Scientific Support for the Wyoming Stream Quantification Tool

Wyoming Stream Technical Team

Army Corps of Engineers
Omaha District

StreamMechanics

Ecosystem Planning & Restoration

Scientific Support for the Wyoming Stream Quantification Tool

October 2018

Prepared by:

Wyoming Stream Technical Team

Paige Wolken, U.S. Army Corps of Engineers, Omaha District, Wyoming Regulatory Office

Julia McCarthy, U.S. Environmental Protection Agency, Region 8

Paul Dey, Wyoming Game and Fish Department

Jeremy ZumBerge, Wyoming Department of Environmental Quality

Will Harman, Stream Mechanics

Cidney Jones, Ecosystem Planning and Restoration

Citation: Wyoming Stream Technical Team. 2018. Scientific Support for the Wyoming Stream Quantification Tool, Version 1.0. U.S. Army Corps of Engineers, Omaha District, Wyoming Regulatory Office. Cheyenne, WY.

Table of Contents

List of Figures

List of Tables

Acknowledgements

The scientific support for the Wyoming Stream Quantification Tool was developed to provide the technical basis for the reference curves included in the Wyoming Stream Quantification Tool. This is an original document developed by the Wyoming Stream Technical Team (authors). Many others provided valuable contributions, including: Mike Robertson and other Aquatic Staff with WGFD who assisted with data collection and provided input on fish metrics, WDEQ Water Quality Division staff who assisted with data collection, LeeAnne Lutz (EPR) and Amy James (EPR). Valuable peer review was provided by Sarah Miller, Eric Somerville, Ken Fritz, Brian Topping, Rachel Harrington, Billy Bunch and Joanna Lemley. This document also benefited from others who provided feedback and comment on the WSQT Beta Version through the Public Notice issued by the Wyoming Regulatory Office. Project funding was provided by USEPA Region 8 and Headquarters (contract # EP-C-17-001). The WSQT v1.0 and accompanying documents are available on the USACE Omaha District, RIBITs and the Stream Mechanics web pages.

http://www.nwo.usace.army.mil/Missions/Regulatory-Program/Wyoming/Mitigation/

https://stream-mechanics.com/stream-functions-pyramid-framework/

Acronyms

BEHI/NBS – Bank Erosion Hazard Index / Near Bank Stress

BHR – Bank Height Ratio

CFR – Code of Federal Register

CN – Curve numbers

Corps – United States Army Corps of Engineers (also, USACE)

CWA 404 – Section 404 of the Clean Water Act

ER – Entrenchment Ratio

FF – Functional Feet

GSR – Greenline Stability Rating

LWD – Large woody debris

LWDI – Large Woody Debris Index

NRCS – Natural Resource Conservation Service

SFPF – Stream Function Pyramid Framework

SGCN – Species of Greatest Conservation Need

USACE – United States Army Corps of Engineers (also, Corps)

USDOI – United States Department of Interior

USEPA – US Environmental Protection Agency

USFWS – United States Fish and Wildlife Service

WDEQ – Wyoming Department of Environmental Quality

WDR – Width Depth Ratio

WGFD – Wyoming Game and Fish Department

WSQT – Wyoming Stream Quantification Tool

WSTT – Wyoming Stream Technical Team

Glossary of Terms

Alluvial Valley – Valley formed by the deposition of sediment from fluvial processes.

Catchment – Land area draining to the downstream end of the project reach.

Colluvial Valley – Valley formed by the deposition of sediment from hillslope erosion processes, typically confined by terraces or hillslopes.

Condition – The relative ability of an aquatic resource to support and maintain a community of organisms having a species composition, diversity, and functional organization comparable to reference aquatic resources in the region. (see 33CFR 332.2)

Condition Score –Metric-based index values are averaged to characterize condition for each parameter, functional category, and overall project reach.

ECS = Existing Condition Score

PCS = Proposed Condition Score

Field Value – A field measurement or calculation input into the WSQT for a specific metric. Units vary based on the metric or measurement method used.

Functional Capacity – The degree to which an area of aquatic resource performs a specific function. (see 33CFR 332.2)

Functions – The physical, chemical, and biological processes that occur in ecosystems. (see 33CFR 332.2)

Functional Category – The organizational levels of the stream functions pyramid: Hydrology, Hydraulics, Geomorphology, Physicochemical, and Biology. Each category is defined by a functional statement.

Functional Feet (FF) – Functional feet is the primary unit for communicating functional lift and loss, and is calculated by multiplying a condition score by stream length. ΔFF is the difference between the Existing FF score and the Proposed FF score.

Functional Statement – A description of the functions within each functional category in the Stream Functions Pyramid Framework.

Function-Based Parameter – A structural measure which characterizes a condition at a point in time, or process (expressed as a rate) that describes and supports the functional statement of each functional category.

Index Values: Dimensionless values between 0.00 and 1.00 that express the relative condition of a metric field value, as compared with reference standards. These values are derived from reference curves for each metric. Index values are combined to create parameter, functional category, and overall reach scores.

Measurement Method – A specific tool, equation or assessment method used to inform a metric. Where a metric is informed by a single data collection method, metric and measurement method are used interchangeably. (see Metric)

Metric – A specific tool, equation, measured values or assessment method used to assess the condition of a structural measure or function-based parameter. Some metrics can be

derived from multiple measurement methods. Where a metric is informed by a single data collection method, metric and measurement method are used interchangeably (see Measurement Method).

Performance Standards - Observable or measurable physical (including hydrological), chemical and/or biological attributes that are used to determine if a compensatory mitigation project meets its objectives. (see 33 CFR 332.2)

Reference Aquatic Resources – A set of aquatic resources that represent the full range of variability exhibited by a regional class of aquatic resources as a result of natural processes and anthropogenic disturbances. (see 33 CFR 332.1)

Reference curves – A relationship between observable or measurable metric field values and dimensionless index values. These curves take on several shapes, including linear, polynomial, bell-shaped, and other, to best represent the degree of departure from a reference standard for a given field value. These curves are used to determine the index value for a given metric at a project site.

Reference Standard – The subset of reference aquatic resources that are least disturbed and exhibit the highest level of functions. In the WSQT, this condition is considered functioning for the metric being assessed, and ranges from least disturbed to minimally disturbed or pristine condition.

Stream Functions Pyramid Framework (SFPF) – The Stream Functions Pyramid is comprised of five functional categories stratified based on the premise that lower-level functions support higher-level functions and that they are all influenced by local geology and climate. The SFPF includes the organization of function-based parameters, measurement methods, and performance standards to assess the functional categories of the Stream Functions Pyramid (Harman et al. 2012).

Threshold Values – Criteria used to develop the reference curves and index values for each metric. These criteria differentiate between three condition categories: functioning, functioning-at-risk, and not functioning and relate to the Performance Standards defined in Harman et al. (2012).

Wyoming Stream Quantification Tool (WSQT) – The WSQT is a spreadsheet-based calculator that scores stream condition before and after restoration or impact activities to determine functional lift or loss, and can also be used to determine restoration potential, develop monitoring criteria and assist in other aspects of project planning. The WSQT is based on principles and concepts of the SFPF.

Wyoming Stream Technical Team (WSTT) – The group who worked on the development of function-based parameters, metrics and reference curves for the WSQT and are the authors of this document.

Chapter 1. Background and Introduction

The purposes of this document are to provide the scientific underpinnings of the Wyoming Stream Quantification Tool (WSQT) and the rationale for the conversion of measured stream condition into dimensionless index scores. The WSQT is an application of the Stream Functions Pyramid Framework (SFPF), outlined in 'A Function-Based Framework for Stream Assessment and Restoration Projects' (Harman et al. 2012). Harman et al. (2012) presents the SFPF and provides supporting references and rationale for the organizational framework and its components. The WSQT is one of several Stream Quantification Tools (SQTs) that have recently been developed for use in specific states, including North Carolina (Harman and Jones 2017), Tennessee (TDEC 2017) and Georgia (USACE 2018b).

Here, we expand on the concepts presented in the SFPF and the WSQT to provide the scientific and technical rationale behind selection of the reference curves and metrics included in the WSQT. This document is the first of its kind to be developed in conjunction with a SQT; other versions of the tool include a List of Metrics but have yet to develop detailed supporting documentation. Information on how to use the WSQT or collect data for use in the WSQT is not included in this document, but can be found in the WSQT v1.0 User Manual (USACE 2018a).

Section 1.1 provides a summary of the SFPF, including function-based parameters and metrics.

Section 1.2 provides a summary of the watershed context for determining restoration potential.

Section 1.3 provides a description of reference curve development, and describes how key concepts of reference standard and functional capacity are used in the tool.

Section 1.4 gives an overview of how the WSQT calculates the overall reach condition score, along with weighting considerations.

Section 1.5 discusses the selection of functional feet as the primary unit for communicating functional lift and loss within the tool, and its use in informing debits and credits.

Section 1.6 provides the general criteria used to select function-based parameters and metrics from the SFPF and new metrics developed specifically for the WSQT.

Section 1.7 provides a general summary of the datasets used to develop reference curves and the tool's data gaps and limitations.

Section 1.8 provides key considerations in applying the WSQT.

After the Introduction and Background, the remainder of the document is organized by function-based parameter. Each parameter description includes a summary of why it was included, reasons for selecting the metrics, and in some cases, why other metrics were not selected. Then, the description of metrics used to quantify the parameter are provided. For each metric, the rationale for developing reference curves and any stratifications are provided, followed by data gaps and limitations.

1.1. Background on the Stream Functions Pyramid Framework (SFPF)

In 2006, the Ecosystem Management and Restoration Research Program of the Corps noted that specific functions for stream and riparian corridors had yet to be defined in a manner that

was generally agreed upon and suitable as a basis for which management and policy decisions could be made (Fischenich 2006). To fill this need for Corps programs, an international committee of scientists, engineers, and practitioners defined 15 key stream and riparian zone functions aggregated into 5 categories: system dynamics, hydrologic balance, sediment processes and character, biological support, and chemical processes and pathways (see Table 1 in Fischenich 2006). The committee noted that restoration of hydrodynamic processes, sediment transport processes, stream stability, and riparian buffers could lead to improvements in dependent functions that typically require time to establish, such as diverse biological communities, nutrient processes, diverse habitats, and improved water and soil quality. The SFPF builds on the work completed by Fischenich (2006) by organizing stream functions into a hierarchical structure to create a conceptual model for restoration practitioners to use in communication and the development of function-based assessments.

The SFPF organizes stream and riparian functions into five functional categories: Hydrology, Hydraulics, Geomorphology, Physicochemical, and Biology (Figure 1-1). This organization recognizes that foundational functions, like watershed hydrology and sediment transport processes, generally support higher-level functions like aquatic animal-life histories and that all functions are influenced by local geology and climate. Cause and effect can flow from top to bottom as well, e.g., beavers (biology) can affect hydrology, and riparian communities can influence hydraulics and geomorphology through wood inputs, rooting depths and floodplain roughness. However, the primary thought process for this framework is this: what supporting processes are needed to restore a particular function? With this perspective, the beaver example would change to: what functions are needed to support a healthy beaver population?

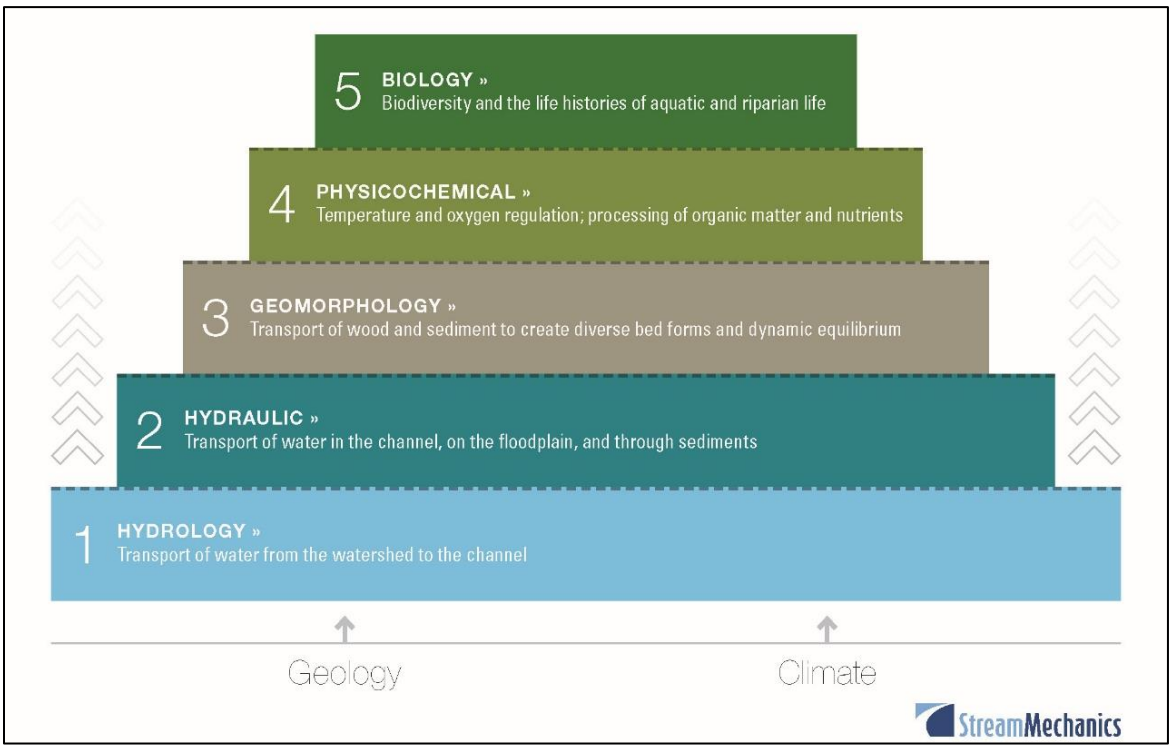

Figure 1-1: Stream Functions Pyramid (Image from Harman et al. 2012)

Within each of the five functional categories, the SFPF outlines parameters and methods to quantify the degree to which a stream ecosystem is functioning (Figure 1-2). In this framework,

function-based parameters describe and support the functional statements of each functional category, and the measurement methods (metrics) are specific tools, equations, measured values and/or assessment methods that are used to quantify the function-based parameter. The SFPF presents two types of function-based parameters and metrics: structural indicators which describe a condition at a point in time, and functions expressed as a rate that tie directly to a stream process (e.g., bank erosion rates). Each metric is compared against performance standards (reference curves) that represent departure from, or attainment of, reference standard. The selection of function-based parameters used in the WSQT and their relationship to reference standards are discussed in more detail in the following sections.

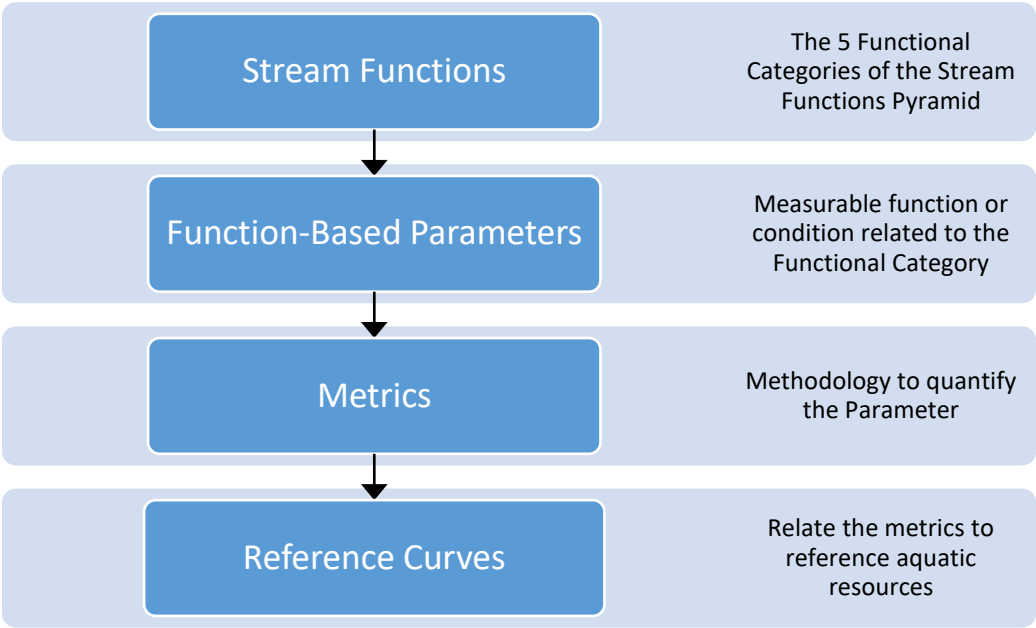

Figure 1-2: Stream Functions Pyramid Framework. Note: terms have been edited to match WSQT application.

The SFPF has informed the development of the WSQT, which is modified from the North Carolina SQT (Harman and Jones 2017). The WSQT is a tool that consolidates the components of the SFPF into an excel workbook to characterize stream ecosystem functions at a specific project reach. The WSQT includes a sub-set of function-based parameters and metrics listed in Harman et al. (2012) along with new parameters and metrics identified as part of the WSQT development and regionalization process, which are relevant to the stream systems found within the state of Wyoming. The Tennessee and North Carolina SQTs calculate an overall reach score from all five functional categories presented in the SFPF, while the Georgia SQT calculates an overall reach score from three of the five (hydraulics, geomorphology and biology). The WSQT merges the original Hydrology and Hydraulics categories into a new combined category (referred to as the Reach Hydrology and Hydraulics Category), leading to an overall reach score calculated using four categories. This change to the WSQT was made due to the small number of parameters and metrics selected in both categories and the consequent disproportionate weighting those parameters were allocated. Differences among the SQTs is primarily due to decisions made at the state-level in consideration of state-specific priorities and resources.

All the metrics selected for the WSQT are measurements of point-in-time condition that serve as indicators of structural or compositional attributes related to underlying stream processes. Assessment data are input into the WSQT, where data for each metric is translated into an index value, thus converting a variety of units into a standardized unitless score. To translate the data into index values, reference curves have been derived for each metric that relate site-specific data to degrees of departure from reference standard.

1.2. Watershed Context

Understanding the watershed processes that support the condition at a particular project site is a critical component to any project. Anthropogenic modification to stream processes can occur via direct and indirect pathways. Direct pathways include effects on reach-scale processes like channel modification, removal of riparian or aquatic vegetation, flow alteration or introduction of non-native species; and indirect pathways are often alterations to watershed-scale processes, like land use changes, that occur away from the stream or distributed throughout a watershed (Roni and Beechie 2013).

The WSQT v1.0 User Manual includes a catchment assessment and a process to determine the restoration potential at a project site, which takes into consideration the watershed context and limiting factors affecting a project site. The catchment assessment was modified by the WSTT from the North Carolina SQT to consider anthropogenic modifications common in Wyoming, including flow alteration. The catchment assessment is a qualitative approach intended to identify watershed-scale factors that may limit the restoration potential at a project site. Restoration potential is defined as the highest level of restoration that can be achieved based on the health of the watershed, project constraints at the reach scale, and the existing condition of the project reach (Harman et al. 2012). Full restoration potential indicates that the project can return biological functions to a reference standard. Partial restoration potential means that improvements can be made, but not all functions can be returned to a reference standard (Beechie et al. 2013).

While the watershed context is critical to understanding the limitations to restoration potential at a site, the focus of this tool is on the “delta” or the change in reach-scale ecological variables following a project. The WSQT focuses primarily on indicators of condition tied to direct pathways of anthropogenic modification within a reach-based approach. Because watershed condition is not likely to change following reach-scale activities, it differs from the metrics that are included in the tool and the WSTT decided to not include it as part of the scoring within the tool itself. Therefore, the catchment assessment is only used to assist the practitioner in determining the restoration potential of the project reach. Information on the catchment assessment and determining restoration potential are included in the WSQT v1.0 User Manual.

1.3. Development of Reference Curves

The metrics within the WSQT are measurements of point-in-time condition. The purpose of the tool is to compare how this condition may change following an impact or restoration activity and draw reach-scale conclusions on changes in functional capacity pre- and post-project. These changes are referred to in the WSQT as functional loss and lift, and relate to the definition of debits and credits in the 2008 Mitigation Rule (33 CFR 332.3). To relate these point-in-time measurements to functional capacity and standardize all metrics to a common scale, reference curves were developed to assign index values that reflect a range of condition. Each numeric

index value range (0.00 to 1.00) was standardized across metrics by determining how field values relate to functional capacity, i.e., functioning, functioning-at-risk and non-functioning condition (Table 1-1).

To account for natural variability among stream systems, reference curves for specific metrics may be stratified by differences in drainage area, stream type, ecoregion, reference community type, or valley type. Stratification varies by metric.

The reference curves in the WSQT are derived from identification of specific benchmarks (thresholds) that represent the degree to which the measured condition departs from a desired or expected condition (Hawkins et al. 2010). Determining where field values lie within the range of conditions that comprise the reference curve can denote whether parameters are attaining a reference standard, and which parameters may require restoration or adaptive management efforts. For purposes of mitigation, these values can also provide a quantitative, objective approach to monitoring, and can be used to inform performance standards and credit release schedules.

Table 1-1: Functional capacity definitions used to define threshold values and develop reference curves for the WSQT

Functional Capacity	Definition	Index Value Range
Functioning	A functioning value means that the metric is quantifying or describing the functional capacity of one aspect of a function-based parameter in a way that supports a healthy aquatic ecosystem. In other words, it is functioning at reference standard condition. The reference standard concept used here aligns with the definition laid out by Stoddard, et al. (2006) for a reference condition for biological integrity. A score of 1.00 does not represent the best attainable condition, but an unaltered or minimally disturbed system. A range of values (0.70-1.00) are provided to account for natural variability and the inclusion of least disturbed and minimally disturbed sites within reference standard datasets.	0.70 to 1.00
Functioning-at-risk	A functioning-at-risk value means that the metric is quantifying or describing one aspect of a function-based parameter in a way that can support a healthy aquatic ecosystem. In many cases, this indicates the function-based parameter is adjusting in response to changes in the reach or the catchment. The trend may be towards lower or higher function. A functioning-at-risk value indicates that the aspect of the function-based parameter, described by the metric, is neither representing an impaired nor least disturbed condition.	0.30 to 0.69
Not functioning	A not functioning value means that the metric is quantifying or describing one aspect of a function-based parameter in a way that does not support a healthy aquatic ecosystem. A value of 0.29 represents a condition that is severely altered or impaired, and a value of 0.00 represents a condition that is indicative of no functional capacity.	0.00 to 0.29

Reference curves were developed by first identifying field values that would serve as thresholds between the categories of functional capacity outlined in Table 1-1. Three approaches were taken to identify these threshold values.

1. Where possible, thresholds were derived from data values already identified in the State of Wyoming's technical publications or the literature (e.g., based on water quality standards, channel classification, or existing indices).
2. Where literature values were not available, threshold values were developed using data from national and regional resource surveys and other available datasets. In evaluating reference datasets, the team considered the degree of departure from reference standard using percentiles of regional reference condition to identify the threshold values. This is similar to other approaches that identify benchmarks or index values (BLM 2017, Nadeau et al. 2018).
3. Where existing data or literature was limited, best professional judgement by the WSTT was used to identify threshold values. In some instances, the decision was made to not identify a threshold value and instead extrapolate index values from a best fit line from data or literature values that were available.

Following the identification of these threshold values, reference curves were fit using linear relationships between threshold values. These continuous curves allow index scores to account for incremental changes in field values, which is important for determining a change in the pre- and post-project condition. If a non-linear fit was used, the rationale for selecting an alternative fit is provided in the metric section below. Reference curves and threshold values were determined for each metric individually. Therefore, a stream reach may achieve a reference standard index value for one parameter, e.g., large woody debris, and not others. Metric index values are then combined to provide a reach score (Section 1.4).

For the WSQT, reference standard is used to define full functional capacity. The greater the departure from reference standard, the lower the index value. The maximum index value equates to the minimally disturbed condition, i.e., the condition expected in the absence of significant anthropogenic influences to the degree that it can be represented by existing data (Stoddard et al. 2006). A range of index values (0.70-1.00) is used for characterizing reference standard to account for natural variability, and recognizing that reference standard datasets include sites that reflect least disturbed condition (i.e., the best available conditions given current anthropogenic influence per Stoddard et al. 2006). In use of existing datasets, the WSTT relied on the definitions of reference condition provided by the authors.

The functioning-at-risk range of index values (0.30-0.69) reflect a condition that can support a healthy aquatic ecosystem, but is not achieving the highest level of function consistent with a reference standard. This range characterizes a grey area, where a resource may be trending towards a higher or lower functional capacity, but is neither attaining a reference standard nor is significantly degraded or impaired. It is important to understand the difference between the attainment of reference standard and functional lift or loss. Functional lift or loss is the difference in the condition scores before and after restoration or a permitted impact. It is possible for a restoration site to have a large lift, but only achieve a functioning-at-risk score because it does not (or cannot) attain a least disturbed condition.

The non-functioning range of index values (0.00-0.29) reflects a degraded or impaired condition that does not support a healthy aquatic ecosystem. Measured field values at the upper end of

this range reflects an impaired or severely altered condition and an index value of zero reflects a condition that provides no functional support for that parameter. Minimum index values were often extrapolated from the best fit lines, and the WSTT considered whether field values would reasonably represent no functional support for that metric.

1.4. Calculating Reach-Scale Condition

The architecture and scoring of the WSQT is simple to allow for flexibility in selecting function-based parameters and metrics, and to allow for additions or exchanges of parameters in the future with advances in stream science. While the WSQT v1.0 User Manual recommends a subset of parameters and metrics be evaluated for all projects (e.g., reach runoff, floodplain connectivity, lateral migration, bedform diversity and riparian vegetation), the tool includes a broader set of parameters that may better align with a project's function-based goals and objectives. For example, a practitioner may choose not to monitor (or receive credit for) physiochemical and biological parameters, and the WSQT would then calculate scores based only on the subset of parameters and metrics that were input into the tool. This approach differs from assessment approaches that rely on rigorous statistical analyses for metric selection, calibration and scoring (Stoddard et al. 2008). There are obvious limitations to this simpler approach, however, a benefit of this approach is the flexible architecture – metrics and parameters can be added to or subtracted from the tool based on new scientific understandings or site-specific considerations without requiring substantial reanalysis of the weighting in the tool. For example, for a specific site or analysis, the same weighting and metrics would be used for each monitoring event to preserve the rigor of the comparison, but additional metrics could be applied at another site based on a different set of site objectives. Because the focus of the tool is on the difference between before and after conditions, flexibility was prioritized over a rigorous approach to weighting (since scoring will be handled the same for before and after conditions).

Index values are generated for each metric, and then combined to provide an overall reach score. Only the metrics selected and assessed for a given project reach are used to calculate the overall score. Metrics not assessed are simply not included in the score; they are not scored as a zero. Metric index values are averaged to create a parameter score, and then parameter scores are averaged to create a functional category score. The category scores are weighted and summed to calculate an overall reach-scale condition score. As noted above, the WSQT combines the hydrology and hydraulic categories from the stream functions pyramid into one category called Reach Hydrology and Hydraulics (H&H). The H&H category is weighted to provide 30% of the overall score; geomorphology provides 30%; and physicochemical and biology each provide 20% of the overall score. The H&H and geomorphology functional categories were weighted at 60% of the total score, reflecting the number and breadth of parameters in these categories. The weighting for the physicochemical and biological categories (20% weighting each) is slightly less than the other two categories because they can be heavily influenced by changes in watershed-scale processes outside of the project reach and often take longer to show improvement post restoration (Fischenich 2006). Functional improvement in these categories often occurs due to improvements in hydrology, hydraulics and geomorphology functions (if catchment-scale stressors do not themselves limit physicochemical or biological improvements). The weighting incentivizes restoration practitioners to attempt to improve and monitor these parameters, even if they may not reach full restoration potential. The maximum overall condition score achievable by monitoring only H&H and geomorphology parameters is 0.60, which is consistent with other SQTs.

The WSQT estimates the change in condition at an impact or mitigation site by calculating the difference between existing (pre-project) and proposed (post-project) condition. This change in condition is referred to as the “delta.” Existing and proposed condition scores are multiplied by stream length to calculate a Functional Feet score. In a pristine stream with an existing condition score of 1.00, one functional foot would equal one linear foot of stream. When condition is less than 1.00, or not all functional categories are measured, the functional feet score is no longer equivalent to stream length.

1.5. Calculating Functional Feet

The change in functional feet pre- and post-project, or the “delta”, is intended to serve as the basis for calculating debits and credits. The functional feet calculation made in the WSQT workbook incorporates measures of stream condition (Existing Condition and Proposed Condition Scores) and stream length (existing and proposed). The integrated functional feet unit is more representative of ecological function than extent (stream length or area only) or activity-based approaches, as it relies on a quantitative approach to assess condition that will change because of impact or restoration activities. Many programs continue to rely on activity or ratio-based approaches to calculate credits or assign credit ratios based on changed to channel geometry (ELI et al. 2016). The goal of the WSQT is to create a unit that better integrates changes in condition into crediting and debiting approaches. The functional feet unit serves as the bridge between the condition assessment and application within a debit/credit policy framework for program implementation because it provides a unit of measure that can be added together and compared across sites better than condition scores alone.

The inclusion of stream length in the functional feet unit adds scale to the condition score. For example, a culvert removal provides a substantial amount of lift (i.e., a large change between the proposed and existing condition score), but a fairly small change in functional feet because the reach is very short. A very long project with moderate condition improvement will produce a bigger change in the functional feet score because of its scale. The use of stream length follows that of other Corps Districts with established compensatory stream mitigation programs, which use a variety of factors that are then multiplied by stream length to create a debit or credit (Table 1-2). This product of quality and length is a common currency for a debit and credit calculations (ELI et al. 2016).

Table 1-2: Debit and credit approaches that consider a combination of condition and stream length, including the approach for Wyoming. Adapted from ELI et al. (2016).

State/Corps District	General Debit Approach	General Credit Approach
Nebraska (2012)	***Impact Units = (Stream Condition Index Score) x (stream length)*** Includes a condition assessment procedure and impact/mitigation calculator predicting proposed condition.	***Mitigation Units = (Stream Condition Index Score) x (stream length)*** Same assessment and calculator used to compare impact and compensation sites.

Table 1-2: Continued from previous page.

State/Corps District	General Debit Approach	General Credit Approach
South Pacific Division (2015)	Mitigation Ratio Checklist factors in several multipliers but can also incorporate functional assessment data via a before-after-mitigation-impact (BAMI) spreadsheet to determine baseline ratio.	Mitigation Ratio Checklist - both impact and mitigation info are input into same checklist
Pennsylvania (2004)	***Compensation Requirement (CR) = (area of impact) x (PE) x (RV) x (CI)*** PE = project effect factor based on severity of impacts RV = resource value based on categories of resource quality CI = condition index value from condition assessment CR is calculated for each aquatic resource function category and summed for total debit.	***Functional Credit Gain (FCG) = (area of project) x (RV) x (CV) x (CI diff)*** RV= same as debits CV = compensation value based on level of benefit (1-3) CI diff = condition index differential value based on difference in existing and predicted condition
Galveston (2013)	***Compensation requirement (CR) = (Δ resource condition) x (impact factor) x (stream length)*** Like Pennsylvania, but CR is not calculated separately for each functional category.	Credits defined by level of effort (e.g., reestablishment is 3 credits/LF, rehabilitation or enhancement is 1 credit/LF) and factors related to the riparian buffer.
Norfolk (2004)	***Compensation Requirement = (length of impact) x (RCI) x (IF)*** RCI = reach condition index is a weighted average of categorical condition indices for four parameters (channel condition, riparian buffer, instream habitat, channel alteration) IF = impact factor based on the severity of impact (0-1)	***Compensation Credit = restoration credit +enhancement credit + riparian buffer credit + adjustment factor credit*** Credits defined by level of effort (e.g., restoration is 1 credit/LF, enhancement is 0.09-0.3 credit/LF, etc.) and other adjustment factors (e.g., T&E or watershed preservation)
West Virginia (2011)	***Unit Score = (Index Score) x (stream length)*** Debit tables used to calculate index score (0-1), considers West Virginia Stream and Wetland Valuation Metric (WVSWVM) which incorporates assessment methods using project specific data, and factors like temporal loss and site protection.	***Unit Score = (Index Score) x (stream length)*** Index score derived from credit tables. Unit scores are calculated at mitigation site over three intervals (existing, post-construction & maturity).
Wyoming (2018)	***Debits = ΔFF x sum[DF]*** ΔFunctional Feet (ΔFF) = (Proposed Condition Score x proposed stream length) – (Existing Condition Score x existing stream length) DF = debit factors identified in the WSMP v2.	***Credits = ΔFF x sum[CF]*** ΔFunctional Feet(ΔFF) same as debits CF = credit factors identified in the WSMP v2.

The WSTT considered several alternatives to the length-based approach, including a functional-area product and a valley-area measure. Both are discussed briefly below:

Functional-area product: This approach would rely on an area-based measure instead of stream length. An approach using stream width by stream length may better account for the size differences between small and large streams, including a greater amount of aquatic habitat in a larger stream. The major challenges with an approach that relies on channel width is that width often changes. In the western U.S., flow alteration has led to substantial changes in hydrology followed by adjustments in channel form, including narrower channels. Where flows cannot be restored, restoration approaches may include accelerating this channel evolution to improve stream condition and underlying processes, and including width in the credit calculation would lead to less potential for credit. In addition, practitioners commonly design a wider width than the final target. The channel narrows during the monitoring years as vegetation becomes established on the streambanks. The vegetation increases boundary roughness, which deposits sediment on the bank and narrows the channel width. So, this natural and positive process would result in the practitioner losing area between the design and monitoring phases. Attempts to predict the final width would be difficult and create more uncertainty than relying on length alone. Because of these implementation challenges, the WSTT decided to not pursue this approach.

Valley area: Another approach that was considered was using valley area instead of stream length. This approach has merit, as it characterizes the stream and floodplain corridor in a more holistic way and better accounts for floodplain functions and stream systems that include stream-wetland complexes and/or multi-thread channels. However, the major challenge with this approach is in accounting for the net loss or gain in stream length, an important consideration in the regulatory program. The Corps currently accounts for permitted impacts in linear feet or aquatic resource area (e.g., Nationwide Permit impact thresholds, data entry into ORM database) and only regulates activities within aquatic resource boundaries (e.g., within a delineated wetland or the ordinary high water mark of streams), and it is unclear how a valley-based approach could align with the current practices for accounting for impacts. Additional discussions and research on implementation are needed before adopting a valley-based approach across all projects. This may be considered for future versions of the WSQT.

The unit of measure in the WSQT v1.0 is functional feet because it conforms with many existing stream mitigation approaches while improving the link between activities and changes in condition. Future versions of the WSQT and WSMP may accommodate alternate or modified approaches, as discussed above, but more consideration on how these approaches could be implemented on the debit and credit side is needed before this selection is made.

1.6. Function-Based Parameters in the WSQT v1.0

The WSQT is designed to consider a suite of functional indicators that are sensitive to anthropogenic modification of reach-scale processes, i.e., the types of activities (both impact and mitigation projects) that are common in the Clean Water Act Section 404 (CWA 404) dredge and fill permitting program. The tool also considers related ecosystem functions that could similarly be affected by these activities, including changes to water quantity, water quality, and biological communities. The WSQT incorporates many of the functions and parameters outlined in Fischenich (2006) and Harman et al. (2012). The WSQT v1.0 User Manual identifies a subset of parameters and metrics included in the tool that should be evaluated for all projects. Recognizing that not all compensatory mitigation projects will have the same objectives or components, the WSQT allows for flexibility in selecting additional parameters for specific

projects. ELI et al. (2016) noted that regulatory protocols should allow for function-based goals and objectives that are project specific, clearly stated, and feasible so that performance standards and monitoring can be targeted for that specific project. Parameters included in the WSQT could assist in setting performance standards for projects with goals to restore instream flows, restore targeted fish communities, improve water quality, or implement other project-specific objectives.

The complete set of function-based parameters and metrics used in the WSQT v1.0 is listed in Appendix A. Rationale for selecting a parameter and its metrics, and why other metrics were not used, is provided in Table 1-3 and throughout this document in the parameter summaries. The overarching criteria used to select parameters and metrics included the following:

- Ability to link the parameters to the functional statement in the SFPF and ability to link the metrics to restoration or impact activities. The metric that informs the functional capacity of the parameter should be responsive to activities.
- Parameters and metrics should be reach-based. Changes in metrics should occur at a reach scale where restoration and impact activities occur. Note, stressors and perturbations that occur at a watershed scale may affect both existing and potential condition scores and are considered in the catchment assessment and determination of restoration potential (see Section 1.2 and the WSQT v1.0 User Manual for details).
- Ability to develop reference curves for each metric. Information needs to be available to characterize the reference aquatic resources and relate conditions to a reference standard.
- Level of effort for data collection and analysis. Corps Guidance (RGL 08-03) articulates that the level of analysis and documentation for evaluating applications under CWA 404 should be commensurate with the scale and scope of a project.
- Applicable and meaningful in Wyoming. Wyoming is a high elevation, headwaters state characterized by low precipitation (6-20” in the basins and plains and 20-70” in the mountains). Wyoming contains variable soils and parent materials and has minimal urban development; abundant federal public lands managed for multiple uses including rangeland, extractive industries and recreation; and highly allocated and diverted surface water rights.

Table 1-3: A summary of the parameters included in Harman et al. (2012) and rationale for their inclusion or exclusion from the WSQT

Functional Category	Parameter from the Function-Based Framework	Included in WSQT (Yes/No)	Rationale
Hydrology	Channel Forming Discharge	No	Not recommended by Function-Based Framework for showing functional lift/loss. *Include in design phase.
	Precipitation/Runoff Relationship	No	
	Flood Frequency	No	
	Flow Duration	No	Difficult to assess and develop reference curves.
	Reach Runoff**	Yes	See Chapter 2.
	Baseflow Alteration**	Yes	See Chapter 3.
Hydraulics	Flow Dynamics	No	Requires consideration of species-specific habitat requirements; may be developed in future versions of the WSQT.
	Groundwater/Surface Water Exchange	No	Difficult to assess and develop reference curves.
	Floodplain Connectivity	Yes	See Chapter 4.
Geomorphology	Channel Evolution	No	Considered in determination of restoration potential and selection of reference stream type.
	Sediment Transport Competency and Capacity	No	Not recommended by Function-Based Framework for showing functional lift/loss. *Highly recommended as part of the design process.
	Large Woody Debris	Yes	See Chapter 5.
	Bank Migration/Lateral Stability	Yes	See Chapter 6.
	Bed Material Characterization	Yes	See Chapter 7.
	Bed Form Diversity	Yes	See Chapter 8.
	Plan Form**	Yes	See Chapter 9.
	Riparian Vegetation	Yes	See Chapter 10.
Physicochemical	Organic Carbon	No	Difficult to develop reference curves.
	Bacteria**	No	Difficult to develop reference curves, WY water quality criteria are more related to human health than aquatic ecosystem function.
	Water Quality (Dissolved Oxygen, pH, and Conductivity)	No	Dissolved Oxygen is related to temperature and was not prioritized for inclusion in this version of the WSQT. Conductivity and pH were also not prioritized for inclusion.
	Water Quality (Temperature)	Yes	See Chapter 11.
	Nutrients	Yes	See Chapter 12.
Biology	Macrophyte Communities	No	Uncommon in stream mitigation monitoring.
	Microbial Communities	No	Uncommon in stream mitigation monitoring.
	Landscape Connectivity	No	Requires assessments beyond the project reach. Reference standards are typically species specific.
	Macroinvertebrate Communities	Yes	See Chapter 13.
	Fish Communities	Yes	See Chapter 14.

** The Function-Based Framework refers to Harman et al. (2012) which provides more information about these parameters and why they are recommended for the design phase and not for characterizing lift or loss.*

*** These parameters were not included in Harman et al. (2012) but were added later to this or other SQT's.*

1.7. Data Sources, Data Gaps, and Limitations

As described in Section 1.3, the development of reference curves implemented in the WSQT v1.0 sometimes relied on data from national and regional resource surveys and other available datasets. Some larger datasets were used to inform reference curves for multiple metrics, and those datasets are introduced here.

Compiled Geomorphic Reference Dataset:

Geomorphic reference datasets collected by the WY Game and Fish Department (WGFD) and the US Forest Service (USFS) were compiled for the WSQT. The dataset from WGFD was collected at approximately 20 sites throughout the mountainous regions of Wyoming between 2003 and 2006. The USFS dataset was collected from the Shoshone National Forest in the Middle Rockies region of Wyoming between 2003 and 2014 and consists of approximately 40 sites. The longitudinal profiles, cross sections, and bed material data from both datasets were reviewed as part of the quality assurance project plan. Sites that passed the review were included in the study. In August 2016 the WSTT revisited several reference sites from the WGFD dataset to apply the proposed WSQT methodology, verify the reference data from the dataset, and confirm bankfull determinations.

This dataset, referred to as the compiled geomorphic reference dataset in the remainder of this document, represents reference standard sites and was used to develop reference curves for metrics that describe floodplain connectivity, bed form diversity, and plan form parameters.

National Rivers and Streams Assessment (NRSA) Dataset:

The 2009 National Rivers and Streams Assessment dataset (NRSA; USEPA 2016), was reviewed to determine which metrics in the dataset could be used to inform the development of reference curves within the WSQT. The NRSA dataset includes a variety of metrics associated with LWD, plan form, and riparian vegetation. Data were compiled from sites in Wyoming and surrounding states and grouped into EPA Level III ecoregions. Specific attributes from the dataset are referred to in this document and descriptions are provided to relate NRSA attributes to stratification or metrics within the WSQT.

The NRSA dataset was used to develop reference curves for metrics that describe large woody debris and plan form parameters. NRSA datasets include sites across a range of condition, from reference standard to degraded. The WSQT Beta Version relied on this dataset to develop reference curves for riparian vegetation metrics, however changes in data collection methods in v1.0 reduced the relevancy of these datasets.

Colorado Natural Heritage Program (CNHP) Dataset:

The WSTT acquired a riparian vegetation dataset from the Colorado Natural Heritage Program (CNHP; Kittel et al. 1999). The purpose of this study was to characterize riparian community types across Colorado. While this dataset does not contain any data points from Wyoming, Colorado and Wyoming have overlapping ecoregions with similar riparian community assemblages. Data from the following ecoregions in Colorado were used in developing reference curves in the WSQT v1.0: Southern Rockies, Wyoming Basin, Colorado Plateau, Arizona/New Mexico Plateau, High Plains and Southwest Tablelands.

The CNHP dataset included condition ratings for all sites, scored as A, B, C and D. For developing WSQT reference curves, (A) sites were considered reference standard based on ecological conditions and (D) sites were considered degraded. Since the dataset was collected over multiple years and the methods were refined as the program progressed, the sites identified as B and C were removed from the analysis following discussions with CHNP. The dataset also identified whether sites were primarily herbaceous or woody, similar to the reference vegetation cover stratification used in the WSQT. There was no distinction between forested and scrub-shrub communities in the CNHP dataset.

The dataset consisted of species level cover data. Reference curves for woody vegetation cover and herbaceous vegetation cover were developed by summing absolute cover values categorized by stratum. Species in the dataset identified as graminoid or forb were grouped into the herbaceous stratum, shrub species cover values were combined with tree species cover values into a woody stratum. This dataset, referred to as the CNHP dataset in the rest of this document, was used to develop reference curves for riparian vegetation cover metrics.

Stratification by Ecoregion

Several metrics described in this document are stratified by ecoregion, but sample sizes within each EPA Level III ecoregion were variable. Wyoming includes three geophysical ecoregions: Central Rocky Mountains, Wyoming Intermountain Basins and Western Great Plains, which are similar to the EPA Level I ecoregions. To improve sample sizes, the WSTT decided to group ecoregion data into broader ecoregions that align with the three geophysical regions described above: Mountains, Basins and Plains. EPA Level III ecoregions were grouped into these broader ecoregion classifications, as shown in Table 1-4.

Table 1-4: EPA Level III Ecoregion Groupings Used for Data Analysis

Mountains	Basins	Plains
Southern Rockies	Wyoming Basin	High Plains
Middle Rockies	Colorado Plateau	Northwest Great Plains
Wasatch/Uinta Mountains	Arizona/New Mexico Plateau	Southwestern Tablelands

Data Gaps and Limitations:

We recognize there are limitations to the approaches outlined herein to develop reference curves. There is a large diversity of stream types in Wyoming, due to differences in landform, climate and geology, which in turn influence the hydrogeomorphic context of streams. We have tried to develop a tool that is broadly applicable across different hydrologic and geomorphic regimes through the stratification process and simple scoring, but recognize that there will always be limitations in this type of approach.

Rigorously accounting for regional variability among sites requires large datasets and statistically derived conclusions. These types of reference datasets were not always available for metrics included in this tool. Over time, it may be possible to revise certain reference curves as more data become available and the WSQT is used in ecoregions throughout the state. It is important to remember, however, that this tool is intended to compare pre- and post-project

conditions at a site. As such, it is not a stand-alone condition assessment; it is a “delta” (change measurement) tool. The difference between existing and future site conditions is the most important element. For example, a site may not attain reference standard condition, but it may show improvements that translate into an accrual of function.

Some metrics and their reference curves are applicable for the entire state. Others are stratified by ecoregion, valley type, stream type, reference community type, etc., with reference curves for each (See Appendix A). In some instances, data were not available for all regions or stream types, and the WSTT recommended not applying metrics in certain areas or types. Specific data gaps and limits to applicability are addressed within each metric description and are identified in Appendix A. Future versions of the tool will benefit from additional data collection and analysis.

In general, not all metrics are applicable or have been tested in ephemeral or intermittent streams, braided and anastomosed streams and beaver-influenced systems. Some parameters, including riparian vegetation, reach runoff, floodplain connectivity, lateral migration, bed material characterization, large woody debris and bedform diversity are applicable within intermittent and/or ephemeral systems, although the reference standards may not be representative because they have been developed from reference sites within perennial systems. In perennial and intermittent braided or anastomosing systems, riparian vegetation, reach runoff, flow alteration, floodplain connectivity (BHR only), bed material characterization, large woody debris, temperature, nutrients, macroinvertebrate (perennial only) and fish parameters are applicable, although reference curves have not been developed specifically for these streams. Additionally, modifications to sampling methods may be needed to accommodate these stream types.

Several metrics rely on bankfull depth to account for differences in stream size. Inaccuracies and/or inconsistencies in determining bankfull dimensions for a site will affect the way these metrics are characterized in the tool. Additional guidance on bankfull identification was added to the WSQT v1.0 User Manual in response to comments received during beta testing. For example, when possible, localized regional curves should be used to verify the bankfull determination. And, once a bankfull feature/stage has been determined, that feature/stage should be used for all future assessments at a project site to improve repeatability.

Some key limitations are highlighted here:

- This tool has largely been tested in single thread, perennial stream systems. Thus, some data collection methods and reference curves may have limited applicability in ephemeral or intermittent streams, stream-wetland complexes, and braided or anastomosing systems. Work is ongoing to consider how to broaden the applicability of the tool in these systems.
- A limited number of Rosgen stream types are used to stratify some metrics and parameters in the tool, and thus the tool is limited in capturing the geomorphic diversity of natural channel types, including multi-thread/anastomosing and natural canyon systems. The tool architecture allows for future changes to be made to accommodate other stratification approaches (e.g., hydrogeomorphic classification approaches). Additional data collection and analyses would be required, but future changes may be made to the tool to accommodate a broader range of stream types.
- Most metrics in the tool rely on wadeable data collection methods, and as such, the sampling efficiency and applicability of reference curves in larger rivers is unknown.
- By design, the WSQT is a reach scale, point-in-time tool. However, through routine monitoring, the WSQT can show trends or changes in condition that can be tied to channel evolution models.

- As a reach-based evaluation, the tool does not evaluate or consider secondary (indirect) effects in reaches upstream or downstream of the sampled reach. Additional analyses may be needed to evaluate these effects associated with a project. For example, restoration of aquatic organism passage can have an important indirect influence outside of a reach.
- The WSQT relies on structural measures and indicators instead of measuring stream processes directly. However, the WSTT has tried to select reasonable surrogates to characterize underlying processes. And, surrogates are only applied within a functional category. For example, bedform diversity is a surrogate for sediment transport. Both are in the geomorphology category. Bedform diversity is not used as a surrogate for biology, i.e, if the bedforms metrics have high index values (a good riffle/pool sequence) it is not assumed that the macroinvertebrate community (biology functional category) is also in good condition.
- The tool allows parameters/metrics to change between sites. Therefore, the same information may not always be collected across all sites. The condition score at one site may not be reflective of the same suite of parameters as a condition score at another site. Thus, the WSQT should be used to characterize condition changes at a specific site and not as part of an ambient monitoring program unless the same parameters and metrics are used consistently across all sites.
- The roll-up scoring for the WSQT has a simple approach to weighting, instead of relying on a more rigorous, statistically derived approach to calibration and scoring.
- Because multiple datasets and sources were used to develop reference curves, sample sizes and the level of uncertainty varies across metrics and across stratified reference curves within metrics. Additional testing and data collection will be beneficial to inform future versions of the tool.

1.8. Key Considerations

The WSQT and scientific support document have been developed to respond to specific regulatory and policy requirements and program needs. It has been tailored to meet the function-based approaches set forth in the 2008 Compensatory Mitigation Rule, as well as the needs of the WGFD and WDEQ for their stream monitoring and restoration programs. As such, there are several considerations that are critical in understanding the applicability of the tool:

- The parameters and metrics in the tool were, in part, selected due to their sensitivity in responding to reach-scale changes associated with the types of activities commonly encountered in the CWA 404 program and commonly used in stream restoration. These parameters do not comprehensively characterize all structural measures or processes that occur within a stream.
- The WSQT is designed to assess the same metrics at a site over time, thus providing information on the degree to which the condition of the system changes following impacts or restoration activities. We refer to the tool as a “delta” tool for this reason – it is intended to detect change at a site over time. Unless the same parameters and metrics are used across all sites, it would not be appropriate to compare scores across sites.
- The WSQT itself does not score or quantify watershed condition. Watershed condition reflects the external elements that influence functions within a project reach. Watershed condition is an important consideration when selecting a project site, determining the restoration potential of a site, and informing project design. The WSQT v1.0 User Manual describes how watershed condition is used to inform site selection and restoration potential.

- The WSQT is not a design tool. Many function-based parameters are critical to a successful restoration design but sit outside of the scope of the WSQT. For example, hydrologic characterization/modeling and sediment transport competency/capacity are critical to understanding what designs are appropriate, and to understand limitations in site potential. These analyses are often necessary in project design. The WSQT does not include these approaches, but instead measures the hydraulic, geomorphological and ecological responses or outcomes at a reach scale.

Chapter 2. Reach Runoff Parameter

Functional Category: Reach Hydrology & Hydraulics

Function-Based Parameter Summary:

While stream restoration project locations may be strategically selected as part of larger watershed plans, projects are limited in their ability to influence the hydrology that transports runoff from upstream in the watershed to the project reach. Land use changes indirectly influence watershed-scale processes, and these changes often occur away from the stream and are distributed throughout a watershed (Beechie et al. 2013). This function-based parameter is within the Reach Hydrology and Hydraulics category, which focuses on the hydrologic transport of water from the watershed that drains laterally to the reach as opposed to the water coming in from upstream of the project. The lateral drainage area is characterized within the WSQT using metrics and reference standards while the upstream contributing watershed is characterized by an analysis performed for site selection that considers the restoration potential of the reach in conjunction with larger watershed planning goals (see Section 1.2). Within the WSQT, catchment hydrology concerns are documented in the catchment assessment. The catchment assessment includes, but is not limited to, sediment supply and flow alteration and is described in greater detail in the WSQT v1.0 User Manual.

