebrate prediction and classification system (RIVPACS) in Australia, and the multilevel concept for fish-based assessment (MuLFA) in Europe, have been developed to provide overall indicators of reach or watershed health (Karr and Chu 1999; Schmutz et al. 2000). These indices require point samples of benthic invertebrates or fish abundance and diversity. Moreover, while they provide useful information on overall watershed and reach condition, like many assessment techniques, they do not necessarily identify causes of degradation. In Pacific Northwest watersheds, fish-based indices of integrity or diversity have not proven overly useful because of the relatively depauperate fish fauna in these systems. Similarly, the B-IBI and other macroinvertebrate measures tend to respond to broader changes in water quality and nutrients and are of limited utility in identifying restoration actions as well as evaluating the success of some restoration measures (Roni et al. 2008, 2015; Kail et al. 2015).

Prioritization Tools

A variety of tools have been created to assist with restoration prioritization at the project, reach, and subwatershed scales. Beechie et al. (2008a) and Roni et al. (2013a) identified several major types of approaches for prioritizing restoration including: project type and effectiveness, refugia, species or habitat, capacity or life cycle models, cost-effectiveness or cost–benefit, complex computer models or conservation planning software, or simple scoring systems (multicriteria decision analysis [MCDA]) to complex computer models. The Conservation Success Index developed by Trout Unlimited is an example of a simple MCDA approach to rank watersheds based on restoration and protection potential for different trout species (Williams et al. 2007). More complex computer models incorporating a variety of models and data layers have also been used to examine different restoration strategies and prioritize habitat types for restoration (e.g., Greene and Beechie 2004; Scheuerell et al. 2006; Fullerton et al. 2010). However, almost all approaches for prioritizing watersheds, reaches, or restoration projects require data from one or more of the other assessments or tools discussed above.

There are three approaches commonly used for prioritization of watersheds, subwatersheds, reaches, and projects in the Columbia River basin and the Pacific Northwest: simple MCDA or scoring systems, EDT, and the Atlas framework recently developed by the Bonneville Power Administration in close collaboration with Columbia River basin partners (Blair et al. 2009; Roni et al. 2013a; BPA 2015; Booth et al. 2016). All three of these use a combination of empirical data and professional opinion to help identify prioritization goals, data needs, and criteria for scoring and prioritization; most use a combination of the approaches discussed by Beechie et al. (2008a, 2008b) and Roni et al. (2013a). Only MCDA and the Atlas framework function at the level of prioritizing restoration projects throughout watersheds. Prioritizing watersheds or restoration projects using MCDA typically involves a relatively simple scoring system (i.e., 5 to 20 criteria) that uses experts to prioritize areas and restoration actions. Because it is largely based on quantitative data combined with some qualitative information, the MCDA approach is transparent and relatively easy to explain, understand, and modify. In practice, most MCDA prioritization methods use a combination of GIS or field data from previously described assessments to provide quantitative information on a variety of factors, along with professional opinion to assist with the scoring of the selected criteria. Common factors for prioritizing watersheds include land use, area or proportion of intact habitat or degraded habitat, road density, barriers to migration, water quality, invasive species, riparian or habitat condition, species present, biodiversity, presence or abundance of rare or endangered species, and habitat condition index based on computer models (Nehlsen 1997; Gellis et al. 2001; Williams et al. 2007; Roni et al. 2013a). Individual projects are often scored using criteria such as area or length of watercourse restored, potential increase in fish, the cost, cost–benefit, cost-effectiveness (cost per fish), land ownership. Scoring is also determined according to whether the project addresses a factor that limits biotic production, restores a sensitive habitat, restores a key process (restoration type), or is a refuge or priority watershed, as well as other factors (Beechie et al. 2008a; Roni et al. 2013a).

The previously described EDT model, which is species- and watershed-specific, ranks reaches in a given watershed for restoration potential based on key model outputs for an individual fish species using attributes such as abundance, productivity, and diversity. While the model has been broadly applied, there has been much debate about its accuracy, in part because it is very complex and proprietary, and because there has been little published validation of the accuracy and precision of EDT model outputs to date. Despite this, two detailed reviews and sensitivity analyses of EDT suggest that its reach rankings are potentially useful (Steel et al. 2009; McElhany et al. 2010). It should be noted that the EDT model does not determine appropriate restoration strategies or underlying causes of degradation. However, resulting reach rankings can be coupled with other assessment data, including limiting factors analysis, to help identify reach-specific restoration measures (Roni and Timm 2016).

