Scientific Support for the Colorado Stream Quantification Tool

CSQT Steering Committee

Army Corps of Engineers
Omaha District

StreamMechanics

Ecosystem Planning & Restoration

Scientific Support for the Colorado Stream Quantification Tool (Beta Version)

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Table of Contents

List of Figures

List of Tables

Acronyms

BEHI/NBS – Bank Erosion Hazard Index / Near Bank Stress
BHR – Bank Height Ratio
CDPHE – Colorado Department of Public Health and Environment
CFR – Code of Federal Register
COMPS – Colorado Mitigation Procedures for Streams
CS – Cold Stream
CSQT – Colorado Stream Quantification Tool
CSQT SC – Colorado Stream Quantification Tool Steering Committee
Corps – United States Army Corps of Engineers (also, USACE)
CPW – Colorado Parks and Wildlife
CWA 404 – Section 404 of the Clean Water Act
DM – Daily Maximum Temperature
DO – Dissolved Oxygen
ER – Entrenchment Ratio
FF – Functional Feet
GSR – Greenline Stability Rating
HSI – Habitat Suitability Indices
LDA – Lateral Drainage Area
LWD – Large woody debris
LWDI – Large Woody Debris Index
MMI – Multi-metric Index
MWAT – Maximum Weekly Average Temperature
NRCS – Natural Resource Conservation Service
NRSA - National Rivers and Streams Assessment (dataset)
O/E – Ratio of Observed/Expected
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
WDR – Width Depth Ratio
WDRS – Width Depth Ratio State
WQCD – Water Quality Control Division
WQCV – Water Quality Capture Volume
WS – Warm Stream
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.

Colorado Stream Quantification Tool and Debit Calculator (CSQT) – The CSQT is a spreadsheet-based calculator that scores stream condition before and after restoration or impact activities to determine functional lift or loss, respectively, and can also be used to determine restoration potential, develop monitoring criteria and assist in other aspects of project planning. The CSQT is comprised of two workbooks, the CSQT and Debit Calculator workbooks, that have some overlapping components; and both are based on principles and concepts of the SFPF. References to CSQT describe concepts that are applicable within both CSQT and Debit Calculator workbooks.

Colorado Stream Quantification Tool Steering Committee (CSQT SC) – The group who worked on the development of the CSQT and contributed to various aspects of this document.

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 CSQT 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 a process (expressed as a rate) that describes and supports the functional statement of each functional category.

Index Value – Dimensionless value between 0.00 and 1.00 that expresses 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 evaluate 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).

Native Flow – For the purposes of the CSQT, native flows are the estimates of the stream flows that would result from natural hydrologic processes such as rainfall-runoff and snowmelt-runoff without any anthropogenic influence at a given location.

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 forms that 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 CSQT, 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 the Stream Quantification Tool that has been adapted and modified for use in Colorado (USACE 2018b).

Wyoming Stream Technical Team (WSTT) – The group who worked on the development of the WSQT v1.0 and associated documents.

Chapter 1. Background and Introduction

The purposes of this document are to provide the scientific underpinnings of the Colorado Stream Quantification Tool and Debit Calculator (CSQT) and the rationale for the conversion of measured stream condition into dimensionless index scores. The CSQT 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 CSQT is one of several Stream Quantification Tools (SQTs) that have recently been developed for use in specific states, including Wyoming (WSQT; USACE 2018b), North Carolina (Harman and Jones 2017), Tennessee (TDEC 2018) and Georgia (USACE 2018c).

This document is based on the Scientific Support for the WSQT v1.0 (USACE 2018a) and has been edited for Colorado with input from the Colorado Stream Quantification Tool Steering Committee (CSQT SC). The WSQT v1.0 served as the basis for the CSQT Beta Version and many Chapters in this document are reproduced with minor edits from the Scientific Support for the WSQT v1.0 (USACE 2018a). The following modules, parameters and metrics have been added to the CSQT and are discussed in this document but were not included in the WSQT v1.0: flow alteration module; baseflow dynamics and dissolved oxygen parameters; and impervious cover, water quality capture volume, side channels and daily maximum temperature metrics. The following were included in the WSQT v1.0 but have been revised for the CSQT: the maximum weekly average temperature metric, and the macroinvertebrates and fish parameters. Definitions of function-based parameters, metrics, and reference standards are provided below in Section 1.1.

This document expands on the concepts presented in the SFPF and the Colorado Stream Quantification Tool and Debit Calculator (CSQT) Beta Version User Manual (User Manual; CSQT SC 2019) to provide the scientific and technical rationale behind selection of the reference curves and metrics. Information on how to use the CSQT or collect data for use in the CSQT is not included in this document but can be found in the User Manual.

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

Section 1.2 provides a summary of the catchment 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 CSQT 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 included in the CSQT.

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 information on the process for revising reference curves and metrics.

Section 1.9 provides key considerations in applying the CSQT.

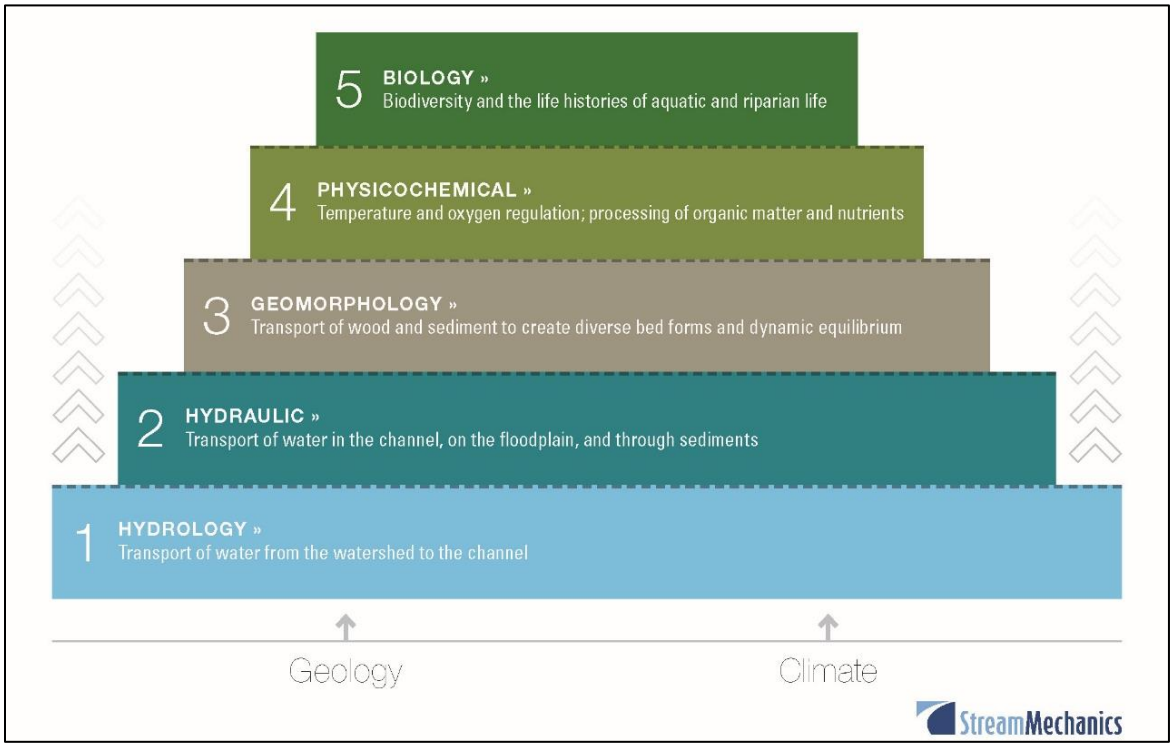
After the Introduction and Background, the remainder of the document is organized by function-based parameter. Each parameter description includes a summary of why it was included, reasons for selecting the metrics, and in some cases, why other metrics were not selected. Then, a description of metrics used to quantify the parameter is provided. Each metric section provides the rationale for developing reference curves and any stratifications, 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 could be used as a basis for management and policy decisions (Fischenich 2006). To address this need, an international committee of scientists, engineers, and practitioners defined 15 key stream and riparian zone functions aggregated into five 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.

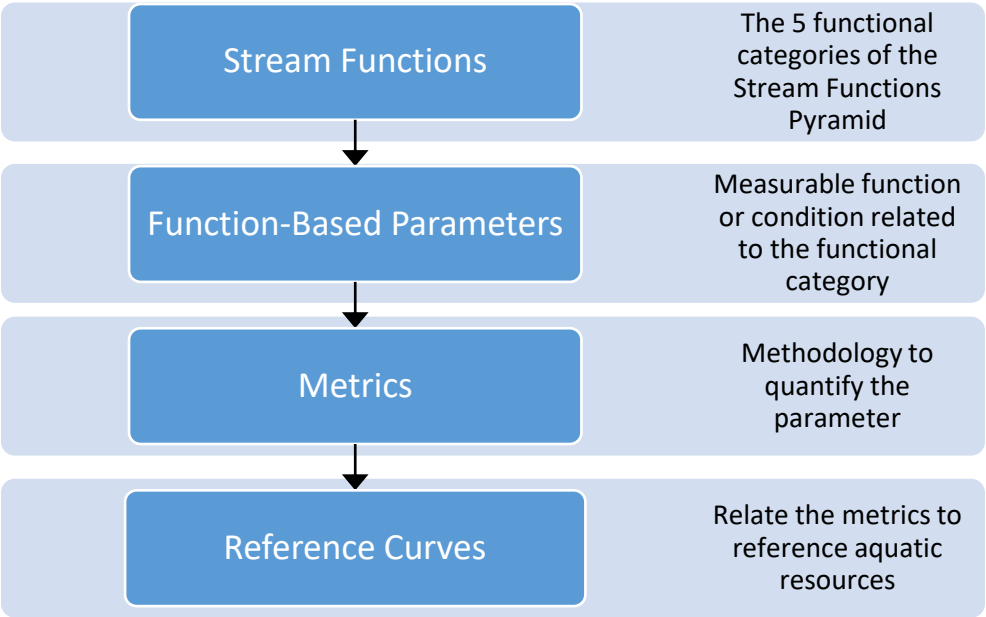
The SFPF organizes stream and riparian functions into five functional categories: Hydrology, Hydraulics, Geomorphology, Physicochemical, and Biology (Figure 1-1). This organization recognizes that foundational functions, like watershed hydrology and sediment transport processes, generally support higher-level 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 CSQT 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 CSQT application.