The reach runoff parameter assesses the infiltration and runoff processes of the land that drains directly into the stream reach from the lateral drainage area; it does not include the upstream catchment. This parameter characterizes the land use in the lateral drainage area adjacent to the project area, which could be altered at the reach or project scale. Land use practices in the lateral drainage area impact the amount of runoff and the pollutants entrained and transported to the receiving stream reach. For example, multiple studies have shown that increases in impervious cover are linked to decreased stream health (Scheuler et al., 2009), while agricultural practices can contribute sediment, nutrients, and other pollutants (USEPA, 2005). The lateral drainage area plays a role in multiple primary functions of healthy stream ecosystems described by Fischenich (2006): maintaining surface water storage processes, surface/subsurface water exchange, quality and quantity of sediments, necessary aquatic and riparian habitats, water and soil quality, and landscape pathways.

There are limitations of scale associated with this parameter in that the larger the contributing watershed area upstream, the less of an influence the lateral drainage has in maintaining stream functions within the project reach. In addition to the limitations of scale associated with the catchment, the size of the lateral drainage area can vary widely between projects. Larger stream reaches and unconfined valleys can have large lateral drainage areas and typical restoration practices are not likely to change the land use coefficient field value in the WSQT. However, larger lateral drainage areas are likely to have more concentrated flow points, and thus the metrics presented for this parameter are intended to be applied together. Including this parameter in the WSQT incentivizes stormwater management and land management practices on a reach-scale that can contribute to cumulative progress in a larger watershed.

The metrics in the WSQT characterize the land use and the number of anthropogenic concentrated flow points (storm water outfalls, ditches, etc.) in the lateral drainage area.

Metrics:

- Land Use Coefficient
- Concentrated Flow Points

2.1. Land Use Coefficient

Summary:

The WSQT uses an area weighted land use coefficient to numerically quantify the impact of various land uses on reach runoff (NRCS 1986). An area-weighted land use coefficient is calculated by delineating areas of different land uses within the lateral drainage area of a stream reach, assigning a land use coefficient to these areas and then calculating an area-weighted coefficient by multiplying the coefficient by the area and dividing the sum of the products by the total lateral drainage area (see WSQT v1.0 User Manual for detailed methods). Land use coefficients are based on curve numbers (CN) developed by the NRCS in Urban Hydrology for Small Watersheds (NRCS 1986), commonly known as the TR-55. Curve number values presented in TR-55 are determined based on soil type, land use, and surface condition. Higher CN values, nearer 100, indicate more runoff potential and lower values, nearer 0, indicate less runoff potential. To focus on land use change rather than infiltration capacity of soils, land use coefficients used in the WSQT are derived solely from CN within hydrologic soil group B[1]. These land use coefficients are not intended to predict changes in runoff, but to serve as an indicator of land use change and the potential for generating runoff in the lateral drainage area.

Reference Curve Development:

TR-55 provides land use coefficients for various natural, agricultural and urban land uses across a range of condition. For example, woods that are protected from grazing and have litter and brush covering the soil are considered good condition and the land use coefficient is 55. For comparison, the land use coefficient for woods in poor condition, i.e. where forest litter, small trees, and brush are destroyed by heavy grazing or regular burning, is 66. Urban land uses have higher land use coefficients as the percent of impervious surfaces increases. For example, commercial and business districts have a land use coefficient of 92 while residential districts with 1/4 acre lots have a land use coefficient of 75.

Land use coefficients for natural land cover types in good condition are always less than 68 and often less than 60, while land use coefficients for agricultural lands typically range from 70 to 80. The land use coefficients for urban land uses trends higher than agricultural lands depending on the percent of impervious cover associated with various cover type descriptions. Therefore, as the lateral drainage area is cultivated or developed, an area weighted land use coefficient will increase.

To develop a reference curve associated with land use changes, land use coefficients that correspond to natural land cover (40-68) were considered to represent a functioning range of index values (0.70-1.00), with the lower coefficient (40) assigned an index value of 1.00. The minimum index value 0.00 equated to a land use coefficient of 80, as this value indicates a significant amount of developed lands within the lateral drainage area, and this level of land use

[1] Soils classified as hydrologic soil group B have moderate infiltration rates.

change likely contributes to substantially altered reach-scale hydrology. Threshold values are presented in Table 2-1.

Stratification by reference riparian vegetation cover type (woody or herbaceous) was considered since herbaceous communities have less roughness and higher runoff potential than forested communities, but this stratification was not used since the reference riparian vegetation community rarely extends to the entire drainage area and thus, would not be representative of the area-weighted land use coefficient for the entire lateral drainage area. A broken-linear curve was applied for this metric (Figure 2-1), and is steeper in the not functioning and functioning-at-risk range of scoring than in the functioning range, allowing for a broader range of land use coefficients within the functioning range to account for natural variability.

Table 2-1: Threshold Values for Land Use Coefficients

Index Value	Field Value
1.00	40
0.70	68
0.00	80

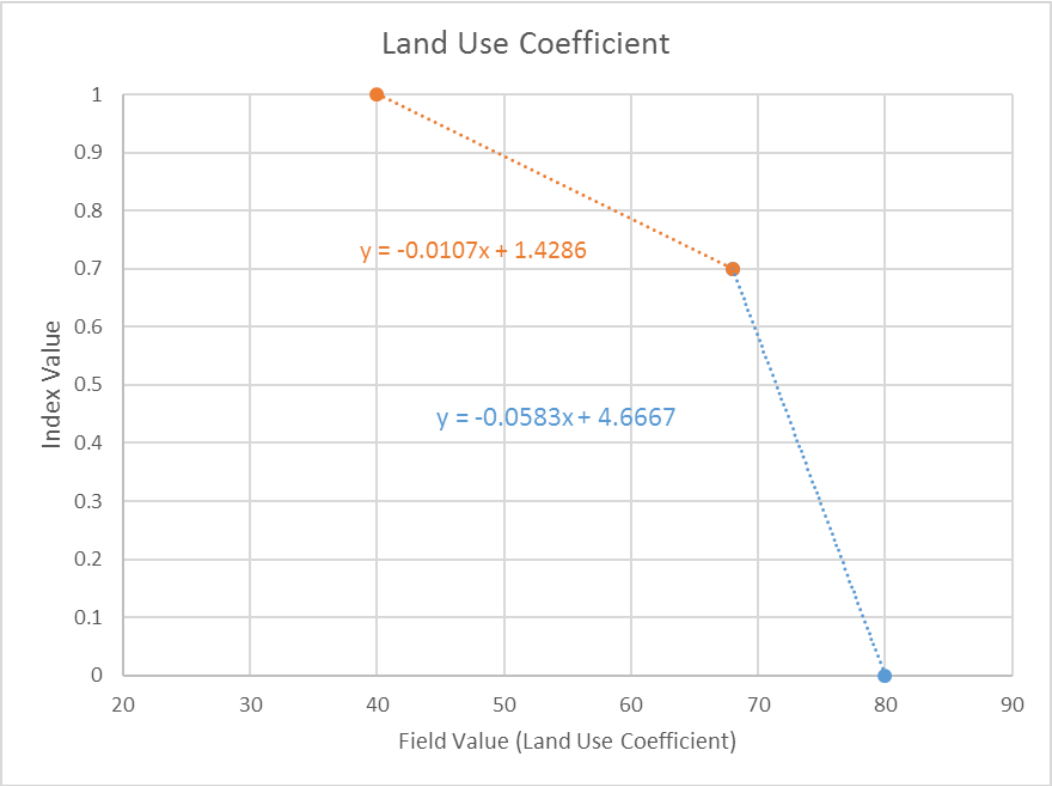

Figure 2-1: Reference Curve for Land Use Coefficient

Limitations and Data Gaps:

The land use coefficients for this metric are derived from curve numbers but assume all soils are moderately drained. Therefore, the metric does not account for the variation in infiltration capacity, impermeable layer depth, or other characteristics important to estimating runoff volumes. Additionally, curve numbers are used to predict runoff volumes and therefore their application as land use coefficients does not accurately account for relative pollution loads coming from different land uses.

There are limitations of scale associated with this metric as the size of the project easement or area compared to the size of the lateral drainage area will influence how much index scores may change in response to land use changes in the project area. . Reaches with larger lateral drainage areas would need to acquire and revegetate more land to achieve a similar amount of lift as a project with a smaller lateral drainage area. Similarly, there are limitations related to the size of the project easement or area compared to the size of the upstream catchment. For example, a reach located far downstream from the headwaters may be more affected by hydrologic changes occurring upstream than from land use change in the lateral drainage area. Alternatively, improving land use condition in small streams near the headwaters may have a greater relative effect. These limitations could be addressed through stratification and development of additional reference curves. Considering relative watershed location (e.g., the proportion of land area within the lateral drainage area compared with the entire watershed area) could account for the relative impact of direct drainage to the channel vs. in-channel delivery from upstream.

Stratification based on natural land use types may also improve this metric. Natural land cover varies in runoff and infiltration potential. For example, natural grasslands function differently than pinyon-juniper forests, with different curve numbers and land use coefficients, but both may represent a pristine or reference standard condition. Also, this metric may be less sensitive to changes between natural land cover types and developed land uses where natural land use coefficients are more similar to developed land use types. Stratification would better account for these differences.

This metric has received limited beta testing and would benefit from additional application and testing in Wyoming. It would also benefit from sensitivity testing and comparison to other indicators of altered stream processes, including percent impervious surface, particularly in areas with more urban development. This is important since even small amounts of impervious cover (e.g., 10%) in a watershed can result in significant loss of stream function (Booth and Jackson 1997, Schueler et al. 2009). As the WSQT is tested and applied, this metric may be updated.

2.2. Concentrated Flow Points

Summary:

This metric assesses the number of concentrated flow points that enter the project reach per 1,000 linear feet of stream. For this method, concentrated flow points are defined as erosional or constructed features (e.g., concrete swales, rills, gullies, ditches, road cuts or other conveyances) created by anthropogenic modifications on the landscape that alter or

concentrate runoff into the stream. These types of features can be caused by agricultural practices that result in irrigation return flow, or cut and fill activities associated with roads or building sites that intercept water otherwise heading downslope as throughflow or groundwater and bring it to the surface. Alterations in runoff processes associated with land use changes are common, particularly due to changes in or removal of vegetation; increased impervious surface area; soil compaction and decreased infiltration; and interception of subsurface flows and routing to streams (Beechie et al. 2013).

Overland flow typically erodes soils relatively slowly through sheet flow; however, anthropogenic impacts can lead to concentrated flows that erode soils quickly, transporting water and sediment into receiving stream channels (Al-Hamdan et al. 2013). Three primary drivers that cause sheet flow to transition to concentrated flow include discharge, bare soil fraction, and slope angle (Al-Hamdan et al. 2013). Anthropogenic changes to runoff characteristics often create new conveyances, where flows are concentrated and routed more quickly to streams. Channels are also constructed to drain the landscape, e.g. agricultural ditches or concrete swales connecting parking lots to stream channels and gutter systems to route rain water away from structures. Even hiking or game trails can intercept and concentrate runoff.

Stream restoration projects can reduce concentrated flow entering the stream by dispersing flow in the floodplain, increasing surface roughness, regrading to flatten slopes, removing roads and ditches, and restoring riparian vegetation. Development can negatively impact streams by creating new concentrated flow points such as stormwater outfalls. Stormwater best management practices can be used to address these outfalls, enhance infiltration and reduce outfall velocity.

<u>Reference Curve Development:</u>

The threshold values for this metric were based on best professional judgement. We could not find literature quantifying relationships between the number of concentrated flow points and stream stability or aquatic life. However, there is a clear negative relationship between concentrated flows and degradation of stream stability and aquatic life (Hammer 1972). The WSTT agreed the absence of anthropogenic concentrated flow points reflected a reference standard, and the presence of one or more concentrated flow points per 1,000ft no longer reflected a reference standard condition. Based on this logic, the threshold values shown in Table 2-2 were created, and a linear curve was fit to these values (Figure 2-2).

Table 2-2: Threshold Values for Concentrated Flow Points

Index Value	Field Value
1.00	0
0.69	1

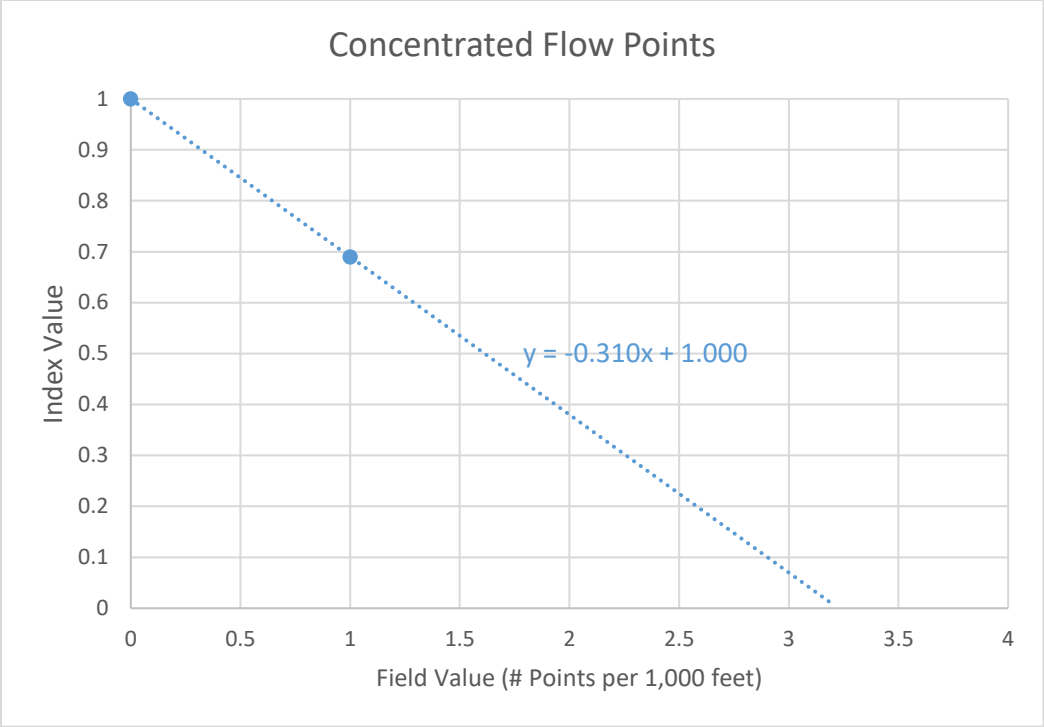

Figure 2-2: Reference Curve for Concentrated Flow Points

<u>Limitations and Data Gaps:</u>

This metric was developed for use in the North Carolina SQT, and was incorporated into the WSQT as well. Because it is a relatively new metric, it will need additional testing and review as it is applied to project sites, particularly in degraded stream reaches and urban areas.

The metric does not consider the type or size of the concentrated flow points, only the number. Considering the cumulative volume of runoff water produced by the flow points, differences in their type, or their contributing drainage area relative to the lateral drainage area would make this a more meaningful metric. For example, one large concentrated flow point may deliver more water (with lower quality) than three or more small conveyances. Some SQT teams around the country are exploring ways to revise and improve the concentrated flow points metric, e.g., to focus on volume rather than number. As the work progresses, the WSQT may be updated.

There are other limitations of using a simple count per linear foot of stream. For example, a practitioner could be incentivized to take three concentrated flow points, merge them together, and create one larger flow point, which may not result in any actual improvements in the stream condition. Alternatively, if the project includes restoration of natural sinuosity but does not reduce the number of concentrated flow points, the tool would show lift solely as a result of the increased channel length, rather than an attempt by the practitioner to reduce the actual number of concentrated flow points. These types of examples will need to be dealt with on the policy side until the metric is modified to address these types of issues.

Chapter 3. Flow Alteration Parameter

Functional Category: Reach Hydrology & Hydraulics

Function-based Parameter Summary:

Dams, water allocation, and effluent discharges can play a significant role in altering the hydrology at a project reach. Altering the magnitude, duration, frequency, timing, and/or rate-of-change of the flow regime can impact geomorphic and ecological functions of the stream (Poff 1997). The resulting effect on a stream ecosystem varies depending on what aspects of the flow regime are modified and the significance of the flow alteration. In a literature review characterizing ecological responses to altered flow regimes, Poff and Zimmerman (2010) found that both macroinvertebrate and fish populations declined with increases and decreases in flow magnitude, yet much of the published literature focused on large flow alterations, i.e., greater than 50% change from natural conditions. The authors noted that the study was “not able to extract any robust statistical relationships between the size of the flow alteration and the ecological responses,” which hints at the difficulty in assigning reference curves for flow alteration metrics.

Flow alteration can have wide-ranging impacts throughout a watershed and some impacts from altered hydrology can take decades or longer to resolve in the system. Understanding historical and current alterations in the flow regime are critical to developing a successful restoration project design, and appropriate hydrologic analyses should be undertaken (Roni and Beechie 2013). This parameter does not substitute for these analyses, nor does it fully characterize the hydrologic condition of a reach. Instead, this parameter characterizes potential changes in condition associated with project-specific changes in hydrology at the reach scale. Examples of project-specific changes in hydrology may include changes in dam operations, increases or decrease in municipal or agricultural water use, and acquisition or preservation of water rights for environmental flows.

The WSQT only includes one metric to evaluate this parameter, a baseflow alteration metric. Due to the effort associated with modeling hydrology, this parameter would likely be applied only in projects where modifications to the baseflow regime are proposed. Alterations in baseflow hydrology include additional withdrawal or augmentation, exchanges or operational changes at a dam or diversion site. While these changes may extend beyond the boundaries of the identified project area, the WSQT currently only addresses the identified project reach. The WSQT does not currently include sufficient metrics to characterize changes beyond baseflow, which may be relevant for projects that propose alteration of other aspects of the flow regime. This is and will continue to be an active area of research for the WSQT.

Additional metrics and approaches are being evaluated for inclusion in the tool and may be incorporated into subsequent versions. The WSTT recognizes that flow alteration is ubiquitous in streams in Wyoming and the western United States, and thus is an important consideration when determining the watershed condition and restoration potential of a site. For projects that do not propose to alter the flow regime, hydrologic modeling and ecological flow analyses will be outside the scope of the WSQT, but should still be considered in project planning and design.

Metric:

- Observed Q Low/ Expected Q Low

3.1. Observed Q Low/ Expected Q Low

Summary:

This metric compares the observed baseflow discharge in the channel to the expected baseflow discharge. Baseflow, or low flow, is defined here as the monthly average flow for August. August and September represent a biologically critical period when flows are likely to be close to their annual minimum and potentially constrained by human water use (Binns 1982). Winter low flow periods are also critical periods for many species, however, August was selected to capture the effects associated with late-summer irrigation withdrawals. In Wyoming, rivers with strong snowmelt signature may not reach baseflow until fall or winter especially during high snowpack years. Highly depleted streams, however, will reach annual lows often during August. This metric requires a reference gage to be identified for the project reach that has similar runoff characteristics to the project site; an assessment of reference gages should consider geology, elevation, and precipitation (Lowham et al. 2009).

The field value for this metric is the observed over expected average low flow for August. Data collection and analysis methods are outlined in the WSQT v1.0 User Manual. The expected value is determined using regional regression curves to predict the average annual discharge in the reach (Lowham 1988; Miselis et al. 1999). A gage analysis is then used to scale the gage data and predict the average low flow for August at the project reach following procedures typically used to perform instream flow studies. Where observed low flow is measured during one season, the value should be accompanied with a gage analysis that determines whether the monitoring year was wet, dry or average.

Reference Curve Development:

In 2010, Poff and Zimmerman published a literature review characterizing ecological responses to altered flow regimes. The authors found that both macroinvertebrate and fish populations declined with increases and decreases in flow magnitude, yet much of the published literature focused on large flow alterations, i.e., greater than 50% change from natural conditions. The study was “not able to extract any robust statistical relationships between the size of the flow alteration and the ecological responses,” which hints at the difficulty in assigning reference curves for flow alteration metrics.

Multiple efforts have been undertaken to determine environmental flow targets, including the Ecological Limits of Hydrologic Alteration (ELOHA) framework, which often rely on extensive hydrologic modeling and stakeholder processes to develop thresholds or benchmarks (Poff et al. 2010). As a starting point for considering flow standards, Richter (2012) proposed a presumptive flow standard for environmental flow protection (Table 3-1) which is applied to daily flow values and requires a hydrologic model to predict daily natural flows (undepleted and unregulated).

Table 3-1: Presumptive Flow Standard. Adapted from Richter (2012)

Deviation from Natural Daily Flow (+/-)	Level of Protection	Description
≤ 10%	High	The natural structure and function of the riverine ecosystem will be maintained with minimal changes.
11 – 20%	Moderate	There may be measurable changes in structure and minimal changes in ecosystem functions.
> 20%	N/A	Likely result in moderate to major changes in natural structure and ecosystem functions, with greater risk associated with greater levels of alteration in daily flows.

In a study on flow alteration at stream gage sites throughout the US, Carlisle et al. (2010) observed biological impairment in some sites with hydrologic alteration of 0-25% and in an increasing percentage of sites beyond 25% hydrologic alteration. Zorn et al. (2008) predicted that adverse resource impacts would occur on most types of rivers with withdrawals greater than 17–25% of index flow. In Wyoming's Habitat Quality Index, Binns (1982) considered the following criteria to determine habitat quality:

- late summer streamflow < 10% of average daily flow is inadequate to support trout,
- 10-15% of average daily flow has very limited potential to support trout,
- 16-25% of average daily flow has limited potential to support trout,
- 26-55% of average daily flow has moderate potential to support trout
- and >55% of average daily flow is completely adequate to support trout.

In developing reference curves, the WSTT tried to reconcile limitations associated with data availability and natural variability of flow regimes. Recognizing the level of effort that is typically undertaken to develop flow standards, the WSTT decided to conservatively rely on the Richter (2012) presumptive standard and the Binns (1982) reference to define threshold values. The following criteria were used to develop reference curves and index values (shown in Table 3-2):

- The maximum index score (1.00) equates to a natural, or expected average low flow value for August, or an O/E value of 1.0. In line with the presumptive standard, 90% to 110% of the expected average low flow for August would yield an index value of 1.00 in the WSQT.
- The minimum index score (0.00) equates to a baseflow loss of greater than 74% to align with the threshold between a limited and moderate capacity to support trout (O/E of 0.26).
- A minimum index score of 0.00 was also equated to a baseflow augmentation of 100% (O/E value of 2.0) based on best professional judgement, recognizing that higher than normal low flows may impact sediment transport, riparian vegetation recruitment and succession, and aquatic biological processes.

A linear curve was fit between the defined values. With this curve, the functioning range of scores aligns with the presumptive standard by Richter (2012). The slope of the reference curve differed between withdrawal and augmentation, as it was assumed that during the low flow season, adding water to a reach is not as likely to limit ecological functioning as quickly as removing water from a reach.

Table 3-2: Threshold Values for Observed Q Low/ Expected Q Low (O/E)

Index Value	Field Value
1.00	0.90-1.10
0.00	0.26, 2.00

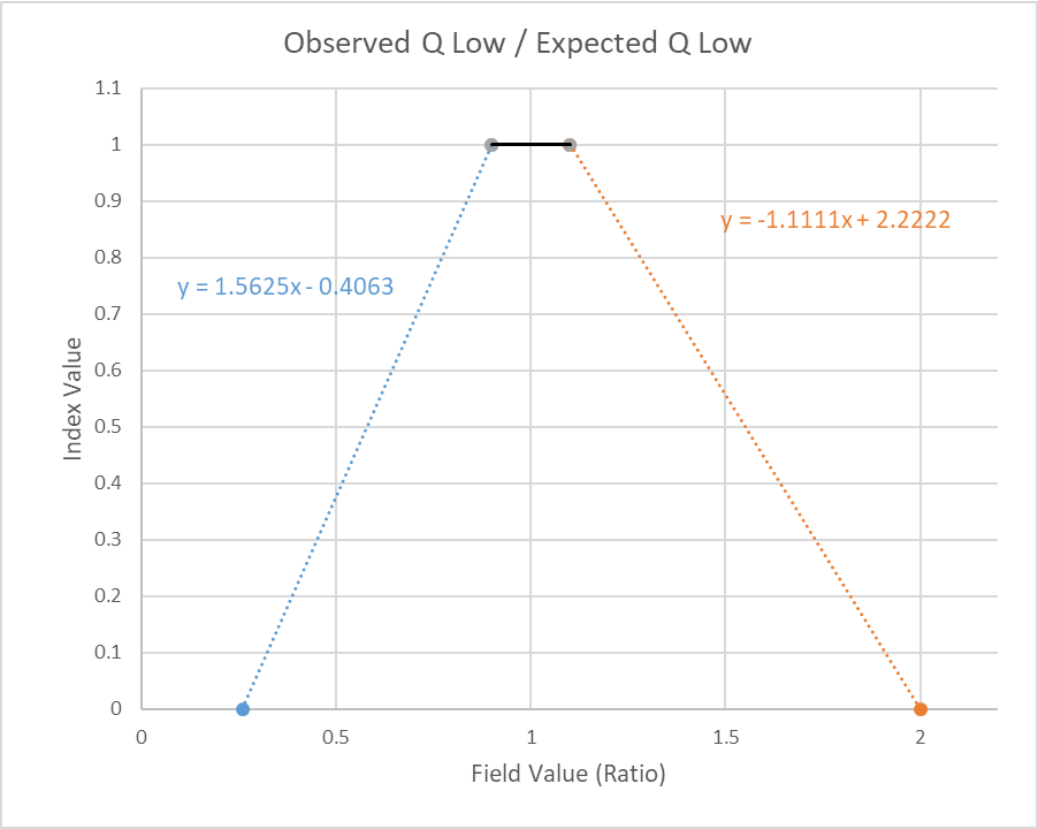

Figure 3-1: Observed Q Low/ Expected Q Low Reference Curve

<u>Limitations and Data Gaps:</u>

This metric can only be applied where there is gage data available to scale field values and calculate an expected average low flow for August. A prediction based on a regional relationship may vary substantially, depending on the standard error of the regression equation (Lowham 1988, Miselis 1999). Additional error may result from the point measurements as well. Testing is needed to determine whether adjustments to the reference curves are needed to account for the model precision.

As noted in the literature referenced in the discussion above, the reference curve may not be sufficient to support geomorphic, physicochemical and biology functioning in all streams. Zorn (2008) noted that some rivers in Michigan are more sensitive to withdrawals than others and Richter (2012) notes that the presumptive standard may not be sufficient to protect ecological values in smaller or intermittent streams.

The metric is also limited in that it characterizes one aspect of the flow regime in a single month of the year. Other aspects of the flow regime are important for function (Poff et al. 1997). For example, base flow hydroperiod and patterns during the winter can have important ramifications for the biological community, especially fish populations. A more comprehensive approach to evaluating flow alteration is likely needed to adequately characterize other aspects of the flow regime (Annear et al. 2004, Poff et al. 2010). The WSTT recognizes this as a major limitation and is evaluating other options that better characterize and incorporate flow alteration into future versions of the WSQT.

Chapter 4. Floodplain Connectivity Parameter

Functional Category: Reach Hydrology & Hydraulics

Function-based Parameter Summary**:**

Floodplain connectivity is one of the most important function-based parameters for stream restoration work (Fischenich 2006), because it is a driver for many geomorphic and ecological functions (Wohl 2004). The floodplain of a stream is the area commonly inundated during high flows or floods. Harman et al. (2012) provide detailed definitions and examples of floodplains and flood prone areas. For example, floodplains consist of alluvium are associated with meandering streams in alluvial valleys. Flood prone areas and bankfull benches are narrower than floodplains and exist in confined or colluvial valleys. Flood prone areas and bankfull benches are flat depositional features that provide some energy dissipation for higher flows. Floodplains, bankfull benches and flood prone areas are assessed as floodplain connectivity in the WSQT.

When a channel is connected to its floodplain, flood flows can inundate the floodplain and spread out across the landscape while in-channel velocities can maintain bed forms without excessive erosion. While it is a common perception that a straight and deep channel can move flood waters quickly downstream, channelization often displaces flooding and increases flood damage downstream of the channelization (Schoof 1980). Channels that are not connected to their floodplain lose the capacity to store water and sediment in the floodplain during large storm or snowmelt events. The functional loss associated with channelization and berm or levee construction is not limited to displaced flooding, but can also lead to loss of bedform diversity, downcutting and incision, increased erosion, and loss of fish species and biomass (Darby and Thornes 1992; Hupp 1992; Kroes and Hupp 2010). Severely incised channels can also lower the local water table, draining riparian wetlands or otherwise impacting the local riparian community (Harman et al. 2012). In a comparison between an incised stream and a similar, non-incised stream, the incised stream had significantly higher turbidity, solids, total nitrogen and phosphorous and chlorophyll concentrations, and lower fish diversity and biomass than the non-incised stream (Shields et al. 2010).

The SFPF (Harman et al. 2012) describes three measurement methods for the floodplain connectivity parameter: bank height ratio (BHR), entrenchment ratio (ER), and stage-discharge relationships. BHR is a measure of channel incision and the relative frequency that flood flows could reach the floodplain, while ER estimates the lateral extent of floodplain inundation (Rosgen 1996). Together these metrics characterize floodplain connectivity, but do not assess whether flood flows are observed. Stage-discharge relationships apply a hydrologic model to predict various discharges (e.g. the 2-year, 5-year, 10-year return interval events), and channel dimensions are used to predict the water stage, or elevation, associated with each discharge value. The value obtained from the stage-discharge relationship would be the flow (Q) contained within the banks of the channel. If that flow is a large and infrequent flood event, then the channel is not connected to its floodplain. For example, a channel that conveys a 5-year flood event is not well-connected to the floodplain. Typical return intervals for bankfull discharge range from 1.1 to 2.0-year return intervals (Mulvihill and Baldigo 2012; Moody et al. 2003; Emmert 2004).

As two-dimensional modeling becomes more used in stream restoration projects, hydraulic parameters like Froude and Reynolds numbers may be added to SQTs. For example, studies have shown Froude and Reynolds number preferences by wild and hatchery-raised cutthroat trout, which can be used as an aide in developing reference standards (Bates 2000).

The BHR and ER metrics were selected for use in the tool over stage-discharge relationships, because they are physically based (practitioners and regulators can measure in the field), and rely on a bankfull indicator or regional curve. A gage station or model is not required.

Metrics:

- Bank Height Ratio
- Entrenchment Ratio

4.1. Bank Height Ratio

Summary:

The bank height ratio (BHR) is a measure of channel incision and indicates whether a stream is connected to an active floodplain or bankfull bench. The BHR is the depth from the top of the low bank to the thalweg divided by the depth from the bankfull elevation to the thalweg (Rosgen 1996). For the WSQT, BHR is measured at every riffle in the sampling reach, and a weighted BHR is then calculated from these measurements. Methods for data collection and metric calculation are in the WSQT v1.0 User Manual.

Because of their lack in floodplain connectivity, incised streams contain flows of greater recurrence intervals than other, non-incised channels within the same region, which dissipate high flows across the floodplain (Simon and Rinaldi 2006). The BHR represents the magnitude of flow required for flows to access the adjacent floodplain (in alluvial valleys) or bankfull bench (in colluvial valleys). A BHR of 1.0 means that all flows greater than bankfull are spreading onto a floodplain or bankfull bench (Rosgen 2009). A BHR of 2.0 means that it takes a stage of two times the bankfull stage to access the former floodplain and the stream is highly incised or disconnected from its former floodplain. A high BHR correlates to a high return interval for flows leaving the channel. For example, a BHR of 2.0 can correspond to a 50-year return interval and a BHR of 1.0 will likely correspond to a 1 to 2-year return interval (Rosgen, 1996; Dunne and Leopold, 1978). Bank height ratios increase as the streambed lowers or degrades. Degradation is often caused by head cutting (bed erosion) processes which further increase the BHR and result in larger floods being contained in the channel and decreased floodplain connectivity. Sullivan and Watzin (2009) found that measurements of bank height ratio, as an indicator of floodplain connectivity, were significantly correlated to fish assemblage diversity. The BHR measurement method is a physically based method (practitioners and regulators can directly make the measurement in the field at any given time), and can be made in any stream with a bankfull indicator or regional curve.

Reference Curve Development:

The BHR metric was developed by Rosgen (2009) as a measure of channel incision as shown in Table 4-1. Harman, et al. (2012) translated channel incision descriptions from Rosgen (2009) into functioning, functioning-at-risk, and not functioning categories that indicate the degree of incision and the relative functional capacity of incised streams (Table 4-1).

Table 4-1: Bank Height Ratio Categories

Channel incision descriptions by Rosgen (2009)		Performance standards by Harman et al. (2012)	
BHR	**Degree of Channel Incision**	**BHR**	**Functional Capacity**
1.0 – 1.1	Stable	1.0 – 1.2	Functioning
1.1 – 1.3	Slightly Incised		
1.3 – 1.5	Moderately Incised	1.3 – 1.5	Functioning-At-Risk
1.5 – 2.0	Deeply Incised	> 1.5	Not Functioning

The BHR categories from Rosgen (2009) and Harman et. al (2012) were evaluated for Wyoming using the compiled geomorphic reference dataset described in Section 1.5. The compiled geomorphic reference dataset consists of 61 sites that report BHR (Table 4-2). Because bank height ratio was used as a quality assurance measure in compiling the dataset, sites that would be considered deeply incised (BHR greater than 1.5) were not included in the reference dataset.

Stratification by stream size is built into the metric by using the bankfull depth as the denominator. Bankfull depth varies throughout the country due to differences in climate and runoff characteristics, however, there are predictable, documented relationships that predict bankfull dimensions for streams in the same physiographic or hydrologic region (Dunne and Leopold 1978, Blackburn-Lynch et al. 2017). Stratification by valley type was considered to address differences in floodplains, e.g., between alluvial and colluvial valleys. However, because this metric focuses on the ability of flood flows to access areas outside the channel and not the extent of floodplain inundation, the decision was made not to stratify by valley type.

Table 4-2: Statistics for BHR from the Compiled Geomorphic Reference Dataset

Statistic	BHR
Number of Sites (n)	54
Average	1.09
Standard Deviation	0.11
Minimum	1.00
25th Percentile	1.00
Median	1.00
75th Percentile	1.19
Maximum	1.50

Most sites in the compiled geomorphic reference dataset had BHR less than 1.2 (Table 4-2). Because the 75th percentile from the dataset aligned well with the criteria identified in Table 4-1,

a BHR of less than 1.2 was used to define the lower threshold for the functioning range of index values (Table 4-3). Sites within this dataset with BHRs greater than 1.2 were considered functioning-at-risk for BHR, since they fell outside the 75th percentile of reference standard sites, even though the reach may reflect functioning condition for other metrics.

A threshold of 1.5 was used to differentiate index values within the functioning-at-risk and non-functioning ranges. BHRs of greater than 1.5 were considered non-functioning, consistent with the supporting literature classifying these as deeply incised channels with a greater likelihood of vertical instability (Rosgen 2009). Deeply incised streams (e.g., BHR > 1.7) provide extremely rare or no floodplain connectivity. A channel that contains any significant flood event, e.g., a 10-year or 25-year recurrence interval, is likely to experience significant erosion during a large precipitation event and transport water and sediment downstream instead of dispersing them across the floodplain.

The thresholds identified in Table 4-3 were plotted and a best-fit line was derived to provide a single equation to calculate index values from field values (Figure 4-1).

Table 4-3: Threshold Values for Bank Height Ratio

Index Value	Field Value
1.00	1.0
0.70	1.2
0.30	1.5

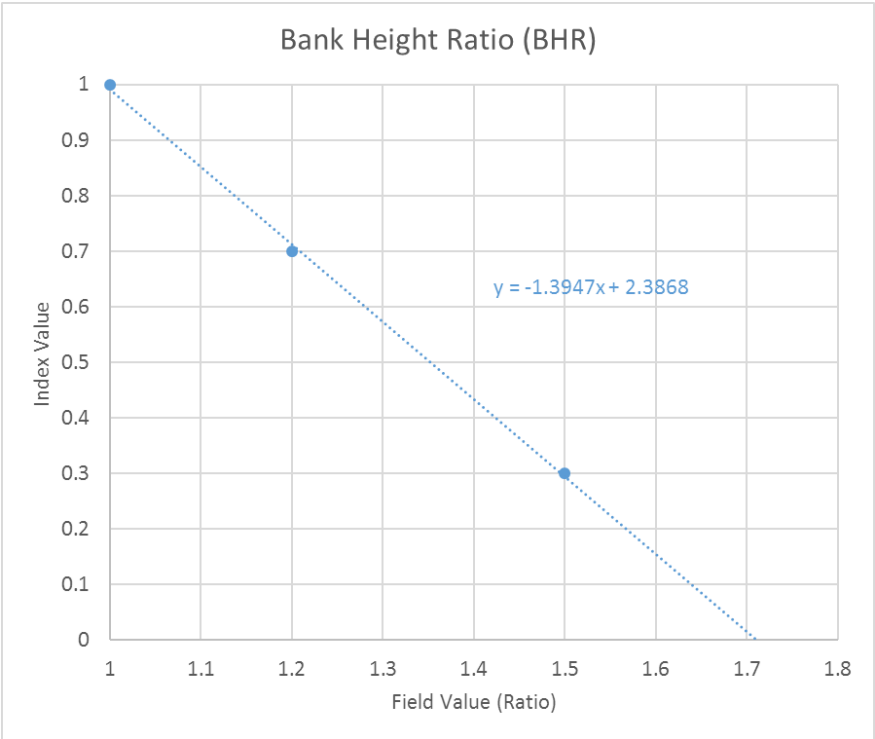

Figure 4-1: Bank Height Ratio Reference Curve

Limitations and Data Gaps:

If bankfull dimensions are not accurately determined for a site, then the bank height ratio will not accurately represent the incision processes. When possible, localized regional curves and flood frequency analysis should be used to verify the field indicators of bankfull. Additional information on verifying bankfull information was added to the WSQT v1.0 User Manual in response to comments received during beta testing.

Bank height ratio often relates to the stage and corresponding return interval where water leaves the channel and inundates a floodplain or terrace. By contrast, in watersheds where the hydrology has been severely altered, the return interval associated with a floodplain surface may dramatically increase (or decrease). For example, the return interval may increase from 1.5 years (the average for a bankfull flow) to 5 years downstream from new impoundments that reduce the frequency (increase the return interval) of flood events. This process converts the active floodplain to a terrace. The BHR will not detect this change initially because the floodplain appears to be intact; and the stream does not appear to be incised because the depth from the streambed to the top of the bank has not changed, though eventually a smaller channel will develop within the former channel as reduced flood flows fail to scour riparian areas and transport less bed sediment. In the scenarios where flow alteration rather than incision has reduced floodplain connectivity, a return interval approach would be preferable. The WSTT is currently evaluating reference curves that could be used to characterize the return interval of flood flows, and this type of metric may be incorporated into a future version of the tool.

4.2. Entrenchment Ratio

Summary:

The entrenchment ratio (ER) is a ratio of the flood prone area width divided by the bankfull riffle width. The flood prone area width is the width of the floodplain at a depth that is twice the bankfull maximum riffle depth (Rosgen 2009). Instructions for collecting and calculating the field value for this metric are provided in the WSQT v1.0 User Manual.

ER estimates the lateral extent of floodplain inundation for a large and infrequent flood event. A stream is considered entrenched when flooding is horizontally confined, i.e. the channel floodplain is narrow compared to other channels in a similar physiographic setting. Because ERs vary naturally by valley shape, they have been used as a primary metric in differentiating stream types (Rosgen 1996). ER can be a useful indicator of functional capacity as many anthropogenic alterations (e.g. levees) constrict the natural extent of floodplains and decrease floodplain connectivity. The ER metric is physically based (i.e., can be measured in the field at any time) method, and can be assessed in any stream with a bankfull indicator or regional curve.

Reference Curve Development:

Entrenchment Ratio (ER) is a primary metric in determining the Rosgen stream type: entrenched stream types (A, G and F streams) have ER values less than 1.4 ±0.2; slightly entrenched stream types (E and C stream types) have ER values greater than 2.2 ±0.2, and those in between are considered moderately entrenched (B stream types; Rosgen 1996). The values used to delineate between stream types were empirically based on data collected by

Rosgen and by modeling a bankfull discharge and 50-year recurrence interval flood through typical cross sections representing various stream types. The ratio of the depth of the 50-year flood to the bankfull depth ranged from 1.3 to 2.7 for all stream types except Da's, with less confined streams like E's having lower ratios (the larger the horizontal area floodwaters can occupy, the lower the difference in stage between a small flood and a large one). A "typical" ratio of 2.0 was selected to calculate the elevation of the flood prone width for all stream types, as a generalized comparison of confinement (Rosgen 1996).

Harman et al. (2012) translated the adjective descriptions of entrenchment used by Rosgen (1996) into functioning, functioning-at-risk, and not functioning categories as shown in Table 4-4 after considering the differences among stream types. The performance standards were based on the stream type delineations listed above and the ±0.2 that "allows for the continuum of channel form" (Rosgen 1996).

Table 4-4: Entrenchment Ratio Performance Standards from Harman et al. (2012)

ER for C and E Stream Types	ER for B and Bc Stream Types	Functional Capacity
> 2.2	> 1.4	Functioning
2.0 – 2.2	1.2 – 1.4	Functioning-At-Risk
< 2.0	< 1.2	Not Functioning

The criteria proposed by Harman et al. (2012) were evaluated for Wyoming using the compiled geomorphic reference dataset described in Section 1.5 of this manual. The compiled geomorphic reference dataset consists of 61 sites that report ER. Of these sites, three were identified as outliers and removed from the analysis and three sites were classified as F channels and were also removed from the analysis. The statistics for ER stratified by stream type are provided in Table 4-5 and Figure 4-2.

Table 4-5: Statistics for ER from the Compiled Geomorphic Reference Dataset

Statistic	ER by Stream Type			
	B	C	E	C & E
Number of Sites (n)	22	25	8	33
Average	1.8	3.6	5.5	4.0
Standard Deviation	0.5	1.3	3.3	2.1
Minimum	1.2	1.5	2.3	1.5
25th Percentile	1.5	2.8	3.3	2.8
Median	1.8	3.3	5.0	3.5
75th Percentile	2.2	4.2	6.5	4.6
Maximum	2.8	6.9	12.5	12.5

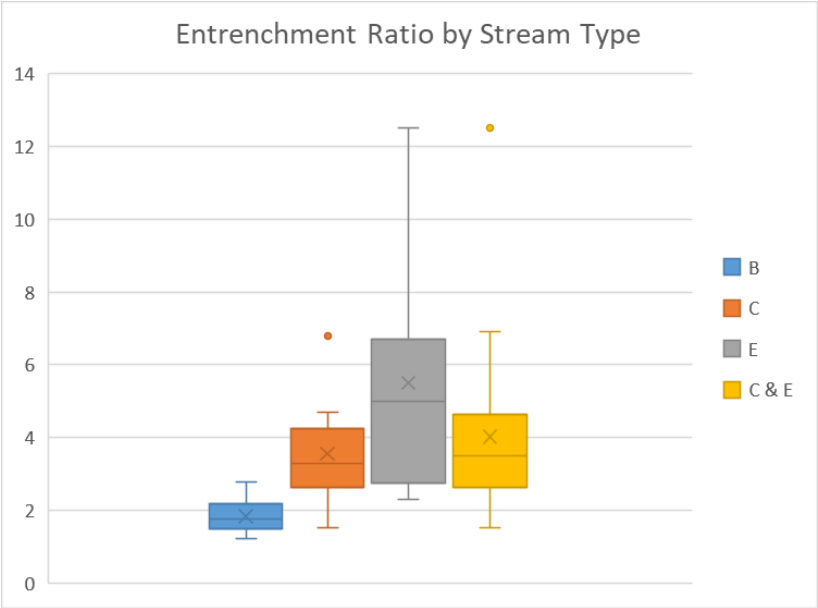

Figure 4-2: Box Plots for ER from the Compiled Geomorphic Reference Dataset

Bankfull width was used as a denominator of this metric, and thus stratification by stream size was not needed. *S*caling by bankfull width accounts for the differences in stream size that may otherwise be relevant in determining flood prone width. Bankfull dimensions vary greatly throughout the country due to differences in climate and runoff characteristics; however, bankfull regional curves can be used to calibrate field identifications (Blackburn-Lynch et al. 2017).

Stratification was needed to account for the natural variability in flood prone width, and therefore entrenchment ratios, across stream and valley types. Stream type was used to stratify the reference curves, and stream types were grouped into relevant valley types. Stream types in confined valleys naturally have low entrenchment ratios and include the following stream types: A, B, Ba, and Bc. Stream types in wider, alluvial valleys include C and E stream types. The compiled geomorphic reference dataset did not include A stream types, but they are likely represented by confined-valley stream types as they naturally occur in confined valleys.

The WSTT evaluated the performance standards in Table 4-4 using the compiled geomorphic reference dataset to develop the threshold values in Table 4-6.

For C and E stream types:

- C and E stream types are grouped together since they typically occur in the same valley types and C stream types have the potential to evolve into an E stream (Rosgen 2009).
- The ER of 2.0 proposed by Harman et al. (2012) as the threshold for not functioning was considered reasonable as streams with an ER less than 2.0 do not have room to dissipate energy laterally through a meandering planform. Only one reference site in the dataset had an ER less than 2.0. This site is a Cb stream with a drainage area of 41 sq. mi. and ER of 1.5 (Cb stream types are located in confined alluvial or colluvial valleys).
- An ER of 2.4 was used to define the threshold between functioning and functioning-at-risk for C and E stream types (Table 4-4). The geomorphic reference dataset supported this, as an ER of 2.4 was below the 25th percentile. Therefore, it was considered reasonable to use the upper end of the range provided by Rosgen (2009) for slightly entrenched stream types.

- In the geomorphic reference dataset, there was quite a bit of variability in the upper bounds of ER values for C and E stream types, so best professional judgement was used to identify an ER of 5.0 for the maximum index score. This value is between the 75th percentile value for C stream types (4.4) and the median value for E stream types (5.2).

For B stream types:

- The threshold for not-functioning identified in Table 4-4 was consistent with the minimum ER observed for B stream types in the compiled geomorphic reference dataset. Values less than 1.2 are outside the natural range of variability observed in stable B-type streams (Rosgen 2009; Harman et al. 2012), and thus an ER of 1.2 was assigned as the threshold between functioning-at-risk and not functioning.
- An ER of 1.4 was selected as the threshold between functioning and functioning-at-risk by Harman et al. (2012) and was consistent with the 25th percentile value from the compiled geomorphic reference dataset (1.5).
- The ER value that yields the maximum index value in the WSQT Version 1.0 was set at 2.2, the 75th percentile value from the compiled geomorphic reference dataset and the typical value used in the stream classification system as a break between B stream types and C and E stream types (Rosgen 2009).

The best-fit line for the plotted threshold values was derived using multiple linear relationships. The final reference curves were reviewed by the WSTT and are shown in Figure 4-3.

Table 4-6: Threshold Values for Entrenchment Ratio

Index Value	Field Values by Stream Type	
	A, B, Ba, Bc	C, Cb, E
1.00	2.2	5.0
0.70	1.4	2.4
0.30	1.2	2.0

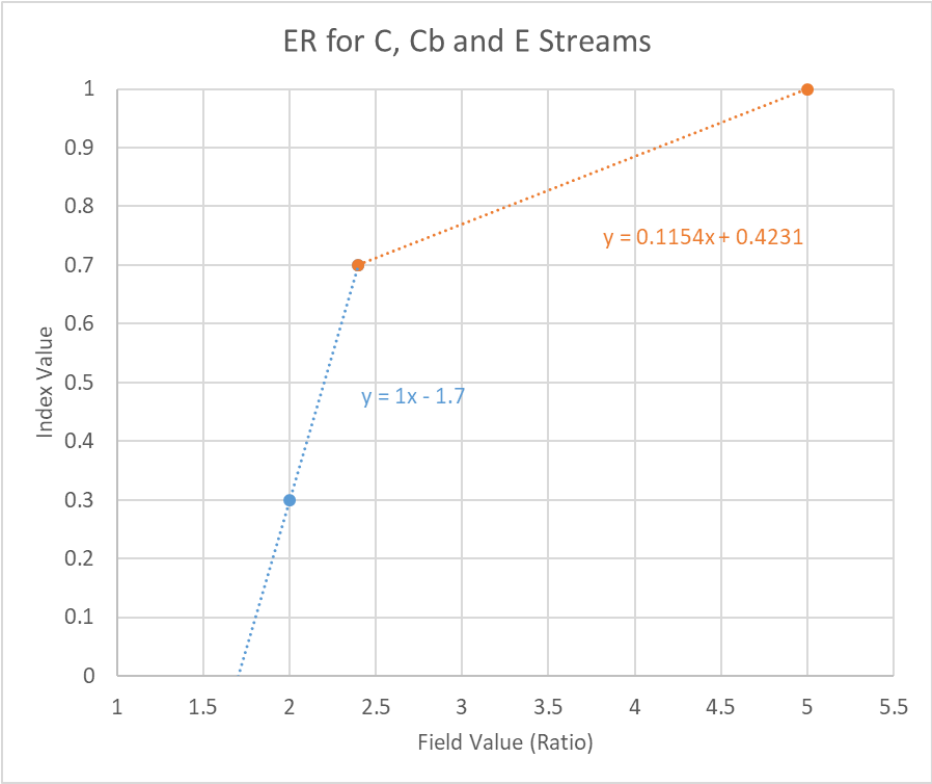

Figure 4-3a: Entrenchment Ratio Reference Curves

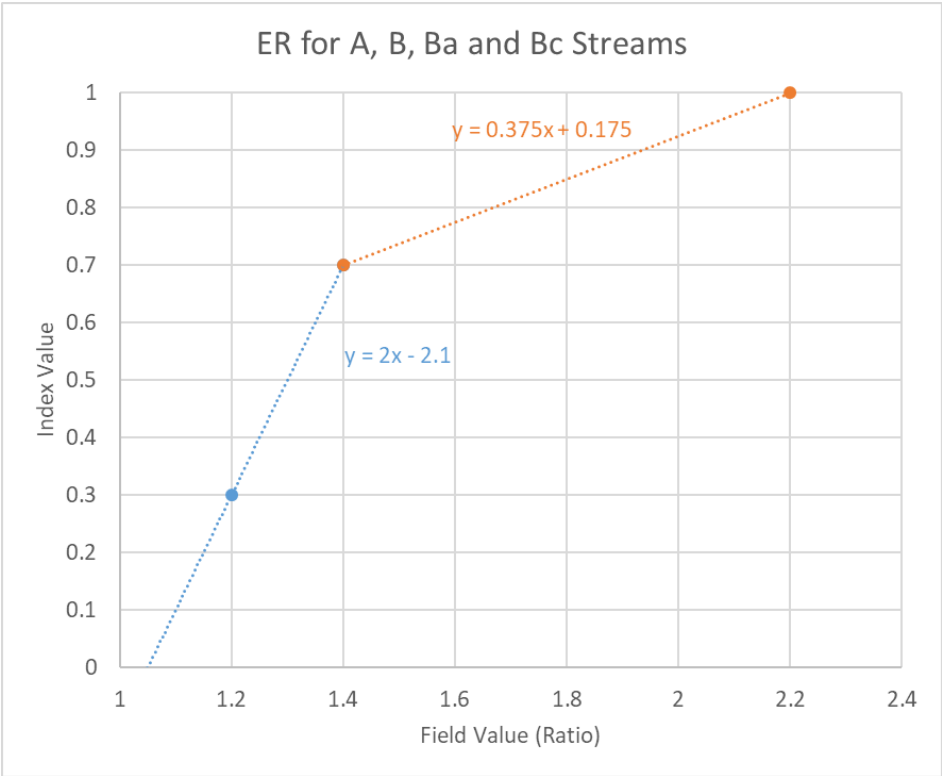

Figure 4-3b: Entrenchment Ratio Reference Curves

Limitations and Data Gaps:

If bankfull dimensions are not accurately determined for a site, then the entrenchment ratio will not accurately represent entrenchment processes. Additional information on verifying bankfull information was added to the WSQT v1.0 User Manual in response to comments received during beta testing.