The Atlas framework provides a consistent and systematic approach to working with local restoration researchers and practitioners to integrate available empirical data, watershed and habitat assessments, models, and professional opinion and consensus to ultimately identify and prioritize subwatersheds, reaches, and restoration actions within a watershed or group of watersheds (BPA 2015; Booth et al. 2016). Local partners in this process include a broad range of interested parties from tribes to federal, state, county, and nonprofit organizations, to research scientists and restoration practitioners. To date, the Atlas framework has been applied in seven Columbia River basin watersheds within Oregon and Idaho and appears to be a promising approach to develop comprehensive and widely accepted strategic restoration plans with clear prioritization of projects. In the end, it uses an MCDA scoring approach to rank subwatersheds, reaches, and restoration projects.

Rather than a simple prioritization strategy, the Atlas approach integrates available assessment data and fish–habitat models to identify subwatersheds and reaches important for salmon recovery and restoration and to ultimately prioritize restoration opportunities. The Atlas method rates a suite of components such as fish species and life stage seasonal use, limiting life stages, limiting habitat factors, and potential restoration actions to address the limiting habitat factors to benefit the limiting life stages. Other components, such as geomorphic potential (stream gradient and lateral confinement), current habitat condition, and projected future habitat condition based on stream temperature and flow modeling under various climate change scenarios, are also used to determine priority subwatersheds in which to sequence restoration actions over time. The assigned rating of each restoration action, the limiting habitat factors it would address, the degree to which a restoration action could lessen the impacts of climate change (Beechie et al. 2013a), and how an action could contribute to the restoration of watershed processes (Beechie et al. 2010) are then used to determine a score for each suite of restoration actions, which becomes a restoration opportunity. Each restoration opportunity is then scoped and mapped throughout the watershed and strategically prioritized for sequenced implementation over a period of 20 or more years (BPA 2015). Because the Atlas process is dependent upon available data and participation of local experts, the results, robustness, and time required to complete the process are likely to vary from watershed to watershed.

DISCUSSION

It is important to realize that no single assessment, model, data set, or analysis includes all the major steps in the restoration process (Figure 1). A thorough restoration plan will require the use of various tools depending on the watershed and its goals. A critical challenge for assessing watershed conditions and developing a detailed and effective restoration plan, and to adequately focus resources, is understanding which assessment tool or tools will help with which step or steps in the restoration process. Here we discuss how each of the major tools can be used to address key steps in the restoration process and discuss key considerations when selecting which assessment tools to use in a particular watershed or region.

Life Cycle and Fish–Habitat Models

Although the various fish–habitat models described above provide useful input for the restoration planning process, they all do not address the same steps or have similar data needs (Tables 2 and 4). Most limiting factors and life cycle models provide information for setting goals and identifying limiting factors and can provide information to use in prioritization of restoration projects (steps 1, 2, 3 and 5; Figure 1) (e.g., Beechie et al. 1994; Scheuerell et al. 2006). Because models such as EDT and intrinsic potential include considerable habitat information, they can also assist with assessing habitat and watershed conditions. Intrinsic potential typically uses a template of what habitat conditions should or could be based on gradient, valley width, confinement, and stream flow (Burnett et al. 2007), while EDT characterizes habitat conditions within in a reach, based on a combination of empirical data and professional opinion (Blair et al. 2009). Thus, it is important to understand the quality of habitat data inputs for EDT to understand the accuracy of habitat assessment information. Food web models can provide useful information on types of habitat that will produce the largest changes in fish production, and thus they can help identify the most relevant types of restoration and help prioritize restoration actions (e.g., Benjamin and Bellmore 2016; Wall et al. 2017). Tools that map predicted changes in stream flow and temperature due to climate change can be used to help identify areas where salmon or steelhead populations are more vulnerable to climate change (Isaak et al. 2010; Wade et al. 2013). They can also provide information to help prioritize reaches and to select appropriate restoration techniques to ameliorate climate change impacts. (Beechie et al. 2013a; Justice et al. 2017).

While the most useful fish–habitat model for a watershed depends on the goals of the individual restoration program and watershed assessment, as well as resources available, simple straightforward limiting-factors models can provide a quick and relatively inexpensive method to identify the limiting habitat and life stages of the species assuming empirical habitat and fish data are available. While adequate data on fish densities are not available for all species and habitats, such information can easily be collected. Limiting-factors models combined with intrinsic potential and climate change models present a relatively simple but valuable approach for broad-scale assessment of bottlenecks and potential habitat that is useful for identifying areas in need of restoration. Life cycle and other fish–habitat models, which are more sophisticated and more labor intensive, can provide more precise estimates of changes in survival and capacity and be used to model different restoration scenarios. However, it is important for managers to understand that these models make hypotheses about how a population is responding to habitat change, and this should be validated with field work and data collection to improve the quality of model inputs.