The SFPF has informed the development of the CSQT, a tool that consolidates the components of the SFPF into an excel workbook to characterize stream ecosystem functions at a specific project reach. The CSQT 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 SQT development and regionalization process, which are relevant to the stream systems found within the state of Colorado. The Tennessee and North Carolina SQTs calculate an overall reach score from the 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 Colorado and Wyoming SQTs merge 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 Colorado and Wyoming SQTs 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 during each state's regionalization process.

All the metrics selected for the CSQT are measurements of point-in-time condition that serve as indicators of structural or compositional attributes related to the function-based parameters selected for a given functional category. That is, the function-based parameters and metrics serve as surrogates for stream processes within the same functional category. For example, bedform diversity is a conditional parameter that is a partial surrogate for sediment transport processes, which is a geomorphology function. Bedform diversity is NOT a surrogate for macroinvertebrate functions because it is in a different functional category (biology). Assessment data are input into the CSQT, where data for each metric are translated into index values, 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. Catchment Context

Understanding the catchment processes that contribute to 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. Indirect pathways are often alterations to catchment-scale processes, like land use changes, that occur away from the stream or are distributed throughout a catchment (Roni and Beechie 2013).

The User Manual includes a catchment assessment and a process to determine the restoration potential at a project site. The catchment assessment was modified from the WSQT v1.0 for Colorado with input from the CSQT SC to consider anthropogenic modifications common in Colorado. The catchment assessment is a qualitative approach intended to identify catchment-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 catchment, 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 catchment context is critical to understanding the limitations to restoration potential at a site, the focus of this tool is on the change to reach-scale ecological variables following a project. The CSQT focuses primarily on indicators of condition tied to direct pathways of anthropogenic modification using a reach-based approach. Because catchment condition is not likely to change following reach-scale activities, it differs from the metrics that are included in the tool; therefore, the CSQT SC decided to not include it as part of the scoring within the tool itself. The catchment assessment is only used to assist the practitioner in assessing limiting factors affecting a project site and determining the restoration potential of the project reaches. Information on the catchment assessment and determining restoration potential are included in the User Manual.

1.3. Development of Reference Curves

The metrics within the Colorado and Wyoming SQTs 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 in functional capacity are referred to in the CSQT as functional loss and lift and can be used to inform debits and credits as defined 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 not-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, biotype, reference community type, or valley type. Stratification varies by metric (See Appendix A).

The reference curves in the Colorado and Wyoming SQTs are derived from identification of specific benchmarks (thresholds) that represent the degree to which the measured condition departs from a reference standard 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 and 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.

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

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 aquatic ecosystem structure and function at a 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 pristine 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 aquatic ecosystem structure and function. 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 aquatic ecosystem structure and function. A value of 0.29 represents a condition that is severely altered or impaired relative to reference standard, 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 Colorado'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 (e.g., BLM 2017; Nadeau et al. 2018).
3. Where existing data or literature was limited, best professional judgement was used to identify threshold values. In some instances, the decision was made to not identify thresholds between all categories and instead extrapolate index values from a best fit line from available data or literature values.

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 specific metric section below. Reference curves and threshold values were determined for each metric individually. Therefore, a reach may achieve a reference standard index value for one metric, e.g., large woody debris index, and not others. Metric index values are then combined to provide a reach score (Section 1.4).

For the Colorado and Wyoming SQTs, 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, 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 the use of existing datasets, this document relied on the definitions of reference standard condition provided by the authors.

The functioning-at-risk range of index values (0.30-0.69) reflects a condition that can support aquatic ecosystem structure and function, but not within the range of reference standard condition. 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 reference standard condition.

The not-functioning range of index values (0.00-0.29) reflects a degraded or impaired condition that does not support aquatic ecosystem structure and function. 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 these were reviewed to determine whether field values would reasonably represent no functional support for that metric.

1.4. Calculating Reach-Scale Condition

The architecture and scoring of the CSQT 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. The User Manual identifies a subset of parameters and metrics that should be evaluated for all projects. These include: reach runoff, floodplain connectivity, lateral migration, bedform diversity, and riparian vegetation. The tool includes additional parameters that users may add to further characterize reach condition and to better align with a project's function-based goals and objectives. Guidance on parameter selection is provided in the User Manual. For example, a practitioner may choose not to monitor (or receive credit for) physiochemical and biological parameters, and the CSQT 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 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 assessed at a given project reach are used to calculate the overall score (refer to the User Manual for guidance on parameter and metric selection). Metrics not assessed are 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 in Section 1.1, the CSQT Beta Version combines the hydrology and hydraulic categories from the stream functions pyramid into one category called Reach Hydrology and Hydraulics (Reach H&H). The Reach H&H category is weighted to provide 30% of the overall score; geomorphology also provides 30%; and physicochemical and biology each provide 20% of the overall score. The Reach 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 catchment-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 maximum overall score achievable by monitoring only Reach H&H and geomorphology parameters is 0.60. Monitoring in all four functional categories would result in a maximum overall score of 1.00. The weighting incentivizes restoration practitioners to attempt to improve and monitor physicochemical and biology parameters, even if they may not reach full restoration potential.

The CSQT estimates the change in condition at an impact or mitigation site by calculating the difference between existing (pre-project) and proposed (post-project) condition. Existing and proposed condition scores are multiplied by stream length to calculate functional feet. 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 are no longer equivalent to stream length. The difference between proposed and existing functional feet values, referred to as the change in functional feet (ΔFF), is the value that is used to identify the amount of functional lift or loss within a project reach.

The CSQT includes a flow alteration module whose structure and scoring are discussed in more detail in the User Manual. The flow alteration module generates functional feet that are added to the functional feet for the project reach. Parameters and metrics that quantify flow alteration were not included in the Reach H&H functional category because flow alteration has the potential to affect longer stream reaches than a project reach evaluated within the Quantification Tool worksheet. The flow alteration module scoring is similar to the Quantification tool: index values are generated for each metric, and then averaged to provide an overall module score. Only the metrics selected and assessed are used to calculate the overall score. The module score is weighted, multiplied by the affected stream length and the change in functional feet is then added to the change in functional feet (ΔFF) calculated for the project reach in the Quantification Tool spreadsheet. Since the flow alteration module only assesses hydrology, the module score is weighted by 20%. This is consistent with other SQTs that weight the hydrology functional category at 20%.

1.5. Calculating Functional Feet

The change in functional feet pre- and post-project (ΔFF) is intended to serve as the basis for calculating debits and credits. The functional feet calculation made in the CSQT and Debit Calculator workbooks 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 changes to channel geometry (ELI et al. 2016). The purpose of the CSQT is to generate 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 difference between the proposed and existing condition score, but a relatively 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 functional feet 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, Page 15). This product of quality and length is a common currency for debit and credit calculations (ELI et al. 2016).

The CSQT SC 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 measures 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. Therefore, 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, this approach was not pursued.

Valley area: Another approach that was considered was using valley area (valley length times valley width) 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 would align with current impact accounting practices. Additional discussions and research on implementation are needed before adopting a valley-based approach across all projects. Therefore, this approach may be considered for future versions of the CSQT.

The unit of measure in the CSQT Beta Version 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 CSQT 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.

Table 1-2: Debit and credit approaches that consider a combination of condition and stream length. 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.
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.

1.6. Function-Based Parameters in the CSQT Beta Version

The CSQT 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 CSQT Beta Version incorporates many of the functions and parameters outlined in Fischenich (2006) and Harman et al. (2012). The 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 CSQT 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 CSQT Beta Version 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 CSQT Beta Version is listed in Appendix A. Rationale for including a parameter and its metrics in the tool, 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 catchment scale may affect both existing and proposed condition scores and are considered in the catchment assessment and determination of restoration potential (see Section 1.2 and the CSQT 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 this range of conditions to a reference standard.
- Flexibility in the 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 Colorado. Colorado is a high elevation, headwaters state characterized by a wide range of precipitation (4-80”). Colorado contains variable soils and parent materials; significant 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 CSQT.

Functional Category	Parameter from the Function-Based Framework	Included in CSQT (Yes/No)	Rationale
Hydrology	Channel Forming Discharge	Yes	See Chapters 2 and 3.
	Precipitation/Runoff Relationship		
	Flood Frequency		
	Flow Duration		
Hydraulics	Flow Dynamics	Yes	Baseflow Dynamics, See Chapter 4.
	Groundwater/Surface Water Exchange	No	Difficult to assess and develop reference curves.
	Floodplain Connectivity	Yes	See Chapter 5.
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 6.
	Bank Migration/Lateral Stability	Yes	See Chapter 7.
	Bed Material Characterization	Yes	See Chapter 8.
	Bed Form Diversity	Yes	See Chapter 9.
	Plan Form**	Yes	See Chapter 10.
	Riparian Vegetation	Yes	See Chapter 11.
Physicochemical	Organic Carbon	No	Difficult to develop reference curves.
	Bacteria**	No	Difficult to develop reference curves, CO water quality criteria are more targeted for human health than aquatic ecosystem function.
	Water Quality (pH, and Conductivity)	No	Conductivity and pH were not prioritized for inclusion in this version of the CSQT.
	Water Quality (Temperature and Dissolved Oxygen)	Yes	See Chapters 12 and 13.
	Nutrients	Yes	See Chapter 14.
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 15.
	Fish Communities	Yes	See Chapter 16.