Reference curves were not developed for naturally occurring F and G stream types. If the stream is a naturally occurring F stream type, e.g., located in a canyon or gorge setting, this metric should not be evaluated, as no reference curves have been developed for this stream type. Additionally, this metric is not applicable to braided (D) stream types since the width of the channels is often the same as the valley width (Rosgen 2009).

For F and G channels that represent degraded streams, these systems should be compared against the proposed, or reference stream type, as informed by channel evolution processes (Rosgen 2009) and described in the WSQT v1.0 User Manual. For example, if the existing stream type is a degraded Gc in an alluvial valley, the proposed / reference stream type and reference curve would be a C or E stream type. Selection of the appropriate reference stream type is important for consistently applying this metric and determining a condition score in the tool. To improve consistency, additional guidance has been added to the WSQT v1.0 User Manual to assist practitioners in identifying the reference stream type.

Chapter 5. Large Woody Debris Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Inputs of large wood, commonly referred to as large woody debris (LWD), provide an important structural component of many streams and floodplains. LWD can take the form of dead, fallen logs, limbs, whole trees, or groups of these components (also known as debris dams) that are transported or stored in the channel, floodplain and flood prone area (U.S. Bureau of Reclamation and U.S. ERDC 2016). LWD influences reach-scale sediment transport and hydraulic processes by: 1) creating sediment and organic matter storage areas; 2) increasing substrate diversity and habitat for benthic macroinvertebrates and cover for fish; 3) creating depth variability where large pieces span the channel and produce pools; 4) sometimes increasing local bank erosion and increasing sediment supply; and 5) providing boundary roughness and flow resistance (Wohl 2000). The LWD parameter is applicable where the upstream watershed or adjacent land area has historically supported (or has the potential to support) trees large enough to recruit LWD. Therefore, this parameter is only applicable to streams with forested catchments, riparian gallery forests, or other streams that naturally have a supply of LWD.

There are numerous metrics available to assess large woody debris. Complex methods include individual piece and jam counts within the channel and floodplain, along with characterization of wood size, type, location and volume (Wohl et al. 2010). The Large Woody Debris Index (LWDI) outlined below provides a similar characterization of LWD in a single index value for a 328-foot (100-meter) reach. Complex approaches like these provide information about how the presence and configuration of wood affects reach-scale functions. For example, large diameter and long pieces of wood and jams within the channel that cannot be readily mobilized, have a greater influence on in-stream functions than a small piece of wood near the top of bank that is easily mobilized. More simplified approaches, such as piece counts, are also used as rapid indicators of LWD. These approaches provide less detailed information on the composition and structure of wood in the channel, but can serve as simple indicators of the influence of wood within the channel.

The WSQT includes two metrics to characterize LWD within streams: 1) the Large Woody Debris Index (LWDI) and 2) the number of pieces per 328 feet (100 meters). Either metric can be applied at a project site; however, users should not enter data for both metrics.

Metrics:

- Large Woody Debris Index (LWDI)
- Number of Pieces per 328 feet (100 meters)

5.1. Large Woody Debris Index (LWDI)

Summary:

This metric is a semi-quantitative measure of the quantity and influence of large woody debris within the active channel, up to and including the top of banks, per 328 feet (100 meters) of channel length. A piece must be at least 10 cm in diameter at one end (Wohl 2000; Davis et al.

2001) and over 1 meter in length (Davis et al. 2001) to be considered LWD. The index does not include LWD beyond the top of bank on the floodplain or terrace. The index was developed by Davis et al. (2001) and evaluates LWD (pieces and debris dams) based on their ability to retain organic matter, provide fish habitat, and affect channel/substrate stability. The LWDI weights this ability for each piece or debris dam by characterizing 1) size (length and width in relation to bankfull dimensions, diameter); 2) location in relation to the active channel or during high flows; 3) type (bridge, ramp, submerged, buried); 4) structure (plain to sticky for organic matter retention); 5) stability during high flows; and 6) orientation (relative to stream bank). Higher scores indicate greater functional influence on instream processes.

The LWDI is a moderately robust measure that is not overly complex. The LWDI requires a moderate level of effort and can typically be completed in one hour or less per project reach. Methods for the LWDI are described in *Application of the Large Woody Debris Index: A Field User Manual* (Harman et al. 2017).

Reference Curve Development:

The WSTT, WDEQ and WGFD collected LWDI data at 22 reference standard sites in Wyoming to develop reference curves for the WSQT. Data were collected at minimally disturbed reference standard sites primarily in mountains, but a few sites were within the basin ecoregion of the state. Table 5-1 shows the statistics for these data. Data collection efforts are continuing to improve the dataset and reference curves. No stratification of this metric was included due to the small reference dataset in Wyoming.

Table 5-1: Statistics for the Wyoming LWDI Reference Standard Dataset. All values are per 328 feet (100 meters) of stream.

Statistic	LWDI Value
Number of Sites (n)	22
Average	689
Standard Deviation	416
Minimum	17
25th Percentile	433
Median	656
75th Percentile	948
Maximum	1583

The following threshold values were proposed based on this dataset (Table 5-2):

- The median of the reference dataset was used to determine the maximum index score (the median value of 656 was rounded up to 660). The median value was used instead of the 75th percentile to account for lower potential for LWD in plains and basins ecoregions. While there are sites from across the state in the dataset, there are more sites in the mountains where higher LWD presence is expected. Also, there are a few sites in the reference dataset

that exhibit LWDI values greater than 1000, and these may have been influenced by recent fires or insect mortality.

- The 25th percentile of the reference dataset was used to inform the threshold between functioning and functioning-at-risk index values. The 25th percentile value of 433 was rounded to 430.
- Due to a lack of LWDI data from degraded sites, no field values were used to define a threshold between functioning at risk and non-functioning index values. Index values within this range are interpolated from the reference curve.

Table 5-2: Threshold Values for the LWDI (per 328 feet or 100 meters)

Index Value	Field Value
1.00	660
0.70	430
0.00	0

A reference curve (Figure 5-1) was derived from the threshold values presented above. A broken linear curve was used to calculate index values. While the reference curve is nearly linear, the shape of the curve follows a conceptual understanding of LWD function and restoration incentives. More lift can be provided in systems characterized as not functioning or functioning-at-risk, represented by the steeper slope on the index curve for that range of scoring, while adding more wood to a system with a functioning amount of wood already would yield less lift.

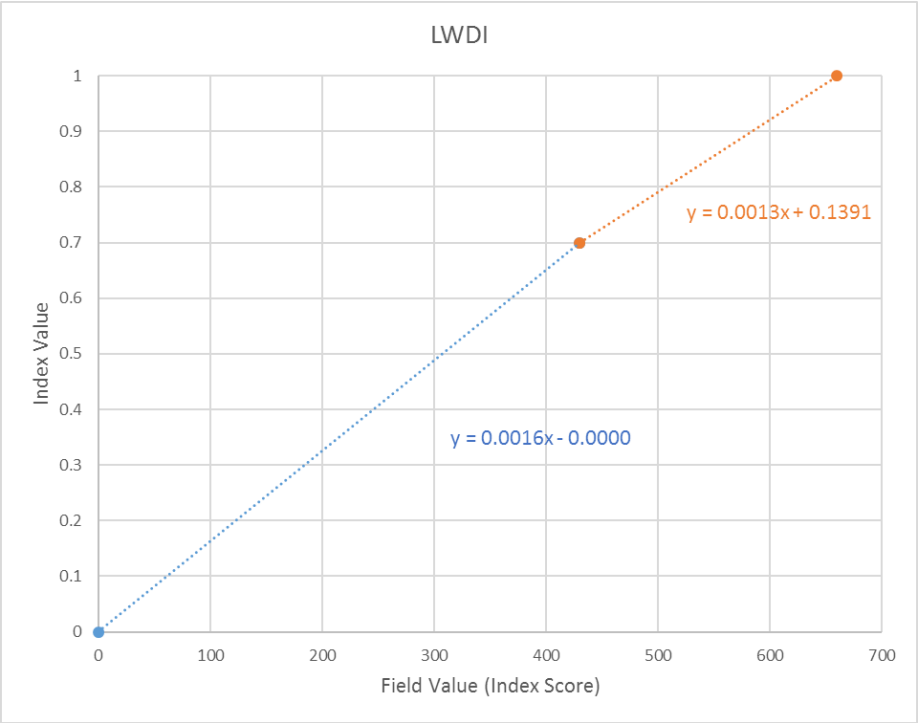

Figure 5-1: LWDI Reference Curve

Limitations and Data Gaps:

The LWDI is a new metric for Wyoming streams and the reference curves in the WSQT Version 1.0 are developed from a relatively small sample size located primarily within the montane ecoregion. As more data are collected, further refinement and stratification of these data and reference curves may be possible. Stratification could consider the role of ecoregion, drainage area, valley type, forest age, canopy type, and other variables (Wohl 2011, Wohl and Beckman 2014). This metric is not applicable to streams without forested catchments, riparian gallery forests, or other streams that naturally have a limited supply of LWD. During beta testing, it was noted that streams in scrub-shrub or willow dominated systems may have wood in the channel associated with willow jams, but the size of the pieces does not qualify as LWD. Additional guidance is provided in the WSQT v1.0 User Manual to address these situations.

5.2. Number of Large Wood Pieces per 328 feet (100 meters)

Summary:

This metric is a count of the LWD pieces in a 100-meter section of the reach, where each piece is counted separately, including within debris dams. To be considered LWD, a piece must be at least 10 cm in diameter at one end (Wohl 2000; Davis et al. 2001) and over 1 meter in length (Davis et al. 2001). This method is a straight-forward, rapid assessment of LWD presence, and is an indicator of its overall structural influence of LWD within the stream.

Reference Curve Development:

Reference curves were developed using the NRSA dataset, described in Section 1.5, which includes a variety of metrics associated with LWD, including the number of pieces per 100 meters. Reference curves were validated using the LWDI data described in the previous section. However, since the LWDI scores dams separately than pieces, the total number of pieces was estimated by assuming all dams only contained three pieces of LWD (Table 5-3). Therefore, these estimated piece counts are likely lower than the actual number of pieces that would be collected with the WSQT methods.

The methods used to collect the NRSA data (i.e., number of LWD pieces in/above the wetted channel within 100m; all sizes) were similar to the LWD piece count developed for the WSQT. There is one notable distinction between the two data collection methods: the NRSA method is an average number of pieces per 100 meters of stream, whereas the WSQT procedure collects data on the 100-meter segment within the reach that would yield the highest value. Therefore, the piece counts from NRSA are likely lower than the number of pieces that would be collected with the WSQT methods.

An effort was made to identify reference standard sites within the NRSA dataset using legacy tree size, riparian vegetation condition, absent canopy, and other attributes available within the NRSA dataset. However, a multivariate analysis was beyond the scope of this analysis and no single attribute was thought suitable to describe reference standards for LWD. As such, the NRSA dataset for this metric includes all reference aquatic resources, including reference standard and degraded sites. Future data analyses and collection efforts will continue to improve the dataset and reference curves.

Stratification of the data by region was explored using the NRSA dataset. Stratification by bankfull width and dominant canopy type (coniferous, deciduous, mixed, or evergreen) were also considered. However, many of the NRSA sites listed the dominant canopy as absent, indicating that there was no canopy at the site. Bankfull width was considered as a surrogate for stream size or drainage area, but no meaningful trend was identified using bankfull width as an independent variable. The dataset was not large enough to stratify by both ecoregion and bankfull width and produce meaningful results. Therefore, the WSTT analyzed data by ecoregion since the ecoregion may represent differences in the riparian community and LWD source material.

The statistics for the NRSA LWD dataset are provided in Table 5-3 and Figure 5-2. Note that the NRSA dataset includes both reference standard and degraded sites, and the 25th percentile and median values for all three ecoregions are low. The average and 75th percentile values indicate that streams in the mountains tend to have the most wood and streams in the basins tend to have the lowest amount of wood. While these differences could be used to produce separate reference curves for the ecoregions, there are multiple sites in both the plains and basins that exhibited large amounts of LWD, as seen in the 95th percentile values in Table 5-3. Some sites within the basins and plains ecoregions may occur in forested areas that provide significant source material, and we were not able to differentiate these sites in the dataset. Thus, we decided to rely on a single reference curve applicable in all ecoregions at sites occurring within naturally forested watersheds or riparian gallery forests.

Table 5-3: Statistics for Number of LWD Pieces from the NRSA Dataset

Statistic	Number of LWD pieces/100m by Ecoregion			Number of LWD pieces/100m	LWDI Estimated Piece Counts
	Basins	Mountains	Plains	All NRSA Data	
Number of Sites (n)	64	38	68	170	22
Average	4	9	5	6	30
Standard Deviation	9	15	8	11	19
Minimum	0	0	0	0	1
25th Percentile	0	0	0	0	23
Median	1	3	1	1	28
75th Percentile	2	13	6	7	45
95th Percentile	24	28	21	26	57
Maximum	49	80	45	80	74

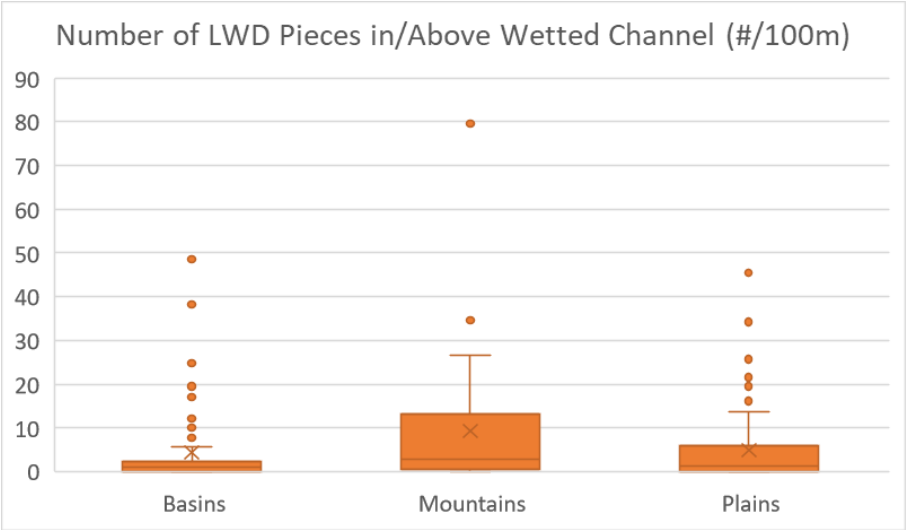

Figure 5-2: Box Plots for Number of LWD Pieces from the NRSA Dataset

Based on the assumption that the LWD parameter would not be applicable for many sites within the basins and plains, the reference curve was developed using the data from the mountains ecoregion. The following threshold values were used to inform the curve (Table 5-4):

- The 95th percentile from the NRSA sites within the mountains matched the median value from the LWDI estimated piece count. The median value from the latter dataset was used to define the maximum index score for the LWDI metric. The 95th percentile from the mountains was used to define the maximum index score for this metric.
- The 75th percentile from the NRSA sites within the mountains was used to define the threshold between functioning and functioning-at-risk. The 25th percentile from the LWDI dataset was used to define the threshold between functioning and functioning-at-risk for the LWDI metric, but because the NRSA dataset contains non-reference standard sites and the LWDI dataset does not, it did not make sense to similarly rely on the 25th percentile.

A broken linear curve was fit to the threshold values (Figure 5-3).

Table 5-4: Threshold Values for the Number of LWD Pieces per 100 meters

Index Value	Field Value
1.00	28
0.70	13
0.00	0

Figure 5-3: Number of LWD Pieces Reference Curve

<u>Limitations and Data Gaps:</u>

This metric is not applicable to streams without forested catchments, riparian gallery forests, or otherwise naturally have a limited supply of LWD. During beta testing, it was noted that streams in scrub-shrub or willow dominated systems may have substantial wood in the channel associated with willow jams, but the size of the pieces does not meet the definition of LWD provided in the LWDI method. Additional guidance is provided in the WSQT v1.0 User Manual to address these situations using the LWDI, but not for the piece count metric. In these instances, it may be beneficial to use the LWDI instead of this metric.

As more data are collected, further refinement and stratification of these data and development of multiple reference curves may be possible. Stratification could consider the within-ecoregion differences associated with drainage area, forest age, valley type, canopy type, and other variables.

Chapter 6. Lateral Migration Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Lateral migration is a function-based parameter used to characterize streambank erosion rates in single-thread, meandering streams. This parameter is included in the Geomorphology functional category because it provides information about sediment supply/transport and dynamic equilibrium processes. Lateral migration rates vary naturally by stream type, and can be affected by changes in sediment processes at the watershed and reach scale (Roni and Beechie 2013). Lateral stability is one of the original parameters described in Harman et al. (2012). Readers should refer to Harman et al. (2012) for additional discussion of bank migration and lateral stability processes, and stream types that are susceptible to lateral migration versus those where migration is naturally constrained.

There are multiple approaches that can be used to measure lateral migration processes and condition (Harman et al. 2012). Some of these approaches include:

- Aerial imagery interpretation of bank retreat, measurements of belt width divided by bankfull width (meander width ratio), and visual assessment of bank cover and stability.
- Semi-quantitative measures of bank cover and stability measured over the entire reach length (BLM 2017; WDEQ 2018; Binns 1982).
- The Bank Erosion Hazard Index/Near Bank Stress approach (BEHI/NBS; Rosgen 2014).
- Measurements of bank erosion using surveyed cross sections, bank profiles or bank pins.
- A modeling program, called BSTEM (Bank Stability and Toe Erosion Model) is an intensive approach if data are not available for model calibration, and a moderately intensive approach if data are available (Simon et. al. 2009).
- Greenline Stability Rating characterizes the live perennial vascular plants and other natural stabilizing elements on or near the water's edge and provides a rating of bank stability for a subsampled section of the reach (Winward 2000).
- Measures of the extent of bank erosion and/or armoring within a reach.

The Lateral Migration parameter includes four metrics: the Greenline Stability Rating, dominant BEHI/NBS, percent eroding streambank and percent armoring. The dominant BEHI/NBS and percent eroding streambank metrics rely on BEHI/NBS assessment, and are intended to be used together. The dominant BEHI/NBS metric characterizes the magnitude of erosion, and the percent eroding streambank characterizes the extent of the problem. The Greenline Stability Rating metric can be collected alone or in conjunction with other lateral migration metrics and is of similar complexity. The percent armoring metric should be used in altered stream segments where rip-rap or other bank stabilization treatments have been or intend to be implemented.

The four metrics in this parameter are measures of channel condition that serve as indicators of altered processes, but do not characterize lateral migration rates or sediment processes themselves. Sediment transport analyses are critical in understanding watershed and reach-scale processes, and should be relied on to evaluate and develop design alternatives (Roni and Beechie 2013). These analyses are not currently incorporated into the tool, although sediment transport and channel evolution models are used to inform restoration potential (Chapter 1.2) and should be included in the design process.

Metrics:

- Greenline Stability Rating
- Dominant BEHI/NBS
- Percent Streambank Erosion (%)
- Percent Armoring (%)

6.1. Greenline Stability Rating

Summary:

There is a strong interrelationship between amount and kind of vegetation along the water's edge and bank stability. Late successional plant communities are indicators of resilience, stability and reference condition (Youngblood et al. 1985; Winward 2000; MacFarlane et al. 2017). Evaluation of the types of vegetation along the greenline provides a good indication of a streambank vegetation's ability to buffer the hydrologic forces of moving water (Winward 2000).

The Greenline Stability Rating (GSR) is collected along the greenline, which is a linear grouping of live perennial vascular plants on or near the water's edge, generally slightly below the bankfull stage. The primary purpose of the GSR is to provide an index rating of the natural capacity of vegetation to protect streambanks against erosion as well as enhancing streambank strength, as they filter sediments and, with the forces of water, they build/rebuild eroded portions of streambanks (Winward, 2000). The metric also characterizes anchored rocks or logs large enough to withstand the forces of water encountered on the greenline edge as a natural, stable percentage of the greenline in place of the vegetation.

The GSR is calculated by multiplying the percent composition of each community type along the greenline by the stability class rating assigned to that type, and calculating the average value for the sample reach. The WSQT allows for two methods to measure GSR: 1) the original data collection procedures described in Winward (2000), or 2) the Modified Winward Greenline Stability Rating procedures described in USDOI (2011). The latter integrates a more systematic approach to collecting data by using plots instead of paces and calculating stability ratings by key species rather than community types to improve precision and includes additional species stability ratings not identified in Winward (2000).

Reference Curve Development:

The threshold values and reference curve for the WSQT were constructed on the index rating classes established by Winward (2000) as shown in Table 6-1.

Table 6-1: Greenline Stability Rating and Functional Capacity

GSR	Stability Description	Functional Capacity
1-2	Very Low	Not Functioning
3-4	Low	
5-6	Mid	Functioning-At-Risk
7-8	High	Functioning
9-10	Excellent	

The WSQT threshold value between Not Functioning and Functioning-At-Risk was set at 5 (between low and mid) and the threshold between Functioning-At-Risk and Functioning was set at 7 (between mid and high) as shown in Table 6-1 and Figure 6-1. A narrow range of the Mid rating class is representing Functioning-At-Risk on the reference curve. A polynomial equation was fit to these threshold values. These curves were compared with the WSTT data collection from 2016 (Table 6-2).

In August 2016, the WSTT visited several sites to apply the proposed WSQT methodology for assessing riparian vegetation. These sites were considered to represent minimally disturbed reference standard sites. However, because they are located on public lands, they have likely been subject to some historical use, including grazing and/or timber removal. In evaluating the datasets and proposed benchmarks, we concluded it was reasonable to characterize these sites as functioning or (high) functioning-at-risk. These sites have the potential to support a healthy aquatic ecosystem, and were not in a clearly degraded state.

Table 6-2: Greenline Stability Rating at Reference Sites Visited by the WSTT

Site	Ecoregion	GSR
Wood River, above Middle Fork	Mountains	6.1
Middle Fork Wood River	Mountains	6.9
Middle Fork Wood River - Upstream	Mountains	7.4
Jack Creek	Mountains	8.0

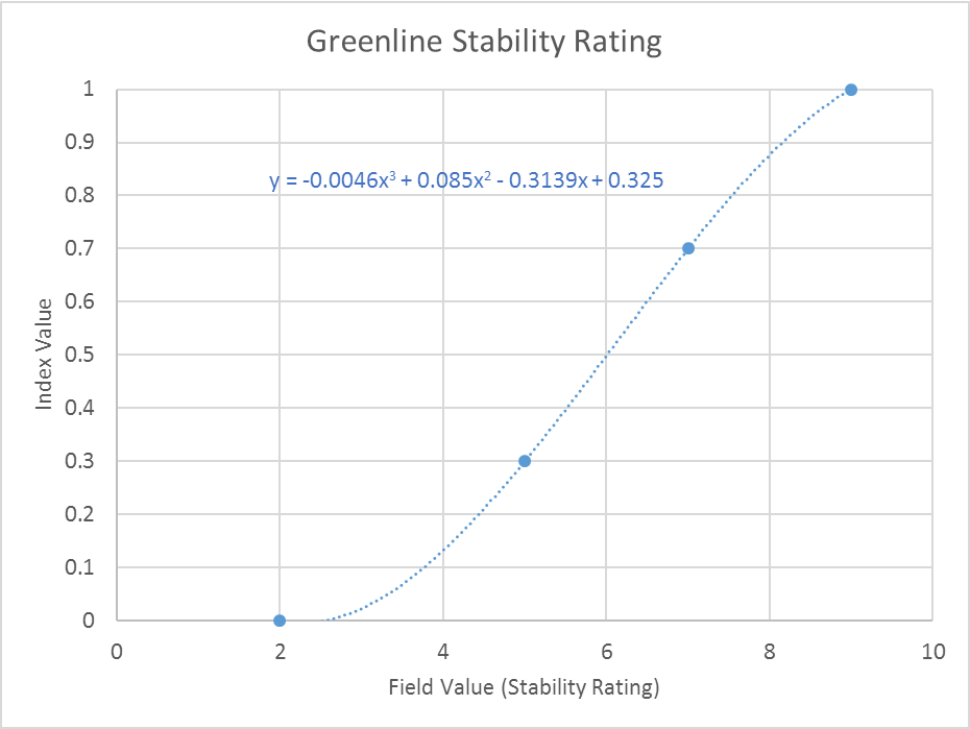

Figure 6-1: Greenline Stability Rating Reference Curve

Limitations and Data Gaps:

As described above, two methods may be employed to produce the GSR. The same methodology must be used for pre- and post-condition/project use.

The original Greenline publication only includes stability class information for riparian (plant) community types of the Intermountain/Rocky Mountain Region (Youngblood et al. 1985), while USDOI (2011) has notably expanded the list of bank stability ratings for other species and community types in the western United States. The MIM Technical Reference (USDOI, 2011 Table H1. p. 136) also outlines procedures for developing a relative stability value based on general rooting characteristics assigned by the authors or other referenced publications.

The number of feet of anchored rocks or logs, large enough to withstand the forces of water, encountered along the greenline edge are counted as a natural, stable percentage of the greenline in place of the vegetation. A potential limitation of this method is differentiation between natural stabilizing elements and unnatural armoring such as exposed riprap that can artificially elevate the stability rating. Armoring treatments in many systems can be considered an adverse impact or form of functional loss. In these cases, use of this metric should be applied in conjunction with the percent armoring metric.

The GSR becomes less valuable in monitoring steeper (greater than 4 percent gradient) streams since the large, permanently anchored rocks are generally less susceptible to management activities. Also, the GSR may be a less valuable measurement on very large rivers where landform features play the dominant role in regulating hydrologic influences compared to vegetation influences (Winward 2000).

6.2. Dominant BEHI/NBS

Summary:

The Bank Erosion Hazard Index (BEHI) and Near Bank Stress (NBS) are two bank erosion estimation tools from the Bank Assessment for Non-point source Consequences of Sediment (BANCS) model (Rosgen 2006). BEHI and NBS ratings are determined based on collecting relatively simple measurements and visual observations. The streambank assessment includes the evaluation of streambank cover, height, depth and density of roots, and bank angle. From the streambank assessment, a categorical BEHI risk rating is assigned, from very low to extreme. Observations of channel flow characteristics, including water-surface slope, direction of velocity vectors and other methods, are used to assign an NBS risk rating, which can also range from very low to extreme.

The dominant BEHI/NBS is the rating that occurs most frequently based on length. For example, a dominant BEHI/NBS rating of High/High means that most of the assessed length, e.g., outside meander bends, has this rating. Instructions on how to measure the dominant BEHI/NBS rating is provided in the WSQT v1.0 User Manual.

Regionalization efforts for the BANCS model have met with mixed results when BEHI/NBS ratings have been used to predict erosion rates (McMillan et al. 2017). The use of BEHI/NBS in the WSQT avoids this problem by using the dominant BEHI/NBS rating to characterize the severity of bank erosion *rather than* trying to predict an erosion rate. The focus is on the

potential for accelerated bank erosion due to geotechnical and hydraulic forces rather than the rate of erosion. BEHI/NBS is included in the WSQT for the following reasons:

1. It is rapid to moderate in terms of time required to collect data depending on the way it is implemented. Rosgen (2014) outlines several data collection approaches to measure BEHI and NBS depending on study objectives and site conditions.
2. By integrating two ratings, the method assesses both geotechnical (BEHI) and hydraulic (NBS) forces, which is unique among rapid methods. This is important because vertical banks devoid of vegetation may visually appear to be eroding, but if the hydraulic forces acting against the bank are very low there may be little to no bank erosion.
3. It is a common method used by practitioners of natural channel design, which is a common approach used in compensatory stream mitigation programs (ELI et al. 2016).

Reference Curve Development:

The BEHI and NBS ratings were tested with field data collected in Colorado and Wyoming, as described in Rosgen (1996). Each combination of BEHI and NBS rating is assigned to one of four stability categories (Rosgen 2008). The WSTT converted these stability categories into functional capacity ratings as follows: stable represents functioning, moderately unstable represents functioning-at-risk, and unstable and highly unstable represent not functioning.

Table 6-3: Dominant BEHI/NBS stability ratings provided in Rosgen (2008). (VL) is very low; (L) is low; (M) is moderate; (H) is high; (VH) is very high; etc.

Stable	Moderately Unstable	Unstable	Highly Unstable
L/VL, L/L, L/M, L/H, L/VH, M/VL	M/L, M/M, M/H, L/Ex, H/VL, H/L*	M/VH, M/Ex, H/M, H/H*, VH/VL, Ex/VL, Ex/L	H/Ex, Ex/M, Ex/H, Ex/VH, VH/VH, Ex/Ex

** Ratings were included in two categories. The erosion rate curves based on data from Colorado were consulted to remove duplicate values from the table.*

Because the metric relies on categorical data, reference curves were not developed. Instead, the ratings and categories from Table 6-3 were assigned index values based on relating the stability ratings to functional capacity as described below (and shown in Table 6-4).

- The ratings within the Stable category were considered to represent a functioning condition (1.00). Stable doesn't mean that functioning streams do not laterally migrate, but they migrate at appropriate rates and maintain their cross-sectional area while their position on the landscape may change.
- The ratings within the moderately unstable category were considered to represent a Functioning-At-Risk range of condition (0.30-0.69).
- The ratings within the Unstable and Highly Unstable categories were considered to represent a Not-Functioning condition (0.00-0.29).
- Within these index ranges, the ratings were assigned an index value based on the severity of the instability, with more unstable rating receiving lower scores.

Table 6-4: Index Values for Dominant BEHI/NBS

Index Value	Field Value
0.00	H/VH, H/Ex, VH/VH, VH/Ex, Ex/M, Ex/H, Ex/VH, Ex/Ex
0.10	M/Ex,
0.20	M/VH, H/M, H/H, VH/M, VH/H
0.30	M/H, Ex/L, Ex/VL
0.40	H/L, VH/L
0.50	H/VL, VH/VL, M/M
0.60	L/Ex, M/L
1.00	L/VL, L/L, L/M, L/H, L/VH, M/VL

Limitations and Data Gaps:

This metric is applicable to single-thread channels where the reference condition is a stable channel. In this context, stable does not mean that lateral migration is not occurring, but rather that the channel maintains dynamic equilibrium. A channel in dynamic equilibrium maintains its cross-sectional area while moving across the landscape; that is, lateral erosion and deposition are approximately equal. Systems naturally in disequilibrium, like some braided streams, ephemeral channels and alluvial fans may naturally experience higher rates of bank erosion as they alternate between aggrading, incising or avulsing states due to natural patterns in sediment and hydrologic processes (Roni and Beechie 2013). For systems with naturally high rates of bank erosion, this metric should not be assessed.

6.3. Percent Streambank Erosion

Summary:

This metric estimates the percent of the streambank within a reach that is actively eroding, according to BEHI/NBS ratings. The percent eroding streambank metric provides a measure of the extent of bank erosion, whereas the dominant BEHI/NBS rating provides the magnitude of active bank erosion. BEHI/NBS ratings that represent actively eroding banks are listed in Table 6-5. These ratings were categorized by the WSTT; all stable and some moderately stable ratings were categorized as non-eroding. The field value is calculated by adding the length of BEHI/NBS ratings that represent actively eroding banks from the left and right banks and dividing it by the total bank length (e.g., reach length times two). Note that riffle sections that are not eroding and depositional areas like point bars are not evaluated in the BEHI/NBS assessment, but these sections are included when calculating the total bank length (denominator) for this metric.

Table 6-5: BEHI/NBS stability ratings that represent actively eroding and non-eroding banks

Non-eroding Banks	***Actively Eroding Banks***
L/VL, L/L, L/M, L/H, L/VH, L/Ex, M/VL, M/L	M/M, M/H, M/VH, M/Ex, H/L, H/M, H/H, H/Ex, VH/VL, Ex/VL, Ex/L Ex/M, Ex/H, Ex/VH, VH/VH, Ex/Ex

<u>Reference Curve Development:</u>

The Wyoming Habitat Quality Index for trout streams (Binns 1982) contains a metric that scores the length of eroding bank according to the following criteria:

- 100% to 75% eroding banks are inadequate to support trout,
- 74% to 50% provide very limited potential,
- 49% to 25% provide limited potential,
- 24% to 10% provide moderate potential to support trout, and
- 9% to 0% eroding banks are completely adequate to support trout

Based on these criteria, a minimum index value of 0.00 was assigned where percent streambank erosion exceeded 75% of bank length. Members of the WSTT that have applied the HQI methods across Wyoming have rarely observed values greater than 10% eroding streambanks among reference standard streams, and thus concluded this to be a reasonable threshold between functioning and functioning-at-risk index scores. The thresholds identified in Table 6-6 were used to develop reference curves (Figure 6-2). It was not possible to fit a single equation to the threshold values, so a broken linear curve was used to differentiate between the functioning range of index values and the not functioning and functioning-at-risk range.

Table 6-6: Threshold Values for Percent Streambank Erosion

Index Value	Field Value (%)
1.00	≤ 5
0.70	10
0.00	≥ 75

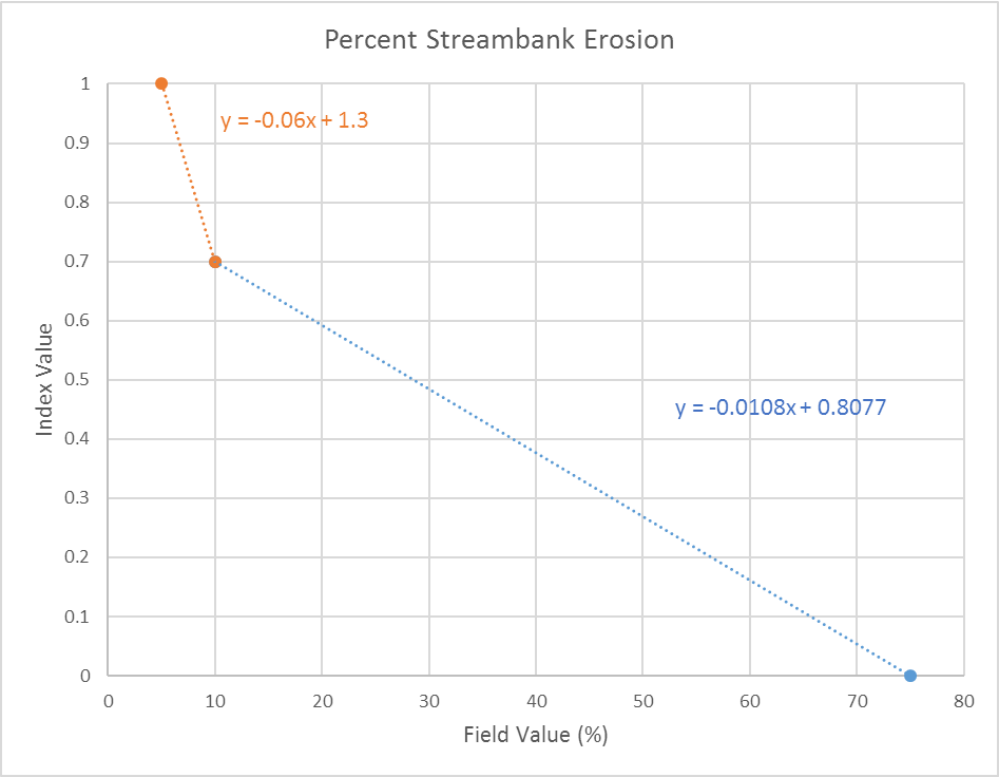

Figure 6-2: Percent Streambank Erosion Reference Curve

Limitations and Data Gaps:

This metric is applicable to single-thread channels where the reference condition is a stable channel. In this context, stable does not mean that lateral migration is not occurring, but rather that the channel maintains dynamic equilibrium. A channel in dynamic equilibrium maintains its cross-sectional area while moving across the landscape; that is, lateral erosion and deposition are approximately equal. Systems naturally in disequilibrium, like some braided streams, ephemeral channels and alluvial fans may naturally experience higher rates of bank erosion as they alternate between aggrading, incising or avulsing states due to natural patterns in sediment and hydrologic processes (Roni and Beechie 2013). For systems with naturally high rates of bank erosion, this metric should not be assessed.

This metric does not distinguish between sections of bank that are naturally stable from those that are anthropogenically hardened or armored. In many systems armoring treatments can be considered an adverse impact or form of functional loss. Where armoring is present, use of this metric should be applied in conjunction with the percent armoring metric.

6.4 Percent Armoring

Summary:

Bank armoring is a common technique to stabilize banks and/or prevent lateral migration, and involves the establishment of hard structures (e.g., rip rap, gabion baskets, concrete or other engineered materials that prevent streams from meandering) along the bank edge. Literature

shows that bank armoring can have positive and negative effects on aquatic functions (Fischenich 2003; Henderson 1986). Beneficial effects of armoring may include the creation of localized fish habitat (pool and cover formation) and the reduction in excessive bank erosion and sediment supply. Negative effects to stream functions include loss of fish habitat, biological diversity, degradation of riparian ecosystems, and impacts to floodplain development and channel evolution by preventing natural rates of lateral migration (Fischenich 2003; Henderson 1986). Bank armoring can also lead to accelerated bank erosion and changes in sediment dynamics in adjacent reaches.

Recognizing the adverse consequences of armoring treatments in streams, the WSTT has included a basic bank armoring metric in the lateral migration parameter. In many systems armoring treatments can be considered an adverse impact or form of functional loss, and the other metrics included to describe this parameter do not adequately capture the functional loss associated with hard armoring practices. The armoring metric should only be used if armoring techniques are present or proposed in the project reach. If banks are not unnaturally armored in the project reach, a field value should not be entered. To calculate the armoring field value, measure the total length of armored banks (left and right) and divide by the total bank length (e.g., project reach length times two). Multiply by 100 to report the percentage of bank length that is armored.

Reference Curve Development:

Even though there are some positive benefits to armoring, the negative impacts to ecological function generally outweigh the positives. Furthermore, hard armoring does not support natural sediment processes and function. The intent of stream restoration/mitigation is to restore/enhance natural processes.

No studies were found showing a relationship between the percent of armoring to functional impairment, so threshold values were proposed by the WSTT (Table 6-7). Because hard armoring would be absent in reference standard sites, a field value of 0% was assigned an index value of 1.00. Thirty percent armored was assigned an index score of 0.00 and a linear curve was established between the two points (Figure 6-3). Setting the minimum index value at 30% armored stream length seemed reasonable to the WSTT, as it means that almost a third of the project reach is armored. At this level of armoring, the reach could be considered channelized and functional loss of channel migration processes could be severe.

If more than 75% of the reach is armored, it is recommended that the other metrics in Lateral Migration not be measured. At this magnitude, the armoring is so pervasive that lateral migration processes would likely have no functional value.

Table 6-7: Threshold Values for Percent Armoring

Index Value	Field Value (%)
1.00	0
0.00	30

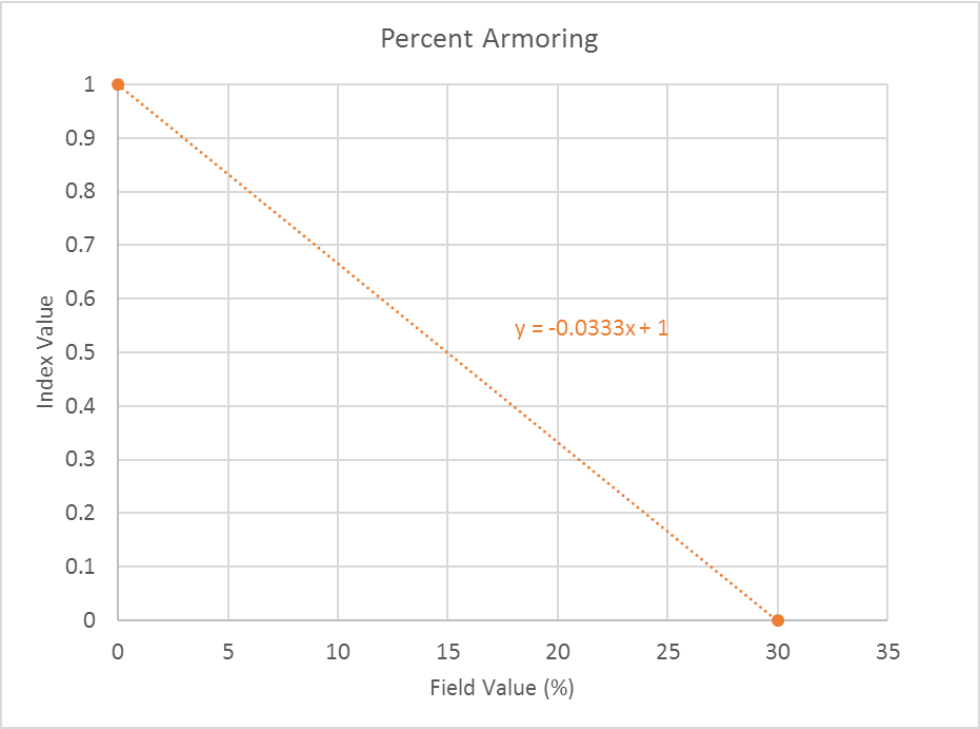

Figure 6-3: Percent Armoring Reference Curve

<u>Limitations and Data Gaps:</u>

While the literature documents a negative relationship between armoring and multiple stream functions, no information could be found relating the extent of armored stream banks to functional loss. Therefore, the reference curves are based solely on best professional judgement. The reference curves for this metric will benefit from validation and testing as the WSQT is implemented.

Chapter 7. Bed Material Characterization Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

The interaction between flowing water and sediment transport creates bed forms, which provide the critical habitats for macroinvertebrates, fish and other organisms (Allan 1995). Streams that are in balance with the hydrologic and sediment transport processes in the watershed are said to be in dynamic equilibrium. This means that the stream bed is not aggrading nor degrading over time, and that lateral adjustments do not change the cross-sectional area, even though its position on the landscape may change (Hack 1960).

Human activities have had substantial and wide-ranging effects on sediment processes in streams (Wohl 2004, Wood and Armitage 1997), including land use activities that have modified and often accelerated the input of sediments into streams. In-channel sources of sediment, including banks, mid-channel and point bars, and fine material deposition areas, can be modified via flow alteration and changes in bank stability. Non-channel sources within the catchment, largely from hillslope erosion processes, can be altered when exposed soils are subject to erosion, when mass failures or landslides occur, where urban development alters the timing and magnitude of runoff events, and when other human activities alter the availability and rate of sedimentation into streams.

The ecological effects of fine-sediment accumulation are ubiquitous and wide-ranging (Wood and Armitage 1997; Table 3). The size and stability of bed material has been linked to macroinvertebrate abundance and diversity (Hussain and Pandit 2012). Additionally, multiple fish species build spawning beds out of gravel; and fine sediment accumulation can reduce the quality of spawning habitats and reduce egg survival (summarized in Wood and Armitage 1997). Characterizing bed material provides insight into sediment transport processes (Bunte and Abt 2001), and whether these processes are functioning in a way that supports suitable habitat for a functioning ecological community (Allan 1995).

There are many ways that sediment transport can be directly measured and modeled, however, many of these approaches are time and data intensive (Harman et al. 2012). Monitoring the ecosystem responses to reach-scale impacts or restoration efforts necessitate a simpler indicator. Evaluating the bed material can provide insight into whether sediment transport processes are functioning to transport and distribute sediments in a way that can support the stream ecosystem.

The WSQT only includes one metric to evaluate this parameter, the size class pebble count analyzer. This metric was developed by the Rocky Mountain Forest and Range Experiment Station to assess watershed cumulative effects of land management practices on changes to grain size distributions (Bevenger and King 1995). The WSTT considered adding an embeddedness metric to this parameter but existing metrics (e.g. Rosgen 2014 and USEPA 2016) are qualitative and it was decided to not include them in Version 1.0 of the WSQT.

Harman et al (2012) lists a second metric to characterize bed material called the Riffle Stability Index (Kappesser 2002). This metric has been used in Rosgen B3 and F3b stream types with slopes ranging between 2 and 4% to show if upstream sediment supply is depositing on riffles. Results are placed into three bins that closely related to functioning, functioning-at-risk, and not

functioning. It is a simpler method than the Size Class Analyzer, however, this method was not included in the WSQT Version 1.0 because it is only applicable to B3 and F3b stream types, and most mitigation/restoration activities occur in C4 and B4c stream types. It is a metric that could be considered in future versions of the WSQT.

There are many other methods for developing grain-size distributions and performing associated calculations (Bunte and Abt 2001). Laub et al (2012) provides several metrics that use grain size distributions to assist in determining bed complexity. These metrics include calculations for heterogeneity, sorting, Fredle index, a gradation coefficient, and a sediment coefficient of variation. These metrics were not used in the WSQT because reference values by metric were unavailable and a tool like the Size Class Analyzer wasn't available to process the data and perform statistics. However, these metrics could be added in the future as reference data and/or processing tools become available.

Metric:

- Size Class Pebble Count Analyzer (p-value)

7.1. Size Class Pebble Count Analyzer

Summary:

The Size Class Pebble Count Analyzer metric is a statistical comparison between the percent of fines in bed material samples from the study reach and a reference reach (Bevenger and King 1995). The Size Class Pebble Count Analyzer spreadsheet tool (v1; USDA 2007) tests the hypothesis that the percent of fines in the study reach is the same as the percent of fines in the reference reach. This metric requires the user to perform a representative pebble count using the Wolman (1954) procedure at the study reach and a reference reach.

This metric is applicable for gravel and cobble bed streams where in-channel or non-channel sediment sources and/or transport of those sediments within the stream have been modified by human activities. Examples include areas with accumulation of fine sediments due to bank erosion or land use change, or where flow alteration may lead to additional fine sediment accumulation or scour and armoring. Projects that reduce bank erosion along a long project reach or restore flushing flows may be able to show a reduction in fine sediment deposition (Harman et al. 2012). Changes in land management practices can result in the delivery of fine sediment to streams, which can impact aquatic habitat bedform features such as pools and riffles. The instructions for the metric require the user to select an appropriate size class to compare. The tool and the WSQT v1.0 User Manual recommend using the minimum size criteria for fine or medium gravel as likely candidates, 4 mm or 8 mm, respectively. Bevenger and King (1995) provide case studies from the Shoshone National Forest that compare the impacts of various disturbances. These case studies define fine sediments as those smaller than 8 mm, citing a study indicating the particles up to 6.4 mm are important to fisheries.

Reference Curve Development:

Based on the outputs available from the Size Class Pebble Count Analyzer spreadsheet tool (v1) (USDA 2007), it was determined that the indicator of statistical significance, p-value, would provide a robust metric to evaluate and monitor changes to the bed material of the reach. The Size Class Pebble Count Analyzer spreadsheet tool (v1) (USDA 2007) tests the hypothesis that

the percent of fines in the study reach is the same as the percent of fines in the reference reach. A small p-value (<0.05) represents a statistically significant difference between the study reach and reference reach, and thus indicates that is it highly unlikely that the percent of fines in the study reach is the same as the percent of fines in the reference reach.

The case studies presented by Bevenger and King (1995) show that highly significant results (p-value < 0.001) were observed in the Shoshone National Forest due to fires, grazing and timber harvesting practices. The WSTT did not collect Wyoming field data to develop reference curves for this metric, but instead relied on typical statistical confidence intervals of 90%, 95% and 99%, corresponding to p-values of 0.10, 0.05 and 0.01 (Haldar and Mahadevan 2000). Typical p-values were used to assign threshold index values based on their degree of departure from the reference site as shown in Table 7-1.

Table 7-1: Threshold Values for Size Class Pebble Count Analyzer p-values

Index Value	*Field Value*
1.00	> 0.10
0.29	0.05
0.00	0.01

A linear reference curve was fit using the 0.05 and 0.01 p-values. Non-significant p-values (>0.10) which correspond to a 90% confidence interval were considered to represent a functioning condition (Figure 7-1).

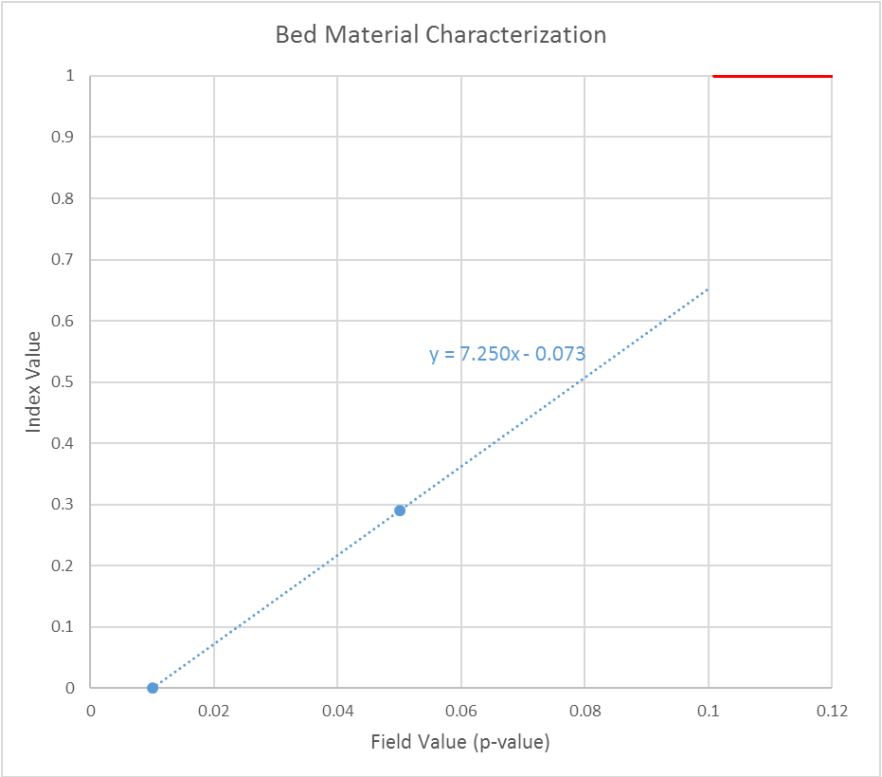

Figure 7-1: Size Class Pebble Count Analyzer (p-value) Reference Curve

Limitations and Data Gaps:

This metric only applies to gravel or cobble bed streams. As noted above, the WSTT did not collect Wyoming field data to develop reference curves for this metric, but instead relied on typical statistical confidence intervals. Wyoming-specific data collection and analysis is still needed to test the sensitivity of this metric to stream restoration practices. In the future, the WSTT would also like to develop a percent fine sediments metric that does not rely on a reference site, but additional data analysis is needed before reference curves are proposed.

Applying this metric requires comparison with a reference standard site with similar stream and watershed characteristics, such as stream type, drainage area, geology, lithology, slope etc. Finding good reference standard sites for comparison with project sites can be challenging, particularly in areas with major land use changes within the watershed. If a suitable reference reach cannot be located, then this bed material characterization metric should not be used. Note that for this metric, it may be possible to identify a reference standard site with respect to sediment transport (bed form diversity, lateral migration, and bed material characterization) that has other watershed impairments, such as water quality impairments associated with wastewater treatment plants or oil and gas development, which affect physicochemical or biology functions.