Watershed-Scale Assessment Tools

Assessing watershed processes such as hydrology, sediment delivery, nutrients, and other factors are some of the more important yet time-consuming tools in the watershed assessment process. Because of this, standardized assessments have not been thoroughly or frequently completed in many watersheds. In reality, watershed process assessments may not be more time consuming than many other assessments and models, though they require collecting data and interpretation by scientists with the proper training in geomorphology, hydrology, and other disciplines. Moreover, inadequate assessment of watershed processes is one of the most common reasons for the failure of reach-scale restoration projects (Roni et al. 2002, 2008; Friberg et al. 2016). The main utility of watershed assessment tools is assessing conditions and processes and identifying degraded processes (Table 5; step 2 in Figure 1). Watershed assessments can also be useful in setting restoration goals. For example, if it becomes clear that changes in stream discharge or timing or in sediment

TABLE 4. Major types of life cycle and fish–habitat models and the key step in the restoration process they address; steps in restoration process are based on Roni and Beechie (2013; Figure 1). Note that none of the models will directly assist with effectiveness monitoring or project design so these steps have been excluded from the table.

	Major step in restoration process					
Model	Goals	Assess condition	Limiting life stage	Problem identification	Select technology	Priorities
Life cycle	X		X			
Limiting-factors	X		X			
EDT	X		X	X		X
Intrinsic potential	X	X				
Climate change	X	X			X	X
Food Web	X				X	X

delivery are severely impaired in many subbasins, the overall restoration goals of the watershed should be revised to incorporate these factors. The various types of watershed-scale assessments can be used in combination to identify priority subwatersheds for restoration (Beechie et al. 2013b), but they do not help one prioritize site-level restoration actions per se. That is, while they can help identify the general types of restoration actions that would be useful, more site-specific surveys are needed to identify specific reaches or areas in need of restoration. For example, if a sediment budget identifies a subbasin as having a fine sediment load that is higher than normal due to a large volume of sediment from forest roads, this helps identify that roads in this subbasin need to be addressed. However, detailed road surveys are needed to identify specific road segments or crossings that are in need of improvement, restoration, or removal.

Reach-Scale Assessment Tools

Reach-scale assessments, such as those for riparian conditions, floodplain and fluvial processes, changes in channel and habitat conditions, and water quality, are critical for identifying lost and degraded habitats and for identifying specific areas in need of restoration on the landscape (Table 5). Riparian assessment techniques similarly focus on identifying degraded riparian condition in different reaches throughout a basin and reaches in need of restoration. Most reach-scale assessments have sufficient resolution to determine whether riparian areas are degraded (steps 2 and 3 in Figure 1) and in need of replanting or protection, but may lack the resolution needed to identify specific treatments (Table 5). It is likely that only a field-based assessment would provide sufficient detail to determine whether specific sites will require replanting, invasive species removal, livestock exclusion, or other treatments. Assessment of current and historical floodplain conditions and habitat is a critical component for restoration planning as it helps determine the amount of floodplain habitat lost—which is typically high in most developed floodplains in the Pacific Northwest—and the current impairments in need of remediation (e.g., levees, roads, bank armoring) (Table 5). Therefore, assessment of floodplain processes and conditions can help identify overall restoration goals and appropriate restoration actions. In addition, floodplain assessments provide important information on current and historical habitat area that is useful for limiting factors, EDT, or other fish–habitat models (Beechie et al. 1994; Blair et al. 2009). Simple surveys of bank armoring and levees across an entire basin, while rarely done, can also prove critical for identifying restoration opportunities.

More detailed reach level modeling and hydraulic analysis, such as HEC RAS models or HSI modeling based on data from topographic surveys, not only provide important information on assessing reach-scale channel conditions and identifying problems and potential restoration actions, but also provide critical information for restoration design (Table 5; steps 2 and 3 in Figure 1). Habitat,

TABLE 5. Major categories of watershed and reach assessment tools and the key step in the restoration process they address; steps in restoration process are based on Roni and Beechie (2013; Figure 1). An asterisk (*) indicates depends upon whether these are continuous surveys or only at selected sites.