** 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 reference curves included in the CSQT Beta Version sometimes relied on data from national and regional resource surveys and other available datasets. Potential data sources were evaluated using the five assessment factors outlined by the Science Policy Council in *A Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information*, including applicability and utility; evaluation and review; soundness; clarity and completeness; and uncertainty and variability (USEPA 2003). Some larger datasets were used to inform reference curves for multiple metrics, and those datasets are introduced here.

WY Geomorphic Reference Dataset:

Geomorphic reference datasets collected by the Wyoming Game and Fish Department (WGFD) and the US Forest Service (USFS) were compiled for the WSQT v1.0. The metrics and reference curves developed from these data sets are also used in the CSQT Beta Version. As additional geomorphic data are made available from CO streams, the reference curves can be further evaluated and updated as needed (see Section 1.8). The dataset consisted primarily of B (22 sites) and C (27 sites) Rosgen stream types. There were nine E type streams in the dataset but only three F streams. Therefore, reference curves derived from these data may be limited in capturing the geomorphic diversity of natural channel types, including multi-thread/anastomosing and natural canyon systems.

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 Wyoming Stream Technical Team (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 WY 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 Wyoming and Colorado SQTs. The NRSA dataset includes a variety of metrics associated with LWD, plan form, and riparian vegetation. Data were compiled from sites in Colorado, Wyoming, and surrounding states and grouped into EPA Level III ecoregions. Specific attributes from the dataset are used in this document and descriptions are provided to relate NRSA attributes to stratification or metrics within the CSQT.

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 conditions, from reference standard to degraded.

Colorado Natural Heritage Program (CNHP) Dataset:

A large riparian vegetation dataset was compiled by the Colorado Natural Heritage Program (CNHP; Kittel et al. 1999) to characterize riparian community types across Colorado. This dataset was provided by CNHP for use in evaluating riparian metrics for the Wyoming and Colorado SQTs.

The CNHP dataset included condition ratings for all sites, scored as A, B, C and D. For developing Wyoming and Colorado SQT 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 Wyoming and Colorado SQTs. 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.

Water Quality Control Division (WQCD) Dataset

The Colorado Department of Public Health and Environment (CDPHE) environmental data unit collects benthic macroinvertebrate, physical habitat, and water chemistry data on waters throughout the state. Data that was collected by CDPHE was downloaded from USEPA STORET [1] and processed in consultation with CPW. Sites within the dataset were identified as reference, other, or stressed based on an assessment of the upstream catchment using GIS and aerial imagery (CDPHE 2017). Impacts considered in the analysis included land use, point sources, water diversion, road density, abandoned mines, oil and gas facilities, and concentrated animal feeding operations. This dataset, referred to as the WQCD dataset in the rest of this document, was used to develop reference curves for nutrient and dissolved oxygen metrics.

Stratification by Ecoregion and Biotype

Several metrics described in this document are stratified by ecoregion, but sample sizes within each EPA Level III ecoregion were variable. To improve sample sizes, EPA Level III ecoregion data were grouped into broader ecoregion classifications, as shown in Table 1-4. Some metrics were derived from datasets with sites in multiple states; Table 1-4 includes all Level III ecoregions included in these datasets and thus not all occur within Colorado.

[1] https://www.waterqualitydata.us/portal/

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

Several metrics described in this document are stratified by biotype. Biotype is similar to, but distinct from the ecoregions described above (CDPHE 2017). Biotype is determined based on EPA Level IV ecoregion, elevation, and stream slope (Table 1-5). To avoid confusion with the ecoregion categories above, biotypes are identified in the CSQT by their number (1, 2, or 3) rather than their name designations (transition, mountains, and plains & xeric, respectively).

Table 1-5: Site Biotype Classification Rules (reproduced from Appendix A of CDPHE 2017)

Criteria	Biotype 1	Biotype 2	Biotype 3
Level IV Ecoregions:	21d, 21h, 21i, 21j, 25l, 26i	21a, 21b, 21e, or 21g	All 25 and 26 Level IV Ecoregions except 25l and 26i
Slope:	21c and slope < 0.04 ft/ft	21c and slope > 0.04 ft/ft	-
Elevation:	21f and elevation < 8,202 ft	21f and elevation > 8,202 ft	Any ecoregion, elevation < 5,085 ft

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

We recognize there are limitations to the approaches outlined herein to develop reference curves. There is a large diversity of stream types in Colorado and 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 to 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 will be possible to revise certain reference curves as more data become available and the CSQT is applied throughout the state (see Section 1.8). It is important to remember, however, that this tool is intended to compare pre- and post-project conditions at a site. As such, 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 thus application of certain metrics may be limited. 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. Table 1-6 shows what parameters are applicable to different stream flow and channel types. Additionally, modifications to sampling methods may be needed to accommodate data collection in multi-thread or non-wadable streams.

Table 1-6: Applicability of metrics across flow type and in multi-thread systems. An 'x' denotes that one or more metrics within a parameter is applicable within these stream types.

Applicable Parameters	Perennial	Intermittent	Ephemeral	Multi-thread Channels
Reach Runoff	x	x	x	x
Flow Alteration	x	x		x
Floodplain Connectivity	x	x	x	x (BHR only)
Large Wood	x	x	x*	x
Lateral Migration	x	x	x	x
Bed Material	x	x	x*	x
Bedform Diversity	x	x		
Planform	x			
Riparian Vegetation	x	x	x	x
Temperature	x	Where baseflows extend through index period		x
Dissolved Oxygen	x			x
Nutrients	x			x
Macroinvertebrates	x			x (perennial only)
Fish	x	x		x

** Developed from perennial reference sites and may not be representative.*

Several metrics rely on bankfull depth or width 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. Therefore, guidance on bankfull identification is provided in the User Manual Appendix A. 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. Alternative approaches to field-derived bankfull can be used to calculate metrics in the CSQT Beta Version and are recommended where the bankfull feature cannot be identified, flow alteration has changed the return interval associated with the bankfull feature, or the practitioner is more experienced with hydrologic modeling than field determination of the bankfull stage.

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.
- Most metrics in the tool rely on wadeable data collection methods, and as such, the sampling efficiency and applicability of reference curves in non-wadeable rivers is unknown.
- The SQT is intended to quantify changes in stream condition and is not intended to compare stream function to the function of another resource type. However, some states are investigating ways to use the SQT for dam removal projects that convert lentic systems back into lotic systems. Others are discussing ways to use the SQT for out-of-kind mitigation, e.g. to create stream restoration credits from wetland impacts.
- By design, the CSQT is a reach scale, point-in-time tool. However, through routine monitoring, the CSQT can show trends or changes in condition that can be tied to channel evolution models.
- As a reach-based evaluation, the tool does not automatically evaluate secondary (indirect) effects in reaches upstream or downstream of the sampled reach. Additional reaches would need to be defined and assessed in order to quantify the changes in condition related to the indirect effects of improving longitudinal connectivity, fish passage projects, beaver re-introduction, and other projects that have effects beyond the original reach limits.
- The CSQT relies on structural measures and indicators instead of measuring stream processes directly. The structural metrics included in the CSQT are reasonable surrogates to characterize underlying processes.
- The tool allows users to add parameters/metrics to the basic suite of metrics based on project-specific setting, goals, and objectives. 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 CSQT should be used to characterize pre- to post-project 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 scoring for the CSQT has a simple approach to weighting, instead of relying on a more rigorous, statistically derived approach.
- 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. Revisions to the CSQT and Reference Curves

Reference curves included in the CSQT Beta Version and this document will be reviewed and updated, as needed, including prior to the release of the CSQT Version 1.0. If additional datasets and/or literature values are provided during the public comment period or in the future, they will be evaluated using the five assessment factors outlined in *A Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information* (USEPA 2003) and considered for inclusion in the tool.

Additionally, the CSQT architecture is flexible and can accommodate additional parameters and metrics that are accompanied by reference curves. If a user is interested in proposing additional parameters or metrics for incorporation into the tool, they should provide a written proposal for consideration. The proposal should include data sources and/or literature references and should follow the framework for identifying threshold values and index scores that is outlined in this document.

Technical feedback may be submitted at any time to the USACE Pueblo Regulatory Office at 201 West 8th Street Suite 350, Pueblo, Colorado, 81003, or contact the office at (719) 744-9119; an email address can be provided on request.

1.9. Key Considerations

The CSQT and this scientific support document have been developed to apply the function-based approaches set forth in the 2008 Compensatory Mitigation Rule. Therefore, the following concepts are critical in understanding the applicability and limitations of this 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 CSQT is designed to assess the same metrics at a site over time, thus providing information on the degree to which the condition of the stream system changes following impacts or restoration activities. We refer to the CSQT as a change 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 CSQT itself does not score or quantify catchment condition. Catchment condition reflects the external elements that influence functional capacity within a project reach and is important to consider when selecting and determining the restoration potential of a site. These considerations are addressed in Chapter 3 of the User Manual.
- The CSQT is not a design tool. Many function-based parameters are critical to a successful restoration design but sit outside of the scope of the CSQT. The CSQT instead measures the hydraulic, geomorphological, and ecological responses or outcomes related to a project at a reach scale.

Chapter 2. Flow Alteration Module

Dams, diversions, groundwater withdrawals, effluent discharges, land use change, and climate change are common sources of flow alteration and can have cascading impacts on aquatic resources (Novak et al. 2015). Given the prevalence of irrigation ditches in the state, flow alteration is ubiquitous throughout Colorado. Flow alteration and habitat fragmentation associated with water delivery infrastructure have contributed to the decline of native fish species and degradation of aquatic habitat in Colorado. Approximately 60% of the fish species native to Colorado have either been extirpated or are in need of conservation actions (CPW 2015). Therefore, it is important to assess the degree of flow alteration in Colorado streams to evaluate impairment of stream functions, as well as inform restoration and mitigation strategies. To this end, the CSQT SC developed a provisional flow alteration module for the CSQT Beta Version to encourage mitigation approaches that restore components of the natural flow regime. The module and metrics are subject to testing and revision. Additional metrics may be considered for inclusion in future versions of the flow alteration module following beta testing.