Chapter 8. Bed Form Diversity Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Bed forms include the various channel units that maintain heterogeneity in the channel form, including riffles, runs, pools and glides. The location, stability and depth of these bed features are responsive to sediment transport processes acting against the channel boundary conditions. Bed form diversity is a function-based parameter used to assess these bed form patterns, specifically riffle-pool and step-pool sequences in alluvial and colluvial valleys. This parameter evaluates bedform pattern in relation to expected patterns in channels with similar morphology. As such, this parameter is not a direct measure of fluvial processes, but is an indicator of altered hydraulic and sediment transport processes (Knighton 1998). It is one of the original parameters described in Harman et al. (2012). Readers should refer to this document for a more detailed description of how sediment transport processes affect the development of sand and gravel bedforms.

Natural streams rarely have flat uniform beds (Knighton 1998). Instead, hydraulic and sediment transport processes shape the stream bed into myriad forms, depending on channel slope, type of bed material (sand, gravel, cobble, boulder, bedrock) and other factors. These bed forms reflect local variations in the sediment transport rate and represent lateral and vertical fluctuations in the stream bed (Knighton 1998), dissipating energy and creating habitat diversity through the formation of riffle-pool sequences.

Numerous classifications of bed form exist (Knighton 1998). At a broad level, bed forms can be grouped into three categories: sand bed forms (ripple, dunes and antidunes), gravel/cobble bed forms (riffle, run, pool and glide) and step-pool bed forms. Bed form diversity is important because channel patterns provide a diversity of habitats that aquatic organisms need for survival. For example, macroinvertebrate communities are often most diverse in riffle habitats, and fish rely on pools for resting, thermal and solar refugia. Without the diversity of riffles and pools, there is also a potential loss of diversity in macroinvertebrates and fish (Mathon et al. 2013; Fischenich 2006).

Harman et al. (2012) list metrics that can be used to assess bed form diversity and can be quantified with field surveys, including: percent riffle and pool, facet (riffle/pool) slope, pool spacing and depth variability. An additional metric, aggradation ratio, was not described in Harman et al. (2012), but is useful in characterizing aggradation processes in riffle sections. Many qualitative methods are also available to assess bedforms and in-stream habitats (Somerville and Pruitt 2004), but were not considered for the WSQT because quantitative measures are available and regularly used by practitioners.

The WSQT includes four metrics to characterize bed form diversity: pool spacing ratio, pool depth ratio, percent riffle and aggradation ratio. These metrics are often used by practitioners in quantitative geomorphic assessments of riffle-pool and step-pool sequences (Knighton 1998; Harrelson et al. 1994; Rosgen 2014; and ELI et al. 2016). Pool spacing ratio, pool depth ratio and percent riffle metrics should be evaluated together to characterize the overall bed form diversity of a stream reach. Aggradation ratio should also be considered where indicators of aggradation are present.

Metrics:

- Pool Spacing Ratio
- Pool Depth Ratio
- Percent Riffle
- Aggradation Ratio

8.1. Pool Spacing Ratio

Summary:

Adequate pool spacing and the depth variability created from alternating riffles supports dynamic equilibrium and habitat-forming processes (Knighton 1998, Hey 2006). The pool spacing ratio metric measures the distance between the deepest location of sequential geomorphic pools (i.e., lateral-scour / meander bend pools or step-pools). The distance between geomorphic pools is divided by the bankfull riffle width to calculate the dimensionless pool spacing ratio. The dimensionless ratio allows for the comparison of values from different sites and drainage areas. For example, a pool spacing of 75 feet is meaningless without an understanding of stream size or drainage area; however, a pool spacing ratio of 4.0 can be compared across drainage areas, as long as the values are from the same valley morphology, bed material, and boundary condition (Hey, 2006). The median pool spacing ratio from a sampling reach is entered as the field value into the WSQT. The median is used instead of the mean because the sample size per reach tends to be small with a wide range of values; it was thought that the median provided a better estimate of central tendency than the mean. Field testing of the SQT has shown that median values in the functioning range allow for pattern heterogeneity and do not incentivize designs with equal pool spacing.

Studies have documented a connection between pool spacing ratios and channel stability and complexity (Langbein and Leopold 1966; Gregory et al. 1994; Laub et al. 2012). If a meandering stream has a low pool spacing, the riffle length is also low and energy is transferred to the banks and sometimes the floodplain. Evaluations of numerous stream restoration and mitigation projects by members of the WSTT in North Carolina, New York, and other states have shown that sites constructed with low pool-spacing ratios resulted in excessive bank erosion and sometimes floodplain erosion.

In addition to the issues caused by low pool spacing outlined above, large pool spacing values are also problematic. A large pool spacing ratio essentially means that there are a small number of geomorphic pools in the reach. In alluvial valleys, this might mean that the reach is overly straight, and the habitat value is diminished because the length of pool habitat has been reduced. In colluvial or otherwise confined valleys, the lack of pools might mean there is not sufficient energy dissipation to achieve dynamic equilibrium.

Reference Curve Development:

Reference curves for Wyoming were based on analysis of the compiled geomorphic reference dataset described in Section 1.5. The compiled geomorphic reference dataset consists of 51 sites that report pool spacing ratio. Data collection methods measured pool spacing ratio between the head, or beginning, of sequential pools rather than between the deepest point of

sequential pools. The pool spacing calculations were revised to match the WSQT methodology based on maximum pool depth locations and station data from longitudinal profiles at each site.

The metric accounts for differences in stream size by using bankfull width as the denominator. Scaling by bankfull width accounts for the differences in stream size that may otherwise be relevant in determining pool spacing. Bankfull dimensions may vary based on differences in climate and runoff characteristics; however, bankfull regional curves can be used to calibrate field identifications (Dunne and Leopold 1978, Blackburn-Lynch et al. 2017).

Stratification by Rosgen stream type was used to account for the natural variability in pool spacing because it combines valley type and slope, which are known drivers of pool spacing (Knighton 1998). The compiled geomorphic reference dataset was assessed to determine whether stratifications based on drainage area or region were also appropriate (see discussion in Section 8.3). Trends in the data were not apparent for these variables, so they were not used to stratify data. Results stratified by stream type are shown in Table 8-1 and Figure 8-1. Note, two reference stream channels were identified as F stream types and two outliers were identified in their stream type groupings and removed from the analysis.

Table 8-1: Statistics for Pool Spacing Ratio from the Compiled Geomorphic Reference Dataset

Statistic	Pool Spacing Ratio by Stream Type				
	E	C	Cb	B	Bc
Number of Sites (n)	9	13	7	15	3
Average	6.9	4.5	3.9	4.5	5.3
Standard Deviation	3.2	1.9	1.6	2.5	3.4
Minimum	3.3	1.7	2.3	1.3	2.8
25th Percentile	4.5	3.2	2.7	2.4	3.4
Median	4.9	4.4	3.1	4.1	4.0
75th Percentile	9.0	6.1	5.1	6.7	6.6
Maximum	12.8	7.3	6.4	9.4	9.2

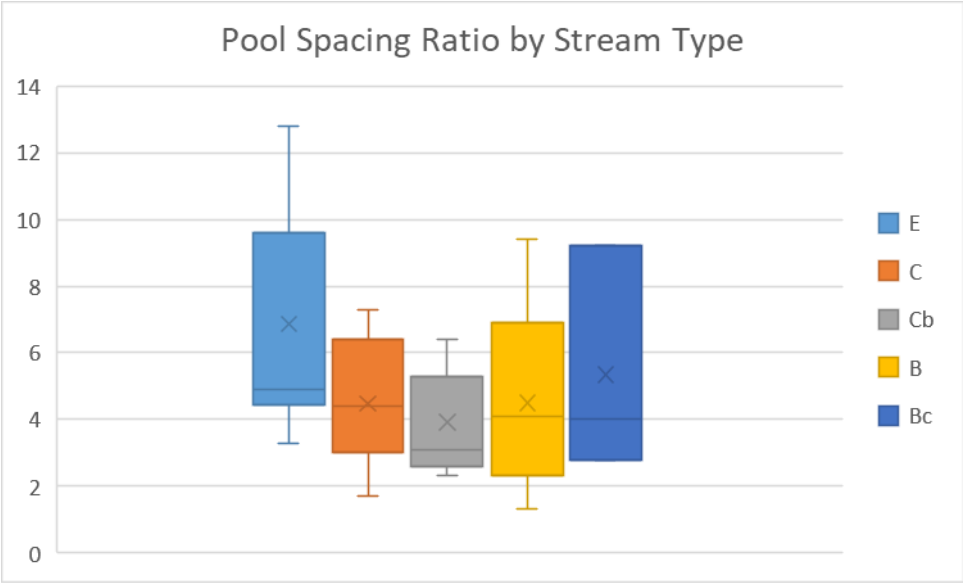

Figure 8-1: Box Plots for Pool Spacing Ratio from the Compiled Geomorphic Reference Dataset

Given that single-thread perennial streams exhibit a range of stable pool spacing, the WSTT combined the data analysis shown in Table 8-1 and Figure 8-1 with best professional judgement to derive the threshold values and reference curves shown in Table 8.2 and Figure 8-2.

The 25th and 75th percentile values for each stream type were used to characterize the reference standard range of index scoring in the WSQT. Modifications to the threshold values were considered using best professional judgement. The WSTT considered adjustments depending on whether the reference curves allowed for natural variability and did not incentivize homogeneous designs. Modifications to the threshold values were made as follows: If a value was considered too low and had the potential to cause stability problems, the value was increased. If the value was considered too high and would limit the number of pools, and therefore habitat, the value was lowered. For example, meandering streams (C and E) can have stability problems if the pool spacing is too low and habitat loss if pool spacing is too high. In moderate gradient streams (B), stability problems occur if the pool spacing is too far apart. However, an exception to this is projects that create a long succession of step-pools without a high enough percentage of riffles, i.e., the length is mostly pool.

Since the compiled geomorphic reference dataset was limited to reference standard streams, the not-functioning range was extrapolated from the reference curves fit to the threshold values identified in Table 8-2. These curves were reviewed to determine if the not-functioning values were reasonable based on stability and habitat considerations.

For C stream types, a two-sided reference curve was developed to account for stability issues associated with low ratios and habitat issues associated with high ratios. Field values of 4.0 and 6.0 were selected to equal an index value of 1.00. A 4.0 was selected rather than the 25th percentile value of 3.2 to provide more certainty that a sinuosity of 1.2 could be achieved. Likewise, a 3.7 was set as the 0.70 index value to equal the low end of the reference condition. As ratios become less than this, pool spacing is reduced and the potential for instability goes up. Experience from the authors have shown that low pool spacing values can lead to instability especially in newly constructed channels. The field value of 6.0 closely equates to the 75th

percentile value of 6.1 from the Compiled Geomorphic Dataset. A pool spacing ratio of 7.0 was set at the 0.70 based on discussions with the WSTT. The team decided that values greater than 7.0 would begin to equal fewer pools per reach and not support fish communities at a reference condition. A 7.0 is also very close to the maximum value observed in the reference data set.

For Cb stream types, field values of 3.8 and 5.0 were selected to equal an index value of 1.00, and 3.0 and 6.0 for the 0.70 index value. These values are slightly lower than the C stream type because steeper streams have a lower sinuosity and closer pool spacing. However, since a sinuosity of 1.2 or slightly higher is a possibility, the 1.00 was set higher than the 25th percentile value to help avoid stability problems. The upper end of 5.0 closely equates to the 75th percentile.

The logic for developing reference standards for E stream types is the same as the C since they exist in similar valley types. However, since the sinuosity is generally higher in E stream types, the pool spacing values can be lower. This is not evident in the reference data shown in Figure 8-1, but has been observed in E's across the country. Figure 8-1 may be different due to the low sample size and the resulting combination of E stream types with greatly different slopes and valley types. Until more data are collected, the WSTT decided that it was more conservative, from a stream stability and habitat perspective, to use the values shown in Table 8-2.

For B and Ba stream types, the 25th and 75th percentile values are 2.4 and 6.7, respectively. Generally, lower pool spacing values are better from a stability and habitat perspective if the riffle percentage is appropriate, e.g., too much pool length has been observed by the authors to create major instability problems. Therefore, the 0.70 index value was reduced to a 4.0 to encourage spacing ratios less than 6.7 and promote channel stability and habitat. Any value under 3.0 was set at an index value of 1.00 to discourage practitioners from over-structuring a stream if it wasn't needed for stability. A 0.00 index value was assigned to any field value over 7.5 due to the lack of pool habitat that this would create and potential instability (headcutting) that could occur. The logic is the same for Bc stream types, but the values were increased slightly to account for the lower slope. Lower slope streams can have pool spacing values that are slightly higher than their steeper counterparts without having stability problems.

Table 8-2: Threshold Values for Pool Spacing Ratio

Index Value	Field Values by Stream Type				
	E	C	Cb	B and Ba	Bc
1.00	3.5 – 5.0	4.0 – 6.0	3.7 – 5.0	3.0	3.4
0.70	3.0, 6.0	3.7, 7.0	3.0, 6.0	4.0	6.0
0.00	1.8, 8.3	3.0, 9.3	-	7.5	-

Linear relationships were fit to threshold values using the above criteria. Since both low and high pool spacing impact stability and complexity in meandering channels, either due to channel instability or reduced habitat length, respectively, the reference curves are parabolic shaped. Low values are not functioning and high values are not functioning. A middle range of values supporting stream stability and pool-habitat quality are considered functioning. These relationships are shown in Figures 8-2a and 8-2b. It is important to remember that the values in

the WSQT are medians; therefore, a range of values can be used in the design process. Field testing of the SQT has shown that median values in the functioning range still allow for pattern heterogeneity and do not incentivize designs with equal pool spacing.

Reference streams with moderate gradients (between 3 and 5%) have naturally lower pool spacing ratios, indicating an inverse relationship between slope and pool spacing (Whittaker 1987; Chin 1989). Unlike meandering streams, moderate gradient systems dissipate less energy laterally and more energy vertically. In moderate gradient streams, low ratios represent functioning conditions from a stability and habitat perspective. Therefore, the reference curves in Figures 8-2c and 8-2d do not show a loss of function with lower index values.

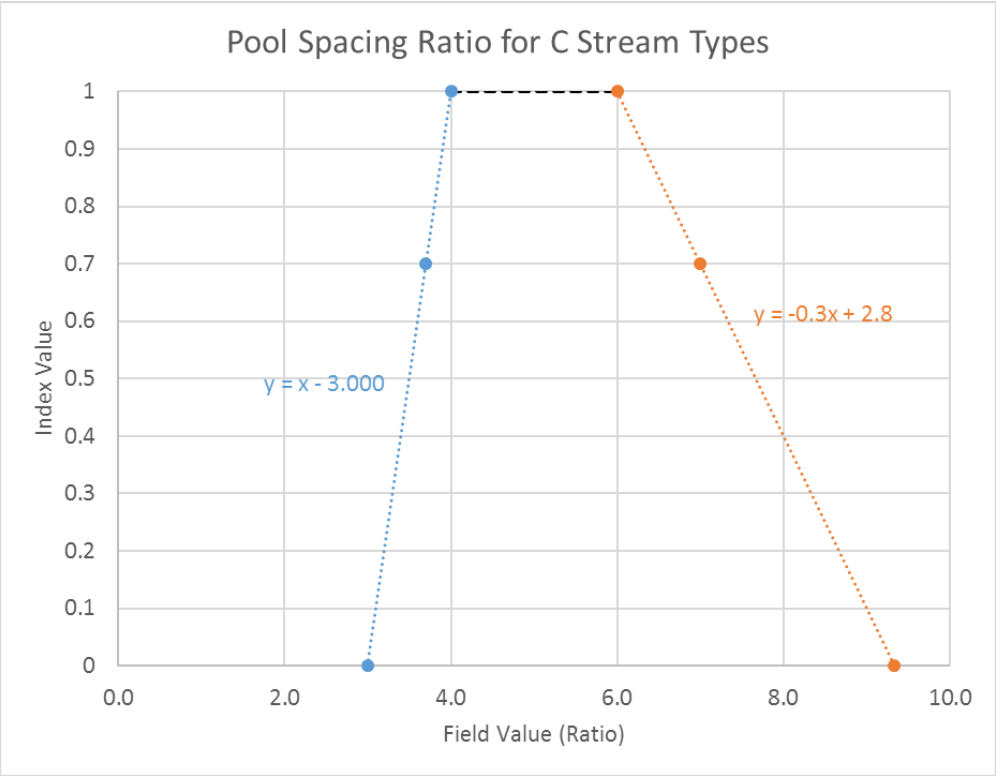

Figure 8-2a: Pool Spacing Ratio Reference Curves

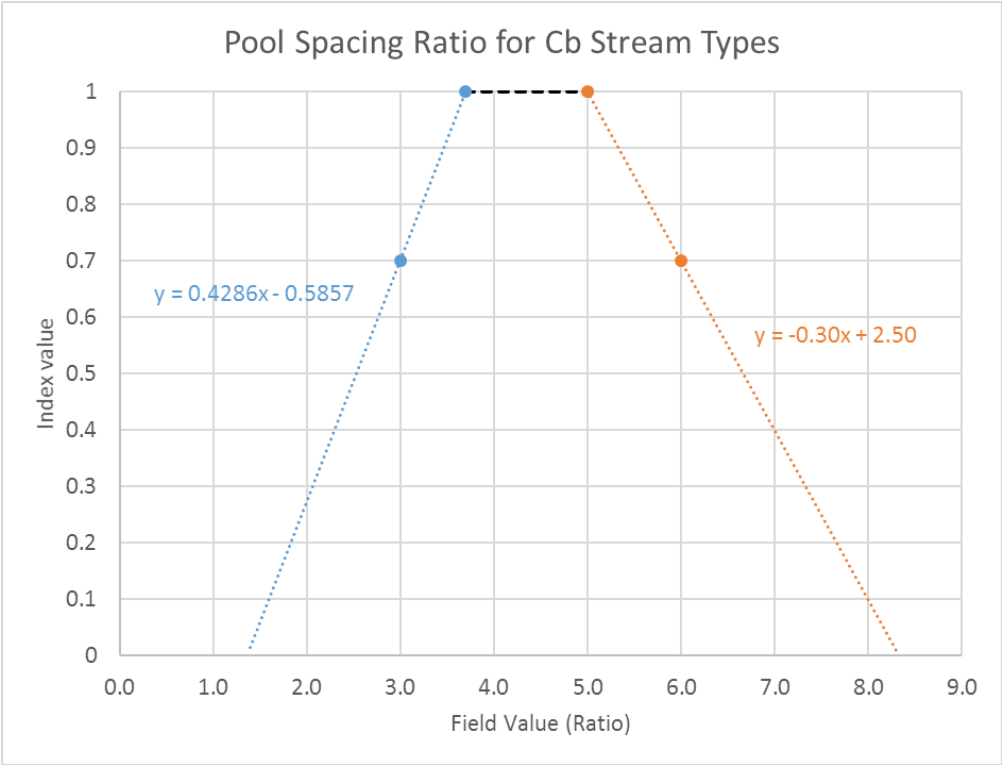

Figure 8-2b: Pool Spacing Ratio Reference Curves

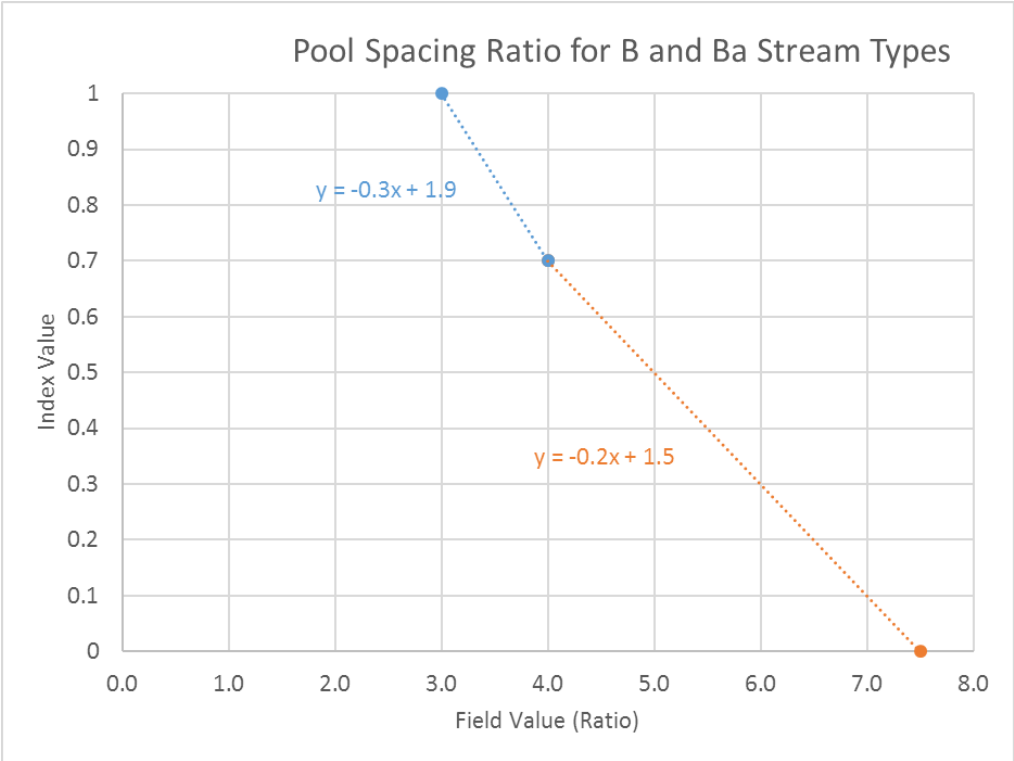

Figure 8-2c: Pool Spacing Ratio Reference Curves

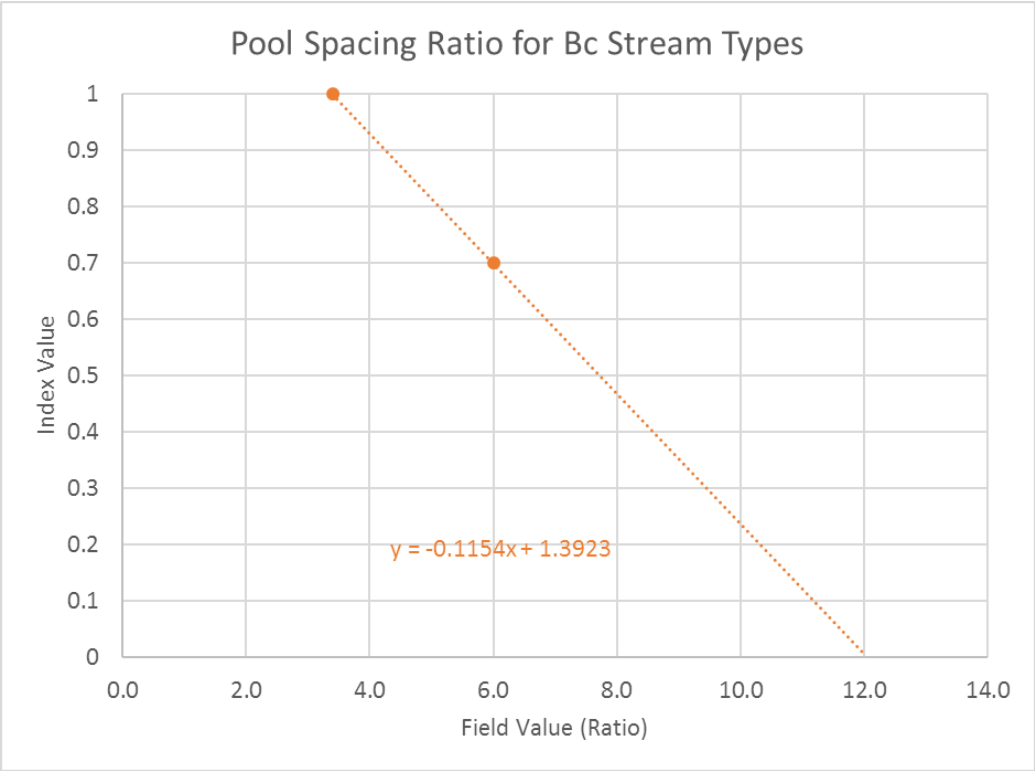

Figure 8-2d: Pool Spacing Ratio Reference Curves

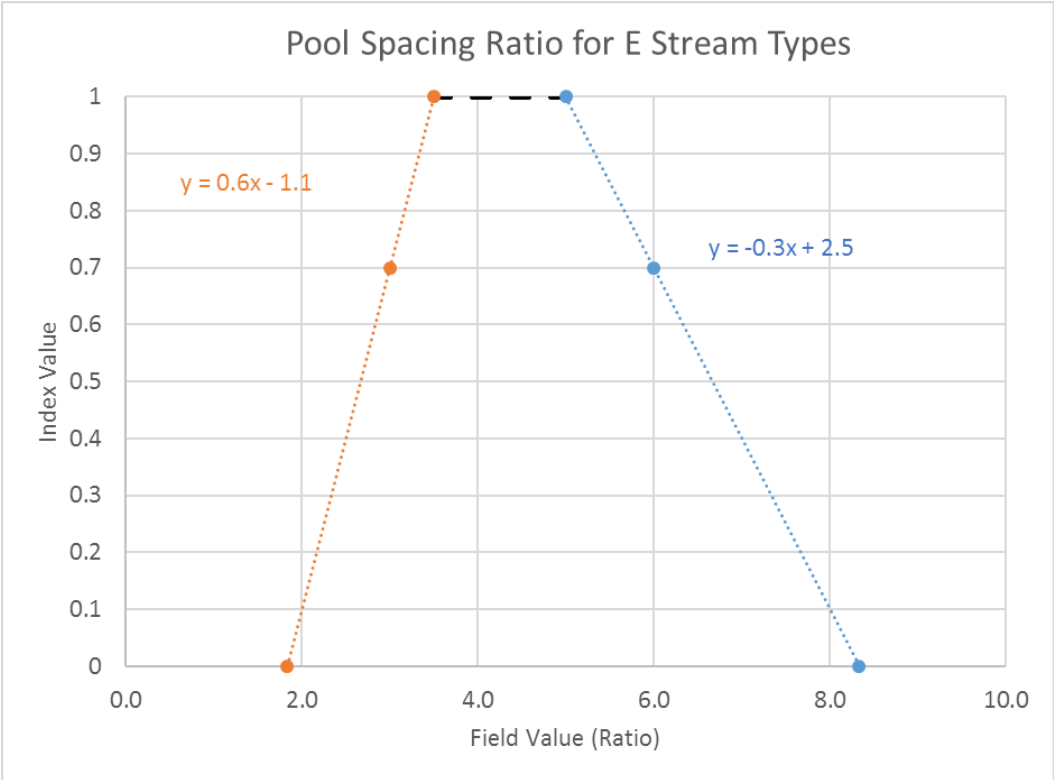

Figure 8-2e: Pool Spacing Ratio Reference Curves

Limitations and Data Gaps:

The presence of bedrock can influence pool spacing, and thus it may not be appropriate to include bedform diversity metrics when evaluating natural bedrock channels in the WSQT. Pool spacing and development in bedrock channels is controlled by the nature of the rock material, e.g., fractures, as opposed to lateral dissipation of energy through a meandering channel. This consideration is only applicable to channels that are dominated by bedrock (e.g., bedrock is the median size of the bed material) and not channels that simply have bedrock outcrops.

If bankfull dimensions are not accurately determined for a site, then the pool spacing ratio may not accurately reflect the bedform diversity. When possible, localized regional curves should be used to verify the bankfull determination. Once a bankfull feature/stage has been determined, that feature/stage should be used for all future assessments. Additional information on verifying bankfull information was added to the WSQT v1.0 User Manual in response to comments received during beta testing.

Reference curves were not developed for naturally occurring F and G stream types. If the stream is a naturally occurring F stream type, e.g., located in a canyon or gorge setting, this metric should not be evaluated, as no reference curves have been developed for this stream type. Additionally, this metric is not applicable to braided (D) stream types with multiple channels or ephemeral channels because a predictable pool spacing is not typically found in these environments (Bull and Kirby 2002).

F and G channels that represent degraded streams should be compared against the proposed, or reference stream type, as informed by channel evolution processes (Cluer and Thorne 2014; Rosgen 2014) and described in the WSQT v1.0 User Manual. For example, if the existing stream type is a degraded Gc in an alluvial valley, the proposed / reference stream type and reference curve would be a C or E. Selection of the appropriate reference stream type is important for consistently applying this metric and determining a condition score in the tool. To improve consistency, additional guidance has been added to the WSQT v1.0 User Manual to assist practitioners in identifying the reference stream type.

The reference curves were derived using a geomorphic reference dataset primarily from the mountainous regions of Wyoming. Additional testing is desirable to determine whether different reference curves will be necessary for the basins and plains regions.

This metric stratifies reference curves by Rosgen stream type. Other geomorphic classification approaches may also be appropriate for stratifying reference curves for this metric (Buffington and Montgomery 2013), and may broaden the applicability of this metric to additional morphologies common to Wyoming. Additional data collection and analyses would be required to adapt the tool and reference curves for use with other classification approaches.

8.2. Pool Depth Ratio

Summary:

This metric measures the bankfull depth of the deepest point of each pool within the sampling reach. All pools, including both geomorphic pools and micro-pools, are included in this metric (note: this is different than the pool spacing metric above). The bankfull pool depth is normalized

by the bankfull mean riffle depth to calculate the dimensionless pool depth ratio. The average pool depth ratio from a sampling reach is entered as the field value into the WSQT. The average is used instead of the median because typically the sample size is larger and the range lower than the pool spacing ratio.

Pools provide fish habitat and thermal refugia, support thermal regulation, provide energy dissipation, and are an indication of how the stream is transporting and storing sediment (Knighton 1998; Allan 1995; Hauer and Lamberti 2007). For example, if the outside meander bend has filled with sediment, this can be an indication of an aggradation problem. The channel cannot transport the sediment load through the meander bend. In combination with pool spacing ratio and percent riffle metrics, the pool depth ratio characterizes the bed form diversity of a stream reach (Harman et. al. 2012).

Reference Curve Development:

Reference curves for Wyoming were based on analysis of the compiled geomorphic reference dataset described in Section 1.5. The compiled geomorphic reference dataset consists of 54 sites that report pool depth ratio. The dataset was assessed to determine whether stratification based on stream type, bed material, slope, or region (see discussion in Section 8.3) were appropriate. Scaling for stream size is accounted for in the metric by using the bankfull mean depth as the denominator.

Differences in slope and region were not apparent, and only slight differences were noted based on stream type or bed material (Figure 8-3). The median values for Rosgen C, B, and E stream types are similar, but there is slightly more variability between the 75th percentiles and the minimum and maximum values. For bed material, there is a slightly higher median value for cobble-bed streams, but the range of depths is higher for the gravel-bed streams. Note that there were no sand bed streams in the dataset.

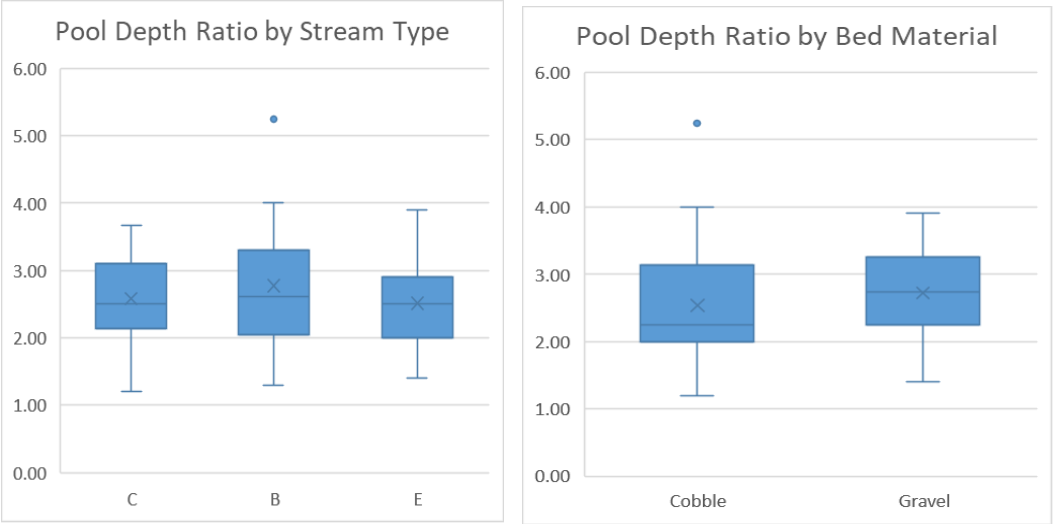

Figure 8-3: Box Plots for Pool Depth Ratio from the Compiled Geomorphic Reference Dataset

Because there were no meaningful differences in pool depth ratio based on stream type or bed material, one reference curve was implemented for all streams without stratification. The statistics for the compiled geomorphic reference dataset are provided in Table 8-3.

Table 8-3: Statistics for Pool Depth Ratio from the Compiled Geomorphic Reference Dataset

Statistic	Pool Depth Ratio
Number of Sites (n)	54
Average	2.7
Standard Deviation	0.8
Minimum	1.2
25th Percentile	2.2
Median	2.5
75th Percentile	3.2
Maximum	5.3

Using the premise that deep pools have greater ecological benefits than shallow pools, the threshold for the lower end of the functioning range was set at 2.2 to match the 25th percentile from the reference standard dataset. The minimum index value of 0.00 was set at 1.0, which means that no pools occurred that were greater than the bankfull mean depth. The maximum index value (1.00) was determined using the 75th percentile of 3.2. Because all data in Table 8-3 came from reference standard reaches, no threshold value was selected for the functioning-at-risk and non-functioning ranges. Threshold values are shown in Table 8-4. A broken linear relationship was fit to the identified threshold values to develop the reference curve (Figure 8-4).

Table 8-4: Threshold Values for Pool Depth Ratio

Index Value	Field Value
1.00	3.2
0.70	2.2
0.00	1.0

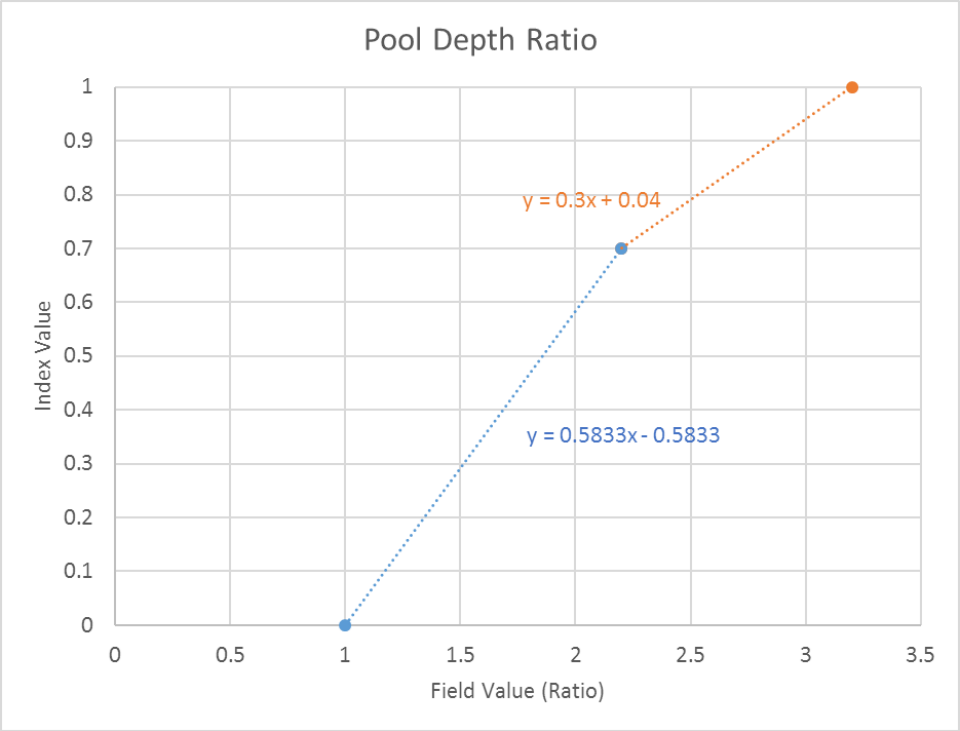

Figure 8-4: Pool Depth Ratio Reference Curve

Limitations and Data Gaps:

If bankfull dimensions are not accurately determined for a site, then the pool depth ratio will not be accurate. Additional discussion on this limitation is provided in Chapter 1.5 and Chapter 8.1.

The compiled geomorphic reference dataset used to derive the reference curves is from single-thread, perennial streams in the mountainous regions of Wyoming. Testing is desirable to determine whether additional or modified reference curves are needed for the basins and plains regions, and in intermittent, ephemeral and braided systems. Sand bed streams may have lower pool depth ratios, but should be evaluated using the WSQT recognizing that the current reference curves may not accurately characterize the level of functioning in these systems.

8.3. Percent Riffle

Summary:

This metric measures the length of riffles (including runs) within the sample reach. The total length of riffles and runs is divided by the total reach length to calculate the percent riffle.

Pools and riffles are valuable habitat and both are needed to support various aquatic species and dissipate energy within a reach. The riffle is the natural grade-control feature of the stream, providing floodplain connection and vertical stability (Knighton 1998). The pool provides energy dissipation, habitat diversity, and more. Much of the discussion regarding stream function presented in the pool spacing ratio and pool depth metric summaries applies to this metric as well. While the pool spacing ratio quantifies the frequency of pools within a reach, this metric

quantifies the relative prevalence of riffle habitat length throughout the reach. Streams that have too much riffle length also have a low percentage of pools. Conversely, streams that have a low percentage of riffle also have a high percentage of pool. The appropriate proportion of riffles and pools is necessary to support dynamic equilibrium and habitat for in-stream biota. Percent riffle works with the pool spacing and pool depth ratio metrics to characterize bed form diversity.

Reference Curve Development:

Reference curves for Wyoming are based on analysis of the compiled geomorphic reference dataset described in Section 1.5. The dataset included profile data that identified bed features, and these data were used to calculate a percent riffle for each of 51 reference sites in the mountainous regions of Wyoming; one site was removed as an outlier.

The compiled geomorphic reference dataset was assessed using various possible stratifications including bioregion, stream type, drainage area, slope and bed material. Streams from the Volcanic Mountains and Valleys bioregion (Volcanic Region) had higher percent riffle values than the rest of the data (Table 8-5). Based on this observation, the decision was made to develop unique reference curves for the Volcanic Region. This stratification was explored for other bed form diversity metrics as well, but the WSTT concluded that the result did not warrant separate stratification for these other metrics.

Once the data were stratified by bioregion, the dataset was evaluated for differences in percent riffle based on other factors. Trends in percent riffle based on stream type, bed material and drainage area were not observed in the data; differences in percent riffle were observed in streams of different slope. For the sites outside the Volcanic Region, channels with higher slopes had more riffle length. A 3% slope break matches well with other literature showing that mountain streams with slopes greater than 3% often have stair-like appearance (Chin 1989; Abrahams et al. 1995) and are riffle dominated. These trends matched professional experience of the WSTT. Stratification for this metric included the volcanic and non-volcanic regions, with streams outside the volcanic region also stratified by slope (Table 8-5).

Table 8-5: Statistics for Percent Riffle from the Compiled Geomorphic Reference Dataset

Statistic	Percent Riffle (%)		Percent Riffle (%) Volcanic Region
	Slope < 3%	Slope ≥ 3%	
Number of Sites (n)	20	6	24
Geomean	52	72	80
Average	55	73	81
Standard Deviation	18	8	10
Minimum	28	60	61
25th Percentile	39	68	76
Median	57	74	82
75th Percentile	69	78	89
Maximum	88	83	95

The WSTT identified the thresholds presented in Table 8-6 using the stratification and data outlined above:

- For stream with low slope (< 3%), the functioning range of scoring was set equal to the interquartile range observed in the compiled geomorphic dataset. The maximum index score within the functioning range was determined using best professional judgement.
- The number of sites with a slope of 3% or greater was limited and the WSTT used best professional judgement to set the functioning range of scoring equal to the range of values observed in the dataset. The maximum index score within the functioning range was set equal to the interquartile range observed in the compiled geomorphic dataset.
- For streams within the volcanic ecoregion, the functioning range of scoring was set equal to the interquartile range observed in the compiled geomorphic dataset. The maximum index score within the functioning range was determined using best professional judgement.
- Since the compiled geomorphic reference dataset was limited to reference streams, the not functioning range was determined by extrapolating the curves.

The best-fit relationship for percent riffle is a two-sided reference curve, which reflects less function in systems where there is both very high or very low percent riffle. Channel stability and macroinvertebrate habitat can be negatively affected by low percent riffle, and fish habitat can be negatively affected by high percent riffle (Clifford and Richards 1992). Linear relationships were fit to identified threshold values (Figure 8-5). The results were reviewed by the WSTT to determine the applicability and appropriateness of percent riffle's role in supporting bedform diversity.

Table 8-6: Threshold Values for Percent Riffle

Index Value	Field Value (%)		
	Slope < 3%	Slope ≥ 3%	Volcanic Region
1.00	50 – 60	68 – 78	73 – 80
0.70	39, 69	60, 83	61, 82

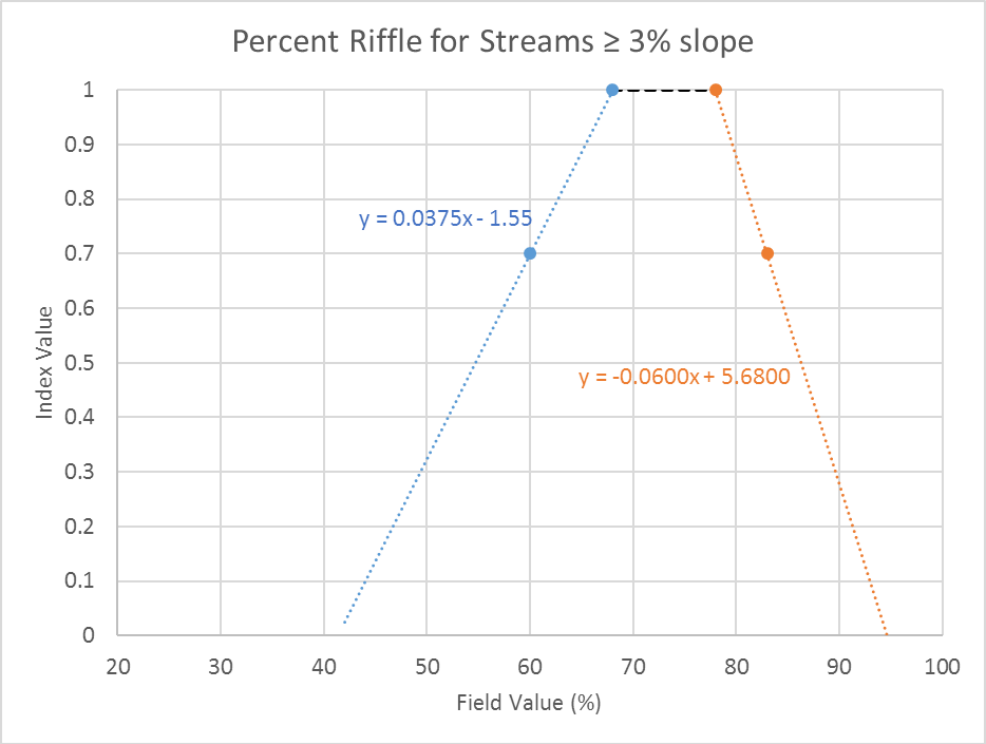

Figure 8-5a: Percent Riffle Reference Curves

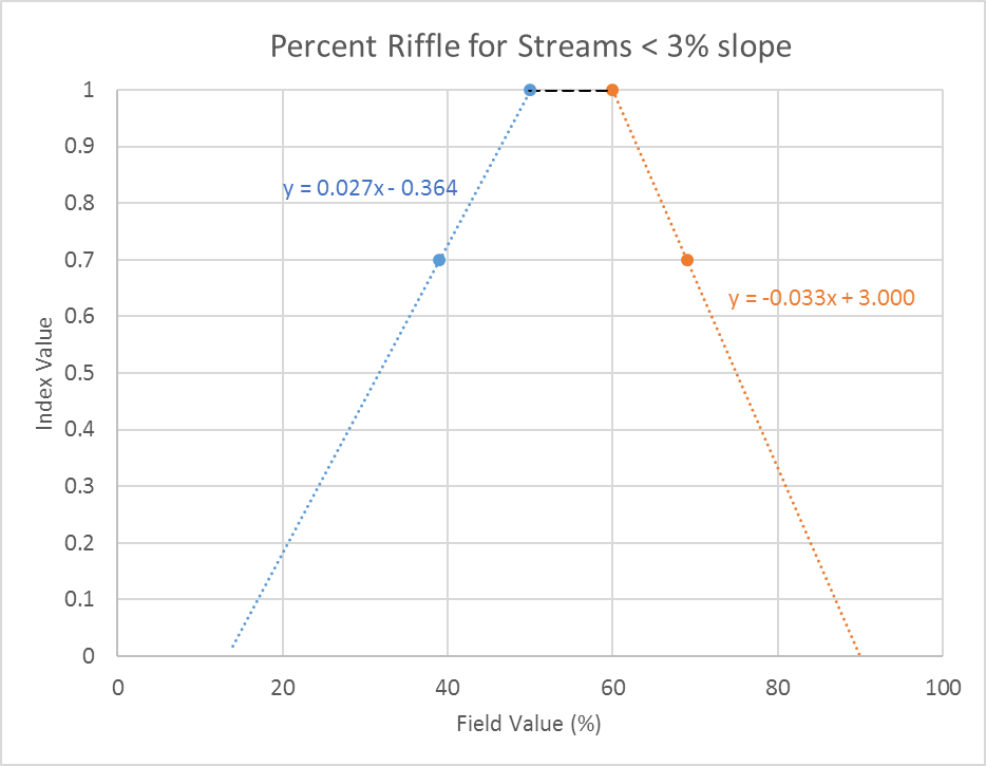

Figure 8-5b: Percent Riffle Reference Curves

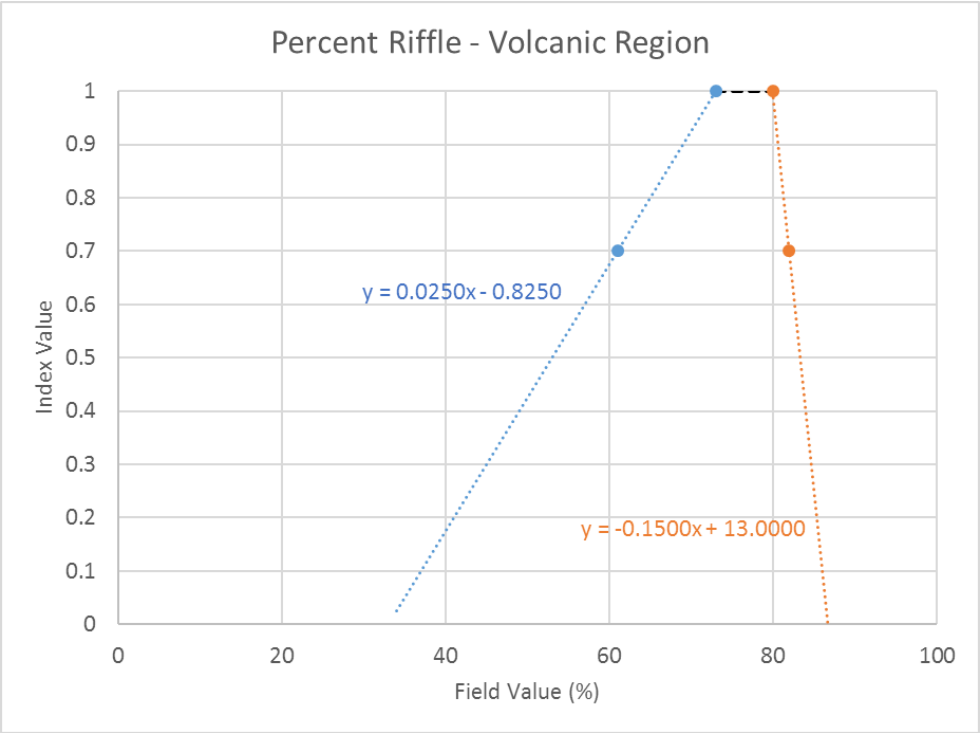

Figure 8-5c: Percent Riffle Reference Curves

Limitations and Data Gaps:

The compiled geomorphic reference dataset that was the primary reference in deriving the reference curves is from the mountainous regions of Wyoming and testing is needed to determine whether different reference curves will be necessary for the basins and plains regions.

8.4. Aggradation Ratio

Summary:

The width depth ratio (WDR) is a ratio of the bankfull width to the mean depth of the bankfull channel (Rosgen 2014). Within the assessment segment, each riffle exhibiting signs of excessive deposition should be surveyed and the WDR calculated. The aggradation ratio is calculated as the WDR measured at the widest riffle in the sampling reach divided by a reference WDR.

The aggradation ratio is described by Rosgen (2014) as the Width Depth Ratio State to assess departure from a reference condition caused by streambank erosion, excessive deposition, or direct mechanical impacts that lead to an over-wide channel. Aggradation of sediments within a channel is a natural fluvial process, but excessive aggradation can be an indicator of sediment imbalance, where upstream sediment supply exceeds the stream's transport capacity. Accumulation of sediments in pools would result in a lower pool depth ratio, and similarly, a higher width depth ratio is an indicator of sediment accumulation in the riffle areas of the reach. Note that channel degradation, where sediment transport exceeds supply, is captured in

floodplain connectivity with the bank height and entrenchment ratios. As such, this metric only evaluates the range of width-depth ratios larger than the reference range even though guidance has been developed for WDRs both larger and smaller than reference WDRs (Rosgen 2014).

Reference Curve Development:

The channel stability descriptions presented by Rosgen (2014) are provided in Table 8-7. These categories were related to functional capacity based on stability, since stable conditions have a low likelihood of riffle aggradation. The translation from stable to functional capacity is as follows: functioning (stable), functioning-at-risk (moderately stable) and non-functioning (unstable and highly unstable). Any aggradation ratio within the highly unstable range of field values was also considered to be non-functioning and assigned an index score of 0.00. Thresholds are shown in Table 8-8. A polynomial relationship was fit to identified threshold values (Figure 8-7).

Table 8-7: Width Depth Ratio State Categories (Rosgen 2014)

Aggradation Ratio	Stability Rating
1.0 – 1.2	Stable
1.2 – 1.4	Moderately Stable
1.4 – 1.6	Unstable
1.6 – 1.8	Highly Unstable

Table 8-8: Threshold Values for Aggradation Ratio

Index Value	Field Value
1.00	1.0
0.69	1.2
0.30	1.4
0.00	1.6

Calculating the field value for this metric requires an expected, or reference value. WDR is a primary metric in determining the Rosgen stream type: streams with a very low WDR (A, G and E streams) have WDR values less than 12; streams with a moderate to high WDR have a WDR >12. Streams with a naturally high WDR greater than 40 are multiple thread channels (Rosgen 1996). A low WDR leads to an increase in shear stress (holding slope and cross-sectional area constant) and sediment transport capacity because a lower WDR increases the hydraulic radius. Conversely, as the WDR increases and the mean depth decreases, sediment transport capacity decreases. This increases the risk of aggradation. The aggradation ratio metric is normalized across stream types by using a reference WDR as the denominator in the ratio.

Statistics from the compiled geomorphic reference dataset are shown in Table 8-9 and Figure 8-6. Statistics from the compiled geomorphic reference dataset are also provided in the WSQT v1.0 User Manual to inform the selection of a reference WDR. At reference sites, at least one riffle is surveyed that represents conditions typical of the reach. The dimensions from that representative, or stable, riffle are used to calculate the dimensionless ratios provided in this manual and in Table 8-9 below. The values provided from the geomorphic reference dataset are stratified by Rosgen stream type. Note that sites within the compiled geomorphic reference dataset that had a WDR greater than 40 were excluded from this analysis.