	Major step in restoration process					
Assessment tool	Goals	Assess condition	Problem identification	Select technology	Priorities	Design
		Watershed-scale assessments				
Sediment budget	X	X	X	X	X	X
Hydrology	X	X	X	X		X
Water quality, nutrients	X	X	X	X		
		Reach-scale assessments				
Riparian mapping	X	X	X	X	X	
Floodplain conditions	X	X	X	X	X	X
Connectivity (e.g., barriers, revetments)	X	X	X	X	X	
BOR reach assessments		X	X			X
HEC RAS 2D						X
HSI						X
Habitat assessment*	X	X	X			
Spawner surveys*		X	X			
Juvenile fish*		X	X			
Effectiveness monitoring				X	X	X

fish, and other status and trend monitoring programs can also provide important information for identifying degraded habitat conditions across a watershed and potential restoration opportunities (steps 2 and 3 in Figure 1), but this is largely dependent upon their coverage (Beechie et al. 2013b). Similarly, effectiveness monitoring programs designed to evaluate the physical and biological effectiveness of restoration actions (Roni et al. 2013b) can provide useful information to help identify the most appropriate restoration type to address a problem as well as prioritize and design restoration actions. Ideally, habitat, spawner, or juvenile fish abundance surveys are conducted across an entire watershed so that differences in habitat conditions and fish use among reaches can be compared. Unfortunately, some habitat monitoring programs sample only a small portion of each watershed using very thorough and intensive methods designed for status and trend monitoring rather than assessment of watershed conditions (CHaMP 2016). Thus, the utility of many types of habitat and watershed monitoring activities for assessing watershed conditions and identifying restoration opportunities is limited unless the site-specific data can be rolled up or extrapolated to broader scales.

Assessments of anthropogenic barriers to fish migration have been completed for many Pacific Northwest watersheds and have clearly identified opportunities for restoring isolated habitat (Table 5). Because these surveys identify the type of barrier, including height and other information, they provide useful information for selecting appropriate restoration techniques. Moreover, when coupled with habitat data, barrier surveys provide important inputs for prioritization of restoration projects. Historically, monitoring water temperature was limited to specific sites, but advances in technology and analytical and mapping tools have allowed either surveys of entire reaches or interpolation of site measurements to larger areas (Torgersen et al. 2001; Vaccaro and Maloy 2006; Figure 4). Moreover, data from the EPA on reaches that have limited water quality are reach-specific and identify areas of degraded water quality and restoration opportunities.

Prioritization Tools

The three major approaches for prioritizing watersheds and restoration actions (MCDA, EDT, and Atlas) vary in whether they simply assist with prioritization of watersheds, reaches, or restoration actions, or assist with other steps in the restoration process. The MCDA approach is designed specifically for prioritizing watersheds, reaches, and restoration projects and is dependent on information from fish–habitat models and watershed and reach assessments to assist with scoring selected prioritization criteria. The EDT model provides an approach for prioritizing reaches for restoration and can assist with setting goals and identifying areas of degraded habitat. This is true in part because instream habitat survey data are the critical input for the model (Blair et al. 2009). However, EDT does not identify underlying causes of degradation or help with restoration design. Finally, the BPA Atlas framework is an approach that assesses conditions based on existing data and professional opinion and thus identifies important reaches for restoration. This approach uses this information and MCDA to prioritize watersheds, subwatersheds, reaches, and restoration projects for longterm, sequenced implementation.

Summary and Conclusions

While dozens of tools exist to assist with watershed assessment and restoration planning, little guidance has been provided on which tools will be most useful for different steps in the restoration process. It is important for restoration practitioners and fisheries scientists to realize that no one tool will provide all the information needed for all steps in the process of developing a comprehensive watershed restoration plan. The most useful assessment tools provide consistent information across a watershed and can be examined at a reach and watershed scale. This renders data from many monitoring or modeling programs, which were designed for other purposes, to be of limited utility for assessing conditions and identifying restoration opportunities. As a general rule, the assessments needed in a specific watershed will depend largely on the goals and objectives of the restoration program. To assist with identifying the required assessments in a particular watershed, we have outlined the major inputs and outputs of various tools for developing a successful restoration plan and have identified which step or steps in the restoration process each tool addresses. While useful for watersheds where there is little information, this review should prove highly useful for guiding restoration in watersheds where there is already considerable data or information. When coupled with the overall goals of the restoration and information on existing watershed assessments, this approach should provide a method for determining which tool or tools will be most useful to address remaining assessment needs for specific watersheds or fish populations.

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