A flow regime is characterized by the magnitude, frequency, duration, timing, and rate of change of these various flow events. Flow alteration can affect a range of environmentally relevant flows, including extreme low flows, baseflows, high flow pulses, small floods, and large floods (Poff et al. 1997). Flow is referred to as discharge, or Q, interchangeably in this document. Multiple approaches have been developed to characterize and evaluate hydrologic alteration (e.g., Richter et al. 1996; Olden and Poff 2003) and identify environmental flows that maintain important stream processes (e.g., Poff et al. 2010; Sanderson et al. 2012). The CSQT SC felt it was important to provide a basic characterization of flow regime to evaluate potential flow changes associated with impact or mitigation projects. Our goal with the flow alteration parameter was to develop a module that builds on previous tools and research to characterize aspects of the flow regime that are associated with the types of activities commonly encountered in the CWA 404 program.

<u>Module Structure and Scoring</u>:

The flow alteration module is included as a worksheet in the CSQT and Debit Calculator workbooks and is described in detail in the User Manual. The module is recommended when a project is expected to alter the existing hydrology.

The module is separate from the reach-scale assessment within the Quantification Tool worksheet because flow alteration has the potential to affect longer stream reaches than a project reach evaluated within the Quantification Tool worksheet. The flow alteration module calculates functional feet based on the stream length affected by the project-specific flow alteration. This length is referred to as the ‘affected stream length’. For example, a user may restore and protect baseflows within a 4-mile reach of river, and the mitigation project site is located on a 1-mile reach within this protected reach. The affected stream length for the flow alteration module would be 4 miles, while the existing stream length for the project reach would be 1 mile.

The architecture and scoring of the flow alteration module follows the architecture and scoring in the CSQT as described in Sections 1.4 and 1.5. Flow alteration module scores are based only on the subset of metric values that are input into the module. Index values are generated for each metric; metric index values are then averaged to create a module score. Metrics that are not assessed are simply not included in the score, i.e. they are not scored as a zero.

The flow alteration module estimates the change in condition at an impact or mitigation site by calculating the difference between existing (pre-project) and proposed (post-project) condition scores. The difference between existing and proposed condition scores is multiplied by affected stream length to calculate the functional feet for the module. The change in functional feet from the affected stream length is then weighted by 20% and added to the functional feet generated by the reach on the Quantification Tool worksheet (refer to Section 1.4 and the User Manual for more detail on scoring).

Metric Selection:

The selection of metrics within the flow alteration module considered limitations of data availability, ecological relevance, and applicability to the types of flow regimes and sources of alteration common in Colorado. The suite of metrics included in the flow alteration module are limited, and the opportunity exists to expand or modify this list of metrics as needed following beta testing and implementation. Additionally, these metrics may not be representative of a critical flow metric within a specific reach or watershed. Substitution of flow metrics may be considered where sufficient information is available to demonstrate a metric's importance to the local native flow regime.

In the development of the Watershed Flow Evaluation Tool (WFET), Sanderson et al. (2012) evaluated ecologically relevant flow metrics (Olden and Poff, 2003) using Indicators of Hydrologic Alteration (Richter et al. 1996). The analyses relied on gage data and Colorado's StateMod hydrologic modeling approach. In their study, Sanderson et al. identified five Indicators of Hydrologic Alteration (IHA) metrics that were compatible and sufficiently accurate to be useful in the flow analysis: mean annual flow, mean August flow, mean September flow, mean January flow, and mean annual peak daily flow. (Note: The WFET derived flow-ecology relationships using these metrics for specific basins within Colorado. For projects proposed in these areas, users should consider the WFET approach in evaluating flow alteration instead of the Flow Alteration Module.)

Multiple aspects of the natural flow regime, including extreme low flows, baseflows, high flow pulses, small floods, and large floods are important to maintain ecological processes (Mathews and Richter 2007). The flow alteration module includes a 7-day minimum metric to characterize extreme low flows; August, September and January mean flow metrics to characterize baseflows; a mean annual peak flow metric to characterize high flow pulses, and a mean annual flow metric to provide a general characterization of flow volume. The six-metric module is primarily focused on flow magnitude, and does not address frequency, duration, timing or rate of change. The monthly averages consider timing to a limited degree but not the timing of high flow events. All metrics included in the tool compare current or proposed conditions to native flow conditions. Because the metrics in the flow alteration module evaluate the departure from native flow conditions, the approach requires the availability of historical gage data and the ability to model native flow regimes (e.g., using StateMOD).

Metrics:

- Mean Annual Q (O/E)
- Mean Aug Q (O/E)
- Mean Sept Q (O/E)
- Mean Jan Q (O/E)
- Mean Annual Peak Daily Q (O/E)
- 7-Day Minimum (O/E)

Reference Curve Development:

While the flow alteration module was developed for the CSQT, reference curves for this metric have been adapted from the WSQT v1.0 for use and testing in CO. Criteria similar to what is outlined below were used to develop reference curves for the observed over expected monthly average flow for August (a metric for the flow alteration parameter in the WSQT v1.0). The reference standards were revised from the WSQT v1.0 as noted below.

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 et al. 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 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.

As a starting point for considering flow standards, Richter et al. (2012) proposed a presumptive flow standard for environmental flow protection (Table 2-1) which is applied to daily flow values and requires a hydrologic analysis to predict daily natural flows (undepleted and unregulated).

Table 2-1: Presumptive Flow Standard. Adapted from Richter et al. (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.

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). The ELOHA approach was also used to derive flow-ecology relationships in some basins in Colorado (WFET; Sanderson et al. 2012). In a study on flow alteration at stream gage sites throughout the U.S., 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. Binns and Eisermann (1979) considered the relationship of late summer baseflows to average annual flows to support trout.

Recognizing the level of effort that is typically undertaken to develop flow standards, the flow alteration module metrics rely on a presumptive standard (Richter et al. 2012) to define threshold values. The presumptive standard identified below considers flow alterations as a proportion of native flow values and is applied to all metrics in the flow alteration module. The following criteria were used to develop the reference curve in the CSQT Beta Version (Table 2-2):

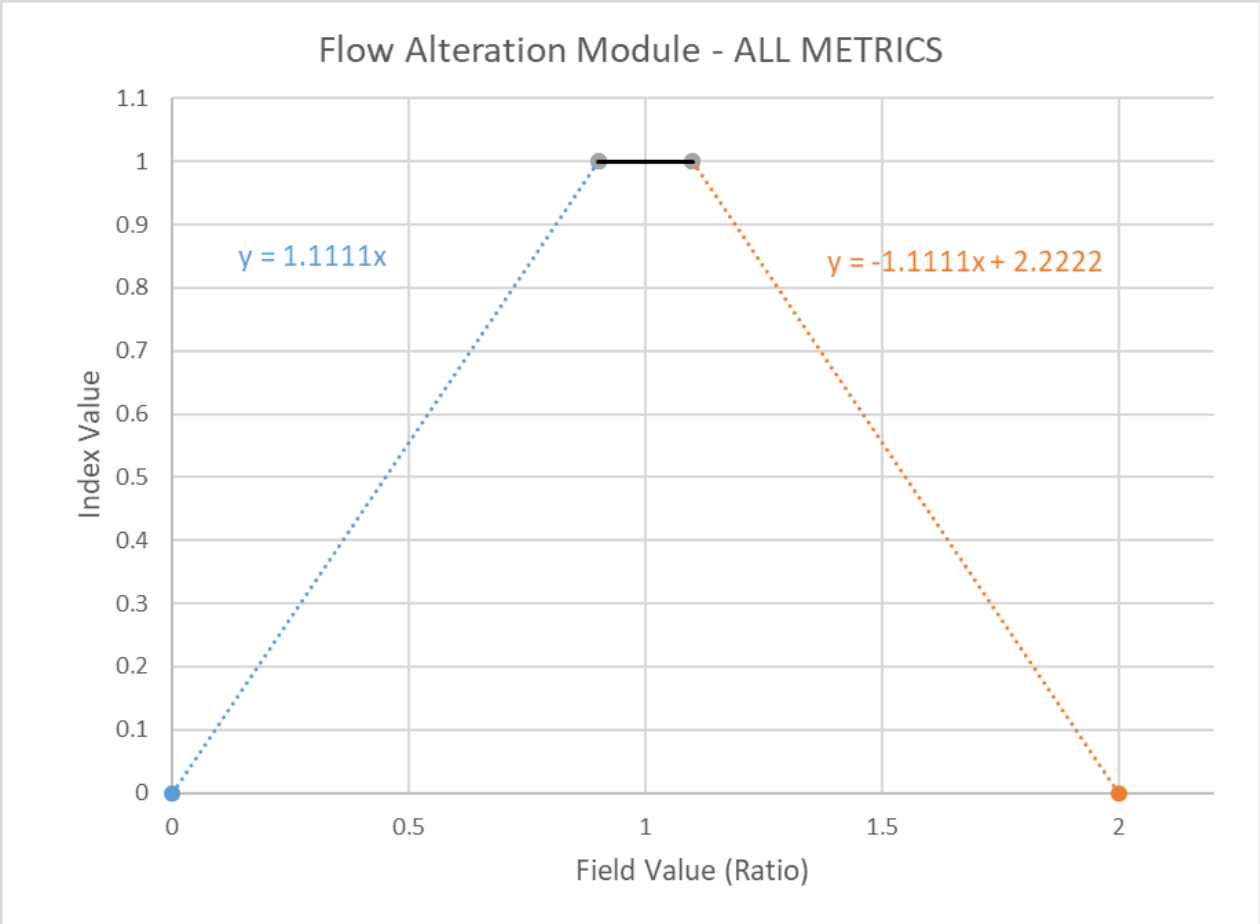
- The maximum index score (1.00) equates to a native flow value (e.g., a deviation of 0 or an O/E value of 1.0). In line with the presumptive standard, 90% to 110% of the expected flow value would yield an index value of 1.00 in the CSQT. Note that as a presumptive standard, this may be too lenient to protect sensitive systems (Richter et al. 2012).
- The minimum index score (0.00) equates to deviation of factor of 1, or an O/E value of 0.0 for flow decreases or 2.0 for flow increases). These values are based on best professional judgement, recognizing that substantial deviation from the native flow regime may impact stream structure and function.