Table 8-9: Statistics for WDR from the Compiled Geomorphic Reference Dataset

Statistic	WDR by Stream Type		
	B	C	E
Number of Sites (n)	17	22	9
Average	20	20	8
Standard Deviation	8	7	2
Minimum	9	6	5
25th Percentile	16	16	7
Median	18	20	8
75th Percentile	23	23	10
Maximum	35	37	12

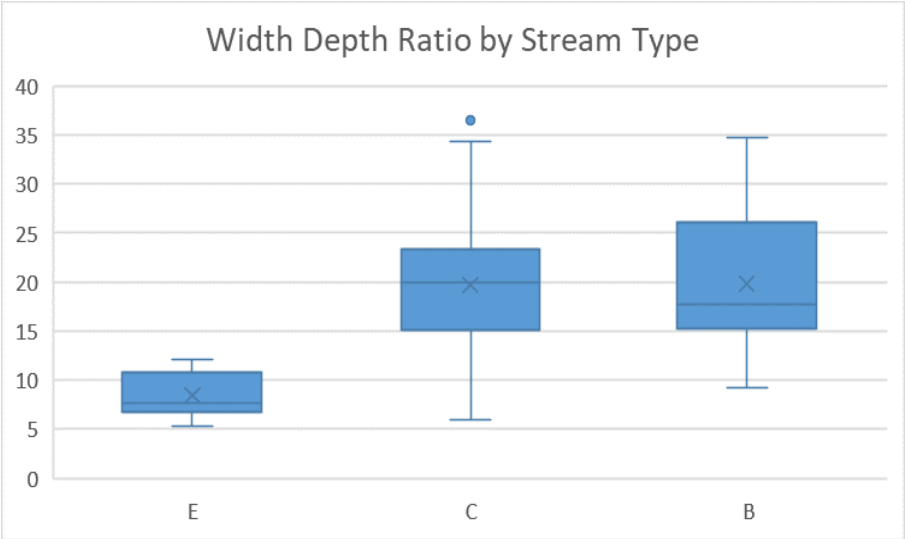

Figure 8-6: Box Plots for WDR from the Compiled Geomorphic Reference Dataset

Since the WDR can play a large role in the design process and is often linked to slope and sediment transport assessments, strict reference curves were not developed. This method requires that a reference WDR be established for the proposed stream restoration design and

that increases from that reference value results in a loss of function according to the reference curve shown in Figure 8-7. A reference WDR will likely require review during the design and/or permitting process. To improve the consistency of this metric, guidance for determining reference values (denominator) for each stream type was developed from an analysis of the compiled geomorphic reference dataset.

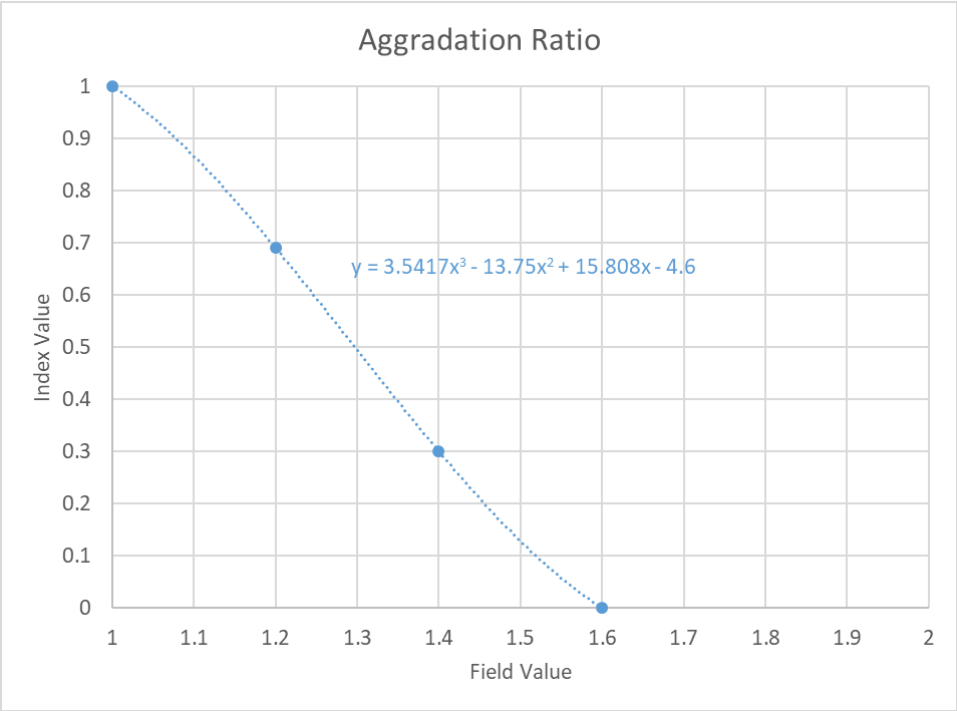

Figure 8-7: Aggradation Ratio Reference Curve

<u>Limitations and Data Gaps:</u>

If bankfull dimensions are not accurately determined for a site, then the pool spacing ratio will not be accurate. When possible, localized regional curves should be used to verify the bankfull determination.

Aggradation ratio is not applicable to braided (D) stream types since the width of the channels is often the same as the valley width (Rosgen 2009). Also, guidance is not provided for selecting a reference value for all stream types. There were not enough, or any, F and G channels in the compiled reference dataset to determine a reference WDR. Unless naturally occurring, these channel types are typically degraded streams. Additional discussion on these limitations is provided in Chapter 1.5 and Chapter 8.1.

Chapter 9. Planform Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Channel pattern or planform is the horizontal positioning of the stream on the landscape; it can be thought of as the aerial or bird's eye view of the channel. There are multiple classification schemes that characterize the nature of the planform geometry, including: straight, meandering, braided, anastomosed, and other sub-classifications (Leopold and Wolman 1957; Schumm 1985). Within these classifications, there are many ways to quantify pattern, including sinuosity, meander wavelength, radius of curvature, belt width, and amplitude (Knighton 1998; Leopold et al. 1992; Copeland et al. 2001; Rosgen 2014).

Channel pattern affects flow resistance primarily by effecting channel slope (Knighton 1998). Sinuosity is a direct and integrating measure of channel pattern and is typically quantified as stream length divided by valley length between two fixed points along the channel. Therefore, as sinuosity increases, the average channel slope decreases. Sinuosity can be measured in each of the classifications listed above, but is mostly used in single-thread meandering streams.

Metric:

- Sinuosity

9.1. Sinuosity

Summary:

Sinuosity is a measure of how much a channel meanders within its valley. This metric is calculated by dividing the stream centerline length by the valley length. Somerville (2010) found sinuosity as one of the most commonly assessed metrics for stream restoration.

The relationship between sinuosity and stream function is better established than other metrics that are often used in the design process for creating planform designs, but which are not as intrinsically linked to stream function. Channelization, which involves the straightening and enlargement of natural channels, has numerous negative effects on stream functions, including increased flow velocity, accelerated erosion of the bed and banks, loss of habitat features (riffles and pools), diminished connection to the floodplain, and reduced nutrient retention and hyporheic exchange (Bernhardt et al. 2005). Reestablishing the appropriate sinuosity in valleys that support meandering streams may not return all of these functions to a reference condition. However, reestablishing sinuosity will support key processes, such as hyporheic exchange, organic matter retention, and the development of in-stream habitats, such as riffle-pool sequences (Winter et al. 1998; Knighton 1998). Furthermore, increasing sinuosity increases stream length and thereby stream habitat. The United States has lost hundreds of thousands of stream miles throughout its history (Wohl 2004), and re-establishing sinuosity is one way to regain stream length/habitat.

Reference Curve Development:

Reference curves for Wyoming were based on analysis of the compiled geomorphic reference dataset and NRSA dataset described in Section 1.5. These datasets were evaluated for

potential stratification by stream type and valley type, as the sinuosity of a stream varies by the shape and slope of the valley.

The compiled geomorphic reference dataset consists of 56 sites that report sinuosity. This dataset was stratified by Rosgen stream type (Table 9-1 and Figure 9-1). Valley characteristics were not included in this dataset. Rosgen identifies three classes of sinuosity: Low sinuosity < 1.2, Moderate 1.5 > sinuosity > 1.2, and High sinuosity > 1.5. E stream types and sometimes C stream types are highly sinuous while B, F, and G stream types have moderate sinuosity, and A stream types have low sinuosity.

Table 9-1: Statistics for Sinuosity from the Compiled Geomorphic Reference Dataset

Statistic	Sinuosity by Stream Type		
	B	C	E
Number of Sites (n)	22	25	9
Mean	1.22	1.29	1.50
Standard Deviation	0.11	0.23	0.38
Minimum	1.07	1.00	1.10
25th Percentile	1.17	1.10	1.20
Median	1.20	1.30	1.40
75th Percentile	1.26	1.40	1.70
Maximum	1.50	2.00	2.10

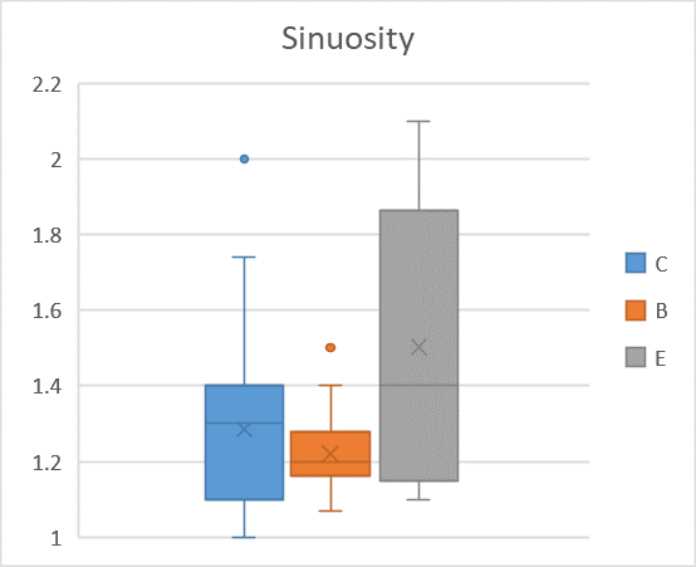

Figure 9-1: Box Plots for Sinuosity from the Compiled Geomorphic Reference Dataset

Analysis using the NRSA dataset was performed on 156 sites from Wyoming and the surrounding states. Sites with a single thread channel pattern (attribute CONPATTERN) were selected and identified as reference or degraded (attribute RT_NRSA). The dataset identifies the channel constraint type (attribute CONSTRAINT) as "…constrained within a narrow valley [CON_VSHAPED], constrained by local features within a broad valley [CON_BROAD], unconstrained and free to move about within a broad floodplain [UNC_BROAD], or free to move about, but within a relatively narrow valley floor [UNC_NARROW]." (USEPA, 2009). The resulting data stratified by valley constraint and condition, reference condition (R) and degraded (D), are shown in Table 9-2 and Figure 9-2. Sites with sinuosity values greater than 3 were considered outliers and removed from the dataset.

Table 9-2: Statistics for Sinuosity from the NRSA Dataset

	Sinuosity by Valley Constraint and Condition							
	CON_BROAD		CON_VSHAPED		UNC_BROAD		UNC_NARROW	
Statistic	**R**	**D**	**R**	**D**	**R**	**D**	**R**	**D**
Number of Sites (n)	46	56	11	22	5	9	1	3
Average	1.48	1.35	1.23	1.23	1.54	1.46	1.07	1.47
Standard Deviation	0.50	0.39	0.45	0.13	0.45	0.51	-	0.34
Minimum	1.01	1.01	1.01	1.05	1.05	1.06	-	1.16
25th Percentile	1.10	1.06	1.04	1.13	1.15	1.08	-	1.29
Median	1.24	1.17	1.05	1.22	1.51	1.20	1.07	1.41
75th Percentile	1.92	1.52	1.14	1.27	1.99	1.61	-	1.62
Maximum	2.76	2.49	2.50	1.50	2.00	2.31	-	1.83

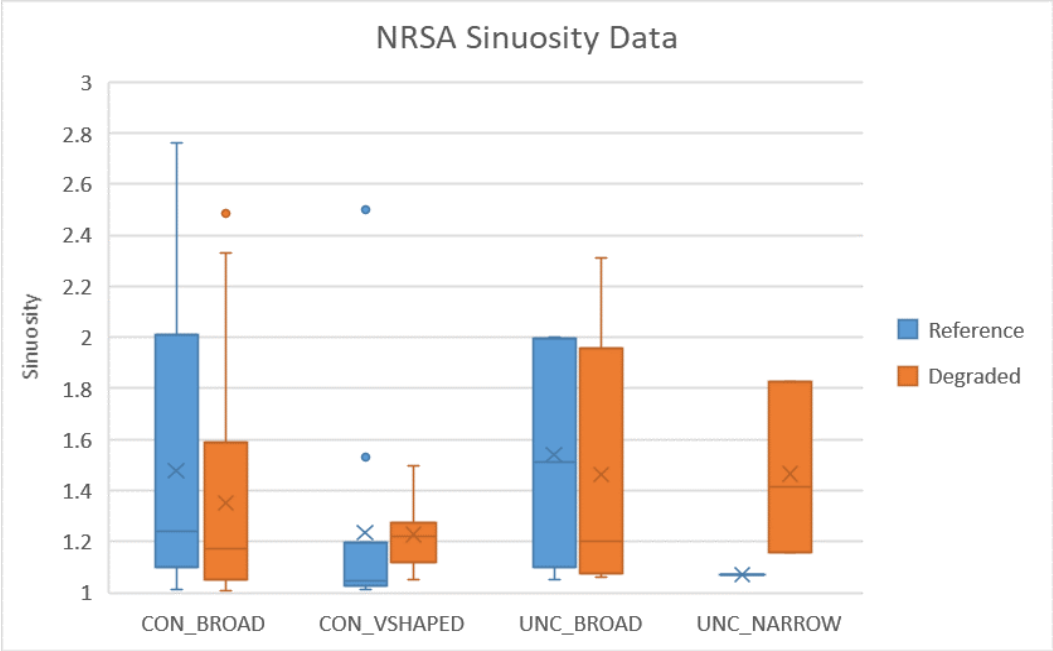

Figure 9-2: Box Plots for Sinuosity from the NRSA Dataset

Since the compiled geomorphic reference dataset included only sites within montane ecoregions, stratification by ecoregion and valley type was investigated using the NRSA dataset. The valley types described in the NRSA dataset were related to the WSQT valley types as follows: unconfined broad valleys are characterized as unconfined alluvial valleys; confined broad valleys are characterized as confined alluvial valleys; and narrow valleys (confined and unconfined) are characterized as colluvial valleys.

The WSTT decided to stratify by valley type, with an exception for highly sinuous channels (E stream types). Stratification by stream type was considered but rejected because streams in naturally confined alluvial valleys may have naturally lower sinuosity and still be classified as C or E stream types. Ecoregion was considered as an additional stratifier, but there were not enough data points to determine meaningful differences when data were stratified by valley type and ecoregion. Sinuosity values from the NRSA dataset within broad valleys in montane ecoregions did not differ substantially from values within basins or plains ecoregions. The range and measurement of sinuosity from reference standard streams in the basin was less than other ecoregions, potentially due to the small number of basin sites or bedrock constraints. In narrow valleys, sinuosity values were higher in reference standard sites in basins and plains, compared with mountains, although the sample sizes were too small to draw meaningful inferences.

Based on the results from the geomorphic reference dataset and NRSA dataset, the WSTT proposed the following thresholds (Table 9-3). Given that both high and low values of sinuosity can lead to stream instability and loss of function, the data were assessed to determine both high and low ranges for reference curves.

For streams that typically occur in unconfined alluvial valleys:

- C type reference standard streams in the geomorphic reference dataset have sinuosity between 1.1 (25th percentile) and 1.4 (75th percentile), with a median value of 1.3.

- The NRSA data do not identify stream type but the trend for reference standard stream sinuosity in broad valleys was similar, with a median value of 1.51 in unconfined alluvial valleys.
- The threshold values for the functioning range of sinuosity values in unconfined alluvial valleys was set to 1.2 and 1.5 based on both datasets. All values within this range are assigned an index value of 1.00. While it is recognized that a sinuosity of 1.5 may have more functional benefit than a 1.2 because it will have a greater stream length and therefore more aquatic habitat, more organic matter and flow retention, greater hyporheic exchange, and potentially increase floodplain inundation, greater sinuosity could also negatively affect sediment transport processes. Ultimately, the WSTT decided to assign a 1.00 to a range of sinuosity values to dis-incentivize practitioners from designing to a specific sinuosity value, which could create homogeneous patterns and potentially lead to stability problems.
- The threshold between functioning-at-risk and non-functioning index values was set at 1.15 based on consideration of the 25th percentile values from both datasets.
- A two-sided reference curve was developed, as streams with naturally higher sinuosity would likely be E stream types which have their own reference curve.

For streams that typically occur in confined alluvial valleys:

- Reference standard streams tend to have greater sinuosity than degraded streams (Figure 9-2). Since natural valley constraints can limit sinuosity in these settings, the reference curve is one sided rather than bell shaped.
- Sinuosity value greater than or equal to 1.2 was considered functioning, based on the median value (1.24) of NRSA reference standard sites in confined alluvial valleys. All values greater than 1.2 are assigned an index value of 1.00.
- A sinuosity of 1.0 was used to define the minimum index value of 0.00.

For streams that typically occur in narrow, colluvial valleys:

- Almost all B type reference standard streams have sinuosity less than 1.30, with a median of 1.20 and 75th percentile of 1.26. The sample size in the NRSA dataset was small but confirms that reference streams in v-shaped valleys have low sinuosity, with a median of 1.05 and 75th percentile of 1.27. The threshold values for the functioning range of sinuosity values in colluvial valleys was set to 1.1 and 1.3 based on both datasets. All values within this range are assigned an index value of 1.00 for the same reasons discussed above for unconfined alluvial valleys.

For E type streams:

- E type reference standard streams in the geomorphic reference dataset have greater sinuosity, with a median of 1.4 and most sites between a sinuosity of 1.2 (25th percentile) and 1.7 (75th percentile). E stream types were considered to represent the upper ranges of sinuosity in confined and unconfined alluvial valleys from the NRSA dataset. The threshold values for the functioning range of sinuosity values in E stream types was set to 1.3 and 1.8 based on both datasets. All values within this range are assigned an index value of 1.00 for the same reasons discussed above for unconfined alluvial valleys.
- Professional judgement was used to synthesize data from both sources and conclude that values between 1.8 to 2.0 and 1.2 to 1.3 were functioning at risk for E stream types.

Linear curves were fit to the thresholds identified in Table 9-3 to create the reference curves shown in Figure 9-3.

Table 9-3: Threshold Values for Sinuosity

Index Value	Field Values by Valley Type			Field Values for E Stream Types
	Colluvial	Confined Alluvial	Unconfined Alluvial	
1.00	1.10 – 1.30	≥ 1.20	1.20 – 1.50	1.30 – 1.80
0.70	1.09, 1.31	1.19	1.19, 1.51	1.29, 1.81
0.30	1.00, 1.40	-	1.15, 1.60	1.20, 2.00
0.00	-	1.00	-	-

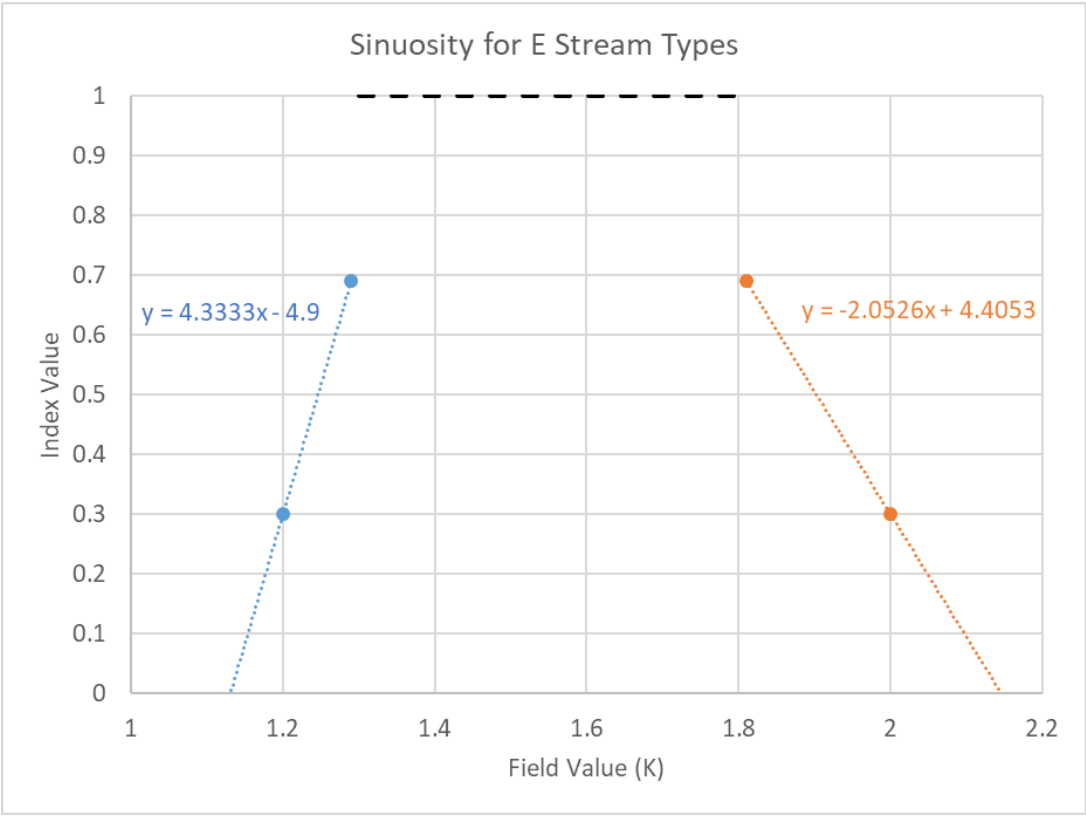

Figure 9-3a: Sinuosity Reference Curves

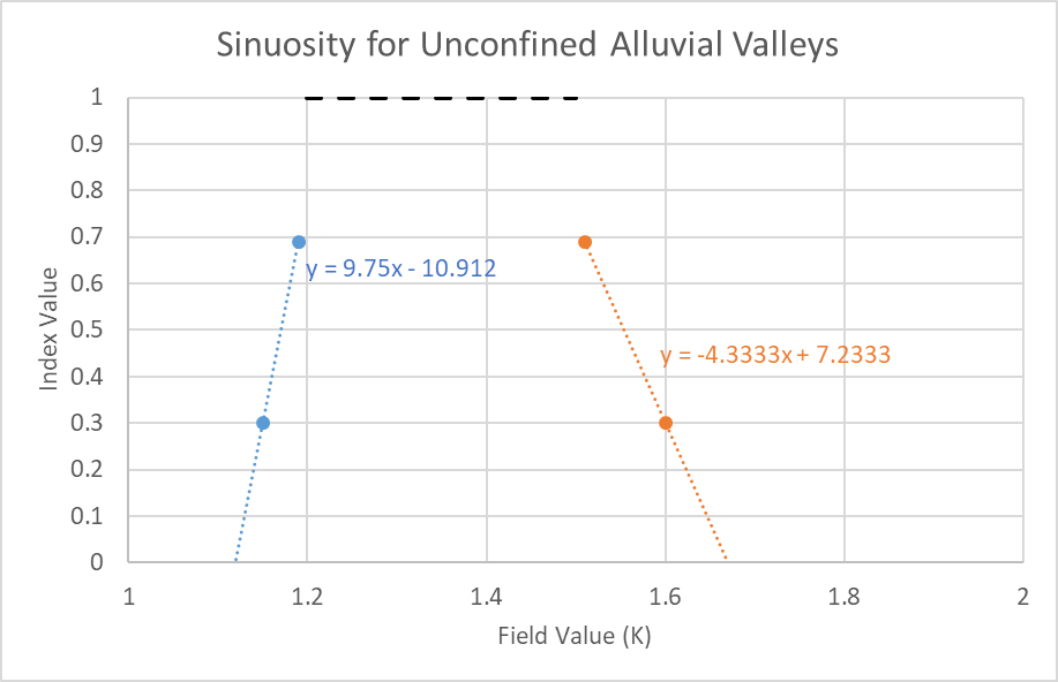

Figure 9-3b: Sinuosity Reference Curves

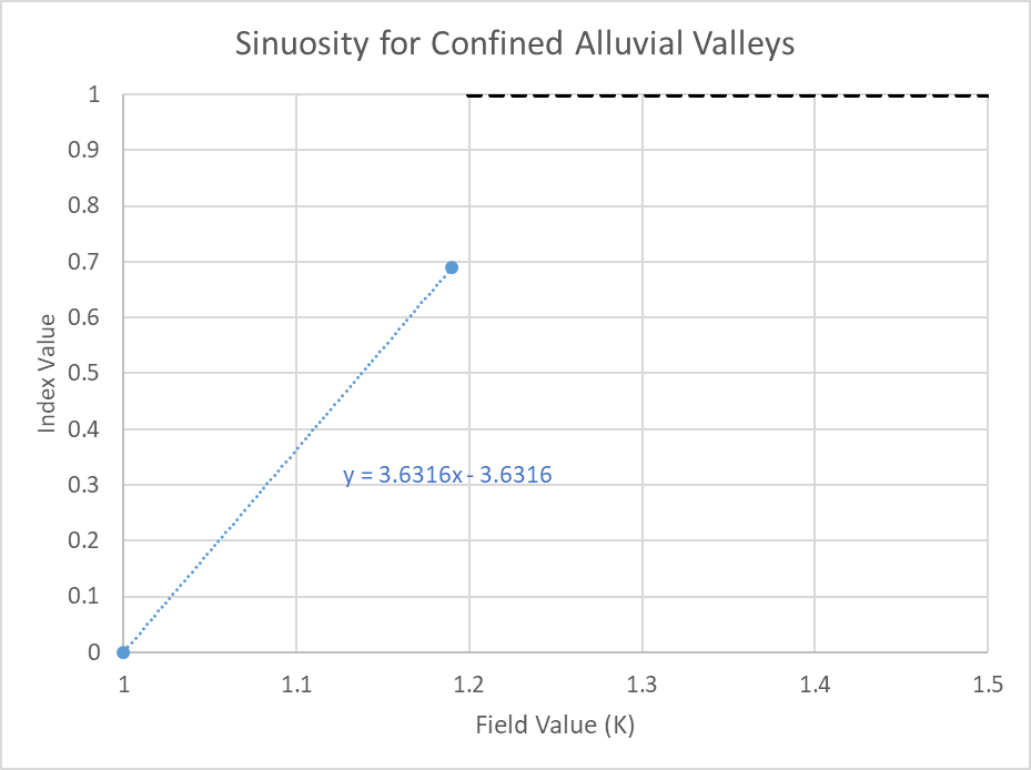

Figure 9-3c: Sinuosity Reference Curves

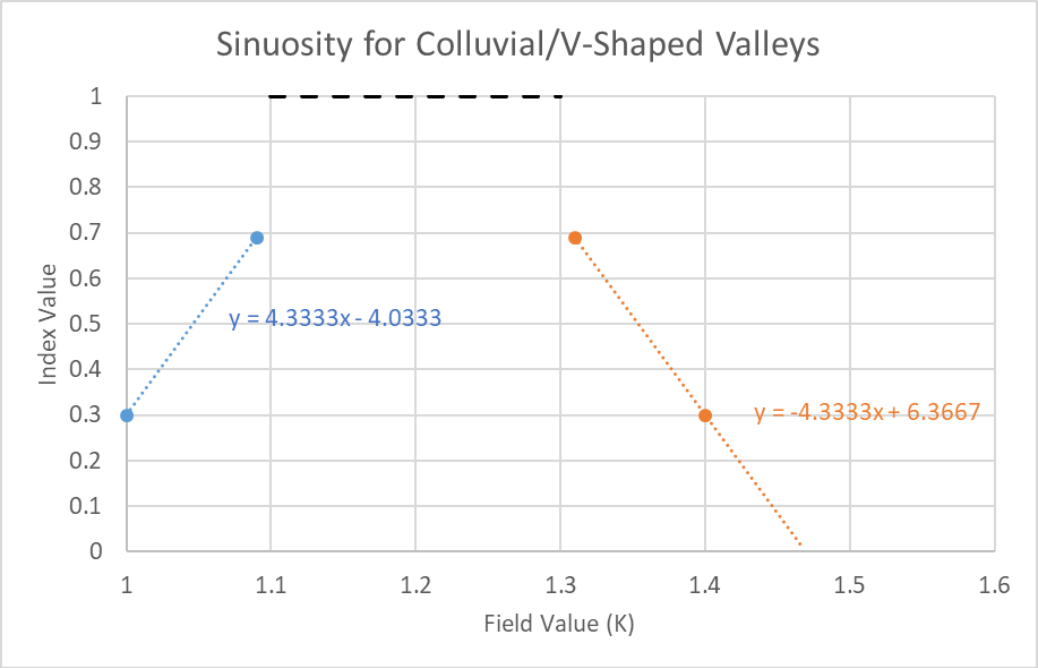

Figure 9-3d: Sinuosity Reference Curves

<u>Limitations and Data Gaps:</u>

In the WSQT, sinuosity is not applicable to braided (D) or anastomosed (DA) stream types. Additionally, F and G channels are typically degraded stream types and therefore reference curves were not included for these stream types. However, if the reference stream type for these channels is a C or E, then those reference curves should be used. See Limitations and Data Gaps section in Chapter 8.1 for additional discussion.

During beta testing, it was noted that additional guidance is needed to consistently characterize planform. To improve repeatability and better align the sinuosity metric with restoration activities, sinuosity should be measured for the length of the project reach. Prior recommendations included assessing sinuosity over a length that is 40 times the bankfull width. This length could extend beyond the reach limits and therefore include stream sections that will not be changed as part of a project, diluting lift and loss calculations. Furthermore, this method led to drastically different answers depending on where along the valley measurements started and ended. Additional information has been added to the WSQT v1.0 User Manual.

Planform metrics have yet to be developed for braided and anastomosing systems in the WSQT, although they may be included in subsequent versions. Some consideration has been given to braid indices (Egozi and Ashmore 2008), although additional work would be needed to select an appropriate metric and develop reference curves before this type of metric could be incorporated into the tool.

Chapter 10. Riparian Vegetation Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Riparian vegetation is a critical component of a healthy stream ecosystem. Riparian vegetation is defined as plant communities contiguous to and affected by surface and subsurface hydrologic features of perennial or intermittent water bodies. While these plant communities are a biological component of the stream ecosystem, riparian vegetation plays such a critical role in supporting channel stability, physicochemical and biological processes that it is included in the geomorphic level of the stream functions pyramid (Harman et al. 2012). Riparian areas support numerous instream and floodplain functions, including:

- Cover and shading
- Channel stability
- Filter excess nutrients, sediments, and pollutants
- Source of woody debris
- Floodplain roughness
- Carbon and nutrient contributions
- Terrestrial habitat
- Plant diversity, species richness, and functional integrity

Seven metrics for riparian vegetation were included in the WSQT Beta version, and all but one assessed the left and right bank separately to account for variations in stream bank ownership and land use. These metrics included:

- Riparian Width Ratio
- Woody Vegetation Cover
- Herbaceous Vegetation Cover
- Non-native Plant Cover
- Hydrophytic Vegetation Cover
- Stem Density
- Greenline Stability Rating

The WSTT prioritized the use or adaptation of existing programmatic methodologies, particularly those with regional datasets or indices that could be used for the development of reference curves. The WSQT Beta Version included a combination of techniques borrowed from 1) USEPA NRSA (USEPA 2009); 2) Bureau of Land Management Assessment, Inventory, and Monitoring (AIM) (BLM 2017); 3) Corps of Engineers Hydrogeomorphic (HGM) Approach (Hauer et al. 2002); 4) Corps of Engineers Arid West Regional Supplement (USACE 2008); and 5) USDA Forest Service Monitoring the Vegetation Resources in Riparian Areas (Winward 2000). In the WSQT Beta Version, the NRSA dataset was used to develop reference curves.

Following beta testing and field training exercises, the decision was made to change data collection methods to improve repeatability and consistency and allow for extrapolation of species information to draw inferences on vegetation composition and/or to apply additional regulatory performance standards at mitigation sites. Availability of species-level data from the CNHP dataset (Kittel; Chapter 1.5) allowed the WSTT the opportunity to more closely align methods with existing protocols required by the Corps for wetland delineations. Data collection includes visual estimates of the percent absolute cover of each plant species within nested plot types to determine vegetation abundance, structure, composition and complexity. Data collection aligns with the methods outlined in the 1987 Wetland Delineaton Manual and regional supplements (USACE 1987, USACE 2008, USACE 2010a, USACE 2010b). Corps field staff and many practitioners are already familiar with these methods, and wetland delineations will likely be required for most stream projects that also contain wetlands in the project area.

Four metrics were carried forward into the WSQT v1.0 riparian parameter: riparian width, absolute woody vegetation cover, absolute herbaceous vegetation cover and percent native cover. These metrics consolidate data collected from both banks into a single WSQT reporting value per metric, which is standard among other methods.

Stem density was eliminated to reduce redundancy in the assessment and characterization of stream and riparian vegetation functions. Both stem density and woody vegetation cover metrics characterize the amount of woody vegetation, yet absolute woody vegetation cover is preferable because of the relation of canopy cover to root cover and the biophysical functions of shrubs as a primary component of western riparian systems (ADD REF). Stem density may be requested by a regulatory agency as an additional regulatory performance standard within the first 5 years of a full restoration project to obtain a better indication of recruitment or establishment of woody vegetation, but has not been incorporated into the tool.

Hydrophytic vegetation cover was eliminated because of the overlap with the riparian width metric, which is defined, in part, using characteristic riparian vegetation, and is thus captured in riparian width, albeit at a coarser scale. The riparian width metric is also preferable because it is an indicator of the extent of hydrologic connectivity. Hydrophytic vegetation may be considered in future versions of the tool, as shifts in vegetation composition can be a valuable, direct indicator of changes in underlying processes (e.g., hydrology, flow regime, floodplain connectivity) associated with a project. Additional data collection and analysis related to hydrophytic vegetation and other compositional metrics is being considered. Hydrophytic vegetation data can be obtained via the data collection methods for WSQT v1.0, and this will allow the WSTT to evaluate how this metric could be developed and applied in the future.

The Greenline Stability Rating metric was retained in the WSQT v1.0, but was moved to the lateral migration parameter. While it is informed by the presence of riparian vegetation, the WSTT felt it was more appropriate as an indicator of bank stability, and was thus moved to serve as an alternative or supplement to the other metrics within the lateral migration parameter.

Data from WSQT Beta Version field testing were used to inform reference curves development. In addition, the CNHP dataset was used to evaluate the WSQT cover metrics and reference curves. The CNHP dataset was selected because it is extensive, overlaps with ecoregions in Wyoming, and had the species-level data that aligns with the selected methods. The CNHP dataset also provided reasonable sample sizes across a range of stream condition. Results from this dataset indicate that herbaceous vegetation cover and native vegetation cover are good predictors of site condition.

<u>Metrics:</u>

- Riparian Width
- Woody Vegetation Cover
- Herbaceous Vegetation Cover
- Percent Native Cover

10.1. Riparian Width

Summary:

The riparian width metric, developed specifically for the WSQT, is the proportion of the expected riparian area width that currently contains riparian vegetation and is free from utility-related, urban, or otherwise soil disturbing land uses and development. This metric characterizes the current width of the riparian area, as compared with the reference expectation for that site. The current, observed riparian width is a measure of the current extent of the riparian zone, and this data is collected in the field at the time of the assessment. The reference expectation, or expected riparian width, is an estimate of the natural or historic extent of the riparian area. Riparian width is driven by valley controls and reach-scale influences, and cannot be easily predicted based upon its location within a river network or the size of the stream. As such, the riparian width metric uses an O/E approach to identify the current extent of the riparian zone compared with the expected extent based on reach-scale processes and drivers. The expected riparian width is determined from hydrologic and geomorphic indicators on the landscape, aerial imagery or meander width ratio. Additional information on data collection methods is provided in the WSQT v1.0 User Manual.

Characterizing the extent of riparian zones is important, as functioning riparian zones influence (and are influenced by) many instream and floodplain processes (Fischer and Fischenich 2000, Mayer et al. 2006). Many existing methodologies focus on fixed buffer widths, yet these approaches can be limited as they don't account for the natural variability in riparian zone widths, and thus may not adequately characterize their functional significance. For example, in high gradient headwater streams, riparian zones are naturally narrow, and may not extend as far as a fixed buffer width. Similarly, in broad, alluvial systems, a fixed buffer width may only characterize a small fraction of the floodplain or riparian area extent. Thus, the approach outlined here is intended to better characterize the natural functional capacity of riparian zones, by comparing the current riparian width against the expected, or reference, width determined from the predominant processes that control riparian zones.

According to Merritt et al. (2017), the edge of a riparian area can be determined using three criteria:

1) substrate attributes—the portion of the valley bottom influenced by fluvial processes under the current climatic regime,
2) biotic attributes—riparian vegetation characteristic of the region and plants known to be adapted to shallow water tables and fluvial disturbance, and
3) hydrologic attributes—the area of the valley bottom flooded at the stage of the 100-year recurrence interval flow (Ries et al. 2004).

Substrate and topographic attributes: The extent of the riparian zone is driven by topographic and geomorphological patterns, as well as the dominant hydrological processes (Polvi et al. 2011, Salo et al. 2016). For example, riparian width will vary based on process domains: steeper gradients yield narrower riparian areas, while lower gradients yield broader floodplains; riparian areas in confined valleys are constrained by hillslope processes while riparian zones in unconfined valleys are defined more by valley processes (e.g., micro topography, groundwater movements, etc.); higher elevation sites with snowmelt-dominated hydrographs provide more consistent flow regime than flashier rain-dominated, lower elevation systems (which may also

have greater propensity for sediment movement due to erosion and flashy regime). Even in altered systems, topographic and geomorphic indicators may be present to determine the expected extent of the riparian zone. These indicators may include breaks in slope between the bankfull and valley edge, fluvial deposited sediments, or a lack of upland soil formation. In the absence of these indicators, e.g., in areas of extensive floodplain development, a meander width ratio based on valley type should be used to determine the expected riparian width.

Biotic attributes: The extent of characteristic riparian vegetation will change in response to changes in floodplain connectivity and hydrologic processes. As such, the extent of the riparian vegetation is used to determine the current, or observed extent of the riparian area. For purposes of this metric, riparian areas are defined as areas with distinctly different vegetation species and/or more robust growth forms than adjacent areas. Riparian areas are often characterized by the predominance of hydrophytic species that have adapted to shallow water tables and fluvial disturbances. It should be noted that in many areas of Wyoming, riparian communities are comprised of a combination of hydrophytic and upland species, including greasewood and sagebrush. The presence of upland species does not preclude an area from being classified as riparian, however, the absence of any hydrophytic species likely would. Similarly, where riparian areas contain species similar to adjacent areas, more vigorous or robust growth forms should be observed in order to classify it as a riparian area (USFWS 2009).

Hydrologic attributes: Riparian extent can relate to flow stage for a specified recurrence interval, although this relationship varies across process domains (Polvi et al. 2011). For example, Polvi et al. (2011) found that in high elevation unconfined valleys the riparian extent is significantly broader than the 100-year stage; in high elevation confined valleys it aligns with the 100-year stage; in unconfined low elevation montane systems, it is not well predicted by 10, 50 or 100-year flow stages; and in confined low elevation montane systems, the riparian extent aligns with the 10-year recurrence interval (and is significantly narrower than the 50 and 100-year stage). Recognizing the challenges in flow-based predictors of riparian extent, Merritt et al. (2017) conservatively recommend the use of the 100-year recurrence interval flow stage to delineate riparian area extent. Where hydrologic attributes have been influenced by anthropogenic modification, the WSQT relies on a meander width ratio to determine expected riparian width.

<u>Reference Curve Development:</u>

This metric was developed specifically for this tool. Limited data and peer reviewed literature were available to inform thresholds and reference curves, as much of the existing literature is related to fixed-width buffers. Thus, reference curves were developed primarily using best professional judgement of the WSTT. These reference curves and thresholds were developed conservatively to encourage and incentivize restoration activities that remove stressors and human land uses from the riparian zone.

Stratification of reference curves took into consideration how hydrologic and geomorphic processes drive riparian zone development. Merritt et al. (2017) recommends stratifying by valley type using a Hydrogeomorphic Valley Classification framework, which identifies nine valley types, but also acknowledges that other simpler classification approaches (e.g., Rosgen 1996) may also be useful to place a stream segment within its watershed context. For this metric, we stratified based on valley confinement, recognizing differences in hillslope and valley bottom processes that influence riparian extent in confined and unconfined valleys (Table 10-1).

Once stratified into valley types, the WSTT considered how potential stressors in the floodplain or adjacent stream area and changes to the hydrologic regime can influence the degree to which riparian zones function, and in turn, support instream functions. For example, whether the extent of riparian zone modification may substantially affect the recruitment of wood and organic matter, nutrient and carbon cycling, flood retention, buffering from sediment and pollutant influxes, and habitat (Fischer and Fischenich 2000, Sweeny and Newbold 2014). In confined and colluvial valleys, where streams and riparian zones are constrained by hillslope processes, riparian width is naturally narrower, and consequently, stressors within that area could be disproportionately higher. A reduction in riparian width of 40% would likely reflect a substantially altered, or not functioning condition, with little remaining flood prone area and a reduced capacity to recruit wood and organic matter and buffer the stream from sediment or pollutant influxes. This magnitude of riparian area loss may no longer support instream and floodplain functions. In unconfined valleys, where riparian areas are naturally broader, a greater proportion of the riparian area may be affected (e.g., 70%) before a similar loss in functionality might occur.

Table 10-1: Threshold Values for Riparian Width

Index Value	Field Value (%)	
	Unconfined Alluvial Valleys	Confined Alluvial and Colluvial Valleys
1.00	100	100
0.00	30	60

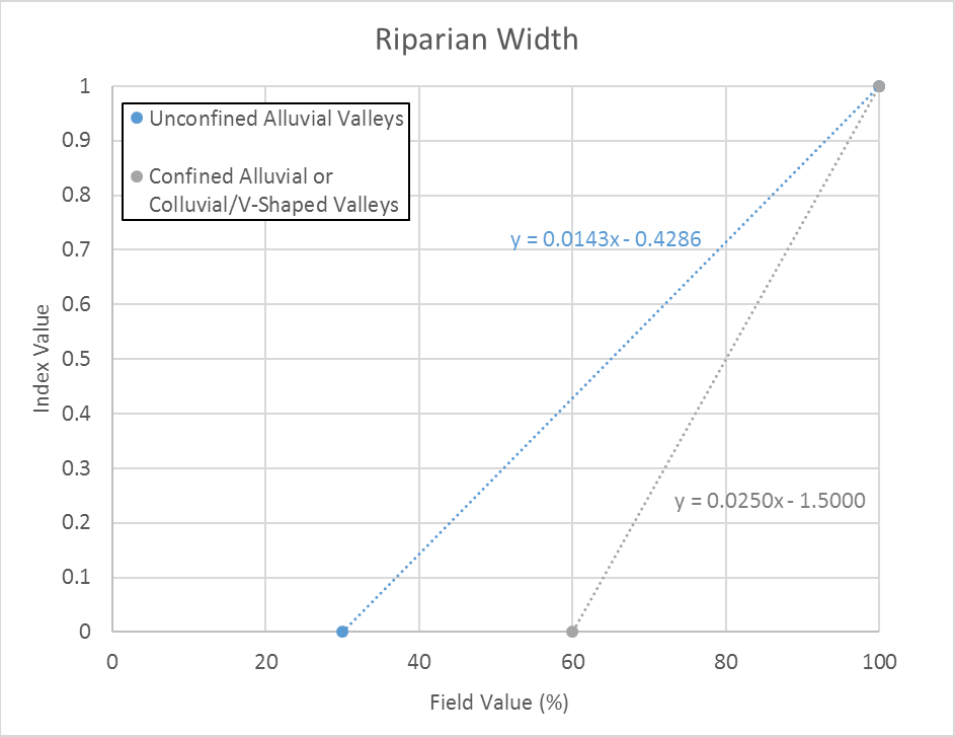

Figure 10-1: Riparian Width Reference Curves

Limitations and Data Gaps:

Because this is a new metric developed for use in the WSQT and reference curves are based on best professional judgement, additional data are needed to test and possibly expand these criteria. Reference curves may benefit from additional stratification that accounts for natural variability in riparian width beyond the valley type approach applied here. The metric would benefit from additional validation, review and refinement as the tool is applied.

Beta testing has revealed challenges in measuring the expected riparian width in the field, including difficulties in accurately measuring straight line distances in dense vegetation and a lack of readily observable geomorphic and hydrologic features in degraded sites, which are often no longer present due to site grading and/or development. As such, the WSQT v1.0 User Manual includes several alternatives for determining the expected riparian width, including use of aerial photography, digital elevation models or calculations of a meander width ratio. Additional testing and review are needed to evaluate the relative accuracy and applicability of these approaches.

10.2. Woody Vegetation Cover

Summary:

Riparian areas in Wyoming are predominately characterized by a woody canopy (Youngblood et al. 1985; Jones and Walford 1995; Walford 1996; Walford et al. 2001; Jones et al. 2001). As noted above, in many areas of Wyoming, riparian communities are comprised of a combination of hydrophytic and upland species. Woody assemblages in Wyoming include willow-dominated scrub-shrub communities, cottonwood gallery forests, birch/alder scrub-shrub communities, spruce woodlands, as well as black greasewood shrub communities and silver sagebrush shrub communities.

Many riparian areas in the western U.S. are heavily influenced by changes in land use, fire regimes, grazing, flow modification and the influx of non-native and invasive species (Macfarlane et al. 2017). Tamarisk and Russian olive have been prolific invaders, and many restoration efforts target the management and eradication of these invasive species (Shafroth et al. 2002). Riparian areas in the plains and basins that historically (pre-European settlement) contained patches of timber or brush were progressively reduced to herbaceous communities for over a century due to the rise of the plains horse culture, migration of white pioneers, and the advance of farming and stock-raising (West and Ruark 2004)

Because of the characteristic role woody vegetation plays in riparian areas, a woody vegetation metric is important to include in the WSQT. Woody vegetation cover provides an indication of the longevity and sustainability of perennial vegetation in the riparian corridor (Kaufmann et al. 1999, Kaufmann and Hughes 2006). The woody cover metric is based on a visual plot-based vegetation assessment. Methods are outlined in the WSQT v1.0 User Manual.

This metric represents absolute cover of woody vegetation. The field value for this metric is summed across all woody species and can be greater than 100% cover.

Reference Curve Development:

In the WSQT Beta Version, the NRSA dataset (USEPA 2016) was used to develop reference curves and inform data collection methods. However, following beta testing and field training exercises, the decision was made to change data collection methods to align with the 1987 Wetland Delineation Manual methods, as the Corps field staff and many practitioners are already familiar with this form of data collection. These methods provide absolute cover by species, which is different than the approach used in the beta version. Because of this, the NRSA dataset, which relies on relative cover by strata, was considered no longer applicable for developing reference curves. The WSTT relied on CNHP datasets (see Chapter 1.5) and a small data collection effort in Wyoming to inform the reference curves for this metric.

Colorado Natural Heritage Program: Woody vegetation cover values were calculated for woody sites in the CNHP dataset, described in Section 1.5. Woody vegetation cover values were developed by summing absolute cover values for all woody species. Shrub species cover values were combined with tree species cover values into a combined woody stratum. Statistics were derived from the CNHP dataset for the reference standard (R) and degraded (D) sites within each ecoregion (Table 10-2). Sample sizes were limited, particularly for degraded sites and for all sites within the Plains ecoregions.

Table 10-2: Statistics for Woody Vegetation Cover from the CNHP Dataset

	Woody Vegetation Cover (%) by Ecoregion and Condition						
	Mountains		Basins		Mountains and Basins	Plains	
Statistic	**D**	**R**	**D**	**R**	**R**	**D**	**R**
Number of Sites (n)	11	336	0	48	384	11	6
95th Percentile	104	166	-	205	177	157	75
75th Percentile	95	117	-	138	122	101	68
Median	71	92	-	119	95	92	59
Mean	68	95	-	118	98	95	57
25th Percentile	46	68	-	90	69	76	53

In the basins ecoregion, the CNHP cover values were substantially higher than the other ecoregions. Cover values may be higher because the data collection within the xeric ecoregions in Colorado followed a sampling methodology using large plot sizes (e.g., $50m^2$-$500m^2$), potentially resulting in an overestimation of cover. Further, Macfarlane et al. (2017) modeled pre-European settlement native land cover and showed that current riparian vegetation showed significant to large (33 to >66%) departure from historic conditions in the Utah and Columbia River basin watersheds, with riparian vegetation conversions being primarily change in native riparian to invasive and upland woody vegetation types. There may be few areas that truly represent reference standard condition on the landscape due to the long history of land use, flow modification and grazing that is prevalent in the Eastern Xeric and Wyoming Basins. Given

these limitations, we decided to combine the Mountains and Basins datasets and develop a single, combined reference curve.

WSTT data collection: In August 2016 and fall of 2017, the WSTT visited several sites to apply the proposed WSQT methodology for assessing riparian vegetation. These sites were considered to represent minimally disturbed reference standard sites. However, because they are located on public lands, they have likely been subject to some historical use, including grazing and/or timber removal. The woody vegetation cover values from these sites are presented in Table 10-3. Note, these data reflect cover values by lifeform, and thus are lower than absolute cover value by species.

Table 10-3: Woody Vegetation Cover at Reference Sites Visited by the WSTT

Site	Ecoregion	Woody Vegetation Cover (%)
Wood River, above Middle Fork	Mountains	53
Middle Fork Wood River	Mountains	47
Middle Fork Wood River - Upstream	Mountains	44
Jack Creek	Mountains	76
Sand Creek (2017)	Basins	46

Analysis: In general, the following criteria were used to establish the thresholds (Table 10-4) between the three functional categories:

- The median and/or 75th percentile of reference standard sites were used to determine the maximum index value of 1.00.
- The 25th percentile values from reference standard sites was used to determine the threshold between functioning and a functioning at risk condition.
- The 75th percentile cover values from degraded sites were used to inform the threshold between functioning at risk and not functioning condition. Where sufficient data were not available, this threshold would not be identified; and values within these index ranges would be determined from the reference curve.
- Minimum index values were set at 0% woody vegetation cover. Even a small amount of woody vegetation recruitment would lead to cover values of 1% or greater.

In the montane and basins ecoregions, the reference standard sample size was 336 from the CNHP dataset. The 75th percentile values from the combined ecoregion dataset was 122% cover, and this value was used to characterize the maximum index value (1.00), meaning any site with absolute woody vegetation cover of 122% or greater would receive a maximum index score. The 25th percentile from the CNHP dataset was 69%, and this was used as the threshold value between functioning and functioning-at-risk index values.

We did not identify the break between non-functioning and functioning-at-risk due to a lack of data resolution. Instead, we decided to fit the reference curve, and allow this break to be extrapolated from the regression equation. The 25th percentile value from the CNHP degraded sites was 46% woody vegetation cover, which falls within the functioning-at-risk range of index

scores from the curve. While it may be appropriate to consider these degraded sites as not-functioning and assign them lower index values, the WSTT considered the natural variability in woody riparian ecosystems and felt that a more conservative approach was appropriate.