A linear curve was fit between the defined values (Table 2-2 and Figure 2-1). Currently, there is a single reference curve to generate index values for all metrics, and no stratification of the reference curve by flow regime. Flow metrics and ecological sensitivity to flow disturbance will vary depending on the natural flow regime.

Table 2-2: Threshold Values for Flow Alteration Module Metrics (O/E)

Index Value	Field Value
1.00	0.90 – 1.10
0.00	0.00, 2.00

Figure 2-1: Flow Alteration Module Metrics Reference Curves



The reference curve was evaluated using data from several gaged sites; results are shown in Table 2-3. Data from 15 sites was collected and analyzed. All data were derived with the IHA application using data from USGS stream gages. The elevation of study gages ranged from 6,280 to 9,990 ft, and the contributing area ranged from 3 to 3,971 square miles. Case studies include sites with reference flow conditions (unaltered) and sites where flow has been altered by reservoirs, trans-mountain diversions, or both.

Table 2-3: Results of Flow Alteration Module Case Study

Statistic	Mean Annual Q (O/E)	Mean Aug Q (O/E)	Mean Sept Q (O/E)	Mean Jan Q (O/E)	Mean Annual Peak Daily Q (O/E)
	Unaltered Sites				
Number of Sites (n)	7	7	7	7	7
Maximum	1.30	1.24	1.18	1.34	1.45
Median	1.09	1.09	1.08	1.10	1.05
Mean	1.11	1.08	1.07	1.08	1.09
Minimum	0.98	0.95	0.95	0.61	0.93
	Altered Sites				
Number of Sites (n)	8	8	8	8	8
Maximum	1.76	3.73	6.97	3.25	1.46
Median	1.08	1.95	2.43	1.15	0.83
Mean	1.04	2.03	2.72	1.53	0.78
Minimum	0.27	0.38	0.43	0.68	0.27

* *Case studies did not include the 7-day minimum metric. A change was made late in the regionalization process that replaced a zero flow days metric with the 7-day minimum.*

Based on the case study results, the single reference curve for flow alteration metrics was included in the CSQT Beta Version. Generally, the altered sites scored not-functioning or functioning-at-risk with module scores ranging from 0.25 to 0.57, but two of the altered sites scored in the functioning range. The highest scoring altered site is the Yampa River at Steamboat Springs, which is altered by mainstem dams at Lake Catamount and Stagecoach reservoir but has a relatively intact flow regime, with moderately diminished peak and increased late summer flow. The irrigation withdrawals are likely buffered by alluvial storage in the floodplain that maintains baseflow into the late season. This was consistent with the module score for that site (0.99). The Paonia Reservoir site on the North Fork Gunnison River also received a functioning module score (0.73). The August and September flows were significantly diminished, scoring 0.43 and 0.21 respectively, while the other three metrics scored 1.00. For the altered sites, mean Annual Q was most altered at sites with trans-mountain diversions and main-channel reservoirs, as were Mean August Q, Mean September Q, and Mean January Q.

Module scores for the unaltered sites ranged from 0.79 to 1.00. Some reference sites exhibited relatively high degrees of altered January Q, but this may be related to data quality associated with ice, seasonally unavailable gage data, or because relative error increases as the magnitude of the flows decreases.

Limitations and Data Gaps:

Because a single, unstratified reference curve was applied for all metrics, the reference curve may not accurately capture the conditions that support geomorphic, physicochemical, and biology functioning in all streams. Zorn et al. (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. Beta testing across multiple flow regimes is needed to determine whether additional stratification or refined reference curves are needed.

The metric is also limited in that it does not characterize all ecologically relevant aspects of the flow regime (Poff et al. 1997, Mathews and Richter 2007). Baseflow 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 may be needed to adequately characterize other aspects of the flow regime, including the frequency and duration of high and extreme low flow events (Annear et al. 2004, Poff et al. 2010).

The flow alteration module requires three separate datasets in its current configuration: native hydrology, current hydrology, and proposed hydrology. In some parts of the state, native hydrology may be difficult to quantify. Sanderson et al. (2012) noted a limitation in the WFET in the Fountain Creek watershed due to the lack of gage data and StateMod coverage. Furthermore, hydrologic assessments range in complexity and magnitude of errors. Designers should have the expertise to perform these assessments and be able to analyze and defend the results.

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 module is intended to compare current and proposed hydrology within a reach to the native flow regime, and does not substitute for hydrologic analyses that support project planning and design.

Chapter 3. Reach Runoff Parameter

Functional Category: Reach Hydrology & Hydraulics

Function-based Parameter Summary:

This function-based parameter is within the Reach Hydrology and Hydraulics category and focuses on the hydrologic transport of water from the portion of the catchment that drains laterally into the reach. Changes in land cover, land use, and stormwater routing within the lateral drainage area impact the magnitude, duration, frequency, timing, and rate of change of runoff hydrographs entering the project reach. This metric only evaluates the lateral drainage area, as projects are often limited in their ability to influence hydrologic changes upstream of the project reach unless stream restoration project locations are strategically selected as part of larger catchment plans.

It is well understood that land use changes indirectly influence catchment-scale processes, and these changes often occur away from the stream and are distributed throughout a catchment (Beechie et al. 2013). While this parameter is limited to lateral drainage area, a broader characterization of the upstream contributing catchment is used to evaluate the restoration potential of a project reach (see Section 1.2) but is not directly scored within the CSQT. For projects that alter hydrology within or extending beyond the project reach, the CSQT also contains a flow alteration module, described in Chapter 2.

The lateral drainage area impacts 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 (Schueler et al. 2009), while agricultural practices can contribute sediment, nutrients, and other pollutants (USEPA 2005). The lateral drainage area plays a role in supporting the structure and function of 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. Including this parameter in the CSQT incentivizes stormwater management and land management practices on a reach-scale that can contribute to cumulative progress in a larger watershed.

The WSQT v1.0 included land use coefficient and concentrated flow points metrics under this parameter. Two additional metrics have been added to the CSQT, impervious cover and water quality capture volume (WQCV), to address some of the limitations of the existing metrics (discussed in Sections 3.1 and 3.3). The land use coefficient and impervious cover metrics quantify anthropogenic land use and land covers that alter the runoff and infiltration processes within the lateral drainage area. These metrics serve as indicators of changes to magnitude, volume, rate of change, and frequency of runoff-generating events.

The concentrated flow points metric was developed to accompany the land use coefficient in the WSQT v1.0 to address stormwater routing from larger drainages. 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 CSQT. However, larger lateral drainage areas are likely to have more concentrated flow, and thus the land use coefficient and concentrated flow metrics are intended to be applied together. Concentrated flow addresses rate of change, duration, and magnitude of the full spectrum of flow events.

While the other three metrics focus on restoration and impact activities that alter what are typically inputs to hydrologic models, e.g. land use, land cover, initial abstraction (depression storage, evapotranspiration and interception from increased canopy, etc.), and hydraulic routing (sheet flow vs. shallow concentrated flow), the water quality capture volume metric characterizes the magnitude, volume, rate of change, and frequency of small runoff-generating events coming from the lateral drainage area. This metric has been adapted from the Tennessee SQT (TDEC 2018) and requires an event-based hydrologic model that targets small but frequent runoff events.

The CSQT SC has decided to include all of these metrics in the CSQT Beta Version to evaluate and compare their performance and utility in calculating change related to the infiltration and runoff processes of the lateral drainage area.

Metrics:

- Land Use Coefficient
- Impervious Cover
- Concentrated Flow Points
- Water Quality Capture Volume (WQCV)

3.1. Land Use Coefficient

Summary:

The CSQT uses an area-weighted land use coefficient to numerically quantify the impact of various land uses on reach runoff. An area-weighted land use coefficient is calculated by first 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.

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 CSQT were selected to adequately represent a land use and reduce subjectivity.

This metric is intended to be used instead of the impervious cover metric in areas where agricultural or other land uses with low impervious cover are contributing runoff to the project reach.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

TR-55 provides curve numbers for various natural, agricultural, and urban land uses across a range of conditions. Curve numbers for natural land cover types in good condition are always less than 68 and often less than 60, while curve numbers for agricultural lands typically range from 70 to 80.[2] The curve numbers 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, curve number and runoff potential increases. A table of single land use coefficients, as opposed to ranges of curve numbers, is provided for CSQT users in the CSQT Beta Version User Manual

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 change likely contributes to substantially altered reach-scale hydrology. Threshold values are presented in Table 3-1.

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.

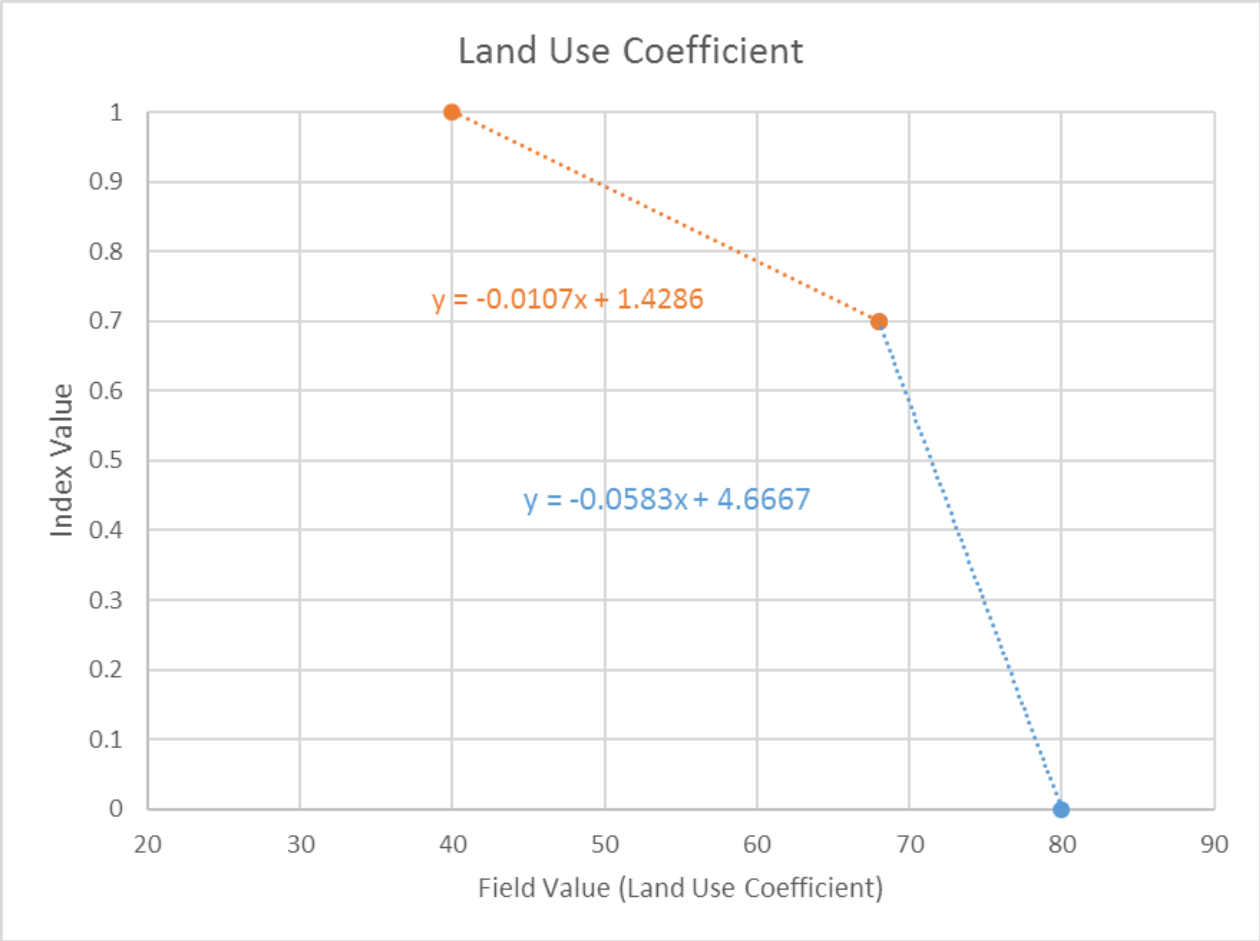
A broken-linear curve was applied for this metric (Figure 3-1). The curve is steeper in the not-functioning and functioning-at-risk range of scoring than in the functioning range, allowing for a broader range of land use coefficients within the functioning range to account for natural variability.

Table 3-1: Threshold Values for Land Use Coefficients

Index Value	Field Value
1.00	40
0.70	68
0.00	80

[2] Curve numbers referenced are for B HSG.

Figure 3-1: Reference Curves for Land Use Coefficient



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

This metric does not account for the variation in infiltration capacity, impermeable layer depth, or other characteristics important to estimating runoff volumes. Additionally, the land use coefficients do not 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 - the larger the contributing catchment area upstream, the less of an influence the lateral drainage has in maintaining stream functions within the project reach. 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 catchment location (e.g., the proportion of land area within the lateral drainage area compared with the entire catchment 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 testing and would benefit from additional application and testing in Colorado. 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.

3.2. Impervious Cover

Summary:

The CSQT uses impervious cover to numerically quantify the potential for hydrologic alteration of runoff coming from the lateral drainage area. Impervious cover is calculated by either 1) delineating all impervious surfaces within a lateral drainage area or 2) delineating larger areas of different land uses within the lateral drainage area of a stream reach and assigning a percent imperviousness to these areas, and then calculating the percent imperviousness for the lateral drainage area (see the User Manual for detailed methods). Land use imperviousness is based on the rational method outlined in the Urban Storm Drainage Criteria Manual Volume 1 (UDFCD 2016).

This metric is intended to be used instead of the land use coefficient metric in areas where urban, suburban or industrial land uses are contributing runoff to the project reach. Note that this metric does not distinguish between effective impervious area and total impervious cover. Effective impervious cover takes into consideration whether impervious surfaces are directly connected to the receiving stream reach (Booth and Jackson 1997). Disconnecting impervious surfaces from a project reach is incentivized using the concentrated flow points metric (Section 3.3).

Reference Curve Development:

Reference curves for impervious cover are based on the body of literature evaluating the effects of urbanization in streams (Walsh et al. 2005). Multiple studies have found that 10% impervious cover within a catchment often results in a loss of stream function (Booth and Jackson 1997, Schueler et al. 2009). Schueler et al. (2009) conducted a meta-analysis of impervious cover studies on stream health and proposed a reformulated impervious cover model that describes streams on a scale of stream quality that ranges from excellent (reference standard) to urban drainage as shown in Table 3-2.

Table 3-2: Reformulated Impervious Cover Model (Schueler 2009)

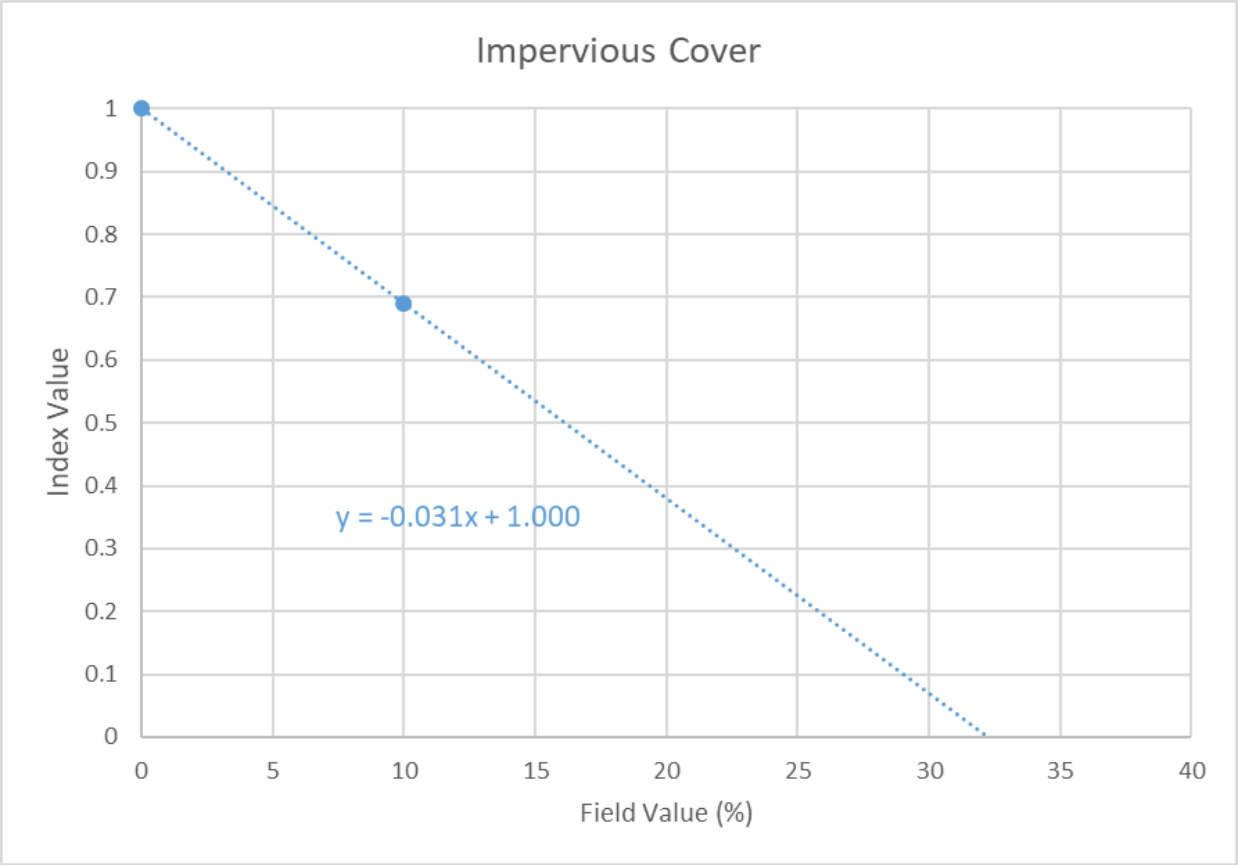
Impervious Cover (%)	Stream Quality	Urban Stream Categories
0-5	Excellent – Fair	Sensitive
5-10		Transition
10-20	Good – Fair	Impacted
20-25		Transition
25-60	Good – Poor	Non-supporting
60-70	Poor	Transition
71+		Urban drainage

The quality of streams in catchments with low impervious cover (<10%) is noted to be more influenced by factors such as riparian continuity and cropping practices (Schueler et al. 2009). Therefore, the absence of impervious cover represents a pristine condition (index = 1.00), and an impervious cover of 10% or greater is outside the reference standard range of functioning index values (index = 0.69). Threshold values are shown in Table 3-3, and a linear curve was fit to these values (Figure 3-2) and extrapolated to an index value of 0.00. Extrapolating the linear reference curve puts the break between functioning-at-risk and not-functioning condition at 23% impervious cover, which is within the transition zone shown in Table 3-2 from impacted to non-supporting. This aligns with the functional capacity definitions provided in Table 1-1.

Table 3-3: Threshold Values for Impervious Cover

Index Value	Field Value
1.00	0
0.69	10

Figure 3-2: Reference Curve for Impervious Cover



Limitations and Data Gaps:

The limitations of scale described in the previous section for land use coefficient are also applicable to this metric.