In developing the reference curve, we took into consideration the data collected from montane and basins field sites by the WSTT in Wyoming, which showed a range of cover values from 44-76%. (Note that these cover values were collected with a different methodology – absolute cover by species values at these sites would be higher). These sites were in good condition, and had healthy, diverse riparian communities. Cover values may be lower than a pristine condition due to historical and current anthropogenic use, including grazing and/or timber harvest. However, the sites in the Wood River basin were characterized by broad, connected floodplains that had micro topography consisting of multiple hummocks, swales, and cobble bars. These conditions support establishment of diverse herbaceous and scrub-shrub floodplain mosaics, which are an ecologically desirable outcome in many ecoregions (Kleindl et al. 2015). The woody vegetation was naturally patchy and interspersed with areas of herbaceous vegetation. In evaluating the datasets and proposed benchmarks, we concluded it was reasonable to characterize these sites as functioning or (high) functioning-at-risk. These sites have the potential to support a healthy aquatic ecosystem, and were not in a clearly degraded state. Therefore, we felt that threshold values and reference curves should be conservatively derived to account for the natural variability that may occur at woody sites in Wyoming.

The CNHP dataset in the plains ecoregions had small sample sizes for both reference and degraded sites. In this dataset, degraded sites consistently had higher woody vegetation cover than reference sites. This could be due to several factors, including the augmentation of flows on the high plains from irrigation practices, or shifts in riparian community type (Richardson et al. 2007; Scott et al. 2000, Macfarlane et al. 2017). Historically, woody communities along streams in the plains would be characterized by cottonwood gallery forests and willows (West and Ruark 2004), and many are now composed of regionally introduced mixed deciduous forest species and/or shrub that tolerate a broader range of environmental factors and land uses (CNHP; Jones and Walford 1995). An evaluation of the NRSA dataset found that non-reference sites had, on average, lower cover values than reference sites, but some sites showed a similar pattern to the CNHP dataset, with much higher cover values than the reference sites. Because of these trends in the data, the WSTT decided to develop a two-sided reference curve that captured woody vegetation cover values that were both lower and higher than reference standard condition.

Due to the small size of the reference dataset, we decided to not define a threshold value between functioning and functioning-at-risk condition, and instead fit a reference curve to a more limited set of threshold values. The median and 75th percentile values from the reference standard sites were used to define the maximum index value of 1.00. Because the CNHP degraded sites were consistently higher cover values than the reference sites, the 75th percentile from the degraded sites in the CNHP dataset (101%) was used to determine the break between functioning-at-risk and not functioning on the right side of the reference curve. On the left side of the reference curve, we did not identify the break between non-functioning and functioning-at-risk due to a lack of data resolution. Instead, we decided to fit the reference curve, and allow this break to be extrapolated from the regression equation. The 25th percentile of the reference standard dataset was 53% cover, which falls within the functioning range of index values in the reference curve. Woody vegetation cover of 0% was assigned a minimum

index values of 0.00, which allows for a range of woody cover values to score within the functioning-at-risk and functioning condition ranges, recognizing ecosystem and flood dynamics that create diverse vegetation floodplain mosaics in a plains environment (Kleindl et al. 2015; Jones and Walford 1995).

Table 10-4: Threshold Values for Woody Vegetation Cover

Index Value	Field Value (%)	
	Mountains and Basins	Plains
1.00	≥ 122	59 – 69
0.70	69	-, -
0.30	-	-, 101
0.00	0	0, -

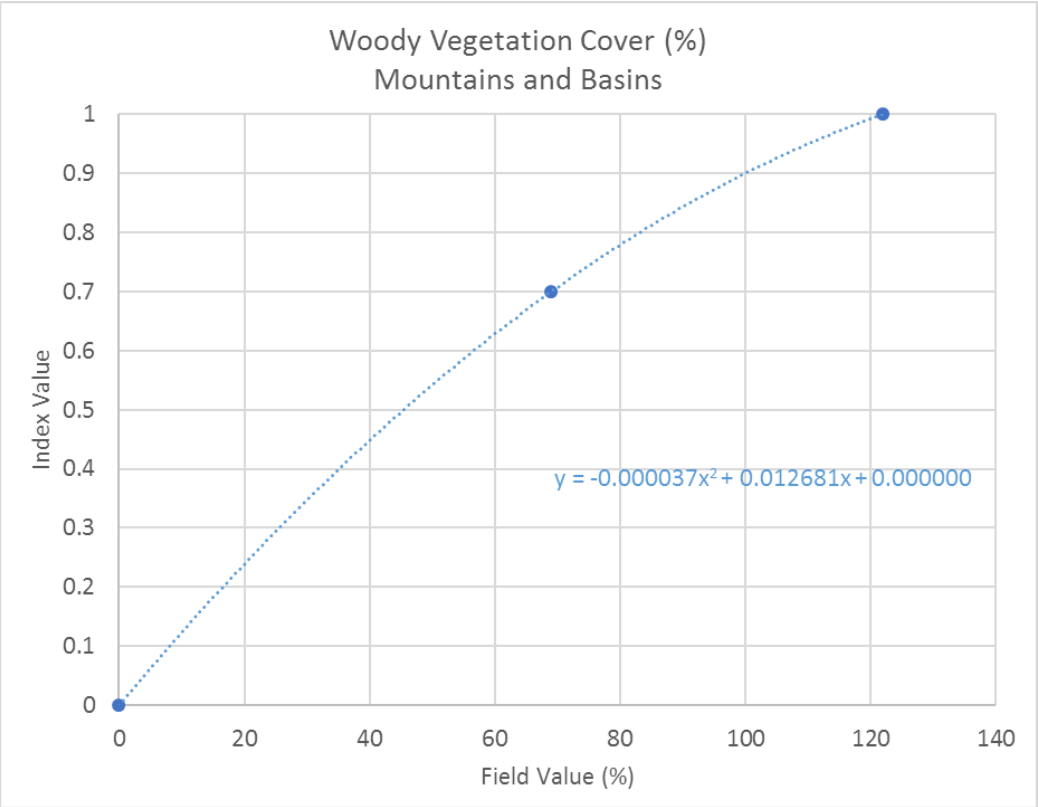

Figure 10-2a: Woody Vegetation Cover Reference Curves

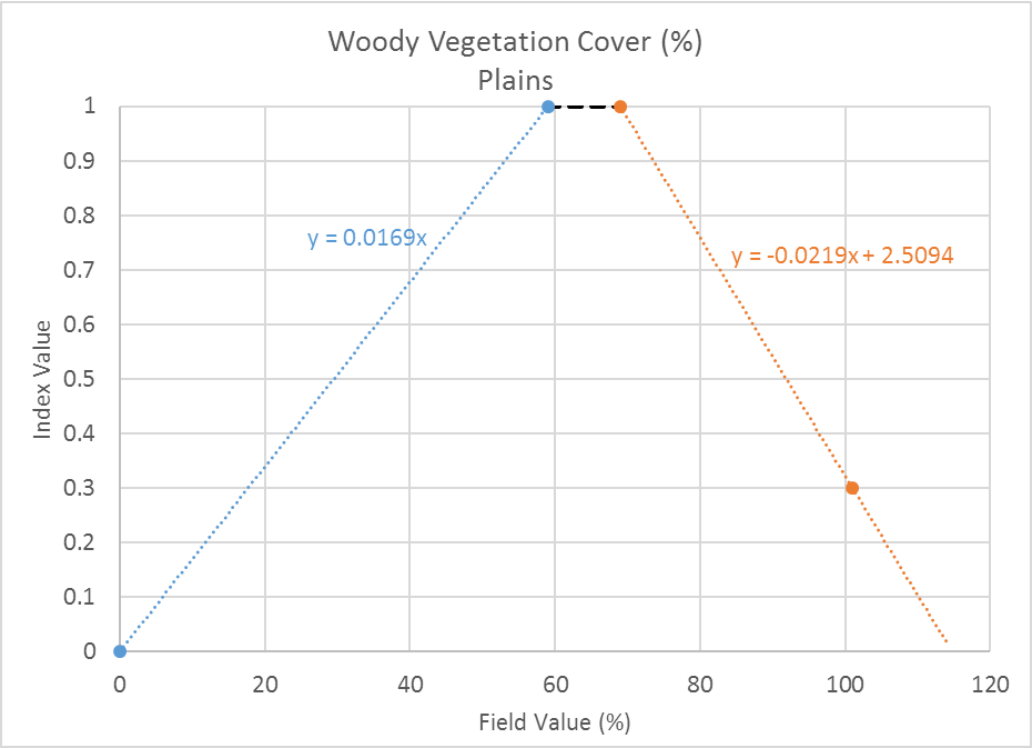

Figure 10-2b: Woody Vegetation Cover Reference Curves

<u>Limitations and Data Gaps:</u>

The CNHP dataset has limitations, including the obvious geographic boundaries of the state. The WSTT assumed that the CNHP dataset would be relevant and translatable to Wyoming due to the overlapping ecoregions and similarity between riparian community types in Colorado and Wyoming. The CNHP dataset includes data collected between 1992 and 1999; with no sites revisited recently. Additional analysis of these sites may be useful to understand whether changes in climate or other large-scale influences have altered the reference expectation for riparian areas in this region.

The reference curve development for the WSQT would benefit from additional Wyoming-specific data to validate the criteria and curves identified above. Additional data would also allow for us to consider whether additional stratification or refinement beyond ecoregion could occur. For example, there was a broad range in cover values across reference standard conditions. The Wood River site had moderate amounts of woody cover due to the naturally patchy nature of the floodplain area. As such, additional stratification within ecoregions, e.g., by valley type, slope, stream size or target community composition, would allow us to further refine these reference curves and identify more specific restoration targets.

The metric does not differentiate between upland and hydrophytic woody vegetation cover. This may attribute a higher level of functioning to degraded systems that have transitioned to an upland dominated woody community. Additional data and research are required to better understand how naturally prevalent upland species are within riparian areas in Wyoming. Many plains and basin riparian systems support upland scrub-shrub communities. While these are

often associated with more degraded, incised systems, they can also occur naturally due to specific soil conditions and in more arid areas with lower water tables.

A major challenge is also differentiating between streams of varying flow permanence. We did not differentiate or evaluate differences in woody riparian vegetation cover across perennial, intermittent or ephemeral systems. It is likely ephemeral streams would naturally sustain lower densities of woody vegetation, and thus would benefit from their own set of reference curves and criteria. We believe this metric should still be applied in ephemeral stream systems, but would expect them to generally score lower than their perennial counterparts.

10.3. Herbaceous Vegetation Cover

Summary:

While riparian areas in Wyoming are predominately characterized by a woody canopy (as noted above), a ground layer of herbaceous vegetation is often also present. These herbaceous species are an important component of the riparian community, as they are often providing surface roughness and cover in the early stages of succession following fluvial disturbances (Youngblood et al. 1985, Winward 2000). Hydrophytic herbaceous vegetation, including sedges and rushes, also contributes to bank stability and floodplain roughness (Winward 2000). Some riparian communities naturally support only herbaceous species, including those that support broad, highly connected floodplains with anaerobic soil conditions; or those that have natural disturbance (flood or fire) regimes that do not favor the persistence of woody species (Youngblood 1985; West and Ruark 2004), although the historical distribution of these communities is not well known.

Many riparian areas in the western U.S. are heavily influenced by changes in land use, fire regimes, grazing, flow modification and the influx of non-native and invasive species (Macfarlane et al. 2017). Many riparian communities contain non-native upland pasture grasses and forage forbs due to agricultural land use and cattle grazing. These species are adapted to a range of moisture regimes and thrive in mesic conditions supported by both connected and disconnected floodplains (Youngblood 1985). Introduced species are very competitive except in highly connected floodplains where anaerobic soil conditions generally support wetland obligate native species (USACE 2008; USACE 2010). Over grazing, frequent fire regimes, woody brush control and channel incision can promote secondary succession and invasion of non-native herbaceous species as well as native upland grasses, which lack root structures to stabilize streams (Youngblood et al. 1985; Jones and Walford 1995; Winward 2000; MacFarlane 2017). Riparian areas dominated by these species can perpetuate degraded conditions.

It is important to include herbaceous vegetation in the WSQT because of the value it provides as a component of riparian communities as well as its sensitivity to disturbance. Consideration was given to including both an herbaceous cover metric and a native/non-native herbaceous species metric, but due to the way scoring rolls up in the tool, it was decided to combine into a single herbaceous vegetation metric. The herbaceous vegetation cover metric is based on a visual plot-based vegetation assessment. This metric represents the sum of absolute aerial cover of herbaceous species collected within 1-meter or 5-meter plots. NOTE: Methods are outlined in the WSQT v1.0 User Manual.

Reference Curve Development:

In the WSQT Beta Version, the NRSA dataset (USEPA 2016) was used to develop reference curves and inform data collection methods. However, following beta testing and field training exercises, the decision was made to apply one method for all vegetation cover metrics that aligns with the 1987 Wetland Delineation Manual, as the Corps field staff and many practitioners are already familiar with this form of data collection. These methods provide absolute cover by species, which is different than the approach used in the beta version. Because of this, the NRSA datasets, which rely on relative cover by strata, were considered no longer applicable for developing reference curves. The WSTT relied on CNHP datasets (as described in Section 1.5), the Northern Rocky Mountain Hydrogeomorphic Manual (Hauer et al. 2002), and a small data collection effort in Wyoming to develop reference curves.

Colorado Natural Heritage Program: Herbaceous vegetation cover values were calculated for sites in the CNHP dataset, described in Section 1.5. Herbaceous vegetation cover values were developed by summing absolute cover values categorized by stratum. Species in the dataset identified as graminoid or forb were grouped together into the herbaceous stratum.

Differences in herbaceous cover across ecoregion were visually assessed to determine if herbaceous vegetation cover should be stratified by this variable. In addition, the CNHP dataset also allowed us to evaluate differences across community types, namely whether the community type was a woody community or herbaceous community. We observed distinct differences between herbaceous-only communities and herbaceous cover within woody communities. We did not observe any meaningful differences between reference standard sites across ecoregions. As such, we decided to stratify this dataset by reference community type, but not by ecoregion. This is consistent with the stratification approach taken in HGM, described below (Hauer et al. 2002). Statistics were derived from the CNHP dataset for the reference (R) and degraded (D) sites for each cover type (Table 10-5).

Table 10-5: Statistics for Herbaceous Vegetation Cover from the CNHP Dataset

	Herbaceous Vegetation Cover (%) by Reference Community Type and Condition			
	Woody		Herbaceous	
Statistic	D	R	D	R
Number of Sites (n)	22	390	5	97
95th Percentile	126	124	97	152
75th Percentile	111	77	82	117
Median	61	54	52	94
Mean	69	56	61	94
25th Percentile	41	27	41	73
5th Percentile	10	6	33	36

HGM: The *Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Riverine Floodplains in Northern Rocky Mountain* (HGM manual; Hauer et al. 2002) scored herbaceous plant coverage according to varying cover types as shown in Table 10-6. The HGM methodology is similar to the SQT methodology in that variables are scored on a scale of 0 to 1 and then combined to assess ecosystem functions. For herbaceous plant coverage, the curves were all linear.

Table 10-6: HGM Manual Performance Standards for Herbaceous Plant Coverage (Hauer et al. 2002)

Cover Type Description[1]	Percent Herbaceous Plant Coverage (%)	
	Variable Sub-Index = 0	Variable Sub-Index = 1 [2]
Mature conifer dominated	0	70
Mature cottonwood dominated	0	85
Mix of willows, alder, shrubs, and interspersed herbaceous cover	0	50
Herbaceous vegetation dominated	0	35

[1] *A list of all cover types and complete descriptions are available from (Hauer et al. 2002).*

[2] *Values rounded.*

WSTT data collection: In August 2016 and fall 2017, the Wyoming Stream Technical Team visited several sites to apply the proposed WSQT methodology for assessing riparian vegetation. These sites were considered to represent minimally disturbed reference standard sites. However, because they are located on public lands, they have likely been subject to some historical use, including grazing and timber removal. The herbaceous vegetation cover values from these reference sites are presented in Table 10-7. Note, these data reflect cover values by lifeform, and thus are lower than absolute cover value by species.

Table 10-7: Herbaceous Vegetation Cover at Sites Visited by the WSTT

Site	Reference Vegetation Cover Type	Herbaceous Vegetation Cover (%)
Wood River, above Middle Fork	Woody	74
Middle Fork Wood River	Woody	43
Middle Fork Wood River - Upstream	Woody	65
Jack Creek	Woody	35
Sand Creek	Woody	80

Analysis: Best professional judgement was used to interpret the datasets to develop threshold values for this metric (Table 10-8). In general, the following criteria were used to establish the breaks between the functional categories:

- The 75th percentile of reference standard sites was used to determine the maximum index value of 1.00.
- The median values from reference standard sites was used to determine the break between functioning and functioning at risk condition.
- Due to small sample sizes, the threshold between functioning-at-risk and non-functioning condition was not identified a priori; and values within these index ranges would be determined from the reference curve. The 25th percentile value from degraded or non-reference sites were used to evaluate the reference curve within these ranges.
- At woody reference sites, minimum index values were set at 0% cover.
- At herbaceous reference sites, minimum index values were derived from the 5th percentile of degraded sites.

For herbaceous vegetation in woody community types, the 75th percentile of the reference data (77% cover) was used to define the maximum index value. The median of the reference data (54%) was used to determine the break between functioning and functioning-at-risk. Herbaceous vegetation cover was slightly higher at some degraded sites, as compared with reference, possibly due to the presence of non-native herbaceous vegetation at degraded sites. We did not identify the break between non-functioning and functioning-at-risk due to the small sample size and the similarity between cover values in the degraded and reference standard datasets. Instead, we decided to fit the reference curve, and allow this break to be extrapolated from the regression equation. The 25th percentile value from the degraded dataset (41% cover) would fall within the functioning-at-risk range of index scores. While it may be appropriate to consider these degraded sites as not-functioning and assign them lower index values, the WSTT felt that more conservative reference curves considered the natural variability in woody riparian ecosystems. The minimum index value was set at 0% herbaceous vegetation cover, recognizing that densely wooded sites may have naturally low herbaceous cover which could be further reduced following recent overbank scouring events. The 5th percentile of the reference standard dataset was 6% herbaceous cover, which is illustrative of the variability in herbaceous vegetation cover that may occur at woody sites. The threshold values identified in Table 10-8 are consistent with the 1.0 variable sub-index scores identified by Hauer (2002) for woody reference community types (Table 10-6).

We had originally considered developing a two-sided distribution for this metric in woody sites, recognizing that higher herbaceous cover in woody sites is occurring at some degraded sites. However, we decided against this approach because in early successional stages at restoration sites, there may be very low cover of woody vegetation and high herbaceous cover. We did not want to deter or adversely influence the natural succession stages by incentivizing lower herbaceous vegetation cover in newly establishing woody sites.

For herbaceous vegetation in herbaceous community types, the 75th percentile of the reference data (117%) was used to determine the maximum index value. The median of the reference data (94%) was used to determine the break between functioning and functioning-at-risk. We did not identify the break between non-functioning and functioning-at-risk due to the small sample size at degraded sites. Instead, we decided to fit the reference curve, and allow this break to be extrapolated from the regression equation. The 25th percentile value from the degraded dataset (41% cover) would fall within the not functioning range of index scores, which seems appropriate within herbaceous communities. We used the 5th percentile value (33%)

from the degraded subset to inform the minimum index value. We felt it was important at herbaceous sites to incentivize a minimum threshold of herbaceous cover.

The threshold values for herbaceous cover in herbaceous community types vary substantially from the variable sub-index scores identified by Hauer (2002), likely because the values in Hauer (2002) are based on select hydrogeomorphic cover types located exclusively in the northern Rocky Mountains. The only herbaceous vegetation cover type (Type 6) is often associated with a filled side channel or abandoned back channel, but may be on any surface type. Herbaceous cover in these HGM locations may be low because of recent depositional events. Hauer also states that "The list of cover types is not exhaustive. Whenever a coverage does not appear on this list, it is at the discretion of the assessment team to appropriately evaluate which coverage type it most closely approximates and apply appropriate levels of impact and weighting to the variable subindex scores."

Table 10-8: Threshold Values for Herbaceous Vegetation Cover

Index Value	Field Value (%)	
	Woody Reference Vegetation Cover Type	Herbaceous Reference Vegetation Cover Type
1.00	> 77	≥ 117
0.70	54	94
0.00	0	30

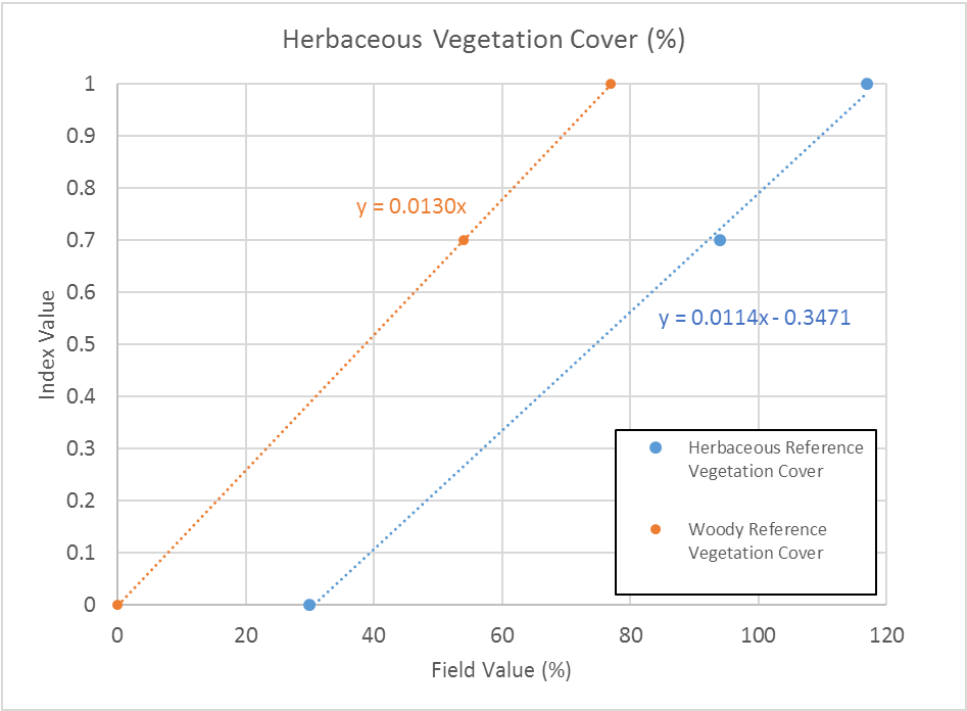

Figure 10-3: Herbaceous Vegetation Cover Reference Curves

Limitations and Data Gaps:

The CNHP dataset had limitations due to sample size and the obvious geographic boundaries of the state. Thus, we had to assume the CNHP dataset would be relevant and translatable to Wyoming due to the overlapping ecoregions and similarity between riparian community types in Wyoming. The CNHP dataset includes data collected between 1992 and 1999; with no sites revisited recently. Additional analysis of these sites may be useful to understand whether changes in climate or other large-scale influences have altered the reference expectation for riparian areas in this region.

Nearly everyone who beta tested the WSQT riparian vegetation metrics commented that it was confusing collecting herbaceous data using the methods proposed. We have simplified the data collection methodology in the new WSQT v1.0 User Manual for herbaceous vegetation cover by identifying herbaceous species absolute areal cover within 1-meter or 5-meter plots.

The reference curve development for the WSQT would benefit from additional Wyoming-specific data to validate the criteria and curves identified above. We are uncertain of the prevalence of naturally occurring herbaceous-only riparian reference communities due to historically altered landscapes, current land uses and altered flow regimes (Jones and Walford, 1995; West and Ruark 2004; Macfarlane et al. 2017). The only certain reference herbaceous communities are those that support broad, highly connected floodplains with anaerobic soil conditions; or those that have natural disturbance (flood or fire) regimes that do not favor the persistence of woody species (Youngblood 1985; West and Ruark 2004). Additional data would also allow for us to consider whether additional stratification or refinement beyond reference community type could occur. This would allow us to consider natural variability in herbaceous cover that would occur across stream sizes, elevations, soil types or between different target herbaceous community composition.

We did not differentiate or evaluate differences in herbaceous riparian vegetation cover across perennial, intermittent or ephemeral systems, and are uncertain if this metric plays a substantial role in differentiating between streams of varying flow permanence. We believe this metric should still be applied in ephemeral stream systems, but would expect they may generally score lower than their perennial counterparts.

10.4. Percent Native Cover

Summary:

Many riparian areas in the western U.S. are heavily influenced by changes in land use, fire regimes, grazing, flow modification and the influx of non-native and invasive species (Macfarlane et al. 2017). Tamarisk and Russian olive have been prolific invaders, and many restoration efforts target the management and eradication of these invasive species (Shafroth et al. 2002). Many riparian areas in the plains and basins that historically (pre-European settlement) contained patches of timber or brush were eventually and progressively reduced to mixed origin herbaceous communities due to the migration of white pioneers, the advance of farming and stock-raising and the introduction of non-native pasture grasses (West and Ruark 2004).

This metric represents relative cover of native species, and is calculated by absolute cover of native species divided by absolute cover of all species at a site. The maximum field value for this metric is 100% cover.

Reference Curve Development:

In the WSQT Beta Version, the NRSA dataset (USEPA 2016) was used to develop reference curves and inform data collection methods. However, following beta testing and field training exercises, the decision was made to use a single, species-level approach for all vegetation cover metrics and change data collection methods to align with the 1987 Wetland Delineation Manual methods, as the Corps field staff and many practitioners are already familiar with this form of data collection. These methods provide absolute cover by species, which is different than the approach used in the beta version. Because of this, the NRSA dataset, which rely on relative cover by strata, were considered no longer applicable for developing reference curves. The WSTT relied on CNHP datasets (see Chapter 1.5) and a small data collection effort in Wyoming to inform the development of reference curves for this metric.

Colorado Natural Heritage Program: Percent native cover values were calculated from sites in the CNHP dataset, described in Section 1.5. Percent native cover was calculated by summing the absolute cover values for all native species and dividing by the total absolute cover value for a site. Statistics were derived from the CNHP dataset for the reference standard and degraded sites (Table 10-9). Sample sizes were limited, particularly for degraded sites and for all sites within the Plains ecoregions.

Table 10-9: Statistics for Percent Native Cover from the CNHP Dataset

Statistic	Percent Native Vegetation Cover (%)	
	Degraded	Reference
Number of Sites (n)	27	487
75th Percentile	91	100
Median	84	99
Mean	77	98
25th Percentile	65	98
5th Percentile	48	89

Percent native cover was consistent across reference standard sites within all ecoregions and reference community types (Figure 10-4), with one exception. In the plains ecoregion at woody reference community types, percent native cover values were lower at both reference and degraded sites than within other ecoregions. This could be reflective land use, flow modification and grazing at the sites included in the dataset. Due to the small sample sizes of this subset, and the consistency across other ecoregions and community types, the WSTT decided not to stratify by ecoregion or reference community type.

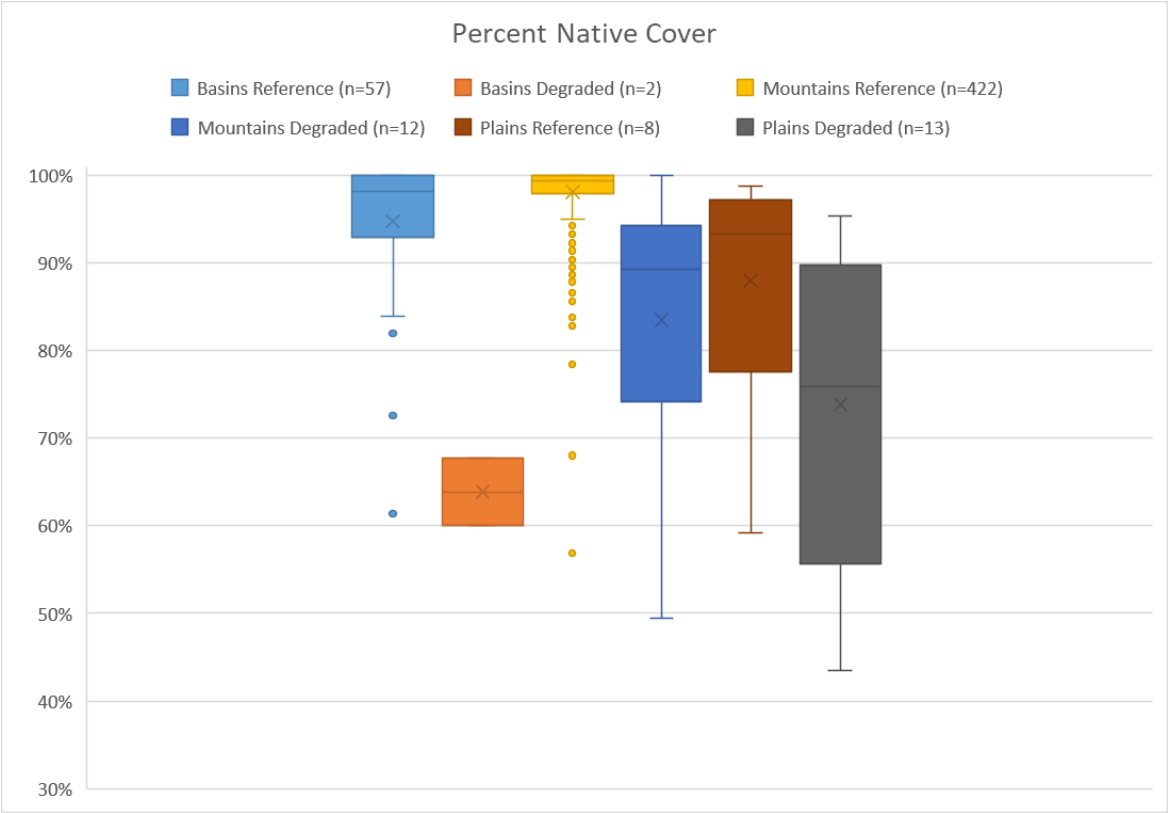

Figure 10-4: Box Plots for Percent Native Cover from the CNHP Dataset. Stratified by Condition (Reference or Degraded) and Ecoregion (Basins, Mountains and Plains)

WSTT data collection: In August 2016 and fall of 2017, the WSTT visited several sites to apply the proposed WSQT methodology for assessing riparian vegetation. These sites were considered to represent minimally disturbed reference standard sites. However, because they are located on public lands, they have likely been subject to some historical use, including grazing and/or timber removal. The percent native cover values from these sites are presented in Table 10-10.

Table 10-10: Percent Native Cover at Reference Sites Visited by the WSTT

Site	Ecoregion	Percent Native Cover (%)
Wood River, above Middle Fork	Mountains	92
Middle Fork Wood River	Mountains	98
Middle Fork Wood River - Upstream	Mountains	100
Jack Creek	Mountains	100

Analysis: In general, the following criteria were used to establish the threshold values using the CNHP dataset. Threshold values are shown in Table 10-11.

- The 75th percentile of reference standard sites were used to determine the maximum index value of 1.00.

- The 75th percentile values from degraded sites was used to determine the threshold between functioning and a functioning at risk condition.
- The 25th percentile cover values from degraded sites were used to inform the threshold between functioning at risk and not functioning condition.
- Minimum index values were extrapolated from the regression equation.

A broken linear curve was used to fit the threshold values. The minimum index value extrapolated from the curve was 46% native cover, which aligns with the 5th percentile from the degraded dataset (48%) and is thus a reasonable minimum value.

Data collected by the WSTT in Wyoming had percent native cover values of 92-100%, which would all fall within the functioning, reference standard range of index scores. As noted above, these sites were in good condition, and had healthy, diverse riparian communities, and the WSTT concluded it was reasonable to characterize these sites as functioning or (high) functioning-at-risk.

Table 10-11: Threshold Values for Percent Native Cover

Index value	Field Value (%)
1.00	100
0.70	91
0.30	65

Limitations and Data Gaps:

The CNHP dataset has limitations, including the obvious geographic boundaries of the state. The WSTT assumed that the CNHP dataset would be relevant and translatable to Wyoming due to the overlapping ecoregions and similarity between riparian community types in Colorado and Wyoming.

The reference curve development for the WSQT would benefit from additional Wyoming-specific data to validate the criteria and curves identified above. Additional data would also allow for us to consider whether stratification is needed.

This metric does not differentiate between upland and hydrophytic native vegetation cover, and as such, may attribute a higher level of functioning to degraded systems that have transitioned to an upland-dominated community. Additional data and research are required to better understand how naturally prevalent upland species are within riparian areas in Wyoming. Many plains and basin riparian systems support upland scrub-shrub communities. While these are often associated with more degraded, incised systems, they can also occur naturally due to specific soil conditions and in more arid areas with lower water tables.

A major challenge is also differentiating between streams of varying flow permanence. We did not differentiate or evaluate differences in native cover across perennial, intermittent or ephemeral systems, or evaluate whether changes in flow regime may facilitate the establishment of non-native species. We believe this metric should be applied in ephemeral stream systems, but would benefit from additional data collection.

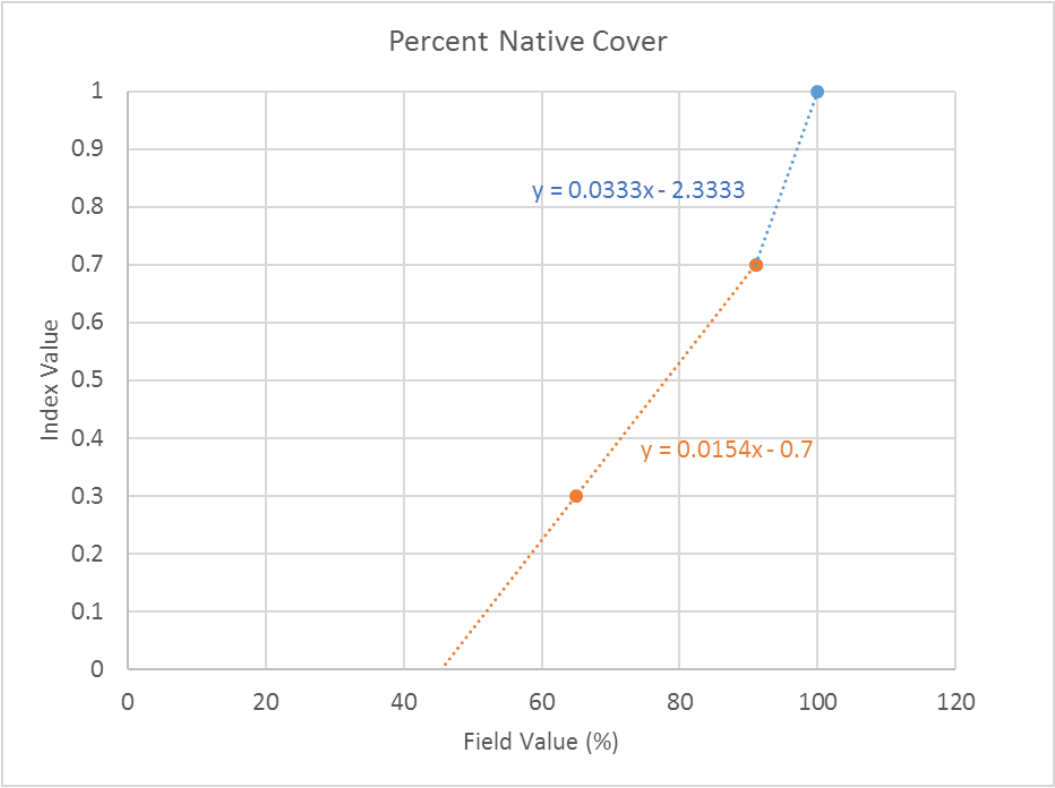

Figure 10-5: Percent Native Cover Reference Curve

Chapter 11. Temperature Parameter

Functional Category: Physicochemical

Function-based Parameter Summary:

Temperature plays a key role in both physicochemical and biological functions. For example, each species of fish has an optimal growth temperature, but can survive a wider range of thermal conditions. Stream temperatures outside of a species' optimal thermal range result in reduced growth and reproduction and ultimately in individual mortality and population extirpation (Cherry et al. 1977). Water temperature also influences conductivity, dissolved oxygen concentration, rates of aqueous chemical reactions, and toxicity of some pollutants. These factors impact the water quality and ability of living organisms to survive in the stream.

Temperature assessments commonly focus on mean and maximum water temperatures, with maximum water temperatures commonly used to inform numeric water quality standards. While comparisons of site condition can be made to numeric standards (e.g., maximum temperature thresholds for aquatic biota), the use of regional reference data can provide a better indication of the degree of degradation and restoration potential than a comparison to temperature standards alone (Roni and Beechie 2013). Emerging monitoring and modeling capabilities are advancing the science on stream temperature, allowing for greater understanding of the temporal and spatial variability of temperature regimes in streams, and expanding the potential range of temperature variables that could inform condition (Steele and Fullerton 2017).

Metric:

- Maximum Weekly Average Temperature (°C)

11.1. Maximum Weekly Average Temperature (MWAT)

Summary:

The Maximum Weekly Average Temperature (MWAT) is a common metric for chronic thermal exposure for fish, and thermal criterion are available for streams throughout Wyoming (Peterson 2017). The MWAT is a chronic criterion that represents the upper bound of the optimum temperature range that supports specific species growth, reproduction, and survival (Brungs et al. 1977). Temperatures that exceed this threshold may limit growth, reproduction, and survival. To calculate the MWAT, first calculate the mean daily temperature for each day in the period of record and then calculate the weekly average temperature on a seven-day rolling basis for the period of record. The MWAT is the largest of these seven-day rolling average values. For the WSQT, the period of record is the month of August. The metric is measured using in-water temperature sensors installed following procedures outlined in the USEPA's 'Best Practices for Continuous Monitoring of Temperature and Flow in Wadeable Streams' (2014).

Reference Curve Development:

Reference curves were derived using data and information presented in Peterson (2017). The values shown in Table 11-1 are the proposed MWAT thermal criteria for Wyoming streams for the five thermal tiers (Peterson 2017). This metric is stratified by ambient stream temperature regime, where Tier I is cold and Tier V is warm. In this study, thermal tiers and associated

thermal criteria were developed using species assemblage data, laboratory-derived thermal tolerance data, and predicted mean August stream temperature as determined by the Air, Water, and Aquatic Environmental Program (AWAE) NorWeST model (Isaak et al. 2017). August was the period used by Issak et al. (2017) to predict summer stream temperature scenarios because of the size of available datasets in August, as well as the strong correlations between August temperatures and other commonly used temperature metrics. Modeled stream temperature data can be accessed through the NorWeST online mapper[2].

Table 11-1: Proposed MWAT Surface Water Thermal Criteria for Wyoming Streams (Peterson, 2017)

Thermal Tier (Stream Classification)	Mean August Stream Temperature (°C)	MWAT Criterion (°C)
Tier I (Cold) Criteria	< 15.5	18.1
Tier II (Cold-Cool) Criteria	15.5 - 17.7	19.3
Tier III (Cool) Criteria	17.7 - 19.9	22
Tier IV (Cool-Warm) Criteria	19.9 - 24.4	26
Tier V (Warm) Criteria	> 24.4	29

The thermal criteria shown in Table 11-1 were used to derive the reference curves for each thermal tier based on the criteria described below and shown in Table 11-2.

- The MWAT criterion identified in Peterson (2017) for each thermal tier were considered to represent the threshold between an index value in the non-functioning range (<0.30) and the functioning-at-risk range. As such, the MWAT criterion equate to an index value of 0.30 as shown in Table 11-1.
- The high (warm) end of the modeled mean August stream temperature ranges shown in Table 11-1 were considered to represent the threshold between an index value in the functioning-at-risk range (0.70) and functioning range. Because an upper temperature extent is not defined for thermal tier V, the mean difference in temperature between index value 0.30 and 0.70 for thermal tiers I-IV (1.7ºC) was used to determine the temperature associated with index value=0.70 (29 - 1.7 = 27.3). A critical assumption made in developing the reference curves for MWAT is that the modeled mean August temperature from Isaak et al. (2017) represents a functioning thermal condition for both physicochemical and biological functions; and in a reference standard site, the MWAT would not exceed the mean August temperature expected for the thermal tier.
- Linear curves fit to the values in Table 11-2 were used to determine temperature values corresponding to index values of 1.00 and 0.00. The final reference curves were reviewed by the Wyoming Stream Technical Team and are shown in Figure 11-1.

[2] https://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html

Table 11-2: Threshold Values for MWAT

Index Value	Field Values (°C) by Thermal Tier (Stream Classification)				
	Tier I (Cold)	Tier II (Cold-Cool)	Tier III (Cool)	Tier IV (Cool-Warm)	Tier V (Warm)
0.70	15.4	15.5	17.7	19.9	27.3
0.30	18.1	19.3	22.0	26.0	29.0

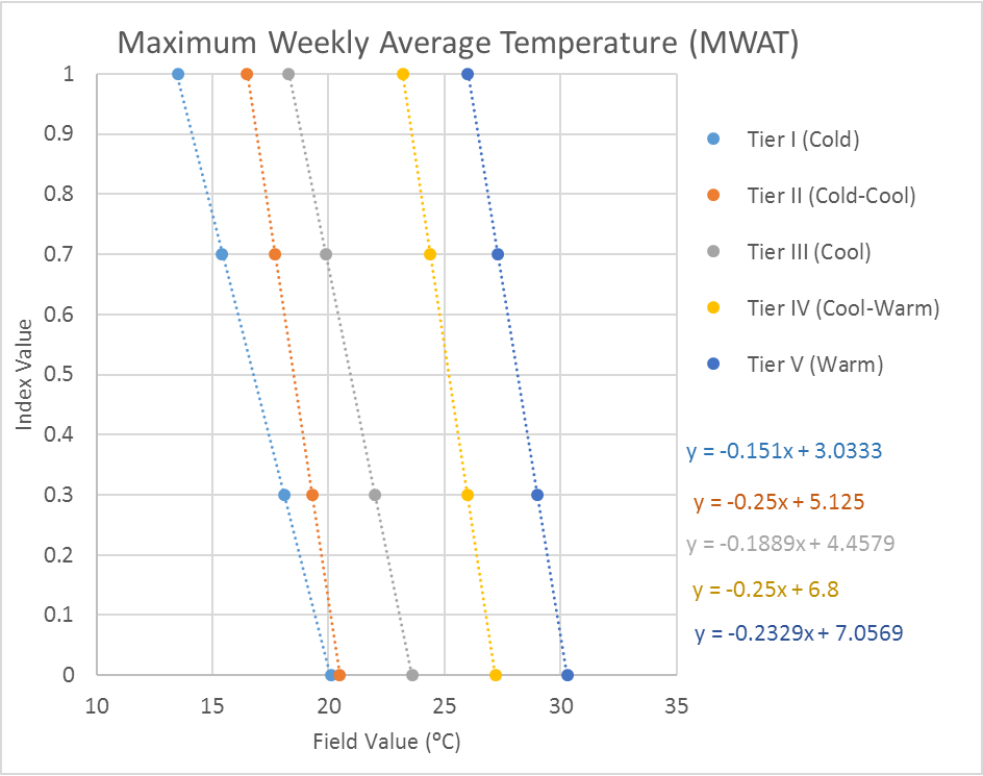

Figure 11-1: MWAT Reference Curves

<u>Limitations and Data Gaps:</u>

This metric relies on the accuracy of the NorWeST historical modeled mean August temperatures for the 1993-2011 baseline period to identify the thermal tier of the stream reach. As such, the historical modeled mean August temperatures are limited by the real-world conditions occurring during the baseline period of the model, and the historical modeled mean August temperature at a particular site may reflect anthropogenic alterations to thermal regimes within temperature monitoring datasets and may not reflect pristine, natural conditions. As a result, some sites may fall within a different thermal tier compared to historical, pristine condition. However, the SQT is primarily used to compare pre- and post- project conditions and this change would still be captured within the tool. Additionally, this metric may not be applied where streams are not included in the NorWeST model unless sufficient monitoring data are

available to determine the thermal tier. The NorWest temperature model data are not available within the Little Missouri, Niobrara, lower North Platte, and South Platte basins.

This metric considers colder MWATs to represent higher functioning for all thermal tiers. Some human activities, such as flow augmentation or hypolimnetic reservoir releases, may cause a stream to be colder than the natural condition. This metric does not capture the potential for reduced condition due to these changes.

Chapter 12. Nutrients Parameter

Functional Category: Physicochemical

Parameter Summary:

Nutrients in stream ecosystems are necessary for growth and survival of aquatic species. Of the nutrients in stream ecosystems, nitrogen and phosphorus are the most important (Allan and Castillo 2007). Excessive nutrients from nonpoint source pollution, particularly runoff from agricultural lands, is one of the leading causes of impairment to streams in the United States (USEPA 2005). While there is a minimum amount of nutrients necessary to support aquatic life, nutrient concentrations often exceed this minimum. This can lead to excess algae growth and resultant degraded aquatic habitat and physicochemical conditions, altered fish and invertebrate communities, occasional fish kills, and aesthetic degradation.

Metric:

- Chlorophyll

12.1. Chlorophyll

Summary:

This metric is a direct measure of the concentration of chlorophyll α (mg/m^2) in stream riffles collected according to procedures outlined in WDEQ/WQD (2017). The sampling index period for this metric has been modified from the WDEQ/WDQ procedure to address late season changes in chlorophyll concentrations. Chlorophyll is the pigment that allows plants (including algae) to use sunlight to convert simple molecules into organic compounds via the process of photosynthesis, and is used in the WSQT as a surrogate for nitrogen (N) and phosphorus (P). Chlorophyll α is the predominant type found in green plants and algae, and concentrations are directly affected by the amount of nitrogen and phosphorus in stream (Dodds and Smith 2016). This metric is preferred to direct sampling of N and/or P because water column N and P concentrations can be misleading when these nutrients are largely assimilated by excess algae and plant growth.

Reference Curve Development:

The Wyoming Department of Environmental Quality (WDEQ) collects nutrient, benthic algae, and chlorophyll data in streams throughout Wyoming as part of its efforts to address nutrient pollution. There are currently no numeric criteria for Wyoming's streams regarding chlorophyll but the WDEQ Water Quality Division provided chlorophyll datasets to develop reference curves for the WSQT. This dataset consisted of 467 samples that were collected between July 2007 and October 2015.

The dataset classified sites as Reference, Non-Reference, or Degraded using the procedure described by Hargett (2011). For this dataset, reference standard sites are considered to approximate best attainable, and not necessarily pristine, conditions for the ecoregion based on presence/absence of anthropogenic stressors in the watershed and reach. Many sites that are identified as degraded were classified as such due to watershed or reach-scale factors that may be unrelated to nutrients. The data were split into two datasets for the subsequent analysis:

sites identified as reference standard (n = 120) and sites identified as non-reference or degraded (n = 347).

A two-sided outlier test was performed on each dataset and identified outliers were removed from the analysis. A visual assessment of the datasets indicated that the data were not normally distributed (Figure 12-1). The natural log of the data points was calculated and the XLSTAT statistical package for Microsoft Excel was used to perform a two-sided Grubbs outlier test with a 5% significance value on the transformed datasets. Twenty-six sites total were removed during the outlier test, eight from the reference standard dataset and eighteen from the non-reference and degraded dataset.

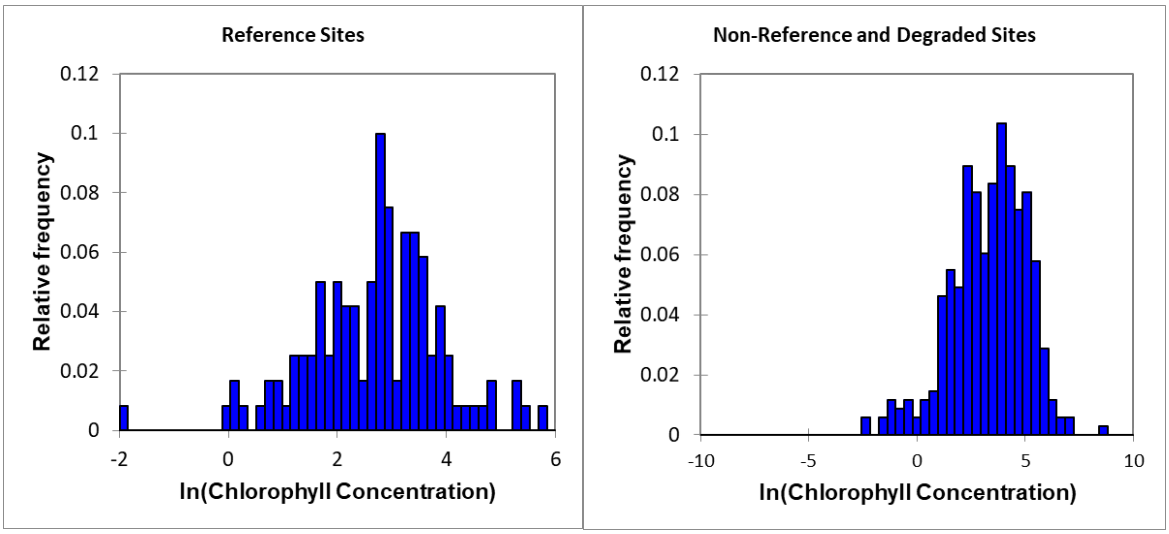

Figure 12-1: Histogram of Chlorophyll Concentrations (mg/m^2) for Reference and Non-Reference Datasets

The datasets contained values obtained using several sampling methods based on different habitat types, e.g., epilithic (coarse substrate), episammic (pea gravel ≤5mm/sand), epidendric (woody snag), and epipelic (silt). The reference data contains mostly epilithic samples, and thus there were inadequate reference data to develop reference curves for episammic, epidendric, and epipelic samples. Therefore, in addition to the outliers, samples collected using methods other than epilithic were removed from the datasets.

WDEQ has observed late season samples with high algal biomass, but not necessarily high chlorophyll concentrations, most likely due to chlorophyll being in a degradation phase because of decreasing water temperatures and shortened photoperiod. Final datasets consisted of samples collected between July 15 to October 1 for the mountains and between June 15 to October 1 for the plains and basins. To address temporal variability, 81 data points were removed that fell outside of this date range.

The dataset was stratified by ecoregion (as defined in Table 1-3) to address geographic variability. Statistics for each dataset, stratified by ecoregion, are provided in Table 12-1.

Table 12-1: Statistics for Chlorophyll Concentrations from the WDEQ Dataset

Statistic	Chlorophyll Concentrations (mg/m^2) by Ecoregion and Condition			
	Mountains		Plains and Basins	
	Non-Reference & Degraded	Reference standard	Non-Reference & Degraded	Reference standard
Number of Sites (n)	50	60	183	13
Geometric Mean	18	12	37	16
Average	42	20	83	24
Standard Deviation	54	20	105	22
Minimum	1.4	1.4	1.2	3.4
25th Percentile	5	6	13	9
Median	19	14	42	16
75th Percentile	53	27	117	29
Maximum	228	100	625	79

The statistics for the two datasets were evaluated using best professional judgement and the following decisions were made to determine the threshold values shown in Table 12-2.

- Given the non-normal distribution of the datasets and the criteria used to identify reference sites, the geometric mean from the reference standard datasets were used to inform the maximum index score.
- The threshold between functioning and functioning-at-risk index scores was determined from the 75th percentile from the reference standard datasets.
- The threshold between not functioning and functioning-at-risk index scores was determined from the 75th percentile from the non-reference and degraded datasets.
- The minimum index value for Mountains is the x-intercept from the best-fit curve.
- Because the curve for Plains and Basins does not intercept the x-axis, the minimum index value for Plains & Basins was based upon a value identified in the literature to represent a threshold for excess benthic chlorophyll independent of landform or ecoregion (Dodds et al. 1998; Suplee et al. 2009; Welch et al. 1988).