3.3. Concentrated Flow Points

Summary:

This metric assesses the number of concentrated flow points that enter the project reach from adjacent land uses per 1,000 linear feet of stream. The adjacent land use is assessed from the upstream to downstream ends of the project reach. Concentrated flow points are defined as erosional or constructed features (e.g., concrete swales, rills, gullies, ditches, 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 that directly enters the project reach by dispersing flow in the floodplain, increasing surface roughness, regrading to flatten slopes, removing roads and ditches, filling 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.

Reference Curve Development:

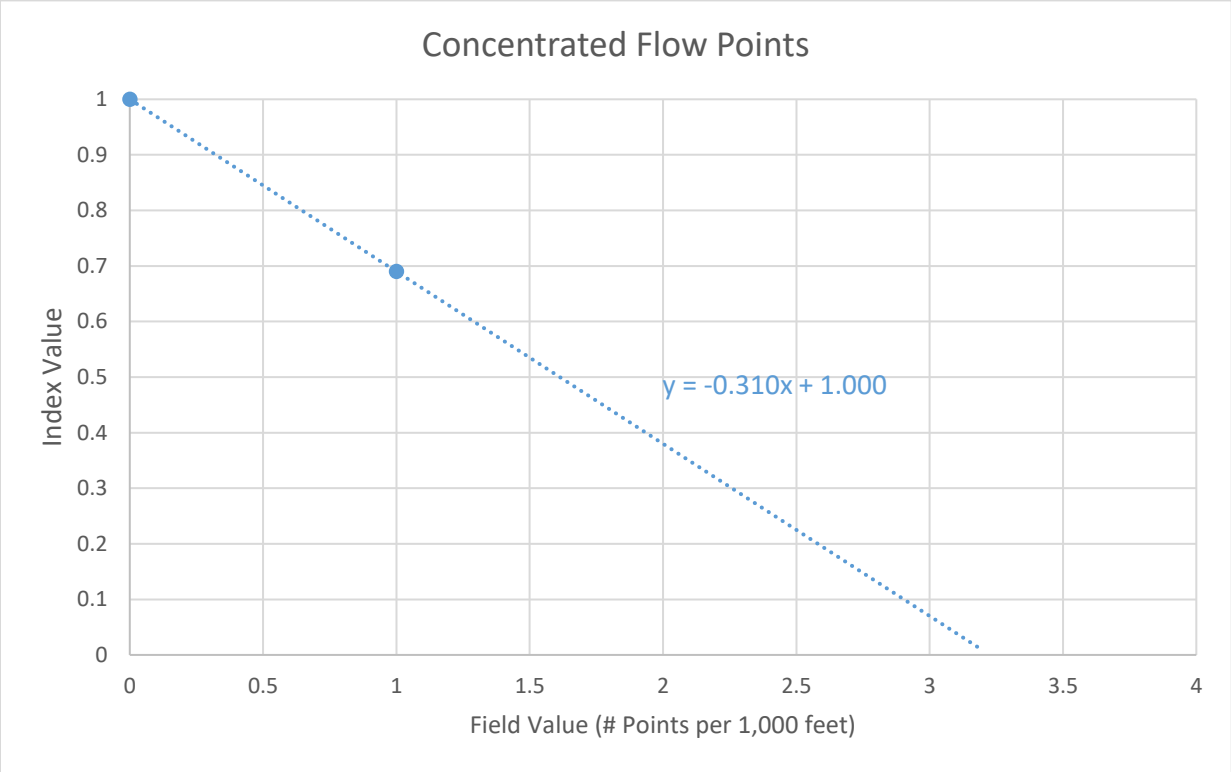
Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

The threshold values for this metric were based on best professional judgement, as literature values were not available that quantified 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). It was assumed that the absence of anthropogenic concentrated flow points reflected a reference standard, and the presence of one or more concentrated flow points per 1,000 ft no longer reflected a reference standard condition. Based on this logic, the threshold values shown in Table 3-4 were created, and a linear curve was fit to these values (Figure 3-3).

Table 3-4: Threshold Values for Concentrated Flow Points

Index Value	Field Value
1.00	0
0.69	1

Figure 3-3: Reference Curve for Concentrated Flow Points



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

This 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.

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.

This metric was developed for use in the North Carolina SQT and was incorporated into the CSQT Beta Version and WSQT v1.0. 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. This metric was initially intended to compliment the land use coefficient metric and incentivize reach scale practices that improve infiltration and runoff processes of the land that drains directly into the stream reach from the lateral drainage area.

3.4. Water Quality Capture Volume

Summary:

Stormwater infiltration of small, frequent rainfall events supports stream functioning by sustaining baseflow, reducing hydromodification, replenishing soil moisture for riparian vegetation, and removing stormwater pollutants. The water quality capture volume (WQCV) is defined as the runoff from frequent storm events (e.g. the 80th percentile storm) that accounts for much of the annual pollutant loads in urban catchments (UDFCD 2011). Volume 3 of the Urban Storm Drainage Criteria Manual describes various best management practices (BMPs) to provide runoff reduction and/or water quality benefits through capturing and/or treating the WQCV (UDFCD 2011). This metric assesses how much of the WQCV coming from runoff source areas (RSA) within the lateral drainage area is controlled through storage, infiltration, or evapotranspiration. Therefore, the field value for this metric is an observed over expected value for the amount of runoff generated during a small but frequent storm event. UDFCD (2011) outlines literature references that support the use of water quality capture volume, and notes that "[c]apturing and properly treating [the 80th percentile runoff event on the CO Front Range] should remove between 80 and 90% of the annual [total suspended sediment] load, while doubling the capture volume was estimated to increase the removal rate by only 1 to 2%."

This metric can capture the benefits of land use changes, disconnecting impervious cover, and removing concentrated flow points where BMPs are installed to remove/stabilize concentrated flow points and addresses some of the limitations of the other metrics for this parameter.

Reference Curve Development:

This metric was initially developed for use in the Tennessee SQT (TDEC 2018) and is adapted for use and testing in the CSQT Beta Version. The threshold values for this metric were based on best professional judgement and the index value for this metric is equal to the field value, as such:

- Two scenarios are representative of a reference standard condition and yield a 1.00 index value: 1) where the lateral drainage area has no anthropogenic runoff source area or 2) facilities are installed to treat 100% of the WQCV within the lateral drainage area.
- If the entire lateral drainage area consists of runoff source areas that are not treated in any way, the index value and field value are 0.00, representing no functional capacity.

Limitations and Data Gaps:

Because WQCV is a 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 procedures to apply this metric can require detailed hydrologic modeling to model the storage, infiltration, and evapotranspiration of stormwater runoff during a rainfall event, which may be onerous for projects that are not already performing hydrologic modeling of the lateral drainage area.

The UDFCD promotes the use of full-spectrum detention "to replicate historic peak flows for a broad spectrum of storm events." (pp. 3-1, UDFCD 2011) As this metric only accounts for small, frequent flow events, it does not characterize runoff changes associated with larger events that contribute to widespread channel erosion and pollutant loads.

Chapter 4. Baseflow Dynamics Parameter

Functional Category: Reach Hydrology & Hydraulics

Function-based Parameter Summary:

Baseflow dynamics was added to the CSQT as a function-based parameter to capture the ecological effects that changing channel dimensions has on baseflow habitat. This parameter will incentivize multi-stage channels and capture the impacts of channel widening. Hydraulic habitat modeling is a useful tool for understanding the baseflow dynamics necessary to maintain optimal habitat for biota and is one of several common approaches to evaluate environmental flow requirements in streams (Acreman and Dunbar 2004). There are many approaches available to characterize useable habitat area and baseflow dynamics (Espegren 1996, Annear et al. 2004), and some newer approaches are being developed to reduce the cost and level of effort associated with many existing methods (Wilding et al. 2014). Several hydraulic metrics have been identified as critical components of habitat maintenance flows and have been used for several decades in Colorado to inform minimum flow guidelines (Nehring 1979; Espegren 1996).

Baseflow is the flow within an intermittent or perennial stream that is sustained between precipitation events. The objective of this parameter is to characterize habitat conditions within the reach during baseflow. Baseflow is defined as the average of the mean daily flow values during the low flow period, typically in the late summer or early fall of the monitoring year. While the flow dynamics above baseflow (small flow pulses, bankfull flows, etc.) shape the channel, baseflow plays an important role in supporting water quality, water supply, and habitat (Price 2011). Baseflow volume and dynamics impact contaminant concentrations, water temperature, and available habitat area.

Two out of the three hydraulic parameters identified in Nehring (1979), average depth and average velocity, are included in the CSQT Beta Version to characterize baseflow dynamics. This parameter requires evaluation of both metrics at riffle features within the reach. The percent of bankfull wetted perimeter was considered for the CSQT Beta Version but was not included. While wetted perimeter is included in many models that determine minimum instream flow recommendations, it is not included in habitat suitability index models (HSI). The metric is considered a useful hydraulic variable in calculating minimum flows but less useful in characterizing changes that result from reach-scale restoration or impact activities.

Metrics:

- Average Velocity (fps)
- Average Depth (ft)

4.1. Average Velocity

Summary:

Baseflow velocity is a critical component of habitat maintenance flows. Riffle velocity has been found to directly affect macroinvertebrate survival and trout egg incubation (Nehring 1979).

The average velocity of a cross section at baseflow is calculated as the baseflow discharge divided by the wetted area at baseflow. For the CSQT, cross section surveys are collected at three riffle features and the results are averaged to determine an average riffle velocity at baseflow in the reach.

Reference Curve Development:

Average velocity minimum criteria is laid out in the *Evaluation of instream flow methods and determination of water quantity needs for streams in the state of Colorado* (Table 4-1; Nehring 1979).