A logarithmic curve best fit the threshold values that were selected from the statistical summary of the data and the single literature value. Figure 12-1 shows the fit of the logarithmic curves to the points identified in Table 12-2. Note that the index values calculated by the WSQT differ from the threshold values identified in Table 12-2, as the threshold values were used as an initial step to define the best fit logarithmic curves. The curve equations were then used to calculate index values from chlorophyll field values in the WSQT.

Table 12-2: Threshold Values for Chlorophyll

Index Value	Field Value (mg/m^2)	
	Mountains	Plains & Basins
1.00	12	16
0.70	27	29
0.30	53	117
0.00	-	150

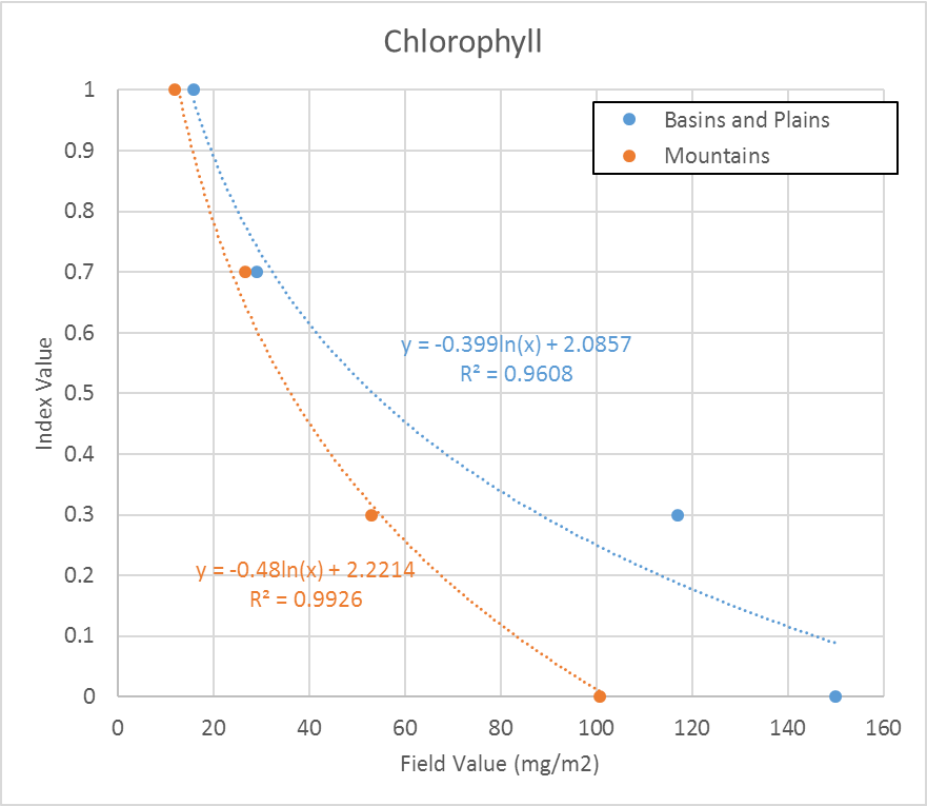

Figure 12-2: Chlorophyll Reference Curves

<u>Limitations and Data Gaps:</u>

The chlorophyll metric is only applicable to stream reaches where epilithic samples can be collected (WDEQ 2018). This is limited to sites where riffles are present and contain gravel or larger bed materials. The dataset contained too few samples collected via episammic (pea gravel ≤5mm/sand), epidendric (woody snag), and epipelic (silt) methods to test comparability between sample sites. As such, these were excluded from the data analysis. Reference curves

may be developed for these substrate types and sampling methods as more data become available.

The reference curves are based on large regional groupings and some sites may be unfairly assessed due to natural variations within regions. This limitation can be addressed as more data are gathered and further statistical analyses performed to determine statistically significant differences between regions throughout the state. Until additional data are available to further stratify and refine the reference curves, this variability should be dealt with on a case-by-case basis within the WSQT.

Use of the chlorophyll metric to represent the nutrient parameter assumes a direct correlation between nitrogen and/or phosphorus and benthic algae growth. Factors such as water clarity, canopy cover, water temperature, and grazing by fish and invertebrates also affect benthic algae biomass, thus also chlorophyll concentrations, but are not directly accounted for by this metric. Site specific conditions need to be considered when applying this metric.

Chapter 13. Macroinvertebrates Parameter

Functional Category: Biology

Function-based Parameter Summary:

Benthic macroinvertebrates are commonly used as indicators of stream ecosystem health, and was included as one of the original parameters described in Harman et al. (2012). Benthic macroinvertebrates are key components of aquatic food webs that link organic matter and nutrient resources (e.g., leaf litter, algae and detritus) with higher trophic levels. They are reliable indicators of condition because they spend all or most of their lives in water, and differ in their tolerance to pollution. Macroinvertebrates respond to environmental stressors in predictable ways, are relatively easy and cost-effective to collect and identify in a laboratory, often live for more than a year and have limited mobility. Unlike fish, macroinvertebrates cannot easily escape pollution, thus they have the capacity to integrate the effects of the stressors to which they are exposed.

Metrics:

- Wyoming Stream Integrity Index (WSII)
- River Invertebrate Prediction and Classification System (RIVPACS)

13.1. Wyoming Stream Integrity Index (WSII)

Summary:

The Wyoming Stream Integrity Index (WSII) is a statewide regionally-calibrated macroinvertebrate-based multimetric index designed to assess biological condition in Wyoming perennial streams (Hargett 2011). Wyoming Stream Integrity Index scores are calculated by averaging the standardized values of selected metrics (composition, structure, tolerance, functional guilds) derived from the riffle-based macroinvertebrate sample (WDEQ 2018). The metrics included in the WSII are those that best discriminate between reference standard and degraded waters. The assessment of biological condition is made by comparing the index score for a site of unknown biological condition to expected values derived from appropriate regional reference sites that are minimally or least impacted by human disturbance. The WSII is one of two biologic indicators of aquatic life use support used by WDEQ (Hargett 2011).

Reference Curve Development:

WDEQ collected 1,488 benthic macroinvertebrate samples from riffles throughout the state between 1993 and 2009 (Hargett 2011) and used this dataset to determine aquatic life use support thresholds for each bioregion (Table 13-1). According to Hargett (2011), “there are three categories of aquatic life use attainment based on biological integrity: ‘full-support’, ‘indeterminate’ and ‘partial/non-support’. The numeric thresholds for these narrative categories vary across bioregions, though all are developed using the same method. For each bioregion, scores that exceed the 25th percentile of reference calibration scores is identified as ‘full-support’ of aquatic life uses. Index scores below the 25th percentile of reference calibration scores are trisected into equal portions. Scores in the upper 1/3 of this trisection are identified as ‘indeterminate’ which is not an attainment category but is rather a designation that would require the use of ancillary information and/or additional data in a weight of evidence evaluation

to determine a proper narrative assignment (e.g. full or partial/non-support). Scores that fall in the lower 2/3 of the trisection are assigned a 'partial/non-support' designation which indicates the resident biota are subjected to substantial anthropogenic stressors."

Table 13-1: WSII Use Support Values for each Bioregion in Wyoming (Hargett, 2011)

Bioregion	WSII Value	
	Partial/ Non-Support	Full-Support
Volcanic Mountains & Valleys	< 46.2	> 69.3
Granitic Mountains	< 40.2	> 60.3
Sedimentary Mountains	< 34.8	> 52.3
Southern Rockies	< 32.6	> 48.8
Southern Foothills & Laramie Range	< 44.5	> 66.7
Bighorn Basin Foothills	< 40.6	> 60.9
Black Hills	< 30.7	> 46.1
High Valleys	< 32.5	> 48.8
SE Plains	< 36.7	> 55.1
NE Plains	< 38.9	> 58.4
Wyoming Basin	< 26.2	> 39.9

Through consultation with WDEQ, the threshold between functioning and functioning-at-risk index values (0.70) was equated to the threshold indicating full-support of aquatic life uses. Similarly, the WDEQ threshold for partial support or non-support of aquatic life uses was equated to the threshold between functioning-at-risk and not functioning in the WSQT (0.30).

The maximum index score (1.00) in the WSQT was set equal to the 75th percentile of WSII scores at reference sites by bioregion. These values represent an attainable biological condition for reference standard sites in each bioregion of the state. The minimum index score (0.00) was set equal to the 5th percentile of the test and degraded sites WSII scores. The 5th percentile of the WSII scores at the test and degraded sites represents the lowest non-outlier value from non-reference standard sites in each bioregion. The WSII threshold values and reference curves are shown in Table 13-2 and Figure 13-1.

To fit the thresholds outlined in Table 13-2, linear reference curves were used. If the coefficient of correlation (R^2) for any linear fit was less than 0.99, the linear curves were broken to ensure the calculated index values fit the threshold values.

Table 13-2: Threshold Values for WSII Scores

Bioregion	Field Values by corresponding Index Value (i)			
	i = 0.00	i = 0.30	i = 0.70	i = 1.00
Volcanic Mountains & Valleys	24.9	46.2	69.3	88.1
Granitic Mountains	32.6	40.2	60.3	74.9
Sedimentary Mountains	16.6	34.8	52.3	70.8
Southern Rockies	5.1	32.6	48.8	82.2
Southern Foothills & Laramie Range	30.7	44.5	66.7	85.3
Bighorn Basin Foothills	3.9	40.6	60.9	80.8
Black Hills	12.8	30.7	46.1	65.7
High Valleys	17.1	32.5	48.8	78.2
SE Plains	10.4	36.7	55.1	87.0
NE Plains	1.6	38.9	58.4	95.8
Wyoming Basin	5.3	26.2	39.9	64.5

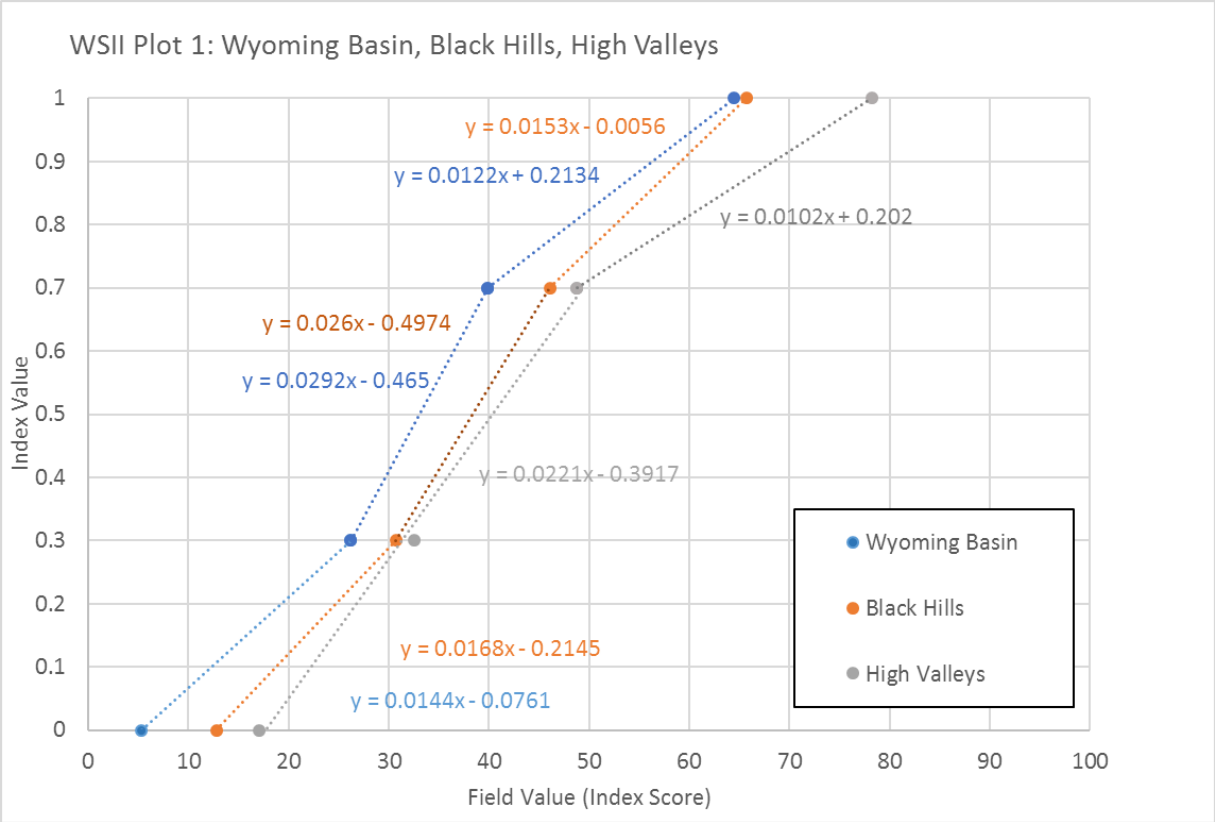

Figure 13-1a: WSII Reference Curves

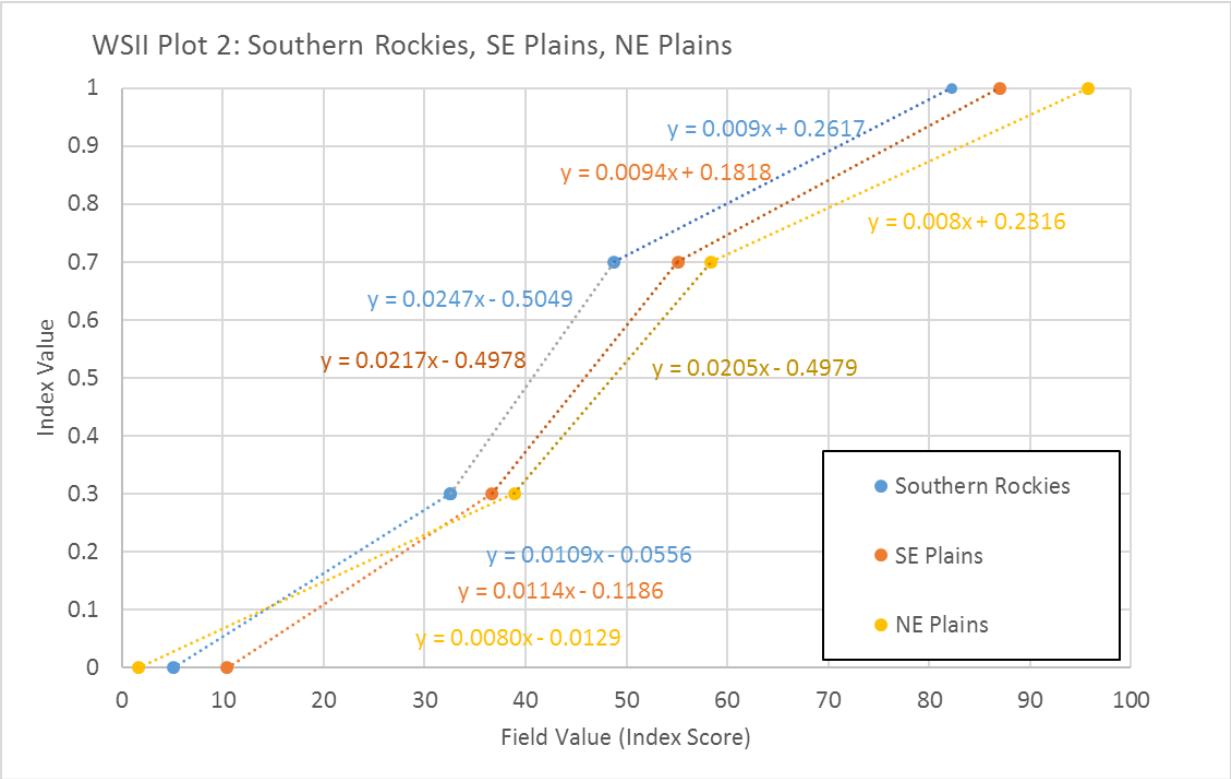

Figure 13-1b: WSII Reference Curves

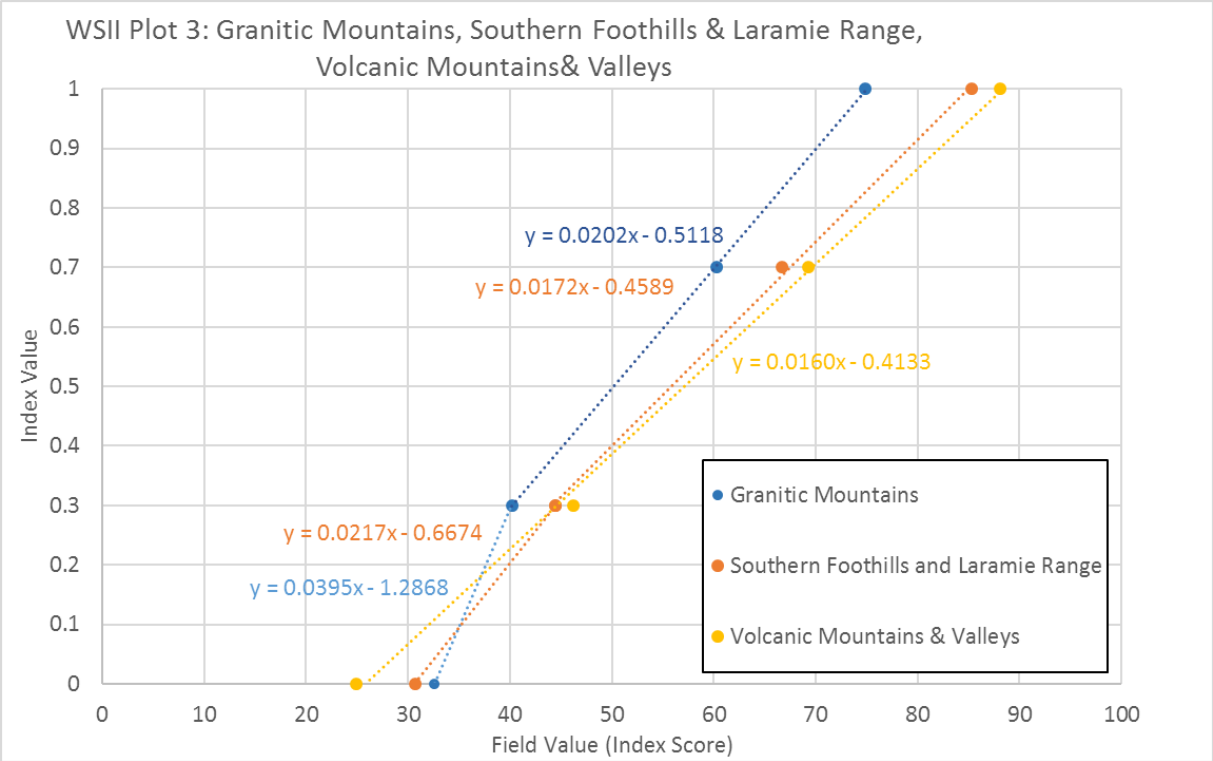

Figure 13-1c: WSII Reference Curves

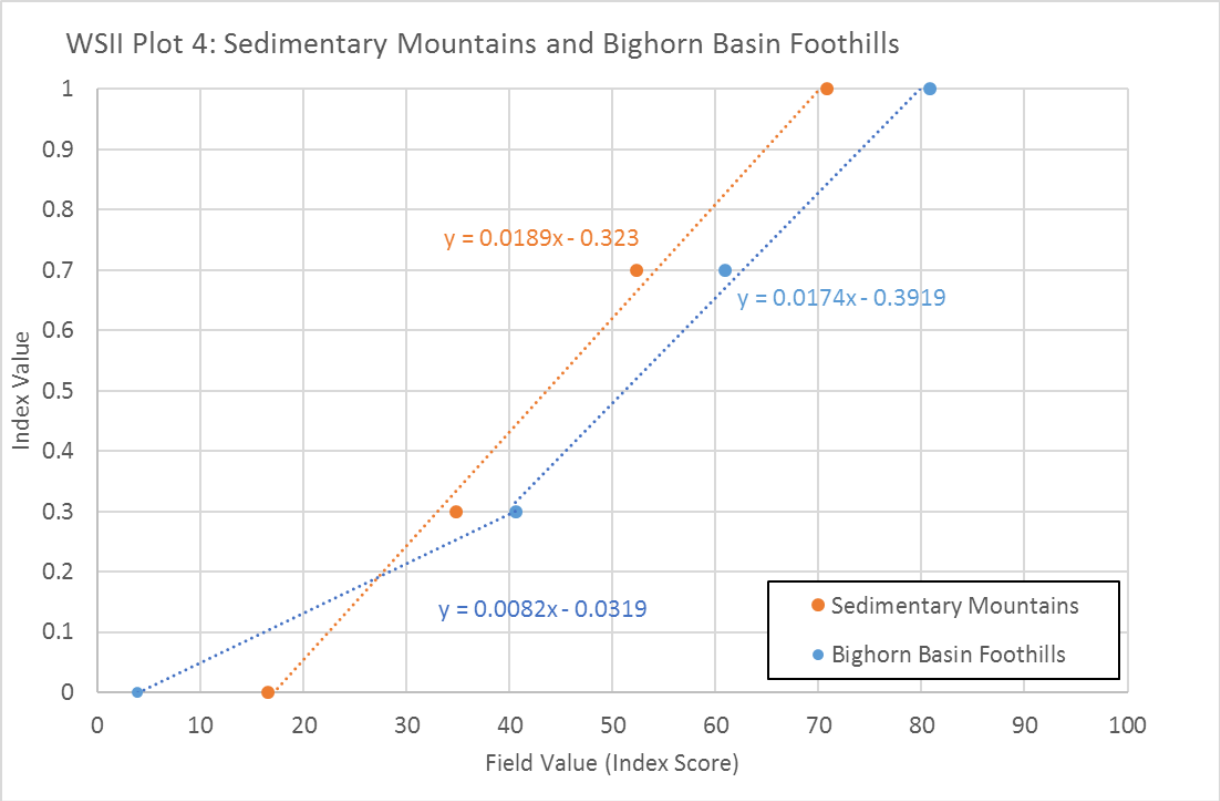

Figure 13-1d: WSII Reference Curves

<u>Limitations and Data Gaps:</u>

A complete description of WSII limitations is included in Hargett (2011). Limitations of the WSII models for the WSQT include:

- Incomplete representation of the reference condition for streams in particular regions will result in less accurate assessments of biological condition for those particular regions. This concern is most apparent in three sub-regions within the greater Wyoming Basin bioregion: streams in the Bighorn Basin of north-central Wyoming, non-montane spring-fed stream segments within the interior of the Wyoming Basin and mixed origin streams of extreme southwest Wyoming.
- Though bioregions are delineated by discrete boundaries, biota and environmental characteristics along the bioregion peripheries (i.e. ecotones) do not always follow these man-made boundaries. Thus, bioregional boundaries should be viewed as transitional, having both similarities and differences with adjacent bioregions. For that reason, expected reference conditions for stream segments located along bioregion peripheries may not necessarily be those represented in the specific bioregion where the segment resides. In addition, a stream segment whose watershed is predominantly located in an adjacent bioregion other than the bioregion where the stream segment resides may best be evaluated to the reference condition of that adjacent bioregion. In these situations, the user is encouraged to deduce the proper expected reference condition for biological condition evaluation using reasonable ecological justifications and a weight-of-evidence approach.

- There is inadequate representation for stream segments with very small watersheds < 12 km^2 (< 5 mi^2) and those located at high montane elevations > 2,740 m (> 9,000 ft.). These stream segments may prove difficult to accurately assess with the WSII.
- Because the WSII was developed with quantitative data collected from targeted riffle/run habitats, it cannot be used to evaluate multi-habitat samples collected with dip nets or other semi-quantitative methods.
- The WSII should not be used to assign attainment category ratings on ephemeral or intermittent streams segments or extremely low-gradient lentic-type systems since the WSII was specifically developed to evaluate the biological condition from perennial lotic systems.

13.2. River Invertebrate Prediction and Classification System (RIVPACS)

Summary:

The WY RIVPACS is a quantitative multivariate biological model that makes site-specific predictions of the benthic macroinvertebrate taxa expected (E) in the absence of anthropogenic stressors for streams and rivers in Wyoming using a network of minimally or least disturbed reference sites. Expectations are based on probabilities of reference group membership using several abiotic predictor variables (latitude, longitude, watershed area, bioregion, and alkalinity) and must be calculated by WDEQ. The ratio (O/E score) of the taxa observed in the stream (O) from the expected (E) taxa is a community-level measurement of biological condition.

Reference Curve Development:

WDEQ collected 1,488 benthic macroinvertebrate samples from riffles throughout the state between 1993 and 2009 (Hargett 2012). WDEQ used this dataset to determine aquatic life use support thresholds for each bioregion (Table 13-3). According to Hargett (2012), “there are three categories of aquatic life use attainment based on biological integrity: ‘full-support’, ‘indeterminate’ and ‘partial/non-support’. Development of numeric thresholds for these three narrative categories is based on interval and equivalence tests described by Kilgour et al. (1998) and applied within each of Wyoming’s eleven bioregions. This EPA-approved statistical methodology establishes ecologically reasonable numeric thresholds based on the mean, variation and 5th percentile of reference site O/E values within a bioregion. O/E values that fall below the interval threshold are considered significantly different from the 5th percentile of reference site O/E values for that bioregion and are assigned a ‘partial/non-support’ status. O/E values that are greater than the equivalence threshold are considered statistically similar to O/E values within the 95% confidence interval of the reference site distribution and are thus assigned a ‘full-support’ status. O/E values that fall between the interval and equivalence thresholds would be considered ‘indeterminate’ which is not an attainment category, but is rather a designation that would require the use of ancillary information and/or additional data in a weight-of-evidence evaluation to determine a proper narrative assignment (e.g. full or partial/non-support).”

Through consultation with WDEQ, the threshold between functioning and functioning-at-risk index values (0.70) was equated to the threshold indicating full-support of aquatic life uses. Similarly, the WDEQ threshold for partial support or non-support of aquatic life uses was equated to the threshold between functioning-at-risk and not functioning in the WSQT (0.30).

Table 13-3: RIVPACS O/E Score Use Support Thresholds for each Bioregion in Wyoming (Hargett 2012)

Bioregion	RIVPACS O/E Score	
	Partial/ Non-Support	Full-Support
Volcanic Mountains & Valleys	0.6456	0.8646
Granitic Mountains	0.6468	0.8832
Sedimentary Mountains	0.6825	0.8234
Southern Rockies	0.6208	0.8917
Southern Foothills & Laramie Range	0.6838	0.8818
Bighorn Basin Foothills	0.6310	0.8445
Black Hills	0.5940	0.8813
High Valleys	0.6847	0.8599
SE Plains	0.5144	0.7813
NE Plains	0.5199	0.7500
Wyoming Basin	0.6351	0.8158

The maximum index score (1.00) in the WSQT was set equal to the 75th percentile of RIVPACS scores at reference standard sites by bioregion. These values represent an attainable biological condition for reference standard sites in each bioregion of the state. The minimum index score (0.00) was set equal to the 5th percentile of the test and degraded sites RIVPACS scores. The 5th percentile of the RIVPACS scores at the test and degraded sites represents the lowest non-outlier value from non-reference standard sites in each bioregion. The RIVPACS threshold values are shown in Table 13-4.

To fit the thresholds outlined in Table 13-4, linear reference curves were used. If the coefficient of correlation (R^2) for any linear trend was less than 0.99, the linear curve was broken to best align with the threshold values. Reference curves are shown in Figure 13-2.

Table 13-4: Threshold Values for RIVPACS O/E Score

Bioregion	Field Values by corresponding Index Value (i)			
	i = 0.00	i = 0.30	i = 0.70	i = 1.00
Volcanic Mountains & Valleys	0.21	0.65	0.86	1.21
Granitic Mountains	0.59	0.65	0.88	1.09
Sedimentary Mountains	0.42	0.68	0.82	1.17
Southern Rockies	0.27	0.62	0.89	1.18
Southern Foothills & Laramie Range	0.29	0.68	0.88	1.20
Bighorn Basin Foothills	0.41	0.63	0.84	0.92
Black Hills	0.37	0.59	0.88	1.08
High Valleys	0.42	0.68	0.86	1.14
SE Plains	0.34	0.51	0.78	1.12
NE Plains	0.11	0.52	0.75	0.98
Wyoming Basin	0.15	0.64	0.82	1.18

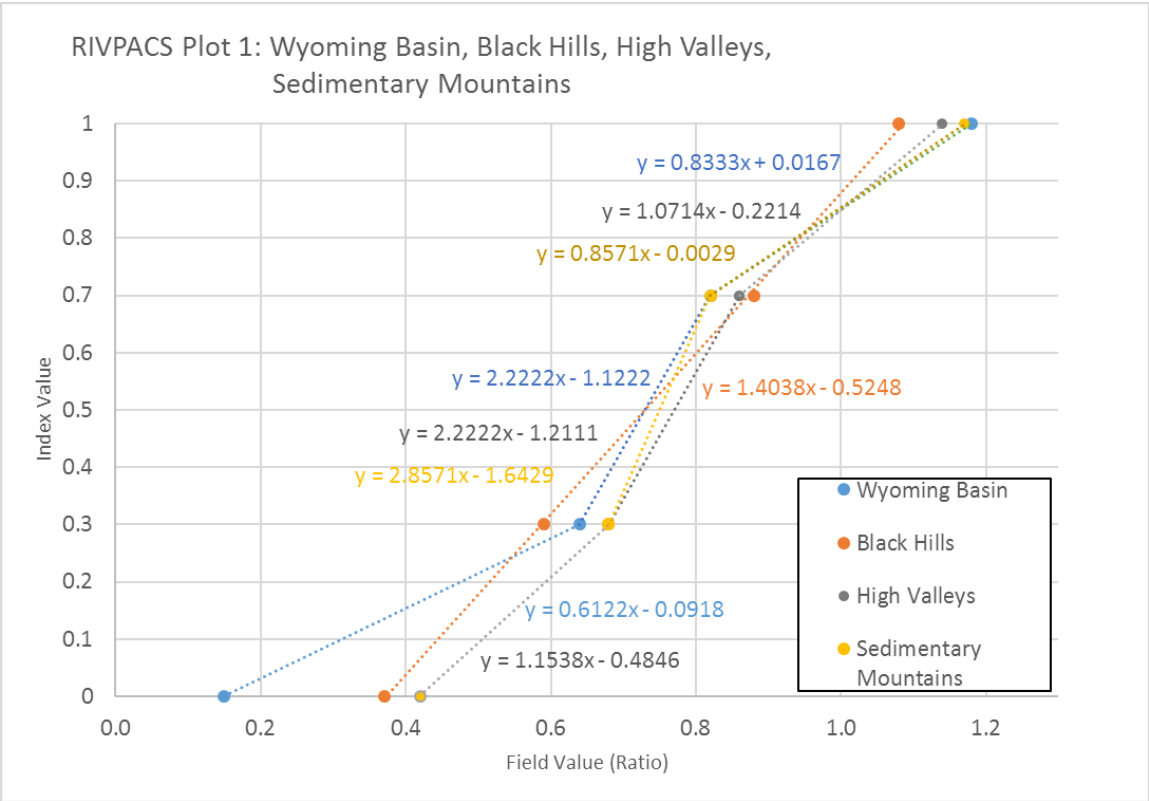

Figure 13-2a: RIVPACS Reference Curves (Note: The High Valleys and Sedimentary Mountains share the same curve for index values 0.00-0.30)

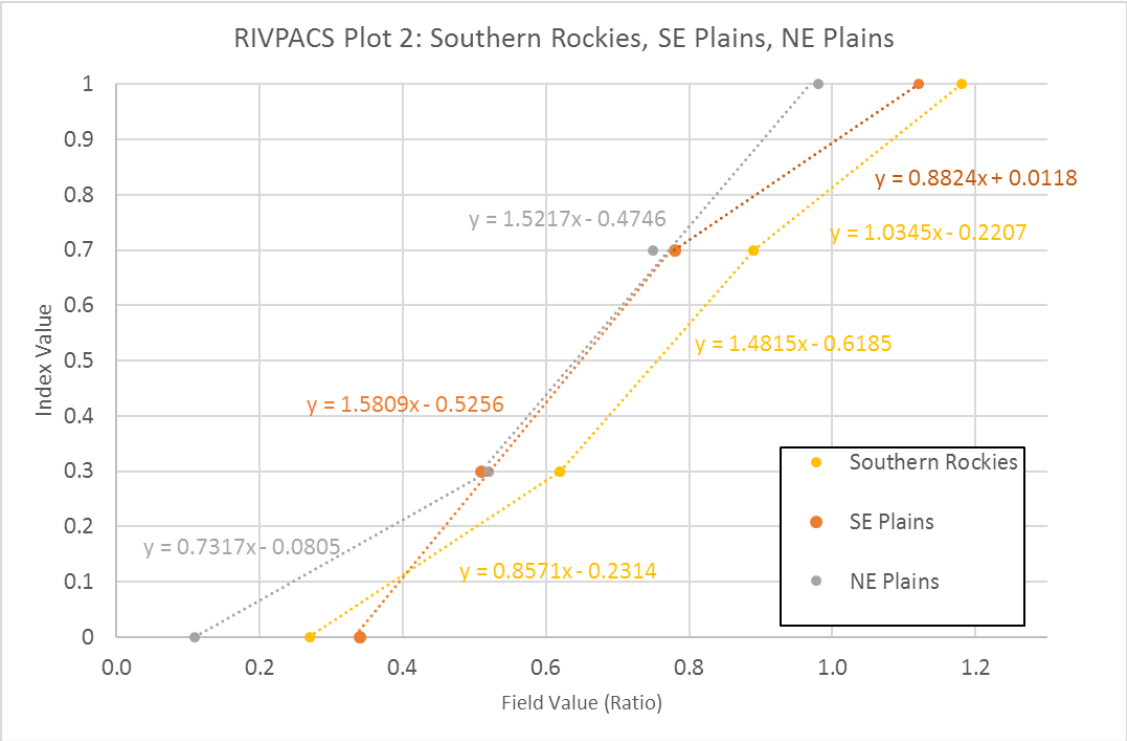

Figure 13-2b: RIVPACS Reference Curves

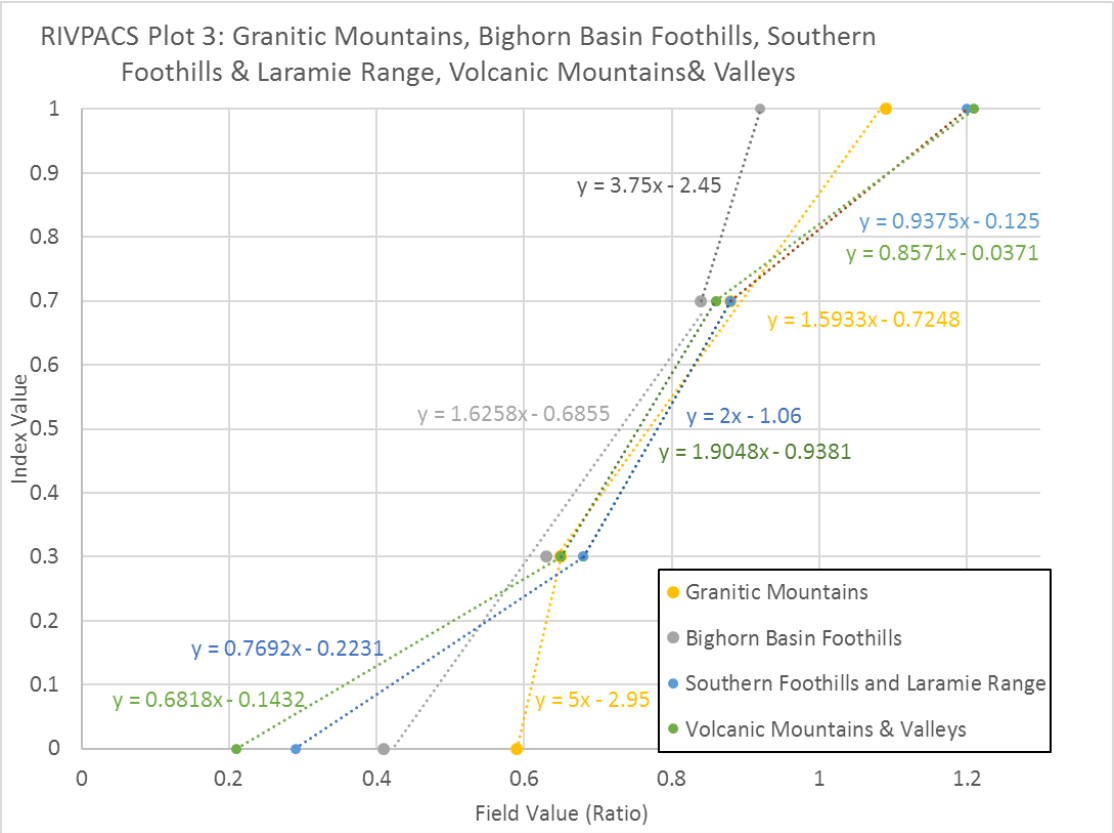

Figure 13-2c: RIVPACS Reference Curves

Limitations and Data Gaps:

WY RIVPACS is designed to flag samples that fall outside the experience of the model and thus prevent extrapolation of predictions to environmental settings beyond those used in model development. In some cases, samples may be within the experience of the model, but biological condition could still be somewhat under or over-predicted by the WY RIVPACS due to:

1. Inadequate reference site representation for stream segments with very small watersheds < 12 km^2 (< 5 mi^2) and those located at high montane elevations > 2,740 m (> 9,000 ft.). These stream segments may prove difficult to accurately assess with the WY RIVPACS.
2. Inadequate reference site representation within three sub-regions of the Wyoming Basin bioregion: the Bighorn Basin of north-central Wyoming (excluding the foothills), non-montane spring-fed stream segments within the interior of the Wyoming Basin and mixed origin streams of extreme southwest Wyoming.
3. Because the WY RIVPACS was developed with quantitative data collected from targeted riffle/run habitats, it cannot be used to assign aquatic life use attainment category ratings to multi-habitat samples collected with dip nets or other semi-quantitative methods.
4. WY RIVPACS should not be used to assign aquatic life use attainment category ratings on ephemeral or intermittent streams segments or extremely low-gradient lentic-type systems since the WY RIVPACS was specifically developed to evaluate the biological condition for perennial lotic systems.

For a more complete description of WY RIVPACS limitations, see Hargett (2012).

Chapter 14. Fish Parameter

Functional Category: Biology

Function-based Parameter Summary:

Fish are an integral part of many functioning stream systems, and are an important management priority within Wyoming. Fish populations require adequate streamflow, water quality and habitat availability to support their life history requirements (Harman et al. 2012). Different species vary in their habitat and life histories, and are adapted to unique stream temperature and flow regimes. Wyoming contains 78 species of fish, and nearly 40% of the State's population are anglers (WGFD 2017). Wyoming streams are managed to support native species as well as native and non-native sport fisheries. Native fish assemblages vary across Wyoming's six major river basins, with the eastern warm water rivers containing a much higher native fish diversity than streams within cold-water, montane regions (WGFD 2017).

This parameter is intended to document several aspects of Wyoming fish assemblages, including the native diversity of the fish community in comparison to reference standards, the presence of Species of Greatest Conservation Need, and the biomass of sportfish populations.

Since there are no existing statewide biological indices used for fish in Wyoming, metrics and reference curves for fish were developed by the WSTT in consultation with regional fisheries biologists at the WGFD. Native fish metrics include a measure of native fish diversity and presence/absence of Species of Greatest Conservation Need (SGCN). Native fish metrics focus on presence/absence metrics instead of abundance metrics due to the large inter-annual variability that naturally occurs in native fish populations. A game species biomass metric is also included to capture post-project increases in biomass following restoration projects. This metric is only intended to be applied at restoration sites where game species are identified as a management priority by WGFD.

Reference standards for native fish species and SGCN are derived from the expected species assemblages within the six major river basins in Wyoming, stratified by differences in stream temperature (cold, transitional, warm) and gradient (WSQT v1.0; Appendix C). These reference standards were based on the species lists provided in the Statewide Wildlife Action Plan (WGFD 2017), and were further refined through consultation with WGFD regional fisheries biologists. The reference standard is defined as the fish species that *should* be naturally present at the site *but for* anthropogenic constraints. Anthropogenic constraints, such as culverts, flow alteration, and downstream barriers, may limit the current presence of native fish species; and may limit the restoration potential at a site if those constraints are not removed as part of a project. The reference standard does not include species that have been extirpated and for which there are no plans or targets for reestablishment. The species assemblage lists provide a preliminary estimate of the expected number of native fish within a particular basin and thermal regime. Given the natural variability in fish assemblages within any basin due to underlying factors such as geology, flow regime, or natural barriers, the expected number of species may need to be modified based on sub-basin characteristics. The WSQT recommends project-specific coordination with WGFD regional fish biologists to account for natural factors that may influence species distribution and any necessary modifications to the species assemblage list.

Metrics:

- Native Fish Species Richness (% of expected)
- Species of Greatest Conservation Need (SGCN) Absent Score
- Game Species Biomass (% increase)

14.1. Native Fish Species Richness

Summary:

This metric measures native fish species richness based on presence/absence data. This metric is calculated as the observed number of native species divided by the expected number of native species.

Native species distributions naturally vary between river basins and within any basin due to underlying factors such as geology, flow regime and duration, water temperatures, or natural barriers. A comparison of the number of native species currently observed to the expected number of species (O/E) is an indicator of anthropogenic disturbance that locally reduces species diversity. Anthropogenic disturbances that could alter the native species assemblages include barriers, flow alteration, water quality impairments, introduction of non-native species, habitat degradation or other disturbances that could alter spawning, rearing or breeding habitats. Reference standards for native fish species are derived from the expected species assemblages within the six major river basins in Wyoming, stratified by differences in stream temperature (cold, transitional, warm) and gradient (WSQT v1.0; Appendix C) and are defined in the Parameter Summary above.

Reference Curve Development:

Reference curves were developed based on best professional judgement of regional fisheries biologists in Wyoming. Achieving 100% native species richness was considered to represent a reference standard condition, and was assigned an index value of 1.00. The absence of one or more native species was no longer meeting the reference standard, and was considered to represent a functioning-at-risk condition, i.e., the system has the potential to support full native species diversity, but does not. The threshold between functioning-at-risk and non-functioning index scores was identified as 75%, meaning if 25% or more of the native species were absent from the site, the index scores would fall within the non-functioning range, i.e., the system does not support native species diversity. The best fit line was extrapolated to zero, and a minimum index score of 0.00 is assigned when less than 58% of native species are present. Threshold value and reference curves are shown in Table 14-1 and Figure 14-1.

Because the total number of native species varies across basins, as well as between the thermal regimes within a basin, the metric is normalized by the expected number of species within that basin and regime. As such, no additional stratification was considered. We recognize that the presence or absence of a single species will more strongly influence the score in basins with naturally lower native species richness, however in basins with naturally low native species richness, each individual species contributes more to the species diversity at the site.

Table 14-1: Threshold Values for Native Fish Species Richness

Index Value	Field Value (% of expected)
1.00	100
0.69	99
0.29	75

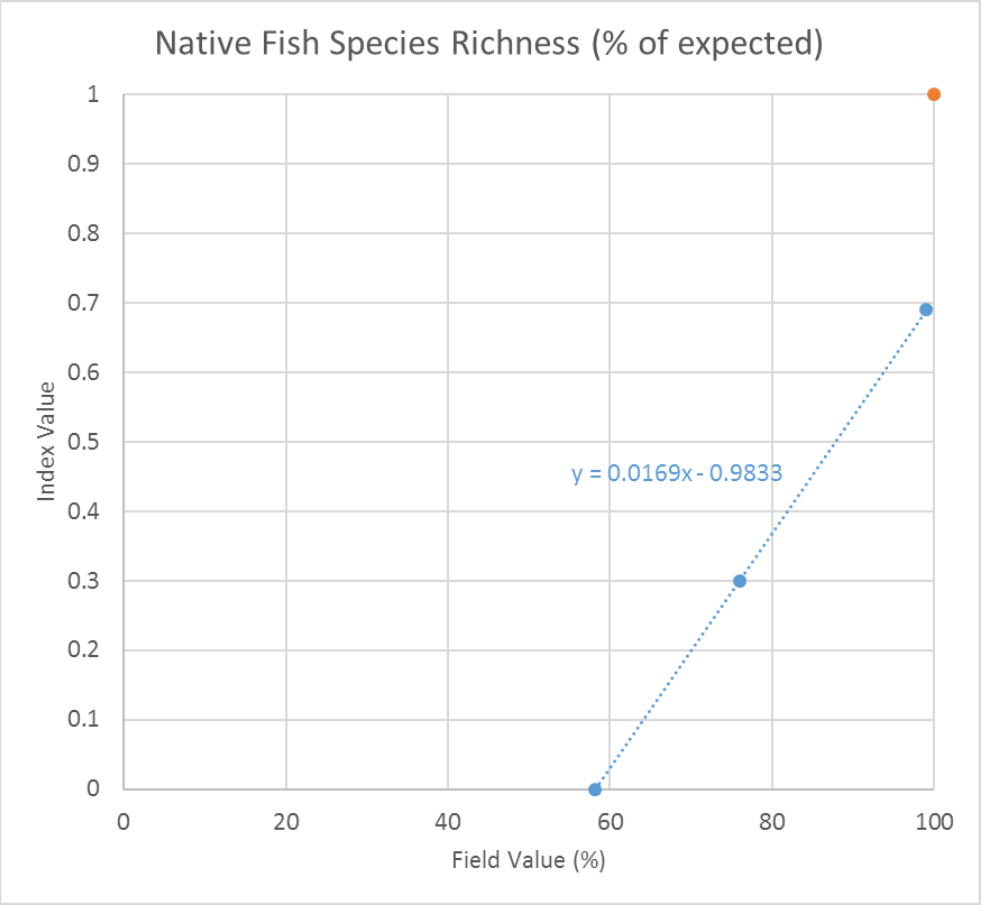

Figure 14-1: Native Fish Species Richness Reference Curve

<u>Limitations and Data Gaps:</u>

There is uncertainty about the fish species that comprise the natural fish community for most locations and it requires a judgment call by a professional fisheries biologist to establish a species list for a location. The assembled native species lists available in Table C.1 of the WSQT v1.0 User Manual provide a starting point for the potential maximum number of species at a location within a given river basin and stream temperature/gradient class. However due to variability in sub-basin geology, flow regime and other natural factors, these lists require coordination with the WGFD prior to finalizing an expected number of native species at any

given site. In addition, the “cold”, “transitional” and “warm” stream systems are not explicitly defined numerically in terms of slope or temperature. Rather, these distinctions are made in a general sense to broadly help differentiate among Wyoming’s mountain, foothill and high plains or desert systems. Ideally, ranges of actual water temperatures and stream channel slopes most often associated with specific fish species occurrence would be identified and used to develop species lists. That information is not available for all Wyoming fish species, necessitating a more general, relative approach.

14.2. Species of Greatest Conservation Need (SGCN) Absent Score

Summary:

This metric is a direct measure of the presence/absence of Species of Greatest Conservation Need (SGCN) within a reach. This categorical metric considers whether an SGCN expected to be present is observed in the reach; the metric also considers the SGCN tier of the species.

Species of Greatest Conservation Need are identified in the State Wildlife Action Plan (WGFD 2017) as those species whose conservation status warrants increased management attention and funding, as well as consideration in conservation, land use and development planning in Wyoming. For any project where this metric is used, the practitioner should consult with the regional fisheries biologist at WGFD to determine whether there is natural potential for SGCN to be present at the site. Natural potential considers natural factors, not anthropogenic constraints, that may restrict the distribution of a SGCN. The State Wildlife Action Plan classifies SGCN species into tiers where Tier 1 species have the highest conservation need, and Tier 3 species have less of a conservation need than Tier 1 or Tier 2 species. The number of species with natural potential to occur at the site in each tier is used to calculate the field value for the WSQT. If no SGCN are expected to occur within the project site, this metric would not be calculated.

Reference Curve Development:

The field value for this metric is a function of the number of expected SGCN that are absent from a site and the Tier of that species. Tier 1 species are weighted 3 times higher than Tier 3 species, while Tier 2 species are valued at twice as much as Tier 3 species (Table 14-2). Note that if there are no species in a tier for the site then there are no species absent for that tier.

Table 14-2: How to Determine the Field Value for SGCN Absent Score

SGCN Species *(A)*	Multiplier *(B)*	Equation
# Tier 1 Species Absent	3	$C_1 = A_1 * B_1$
# Tier 2 Species Absent	2	$C_2 = A_2 * B_2$
# Tier 3 Species Absent	1	$C_3 = A_3 * B_3$
Field Value for the WSQT =		$C_1 + C_2 + C_3$

This weighted approach was considered to reflect the relative importance of species within the tiers and to remain consistent with the management goals and approaches for SGCN by the State of Wyoming. From a restoration perspective, restoration of a Tier 1 species would provide the greatest functional lift to the fish community, and should result in the highest index scores. Similarly, loss of Tier 1 species from a site should be considered a significant functional loss.

The reference criteria for this metric are categorical, and each category was assigned a specific index value score based on best professional judgement after consultation with WGFD regional fisheries biologists. No reference curve was developed. If all SGCN, regardless of Tier, are present, the field value would be 0.00, and this equates to an index value of 1.00, i.e., a reference standard condition. The functioning-at-risk range of index values, representing sites that have the potential to support SGCN, includes reaches where the site lacks one Tier 3 species (index value = 0.69), or either one Tier 2 or two Tier 3 species (index value = 0.30). Sites with one Tier 1 species absent, two or more Tier 2 species absent or three or more Tier 3 species absent, were assigned an index value of zero, and were assumed to not support SGCN.

Table 14-3: Threshold Values for SGCN Absent Score

Index Value	Field Value
1.00	0
0.69	1
0.30	2
0.00	≥ 3

Limitations and Data Gaps:

There is uncertainty about the fish species that comprise the natural fish community for most locations and it requires a judgment call by a professional fisheries biologist to establish a species list for a location. The assembled native species lists available in Table C.1 of the WSQT v1.0 User Manual provide a starting point for the potential maximum number of SGCN at a location within a given river basin and stream temperature/gradient class. However, due to variability in sub-basin geology, flow regime and other natural factors, these lists require coordination with the WGFD prior to finalizing an expected number of SGCN species at any given site. In addition, the “cold”, “transitional” and “warm” stream systems are not explicitly defined numerically in terms of slope or temperature. Rather, these distinctions are made in a general sense to broadly help differentiate among Wyoming’s mountain, foothill and high plains or desert systems. Ideally, ranges of actual water temperatures and stream channel slopes most often associated with specific fish species occurrence would be identified and used to develop species lists. That information is not available for all Wyoming fish species, necessitating a more general, relative approach.

14.3. Game Species Biomass

Summary:

This metric is a direct comparison of pre- and post-project biomass changes and is calculated by comparing the biomass before and after a project, after normalizing the data to a nearby control site. The metric is consistent with the approach undertaken by WGFD in their monitoring of fish habitat improvement projects (Binns 1999).

This metric focuses on the productivity of native or non-native game fish species determined to be a management priority by WGFD. Measurements of biomass can be used to infer whether there have been gains in game species productivity at restoration sites where fisheries goals and objectives have been identified. It is not intended to be applied at impact sites or to draw inferences about reductions in biomass due to anthropogenic activities.

This metric measures the increase in game fish biomass following a restoration project relative to the change observed at a control site. Fish are collected consistent with the approach outlined in Bonar et al. (2009). Fish baseline data from a nearby control reach is required to account for natural inter and intra annual variability in fish populations and reduce the influence of climactic or other external factors in determining increases in biomass associated with a restoration project. The control reach should be at a similar elevation and be similar to the project reach in all other aspects. A control reach can be located upstream or downstream from the project reach, or in a separate catchment within the same river basin as the project reach, but not immediately adjacent to the project reach. A control reach that is geographically proximate to the project reach but outside the influence of the project actions is preferred.

Reference Curve Development:

This metric focuses on the increase in fish biomass following a restoration project, and index values and reference curves are associated with the magnitude of change in biomass (pounds/mile) compared with baseline conditions. As such, reference curves were derived following consideration of the magnitude of change that would be considered marginal and significant.

The change in biomass metric was stratified by WGFD stream classes, recognizing that streams with an already productive fishery may be less likely to see large additional increases in productivity following a restoration project. The WGFD assigns a color-coded classification to Wyoming streams based on measured fish biomass (Annear et.al. 2006). This classification identifies blue, red, yellow, and green ribbon streams and is based on pounds of sport fish per mile. Blue ribbon streams are defined as those with greater than or equal to 600 pounds of sport fish per mile and the lowest category is recognized as green ribbon with less than 50 pounds per mile. Updates to stream classification occur infrequently.

The stream classification was identified as a logical basis for stratifying game species biomass because it is judged to approximately reflect productive potential based on multiple population estimates collected over time and at many sites throughout Wyoming. A driving assumption is that streams identified as “blue ribbon” can be considered the most productive and are most likely to be closest to their biologic potential. As such, it would be relatively difficult to increase biomass in a blue ribbon stream. Conversely, the green ribbon streams are the most common class of streams, have the lowest level of productivity and are more likely expected to be below

their biologic potential. The assumption is that it would be relatively easy to improve biomass in a green ribbon stream. Reference curves were developed to reflect these assumptions, and therefore require less biomass improvement in a blue ribbon stream than in a green ribbon stream.

Results compiled by Binns (1999) from a review of trout habitat restoration projects constructed by WGFD between 1953 and 1998 generally support these assumptions, and show that habitat restoration projects in yellow and green ribbon fisheries yielded greater increases in biomass than in red ribbon fisheries (Table 14-4). However, green ribbon streams did not show greater increases in productivity than yellow ribbon streams and Binns (1999) inferred that these systems may be limited by watershed-scale issues that reduce the potential for greater increases in biomass. Based on these results, we did not stratify between yellow and green ribbon streams, but proposed the same reference curve for both.

Population estimates conducted on natural fish communities are known to vary widely between years due to natural variability in fish populations as well as sampling error (Dey and Annear 2001, House 1995). This background variation was considered in developing the sampling methods for this metric (e.g., multiple sampling events and the use of a control site) and in considering what change in biomass would be detectable. Professional judgment and experience with population data in Wyoming streams suggested that at least a 5% change in biomass would have to occur to be detectable through sampling. Blue ribbon streams were thus assigned a minimum index value (0.00) for changes in biomass less than 5%. Given the assumptions above regarding differences in productivity across stream classes, minimum index values were adjusted upwards in 5% increments for each productivity class to account for the greater potential for increases across stream classes.

Thresholds for determining the reference curves were developed using professional judgment, considering the assumptions about productive capacity and population estimate variability. Binns (1999) evaluated success based on post project changes in several biomass metrics. To define success, he relied on criteria proposed by Hunt (1988), including a post-treatment percent change increase in one of the trout population metrics of 25% or more, and a change of 50%, or more for Level 1 and Level 2 success criteria, respectively. While these are arbitrary criteria, they seem reasonable and related to “the long-term annual benefits from management investments of the kind that have been made to remedy perceived deficiencies in trout carrying capacity and/or the sport fishery” (Hunt 1988, p.4).

We determined that a 25% increase in biomass is a measurable increase that could reasonably represent a substantial lift in a blue ribbon stream. In red ribbon streams, we determined that a 50% increase in biomass could reasonably represent a substantial lift. WGFD habitat improvement projects have exceeded this value in red ribbon streams (Table 14-4), with an average increase of 115% pounds/acre in red ribbon systems. Given the assumption that green and yellow ribbon streams should have the capacity to increase biomass the most, a 75% increase in biomass was identified as a realistic, measurable and substantial improvement. This value is reasonable when compared with Binns (1999), who showed increases well above 200% in yellow and green ribbon streams.

Threshold values and reference curves are shown in Table 14-5 and Figure 14-2.

Table 14-4: Mean empirical values for trout biomass averaged over habitat improvement projects sorted for WGFD stream class. Adapted from Binns (1999).

Stream Class	Number of projects with measurements	Reference (lbs./acre)	Treatment (lbs./acre)	Mean % Change
1 (Blue Ribbon)	0	-	-	-
2 (Red Ribbon)	Wild Trout = 3 *Mixed Pop. = 5	64 52	122 106	104 115
3 (Yellow Ribbon)	Wild Trout = 15 *Mixed Pop. = 23	42 43	78 87	316 303
4 (Green Ribbon)	Wild Trout = 7 *Mixed Pop. = 8	28 31	83 85	248 230

***** *The mixed trout category summarizes all projects combined and include both containing only wild trout and those where fish of hatchery origin were present. (Adapted from Binns 1999)*

Table 14-5: Threshold Values for Game Species Biomass

Stream Class	Field Values by corresponding Index Value (i)			
	No Functional Lift	Substantial Functional Lift		
	i = 0.00	i = 0.30	I = 0.70	i = 1.00
Blue Ribbon and non-trout game fish	< 5	5	25	≥ 40
Red Ribbon	< 10	10	50	≥ 80
Yellow/Green Ribbon	< 15	15	75	≥ 119

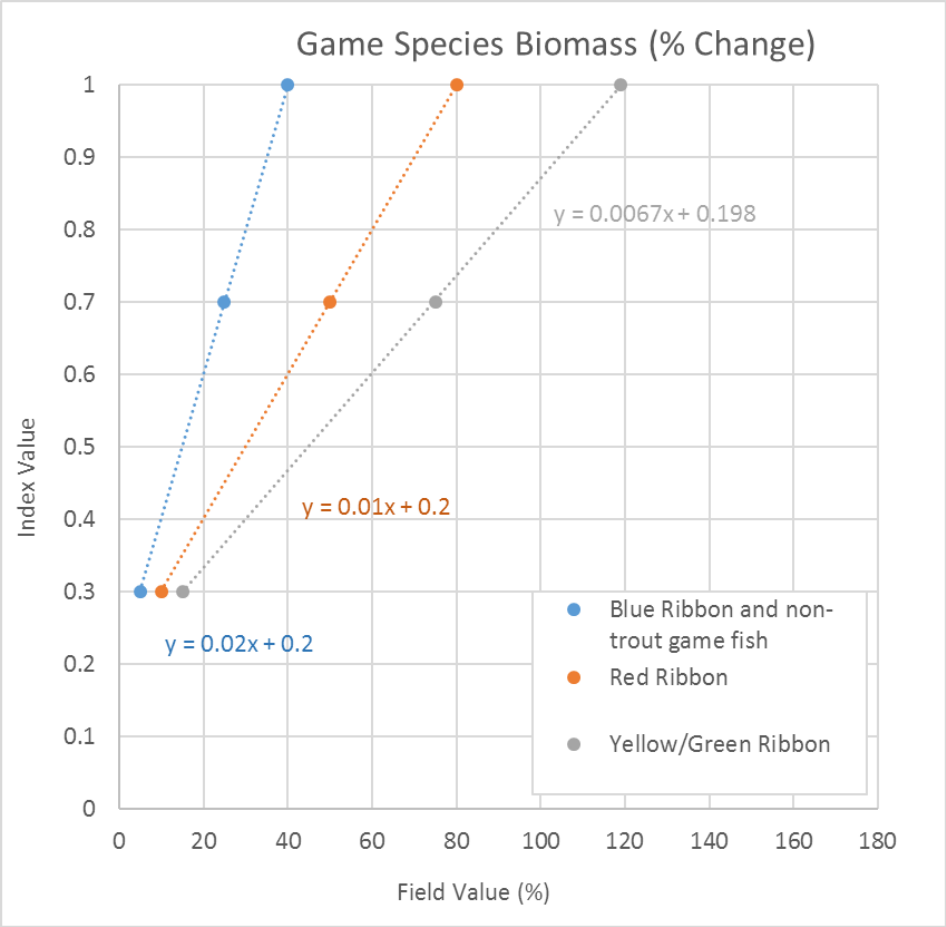

Figure 14-2: Game Species Biomass Reference Curves

<u>Limitations and Data Gaps:</u>

The percentage increases in Table 14-5 forming the basis for the reference curves are based in best professional judgement and supported by previous evaluations in Wyoming. This metric would benefit from additional data analysis and case studies when project information becomes available.

This metric is built on an assumption that restoration work can increase fish biomass permanently, or at least throughout a project monitoring period of 5-10 years. This assumption is not solidly supported in the literature, though examples exist (Pierce et al. 2013). Finally, the approach mathematically ignores error associated with the population biomass estimate. A better, but more complicated approach would include the coefficient of error or other estimate variability measures.

An improvement in non-native game fish biomass could potentially lead to loss or declines in native fish species occurring within a reach. As noted above, this metric is intended to be used where native or non-native game fish species are determined to be a management priority by WGFD. Consultation with regional fish biologists is required before selecting and using this metric in the tool. This consultation should inform metric selection and project design, and reduce the potential for these types of trade-offs between native and non-native species.

Chapter 15. References Cited

Abrahams, A.D., Li, G. and Atkinson, J.F., 1995. Step-pool streams: adjustment to maximum flow resistance. *Water Resources Research* 31, 2593-602.

Al-Hamdan, O.Z., F.B. Pierson, M.A. Nearing, C.J. Williams, J.J. Stone, P.R. Kormos, J. Boll, and M.A. Weltz, 2013. Risk Assessment of Erosion from Concentrated Flow on Rangelands Using Overland Flow Distribution and Shear Stress Partitioning. American Society of Agricultural and Biological Engineers ISSN 2151-0032.

Allan, J.D., 1995. *Stream Ecology*: *Structure and Function of Running Waters.* Chapman and Hall. London, England.

Allan, J.D. and M.M. Castillo, 2007. *Stream Ecology: Structure and Function of Running Waters, Second Edition*. Springer, Dordrecht, The Netherlands.

Annear, T., I. Chisholm, H. Beecher, A. Locke, and 12 other authors, 2004. Instream flows for riverine resource stewardship, revised edition. Instream Flow Council, Cheyenne, WY.

Annear, T., S. Wolff, B.Wiley, R.Keith, K.Johnson, P.Mavrakis, and C.Meyer, 2006. Modification of The Wyoming Game and Fish Department's System For Classifying Stream Fisheries. Wyoming Game and Fish Division Administrative Report. 9 pages.

Bates, D.J. 2000. Comparison of select life history features in wild versus hatchery coastal cutthroat trout and the implications towards species fitness. PhD Dissertation, Department of Biological Sciences, Simon Fraser University, Burnaby, BC, Canada.

Beechie, T., J.S. Richardson, A.M. Gurnell and J. Negishi. 2013. Watershed Processes, Human Impacts, and Process-based Restoration. *Stream and Watershed Restoration: A Guide to Restoring Riverine Processes and Habitats,* First Edition. Editors: Philip Roni and Tim Beechie. John Wiley & Sons, Ltd.

Bernhardt E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Baranas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S.G. Loss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G.M. Kondolf, P.S. Lake, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, B. Powerll, and E. Sudduth, 2005. Synthesizing US River Restoration Efforts. Science, 308:636-637.

Bevenger, G.S. and R.M. King, 1995. A Pebble Count Procedure for Assessing Watershed Cumulative Effects. Research Paper RM-RP-319. US Department of Agriculture Forest Service Research Paper RM-RP-319. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Binns, N.A., 1982. Habitat Quality Index Procedures Manual. Wyoming Game and Fish Department.

Binns, N.A., 1999. A Compendium of Trout Stream Habitat Improvement Projects Done by the Wyoming Game and Fish Department, 1953-1998. Wyoming Game and Fish Department, Fish Division. Cheyenne, WY.

Blackburn-Lynch, W., C.T. Agouridis, and C.D. Barton, 2017. Development of Regional Curves for Hydrologic Landscape Regions (HLR) in the Contiguous United States. Journal of American Water Resources Association (JAWRA), 53(4): 903-928.

Bonar, S.A., W.A. Hubert and D.W. Willis (Editors), 2009. Standard Methods for Sampling North American Freshwater Fishes. Published by the American Fisheries Society.

Booth, D.B. and C.R. Jackson, 1997. Urbanization of Aquatic Systems: Degradation Thresholds, Stormwater Detection, and the Limits of Mitigation. Journal of American Water Resources Association (JAWRA), 33:1077-1090.

Brungs, W.A., and B.R. Jones, 1977. Temperature criteria for freshwater fish: Protocol and procedures. U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, MN. EPA-600/3-77-061.

Buffington, J.M. and D.R. Montgomery, 2013. Geomorphic classification of rivers. In: J. Shroder and E. Wohl (Editors), Treatise on Geomorphology. Academic Press, San Diego, CA. vol. 9, Fluvial Geomorphology, pp. 730-767.

Bull, L.J. and M.J. Kirby (Editors), 2002. *Dryland Rivers*: *Hydrology and Geomorphology of Semi-Arid Channels*. John Wiley and Sons, West Sussex, England.

Bunte, K. and S.R. Abt, 2001. Sampling surface and subsurface particle-size distributions in wadable gravel- and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. US Department of Agriculture Forest Service General Technical Report RMRS-GTR-74. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Bureau of Land Management (BLM), 2017. AIM National Aquatic Monitoring Framework: Field Protocol for Wadeable Lotic Systems. Tech Ref 1735-2. U.S. Department of the Interior, Bureau of Land Management, National Operations Center, Denver, CO.

Carlisle D.M., J. Falcone, D.M. Wolock, M.R. Meador, and R.H. Norris, 2010. Predicting the natural flow regime: models for assessing hydrological alteration in streams. River Research and Applications, 26(2): 118–136.

Cherry, D.S., K.L. Dickson, J. Cairns, Jr. and J.R. Stauffer, 1977. Preferred, avoided and lethal temperatures of fish during rising temperature conditions. Journal of the Fisheries Research Board of Canada 34(2):239-246.

Chin, A., 1989. Step Pools in Stream Channels. Progress in Physical Geography 13:391-407

Cleur, B. and C. Thorne, 2014. A Stream Evolution Model Integrating Habitat and Ecosystem Benefits. River Research and Applications, 30(2):135-154.

Clifford, N.J. and Richards. K.S., 1992. The reversal hypothesis and the maintenance of riffle-pool sequences: are review and field appraisal. In Carling, P.A. and Petts, G.E. (Editors), *Lowland floodplain rivers*. Chichester: Wiley, 43-70.

Copeland, R.R., D.N. McComas, C.R. Thorne, P, J. Soar, M.M. Jonas, and J.B. Fripp, 2001. Hydraulic Design of Stream Restoration Projects. ERDC/CHL TR-01-28. US Army Corps of Engineers, Washington, DC.

Darby, S.E. and C.R. Thornes, 1992. Impact of Channelization on the Mimmshall Brook, Hertfordshire, UK. Regulated Rivers 7:193-204.

Davis, J.C., W.G. Minshall, C.T. Robinson and P. Landres, 2001. Monitoring Wilderness Stream Ecosystems. General Technical Report RMRS-GTR-70. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Dey, P. D. and T.C. Annear, 2001. Inter-annual variation in trout population estimates among six Wyoming Streams and predictions of the PMB and HQI models. Wyoming Game and Fish Department Administrative Report. 11 pages.

Dodds, W.K., J.R. Jones, and E.B. Welch, 1998. Suggested classification of stream trophic state: Distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Wat. Res. 32(5): 1455-1462.

Dodds, W.K., and Val H. Smith, 2016. Nitrogen, Phosphorus, and Eutrophication in Streams. Inland Waters 6: 155-164. Available from: https://www.usgs.gov/news/science-harmful-algae-blooms

Dunne T. and L. Leopold, 1978. *Water in Environmental Planning.* W.H. Freeman and Company, New York, NY.

Egozi, R. and P. Ashmore, 2008. Defining and Measuring Braiding Intensity. Earth Surface Processes and Landforms, 33(14): 2121-2138.

Emmert, B., 2004. Regional Curve Development for Kansas. *In:* Proceeding of the ASAE Conference: Self-Sustaining Solutions for Streams, Wetlands and Watersheds, J. D'Ambrosio (Editor). American Society of Agricultural Engineers, St. Paul, Minnesota, pp. 27-34. 10.13031/2013.17373

Environmental Law Institute (ELI), Stream Mechanics and The Nature Conservancy, 2016. Stream mitigation: Science, policy, and practice. 137pp. https://www.eli.org/sites/default/files/eli-pubs/stream-mitigation-science-policy-and-practice-final-report.pdf.

Fischenich, J.C., 2003. Effects of Riprap on Riverine and Riparian Ecosystems. ERDC/EL TR-03-4. United States Army Corps of Engineers, Washington DC.

Fischenich, J.C., 2006. Functional Objectives for Stream Restoration, EMRRP Technical Notes Collection (ERDC TN-EMRRP-SR-52), US Army Engineer Research and Development Center, Vicksburg, Mississippi.

Fischer, R.A., and J.C. Fischenich, 2000. "Design recommendations for riparian corridors and vegetated buffer strips," EMRRP Technical Notes Collection (ERDC TN-EMRRP-SR-24), U.S. Army Engineer Research and Development Center, Vicksburg, MS. www.wes.army.mil/el/emrrp

Gregory, K.J., A.M. Gurnell, C.T. Hill and S. Tooth, 1994. Stability of the Pool-riffle Sequence in Changing River Channels. Regulated Rivers: Research and Management 9:35-43.

Hack, J.T., 1960. Interpretation of erosional topography in humid temperate regions. American Journal of Science, 258A: 80-97.

Haldar, A. and S. Mahadevan, 2000. *Probability, Reliability, and Statistical Methods in Engineering Design*. John Wiley and Sons, Inc.

Hammer, T.R., 1972. Stream Channel Enlargement Due to Urbanization. *Water Resources Research*, American Geophysical Union. Vol 8, Issue 6. Pp. 1530-1540.

Hargett, E.G., 2011. The Wyoming Stream Integrity Index (WSII) – Multimetric Indices for Assessment of Wadeable Streams and Large Rivers in Wyoming. Wyoming Department of Environmental Quality, Water Quality Division, Cheyenne, WY. Accessed at: http://deq.wyoming.gov/media/attachments/Water%20Quality/Surface%20Water%20Monitoring/Publications/wqd-wpp-monitoring_Wyoming-Stream-Integrity-Index-Multimetric-Indices-for-Assessment-of-Wadeable-Streams-and-Large-Rivers.pdf

Hargett, E.G., 2012. Assessment of Aquatic Biological Condition Using WY RIVPACS with Comparisons to the Wyoming Stream Integrity Index (WSII). Wyoming Department of Environmental Quality, Water Quality Division, Cheyenne, WY. Accessed at: http://deq.wyoming.gov/media/attachments/Water%20Quality/Surface%20Water%20Monitoring/Publications/wqd-wpp-monitoring_Assessment-of-Aquatic-Biological-Condition-Using-WY-RIVPACS-With-Comparisons-to-the-Wyoming-Stream-Integrity-Index.pdf

Harman, W.A., T.B. Barrett, C.J. Jones, A. James, and H.M. Peel, 2017. Application of the Large Woody Debris Index: A Field User Manual Version 1. Stream Mechanics and Ecosystem Planning & Restoration, Raleigh, NC.

Harman, W., R. Starr, M. Carter, K. Tweedy, M. Clemmons, K. Suggs, and C. Miller, 2012. A Function-Based Framework for Stream Assessment and Restoration Projects. US Environmental Protection Agency, Office of Wetlands, Oceans and Watersheds. Washington, D.C. EPA 843-K-12-006.

Harman, W.A. and C.J. Jones, 2017. North Carolina Stream Quantification Tool: Spreadsheet User Manual, NC SQT v3.0. Environmental Defense Fund, Raleigh, NC.

Harrelson, C.C., C.L. Rawlins and J.P. Potyondy, 1994. Stream Channel Reference Sites: An Illustrated Guide to Field Technique. General Technical Report RM-245. US Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Hauer, F.R., B.J. Cook, M.C. Gilbert, E.J. Clairain JR. and R.D. Smith, 2002. A Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Riverine Floodplains in Northern Rocky Mountains. US Army Corps of Engineers, Engineer Research and Development Center. ERDC/EL TR-02-21.

Hauer, F.R. and G.A. Lamberti, 2007. *Methods in Stream Ecology, Second Edition*. Academic Press, Elsevier Inc., Burlington, MA.

Hawkins, C.P., J.R. Olson, and R.A. Hill, 2010. The reference condition: predicting benchmarks for ecological and water-quality assessments. Journal of the North American Benthological Society, 29(1): 312-343.

Henderson, J.E., 1986. Environmental designs for streambank protection projects. *Water Resources Bulletin,* 22(4): 549 – 558.

Hey, R.D., 2006. Fluvial Geomorphological Methodology for Natural Stable Channel Design. Journal of the American Water Resources Association (JAWRA), 42(2):357-374.

House, R., 1995. Temporal variation in abundance of an isolated population of cutthroat trout in western Oregon. North American Journal of Fisheries Management 15:33-41.

Hunt, R. L., 1988. A compendium of 45 trout stream habitat development evaluations in Wisconsin during 1953-1985. Technical Bulletin No. 162, Wisconsin Department of Natural Resources, Madison.

Hupp, C.R., 1992. Riparian Vegetation Recovery Patterns Following Stream Channelization: A Geomorphic Perspective. Ecology 73:1209-1226.

Hussain, Q.A. and A.K. Pandit, 2012. Macroinvertebrate in streams: A review of some ecological factors. International Journal of Fisheries and Aquaculture, 4(7): 114 – 123.

Isaak, D. J., S. J. Wenger, E.E. Peterson, J. M. Ver Hoef, D.E. Nagel, C.H. Luce, S.W. Hostetler, J.B. Dunham, B.B. Roper, S.P. Wollrab, G.L. Chandler, D.L. Horan, and S. Parkes-Payne, 2017. The NorWeST summer stream temperature model and scenarios for the western U.S.: A crowd-sourced database and new geospatial tools foster a user community and predict broad climate warming of rivers and streams. Water Resources Research, 53.

Jones, G.P. and G.M. Walford, 1995. Major riparian vegetation types of eastern Wyoming. A report for the Wyoming Department of Environmental Quality, Water Quality Division. May, 1995. Unpublished report prepared by Wyoming Natural Diversity Database, Laramie, WY.

Jones, G.P., R.S. Smith, W.F. Fertig, D.A. Keinath, M.L. Neighbours, L.A. Welp and G.P. Beauvais, 2001. Rare Species and Riparian Vegetation of the Snake River Basin in Wyoming. Prepared for the U.S. Bureau of Reclamation, by the Wyoming Natural Diversity Database, University of Wyoming. Laramie, WY

Kappesser, G.B., 2002. A Riffle Stability Index to Evaluate Sediment Loading to Streams. Journal of the American Water Resources Association (JAWRA) 38(4): 1069-1081.

Kaufmann, P. R., P. Levine, E. G. Robison, C. Seeliger, and D. V. Peck, 1999. Quantifying physical habitat in wadeable streams. U.S. Environmental Protection Agency, EPA/620/R-99/003, Washington, D.C.

Kaufmann, P.R. and R.M. Hughes, 2006. Geomorphic Anthropogenic Influences on Fish and Amphibians in Pacific Northwest Coastal Streams. American Fisheries Society Symposium, 48:429-455.

Kilgour, B.W., K.M. Somers and D.E. Matthews, 1998. Using the normal range as a criterion for ecological significance in environmental monitoring and assessment. Ecoscience 5:542-550.

Kittel, G., E. VanWie, M. Damm, R. Rondeau, S. Kettler, A. McMullen and J. Sanderson, 1999. A Classification of Riparian Wetland Plant Associations of Colorado: User Guide to the Classification Project. Colorado Natural Heritage Program, Colorado State University, Fort Collins, CO. 80523

Kleindl, W., M. Rains, L. Marshall, F. Hauer, 2015. Fire and flood expand the floodplain shifting habitat mosaic concept. Freshwater Science 34: 1366.

Knighton, D., 1998. *Fluvial Forms and Processes: A New Perspective*. Oxford University Press Inc., New York, New York.

Kroes, D.E. and C.R. Hupp, 2010. The Effect of Channelization on Floodplain Sediment Deposition and Subsidence Along the Pocomoke River, Maryland. Journal of the American Water Resources Association 46(4):686-699.

Langbein, W.B. and L.B. Leopold, 1966. *River Meanders – Theory of Minimum Variance*. U.S. DOI Geological Survey Professional Paper 422-H, Washington DC.

Laub, B.G., D.W. Baker, B.P. Bledsoe, and M.A. Palmer, 2012. Range of variability of channel complexity in urban, restored and forested reference streams. Freshwater Biology, 57: 1076–1095. doi:10.1111/j.1365-2427.2012.02763.x

Leopold, L.B. and M.G. Wolman, 1957. River Channel Patterns – Braided, Meandering and Straight. United States Geological Survey Professional Paper 282A.

Leopold, L.B., M.G. Wolman, and J.P. Miller, 1992. *Fluvial Processes in Geomorphology*. Dover Publications, Inc. New York.

Lowham, H.W., 1988. Streamflows in Wyoming: U.S. Geological Survey Water-Resources Investigations Report 88-4045, 78 p.

Lowham, H.W., T. Wilkinson, J.J. Brown, and A. Strike, 2009. Estimating Streamflow from Concurrent Discharge Measurements. Wyoming Water Development Commission. Lander, Wyoming.

Macfarlane, W.W., J.T. Gilbert, M.L. Jensen, J.D. Gilbert, N. Hough-Snee, P. A. McHugh, J.M. Wheaton, S.N. Bennett, 2017. Riparian vegetation as an indicator of riparian condition: Detecting departures from historic condition across the North American West. Journal of Environmental Management 202(2): 447-460.

Mathon, B.R., D.M. Rizzo, M. Kline, G. Alexander, S. Fiske, R. Langdon, and L. Stevens, 2013. Assessing Linkages in Stream Habitat, Geomorphic Condition, and Biological Integrity Using a Generalized Regression Neural Network. Journal of the American Water Resources Association (JAWRA) 49(2):415-430. DOI: 10.1111/jawr.12030.

Mayer, P.M., S.K. Reynolds, M.D. McCutchen, and T.J. Canfield, 2006. Riparian Buffer Width, Vegetative Cover, and Nitrogen Removal Effectiveness: A Review of Current Science and Regulations. US Environmental Protection Agency EPA/600/R-05/118, Washington, D.C.

McMillan, M., J. Liebens, and C. Metcalf, 2017. Evaluating the BANCS Streambank Erosion Framework on the Northern Gulf of Mexico Coastal Plain. Journal of the American Water Resources Association (JAWRA) 53(6):1393-1408. https://doi.org/10.1111/1752-1688.12572

Merritt, D.M., M.E. Manning, N. Hough-Snee, (Editors), 2017. The National Riparian Core Protocol: A riparian vegetation monitoring protocol for wadeable streams of the conterminous United States. Gen. Tech. Rep. RMRS-GTR-367. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 37 p.

Miselis, D. V., T. A. Wesche and H. W. Lowham, 1999. Development of hydrologic models for estimating streamflow characteristics of Wyoming's mountainous basins. Wyoming Water Resource Center Report, University of Wyoming, Laramie.

Moody, T., M. Wirtanen, and S.N. Yard, 2003. Regional Relationships for Bankfull Stage in Natural Channel of the Arid Southwest. Natural Channel Design, Inc. Flagstaff, AZ.

Mulvihill, C.I. and B.P. Baldigo, 2012. Optimizing Bankfull Discharge and Hydraulic Geometry Relations for Streams in New York State. Journal of the American Water Resources Association (JAWRA) 48(3): 449-463.

Nadeau, T-L., C. Trowbridge, D. Hicks, and R. Coulombe, 2018. A Scientific Rationale in Support of the Stream Function Assessment Method for Oregon (SFAM, Version 1.0).

Oregon Department of State Lands, Salem, OR, EPA 910-S-18-001, U.S. Environmental Protection Agency, Region 10, Seattle, WA

Natural Resources Conservation Service (NRCS), 1986. Urban Hydrology for Small Watersheds Technical Release No. 55. United States Department of Agriculture Natural Resources Conservation Service, Conservation Engineering Division, Washington, D.C. https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044171.pdf

Peterson, C.M., May 2017. Development of thermal tiers and regulatory criteria for Wyoming stream fishes. M.S., Wyoming University Department of Zoology and Physiology.

Pierce, R., C. Podner, and K. Carim, 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 142:68-81.

Poff, N.L., D.A. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D Richter, R.E. Sparks, and J.C. Stromberg, 1997. The Natural Flow Regime: A paradigm for river conservation and restoration. BioScience, 47(11): 769-784.

Poff, N.L., B.D Richter, A.H. Arthington, S.E. Bunn, R.J. Naiman, E. Kendy. M. Acreman, C. Apse, B.P. Bledsoe, M.C. Freeman, J. Henriksen, R.B. Jacobson, J.G. Kennen, D.M. Merritt, J.H. O'Keefe, J.D. Olden, K. Rogers, R.E. Tharme, and A. Warner, 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional flow standards. Freshwater Biology. doi:10.1111/j.1365-2427.2009.02204.x

Poff, N.L. and J.K.H. Zimemrman, 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology, 55(1): 194-205.

Polvi, L.E., E.E. Wohl, and D.M. Merritt, 2011. Geomorphic and Process Domain Controls on Riparian Zones in the Colorado Front Range. Geomorphology, Volume 125(4): 504-516.

Richardson, D.M., P.M. Holmes, K.J. Esler, S.M. Galatowitsch, J.C. Stromberg, S.P. Kirkman, P. Pysek, R.J. Hobbs, 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Diversity Distributions 13: 126-139.

Richter, B.D., 2012. A presumptive standard for environmental flow protection. River Research and Applications, 28(8): 1312-1321.

Ries, K.G., III; Steeves, P.A.; Coles, J.D.; [et al.], 2004. StreamStats: A U.S. Geological Survey web application for stream information. USGS Fact Sheet FS 2004-3115. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 4 p. https://pubs.usgs.gov/fs/2004/3115/pdf/fs200443115.pdf

Roni, P. and T. Beechie (Editors), 2013. *Stream and Watershed Restoration: A Guide to Restoring Riverine Processes and Habitats,* First Edition. John Wiley & Sons, Ltd.

Rosgen, D.L., 1996. *Applied River Morphology.* Wildland Hydrology Books, Fort Collins, Colorado.

Rosgen, D.L., 2006. A Watershed Assessment for River Stability and Sediment Supply (WARSSS). Wildland Hydrology Books, Fort Collins, Colorado.

Rosgen, D.L., 2008. *River Stability Field Guide*. Wildlands Hydrology Books. Fort Collins, Colorado

Rosgen, D.L., 2009. *Watershed Assessment of River Stability and Sediment Supply (WARSSS)*, *Second Edition*. Wildland Hydrology Books, Fort Collins, Colorado.

Rosgen, D. 2014. *River Stability Field Guide, Second Edition*. Wildlands Hydrology Books. Fort Collins, Colorado

Salo, J.A., D.M. Theobald, and T.C. Brown. 2016. Evaluation of Methods for Delineating Riparian Zones in a Semi-Arid Montane Watershed. Journal of the American Water Resources Association (JAWRA) 52(3):632–647. DOI: 10.1111/1752-1688.12414

Schoof, R., 1980. Environmental Impact of Channel Modification. Water Resources Bulletin 16(4): 697-701.

Schueler, T.R., L. Fraley-McNeal, and K. Cappiella, 2009. Is impervious cover still important? Review of recent research. Journal of Hydrologic Engineering 14(4): 309-315.

Schumm, S.A. 1985. Patterns of Alluvial Rivers. Annual Review of Earth and Planetary Sciences, Vol 13, 5-27.

Scott, M.L., G.C. Lines, G.T. Auble. 2000. Channel incision and patterns of cottonwood stress and mortality along the Mojave River, California. Journal of Arid Environments 44: 399-414.

Shafroth, P.B., J.C. Stromberg, D.T. Patten. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecol. Appl. 12, 107e123.

Shields, Jr., F.D., R.E. Lizotte, Jr., S.S. Knight, C.M. Cooper, and D. Wilcox, 2010. The Stream Channel Incision Syndrome and Water Quality. Ecological Engineering 36:78-90.

Simon, A. and M. Rinaldi. 2006. Disturbance, stream incision, and channel evolution: The roles of excess transport capacity and boundary materials in the controlling channel response. Geomorphology 79: 361-383.

Simon, A., N. Pollen-Bankhead, V. Mahacek, and E. Langendoen, 2009. Quantifying Reductions of Mass-Failure Frequency and Sediment Loadings From Streambanks Using Toe Protection and Other Means: Lake Tahoe, United States. Journal of the American Water Resources Association (JAWRA) 45(1):170-186. DOI: 10.1111/j.1752-1688.2008.00268.x

Somerville, D.E. and B.A. Pruitt. 2004. Physical Stream Assessment: A Review of Selected Protocols for Use in the Clean Water Act Section 404 Regulatory Program. Prepared for the U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Wetlands Division (Order No. 3W-0503-NATX). Washington, D.C. 213pp.

Somerville, D.E. 2010. Stream Assessment and Mitigation Protocols: A Review of Commonalities and Differences, May 4, 2010, Prepared for the U.S. Environmental protection Agency, Office of Wetlands, Oceans and Watersheds (Contract No. GS-00F-0032M). Washington, D.C. Document No. EPA 843-S-12-003.

Steele, E.A. and A. Fullerton, 2017. Thermal Networks – Do You Really Mean It? USDA, The Technical Newsletter of the National Stream and Aquatic Ecology Center. November 2017.

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications, 16(4): 1267-1276.

Stoddard, J.L., A.T. Herlihy, D.V. Peck, R.H. Hughes, T.R. Whittier, and E. Tarquinio. 2008. A process for creating multimetric indices for large-scale aquatic surveys. Journal of the North American Benthological Society, 27(4): 878-891.

Sullivan, S.M.P and M.C Watzin. 2009. Stream-floodplain connectivity and fish assemblage diversity in the Champlain Valley, Vermont, U.S.A. Journal of Fish Biology. 74: 1394-1418.

Suplee, M.W., V. Watson, M. Teply, and H. McKee, 2009. How green is too green? Public opinion of what constitutes undesirable algae levels in streams. Journal of the American Water Resources Association (JAWRA) 45(1):123-140.

Sweeney, B.W. and J.D. Newbold, 2014. Streamside Forest Buffer Width Needed to Protect Stream Water Quality, Habitat, and Organisms: A Literature Review. Journal of the American Water Resources Association (JAWRA) 50(3): 560-584.

TDEC, 2017. Tennessee Stream Quantification Tool: Spreadsheet User Manual, TN SQT v1.0. Tennessee Department of Environment and Conservation, Nashville, TN.

USACE, 1987. Corps of Engineers Wetland Delineation Manual. Technical Report Y-87-1. U.S. Army Corps of Engineers Waterways Experiment Station, Environmental Laboratory, Vicksburg, MS.

USACE, 2008. Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Arid West Region (Version 2.0) ERDC/EL TR-08-28. U.S. Army Engineer Research and Development Center, Vicksburg, MS.

USACE, 2010a. Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Great Plains Region (Version 2.0), ed. J. S. Wakeley, R. W. Lichvar, and C. V. Noble. ERDC/EL TR-10-1. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

USACE, 2010b. Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Western Mountains, Valleys, and Coast Region (Version 2.0), ed. J. S. Wakeley, R. W. Lichvar, and C. V. Noble. ERDC/EL TR-10-3. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

USACE, 2018a. Wyoming Stream Quantification Tool (WSQT) User Manual and Spreadsheet Version 1.0. USACE Omaha District, Wyoming Regulatory Office. Cheyenne, WY.

USACE, 2018b. *Georgia Interim Stream Quantification Tool and User Manual.* In 2018 Standard Operating Procedures for Comepnsatory Mitigation. USACE Savannah District, Georgia Regulatory Branch. Burford, GA.

USBR and ERDC, 2017. *National Large Woody Manual: Assessment, Planning, Design, and Maintenance of Large Wood in Fluvial Ecosystems: Restoring Process, Function, and Structure.* U.S. Bureau of Reclamation, Boise, ID and U.S. Army Engineer Research and Development Center, Vicksburg, MS.

USDA Forest Service, 2007. Analyzing pebble count data collected by size classes [Microsoft Excel spreadsheet]. USDA Forest Service National Stream & Aquatic Ecology Center. Available from: https://www.fs.fed.us/biology/nsaec/index.html

USDOI, 2011. Riparian area management: Multiple indicator monitoring (MIM) of stream channels and streamside vegetation. Technical Reference 1737-23. BLM/OC/ST-10/003+1737+REV. Bureau of Land Management, National Operations Center, Denver, CO. 155 pp.

USEPA, 2005. Protecting Water Quality from Agricultural Runoff. EPA Nonpoint Source Control Branch, EPA 841-F-05-001. Washington DC. Available from: https://nepis.epa.gov/Exe/ZyPDF.cgi/P10039OH.PDF?Dockey=P10039OH.PDF

USEPA, 2009. National River and Streams Assessment Field Operations Manual. U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Office of Research and Development. Washington, D.C. EPA-841-B-07-009

USEPA, 2014. Best Practices for Continuous Monitoring of Temperature and Flow in Wadeable Streams. Global Change Research Program, National Center for Environmental Assessment, Washington, D.C; EPA/600/R-13/170F.

USEPA, 2016. National River and Streams Assessment 2008-2009 Technical Report. U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Office of Research and Development. Washington, D.C.

USFWS, 2009. A System for Mapping Riparian Areas in the Western United States. Division of Habitat and Resource Conservation, Branch of Resource and Mapping Support, Arlington, VA.

Walford, G.M., 1996. Statewide classification of riparian and wetland dominance types and plant communities - Bighorn Basin segment. A report submitted to the Wyoming Department of Environmental Quality, Water Quality Division. Cooperative Agreement #WET04, Grant #CD998066-01-0. Wyoming Natural Diversity Database, Laramie WY. 185 pp.

Walford, G., G. Jones, W. Fertig, S. Mellman-Brown, and K.E. Houston, 2001. Riparian and wetland plant community types of the Shoshone National Forest. USDA Forest Service General Technical Report RMRS-GTR-85. Rocky Mountain Research Station, Odgen UT. 122 pp.

WDEQ, 2018. Manual of Standard Operating Procedures for Sample Collection and Analysis. Wyoming Department of Environmental Quality, Water Quality Division, Watershed Program, Cheyenne, WY. Available from: http://deq.wyoming.gov/wqd/qaqc/resources/manual/

Welch, E.B., J.M. Jacoby, R.R. Horner, and M.R. Seeley, 1988. Nuisance biomass levels of periphytic algae in streams. Hydrobiologia 157: 161-168.

West, E. and G. Ruark, 2004. Historical evidence of riparian forests in the Great Plains and how that knowledge can aid with restoration and management. Journal of Soil and Water Conservation 59(5): 105A-110A.

Whittaker, J.G., 1987. Sediment Transport in Step-pool Streams. In: Sediment Transport in Gravel-Bed Rivers, C.R. Thorne, J.C. Bathurst, and R.D. Hey (Editors). John Wiley, New York, New York, pp. 545-579

Winter, T.C., W.H. Judson, O.L. Franke, and W.M. Alley, 1998. Groundwater and Surface Water A Single Resource. Circular 1139, US Geological Survey, Denver, CO.

Winward, A.H., 2000. Monitoring the vegetation resources in riparian areas. Gen. Tech. Rep. RMRSGTR-47. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 p.

Wohl, E.E., 2000. Mountain Rivers. Water Resources Monographs 14, American Geophysical Union, Washington, D.C.

Wohl, E., 2004. *Disconnected Rivers: Linking Rivers to Landscapes*. Yale University Press. New Haven and London.

Wohl, E.E., D.A. Cenderelli, K.A. Dwire, S.E. Ryan-Burkett, M.K. Young, and K.D. Fausch, 2010. Large in-stream wood studies: a call for common metrics. Earth Surface Processes and Landforms, 35: 618-625.

Wohl, E. and N. Beckman, 2014, Controls on the longitudinal distribution of channel-spanning logjams in the Colorado Front Range, USA. River Research and Applications, 30(1): 112-131.

Wohl, E., 2011. Threshold-induced complex behavior of wood in mountain streams. Geology, 39(6): 587-590.

Wolman, M.G., 1954. A Method of Sampling Coarse River-bed Material. Transactions of the American Geophysical Union 35(6):951-956.

Wood, P.J., and P.D Armitage, 1997. Biological Effects of Fine Sediments in Lotic Environments. Environmental Management Vol. 21(2).

WGFD, 2017. State Wildlife Action Plan. Wyoming Game and Fish Department, Habitat Program. Cheyenne, WY.

Youngblood, A. P.; Padgett, W. G.; Winward, A. H., 1985. Riparian community type classification of eastern Idaho-western Wyoming. R4-Ecol-85-01. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 78 p.

Zorn, T.G., P.W. Seelbach, E.S. Rutherford, T.C. Wills, S.T. Cheng, and M.J. Wiley, 2008. A Regional-scale Habitat Suitability Model to Assess the Effects of Flow Reduction on Fish Assemblages in Michigan Streams. Michigan Department of Natural Resources, Fisheries Division Research Report 2089.

Appendix A: WSQT List of Metrics

Functional Category	Function-Based Parameters	Metrics/Units	Stratification		Threshold Index Values				Literature and data sources used to develop Reference Curves	Applicability
			Type	Description	i= 0.00	i= 0.30	i= 0.70	i= 1.00		
Reach Hydrology and Hydraulics	Reach Runoff ●	Land Use Coefficient ●	-		≥ 80	*	68	≤ 40	Literature values from NRCS, 1986.	Applicable in all streams. Additional testing is needed in all stream types.
		Concentrated Flow Points ●	-		*	*	1.0	0.0	Developed by WSTT and Stream Mechanics	
	Flow Alteration	Baseflow Alteration (O/E)	-		≤ 0.26 ≥ 2.00	*	*	0.90 - 1.10	Literature values from Binns (1982) and Richter (2012)	Applicable in all streams with baseflow extending through August.
	Floodplain Connectivity ●	Bank Height Ratio (ft/ft) ●	-		*	1.5	1.2	1.0	Literature values from Rosgen (2008), Harman et al. (2012) and data from the Combined Geomorphic Reference Dataset.	Applicable in all streams. Additional testing is needed in intermittent/ ephephemeral streams and braided or anastomosing systems.
		Entrenchment Ratio (ft/ft) ●	Reference Stream Type	C, Cb or E	*	2.0	2.4	≥ 5.0		Not applicable in naturally occuring canyon systems (e.g., F type streams) or braided (D) stream types.
				A, B, Ba or Bc	*	1.2	1.4	≥ 2.2		
Geomorphology	Large Woody Debris	LWD Index (Dimensionless)	-		0	*	430	≥ 660	Data collected by WGFD, WDEQ and WSTT.	Applicable to all streams with naturally forested catchment or riparian gallery forests. Additional data is needed to determine stratification.
		# Pieces	-		0	*	13	≥ 28	NRSA dataset (USEPA 2016) and data collected by WGFD and WSTT.	
	Lateral Migration ●	Greenline Stability Rating	-		< 2	5	7	≥ 9	Literature values from Winward (2000).	Applicable in all streams statewide with slopes less than 4%
		Dominant BEHI/NBS	-		H/VH, H/Ex, VH/VH, VH/Ex, Ex/M, Ex/H, Ex/VH, Ex/Ex	M/H, Ex/L, Ex/VL	-	L/VL, L/L, L/M, L/H, L/VH, M/VL	Literature values from Rosgen (2014) and Harman et al. (2012).	Applicable to single-thread channels. For systems naturally in disequilibrium, like some braided streams, ephemeral channels, alluvial fans or other systems with naturally high rates of bank erosion, this metric should not be assessed.
		Percent Streambank Erosion (%)	-		≥ 75	*	10	≤ 5	Literature values from Binns (1982).	
		Percent Armoring (%)	-		≥ 30	*	*	0	Developed by Stream Mechanics.	Applicable only when armoring techniques are present or proposed in the project reach. Additional testing is needed in all stream types.
	Bed Material Characterization	Size Class Pebble Count Analyzer (p-value)	-		≤ 0.01	0.05	*	> 0.10	Developed using approach presented in Bevenger and King (1995).	Applicable in gravel and cobble bed streams only. Requires comparison with reference reach.
	Bed Form Diversity ●	Pool Spacing Ratio ●	Reference Stream Type	C	≥ 9.3 ≤ 3.0	*	7.0 3.7	4.0 - 6.0	Data from the Compiled Geomorphic Reference Dataset.	Not applicable in natural bedrock systems, naturally occuring canyon systems (e.g., F type streams), ephemeral streams or braided (D) stream types. Reference curves developed using data from single-thread, perennial, gravel/cobble streams in the mountainous regions of Wyoming. Additional data would be useful to determine whether additional or modified reference curves are needed for basins and plains regions.
				Cb	*	*	3.0 6.0	3.7 - 5.0		
				B & Ba	≥ 7.5	*	4.0	≤ 3.0		
				Bc	*	*	6.0	≤ 3.4		
				E	≤ 1.8 ≥ 8.3	*	3.0 6.0	3.5 - 5.0		
		Pool Depth Ratio ●	-		1.0	*	2.2	≥ 3.2		Reference curves developed using data from single-thread, perennial, gravel/cobble streams in the mountainous regions of Wyoming. Additional data and testing would be useful to determine whether additional or modified reference curves are needed for basins and plains regions, in intermittent, ephemeral and braided systems or sand bed streams.
		Percent Riffle (%) ●	Slope	S < 3%	*	*	39 69	50 - 60		
				S ≥ 3%	*	*	60 83	68 - 78		
			Bioregion	Volcanic Mountains & Valleys	*	*	61 82	73 - 80		
		Aggradation Ratio	-		≥ 1.6	1.4	1.2	1.0	Literature values from Rosgen (2014) and modified using regional data from NRSA and Combined Geomorphic Dataset.	Not applicable to braided (D) stream types.
	Plan Form	Sinuosity	Valley Type	Unconfined Alluvial	*	1.15 1.60	1.19 1.51	1.20 - 1.50	Literature values from Rosgen (2014) and modified using regional data from NRSA and Combined Geomorphic Dataset.	Only applicable in single-thread systems; reference curves have not been developed for natural F and G channels.
				Confined Alluvial	1.00	*	1.19	≥ 1.20		
				Colluvial/V-Shaped	*	1.00 1.40	1.09 1.31	1.10 - 1.30		
			Reference Stream Type	E	*	1.20 2.00	1.29 1.81	1.30 - 1.80		
	Riparian Vegetation ●	Riparian Width Ratio ●	Valley Type	Unconfined Alluvial	30	*	*	100	Developed by WSTT.	Applicable to all streams. Additional testing is needed in all stream types.
				Confined Alluvial or Colluvial	60	*	*	100		
		Woody Vegetation Cover (%) ●	Ecoregion	Mountains & Basins	0	*	69	≥ 122	Data from the CNHP dataset (Kittel et al. 1999) and WSTT data.	
				Plains	0 *	* 101	* *	59-69		
		Herbaceous Vegetation Cover (%) ●	Cover Type	Herbaceous	30	*	94	≥ 117		
				Forested, Scrub-Shrub	0	*	54	> 77		
		Percent Native Cover (%) ●	-		*	65	91	100		
Physicochemical	Temperature	MWAT 7-day Average (°C)	Steam Temperature	Tier 1 (Cold)	*	18.1	15.4	*	Literature values from Peterson (2017).	Applicable to all streams statewide, however NorWeST temperature models are not available within the Little Missouri, Niobrara, Lower North Platte and South Platte Basins - sufficient monitoring data are needed in these areas to determine the thermal tier.
				Tier 2 (Cold-Cool)	*	19.3	15.5	*		
				Tier 3 (Cool)	*	22.0	17.7	*		
				Tier 4 (Cool-Warm)	*	26.0	19.9	*		
				Tier 5 (Warm)	*	29.0	27.3	*		
	Nutrients	Chlorophyll (mg/m2)	Ecoregion	Mountains	*	53	27	< 12	Developed using data and guidance from WDEQ-Water Quality Division.	Applicable to all streams statewide where epilithic samples can be collected.
				Plains or Basin	≥ 150	117	29	< 16		
Biology	Macroinvertebrates	WSII	Bioregion	Volcanic Mountains & Valleys	< 24.9	46.2	69.3	> 88.1	Literature values from Hargett (2011).	Not applicable to intermittent or ephemeral streams or low-gradient lentic systems; data must be collected using targeted riffle/run quantitative methods. The following streams will be diffcult to assess with this method due to inadequate reference site representation: streams with very small watersheds < 12 km2 (< 5 mi2), streams located at high montane elevations > 2,740 m (> 9,000 ft.), streams within the Bighorn Basin of north-central Wyoming (excluding the foothills), non-montane spring-fed stream segments within the interior of the Wyoming Basin and mixed origin streams of extreme southwest Wyoming.
				Granitic Mountains	< 32.6	40.2	60.3	> 74.9		
				Sedimentary Mountains	< 16.6	34.8	52.3	> 70.8		
				Southern Rockies	< 5.1	32.6	48.8	> 82.2		
				Southern Foothills & Laramie Range	< 30.7	44.5	66.7	> 85.3		
				Bighorn Basin Foothills	< 3.9	40.6	60.9	> 80.8		
				Black Hills	< 12.8	30.7	46.1	> 65.7		
				High Valleys	< 17.1	32.5	48.8	> 78.2		
				SE Plains	< 10.4	36.7	55.1	> 87.0		
				NE Plains	< 1.6	38.9	58.4	> 95.8		
				Wyoming Basin	< 5.3	26.2	39.9	> 64.5		
		RIVPACS	Bioregion	Volcanic Mountains & Valleys	≤ 0.21	0.65	0.86	> 1.21	Literature values from Hargett (2012).	
				Granitic Mountains	< 0.59	0.65	0.88	> 1.09		
				Sedimentary Mountains	< 0.42	0.68	0.82	> 1.17		
				Southern Rockies	< 0.27	0.62	0.89	> 1.18		
				Southern Foothills & Laramie Range	< 0.29	0.68	0.88	> 1.20		
				Bighorn Basin Foothills	< 0.41	0.63	0.84	> 0.92		
				Black Hills	< 0.37	0.59	0.88	> 1.08		
				High Valleys	< 0.42	0.68	0.86	> 1.14		
				SE Plains	< 0.34	0.51	0.78	> 1.12		
				NE Plains	< 0.11	0.52	0.75	> 0.98		
				Wyoming Basin	< 0.15	0.52	0.75	> 1.18		
	Fish	Native Fish Species (O/E)	-		≤ 58	75	99	100	Developed by Wyoming Game and Fish and WSTT.	Applicable in all streams statewide, except naturally fishless streams.
		SGCN Absent Score	-		≥ 3	2	1	0		
		Game Species Biomass (% Increase)	Stream Productivity Rating	Blue Ribbon and non-trout game fish	< 5	5	25	≥ 40	Developed by Wyoming Game and Fish and WSTT, and evaluated using data from Binns (1999)	
				Red Ribbon	< 10	10	50	≥ 80		
				Yellow/Green Ribbon	< 15	15	75	≥ 119		

* Threshold Index Values were not assigned to generate the reference curve

● Basic Assessment required elements per WSMP v2