Table 4-1: Minimum Flow Requirements Over Riffles (Nehring 1979)

Bankfull Width (ft)	Average Depth (ft)	Percent Wetted Perimeter* (%)	Average Velocity (fps)
1-20	0.2	50	1.0
21-40	0.2-0.4	50	1.0
41-60	0.4-0.6	50-60	1.0
61-100	0.6-1.0	≥70	1.0

**Baseflow wetted perimeter as a percent of bankfull.*

There are no reference standards for the velocity metric; this metric does not yield an index value and is considered only in calculating the parameter score. Where velocities are less than 1.0 fps, the parameter will score a 0.00, regardless of the field values for average depth metric. Where velocities exceed the 1.0 fps, they will not influence or inform the parameter score.

Limitations and Data Gaps:

This metric has received limited testing and would benefit from additional application and testing in Colorado. Future versions of the tool may consider whether a more comprehensive approach to characterizing velocity may be useful, particularly to characterize velocities across structures that have the potential to exceed the swim ability of important fish species.

Nehring (1979) notes that transect-based R2CROSS methods do not have a direct link to the biological condition of a stream.

An oft cited disadvantage to hydraulic habitat modeling is the cost (Nehring 1979, Acreman and Dunbar 2004) related to expenses associated with field data collection.

4.2. Average Depth

Summary:

Depth is one of the most important hydraulic criteria for maintaining fish passage (Nehring 1979). Nehring (1979) concluded that "average depth should be the primary criterion on which minimum flow recommendations are determined since it is the first factor to become limiting in almost twice as many instances as average velocity and wetted perimeter combined."

The average depth of a cross section at baseflow is calculated as the wetted cross-sectional area divided by the wetted top width. For the CSQT, cross section surveys are collected at three riffle features and the results are averaged to determine an average depth at baseflow in the reach.

Reference Curve Development:

Reference curves were developed using established minimum flow criteria, as well as average depth criteria from Habitat Suitability Index models. The minimum flow criteria shown in Table 4-1 were used to define the minimum index value of 0.00, as these criteria consider the maximum body depth of the largest fish present.

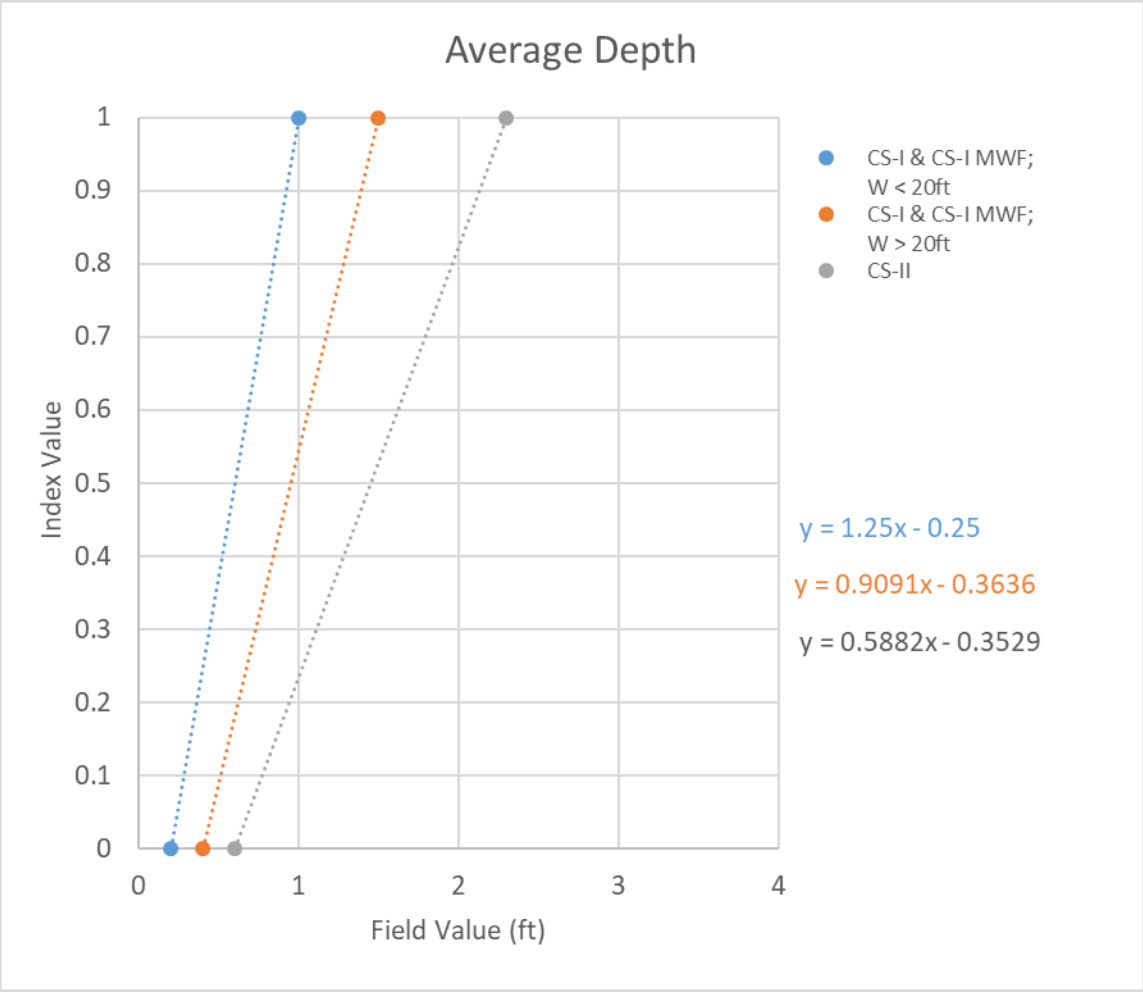
Habitat Suitability Indices (HSI) are similar to the SQT in that they score metrics on a 0.0 to 1.0 scale. HSI for multiple species were reviewed (Hickman and Raleigh 1982, Raleigh 1982, Raleigh et al. 1984, Raleigh et al. 1986, Wesche et al. 1987, Shuler and Nehring 1993) and the 1.00 scores from the average depth HSI variable were used to inform the maximum index value of 1.00 in the CSQT (Table 4-2). Where differences between HSI results occurred between species or studies, maximum index values were selected following consultation with CPW fisheries biologists.

Reference curves are stratified by stream width and temperature tier as shown in Table 4-2. Additional consideration may be given to additional stratification by life stage following beta testing. For the beta version, reference curves were only developed for cold water streams (CS).

Table 4-2: Threshold Values for Average Depth

Index Value	Field Values by Bankfull Width and Temperature Tier		
	CS-*I* & W < 20 ft	CS-*I* & W > 20ft	CS-II
1.00	1.0	1.5	2.3
0.00	0.2	0.4	0.6

Figure 4-1: Reference Curves for Average Depth



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

Average depth reference curves have not yet been developed for warmwater streams. Following beta testing, these curves will be developed for warmwater streams if this parameter remains in the tool.

Nehring (1979) notes that transect-based R2CROSS methods do not have a direct link to the biological condition of a stream. For this metric, we have relied on habitat preferences identified in Habitat Suitability Index models, recognizing that some of these curves may be outdated or not representative of the habitat needs for all life stages of fish. Newer curves are available for some species and life stages and additional updates to this parameter may be needed as new information is available in the literature, including stratification or additional metrics to capture the habitat suitability for different life stages. For example, Allyón et al. (2010) does not propose upper limit depth criteria for adult brown trout, while Louhi et al. (2008) suggest shallower depths are necessary to support brown trout spawning.

The cost of hydraulic habitat modeling noted above in the velocity metric section applies to this metric as well.

Chapter 5. 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 and are associated with meandering streams in alluvial valleys. Bankfull benches are narrower than floodplains and exist in confined or colluvial valleys. Bankfull benches are flat depositional features that provide some energy dissipation for higher flows. Floodplains and bankfull benches (also called flood prone areas) are assessed as floodplain connectivity in the CSQT.

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; Richer et al. 2015; Kondratieff and Richer 2018). 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 can be used to characterize floodplain connectivity.

Stage-discharge relationships use cross section geometry, channel slope and roughness to relate a water surface elevation, or stage, to a flow rate. The CSQT implemented stage-discharge relationships as a return interval metric that calculates the return interval of the discharge that is contained within the banks of the channel. If overbank flows are occurring only during infrequent flood events, then the stream is not well connected to its floodplain. For example, a channel that conveys a 5-year flood event with no overbank flows is not well-connected to the floodplain. Typical return intervals for bankfull discharge range from 1.01 to 2.0-year return intervals (Mulvihill and Baldigo 2012; Moody et al. 2003; Emmert 2004).

Recognizing the importance of side channels in contributing to the hydraulic and geomorphic functioning of streams in unconfined alluvial valleys, a side channel metric has been developed for the floodplain connectivity parameter in the CSQT Beta Version. While side channels provide many habitat functions, they also connect the main channel to the floodplain, e.g. sloughs and side channels, natural chute cut offs, and connecting oxbow ponds. Including side channels as a metric for floodplain connectivity emphasizes their role in distributing water on the floodplain.

The return interval, BHR, ER, and side channel metrics were selected for use in the tool. The BHR and ER combination are physically based (practitioners and regulators can measure in the field) and rely on a bankfull indicator or regional curve. These metrics do not require a gage or model. The return interval metric can be used instead of BHR to quantify the level of channel incision where a bankfull feature is not present or the user prefers a modeling approach. Additionally, in the scenarios where flow alteration rather than incision has reduced floodplain connectivity, the return interval metric should be used instead of BHR. The percent side channels metric would be used in addition to the other metrics when side channels are a desirable and appropriate end state for the project reach or would naturally exist in the given valley type (e.g., likely applicable in response reaches only). Each metric is described in more detail below.

Metrics:

- Return Interval
- Bank Height Ratio
- Entrenchment Ratio
- Percent Side Channels

5.1. Return Interval

Summary:

The return interval is the estimated likelihood of an event of a specific magnitude. A flood flow that has a 1% annual chance of occurring is commonly referred to as the 100-year flood since it has a 1/100 chance of occurring in a given year (1/100 = 0.01 exceedance probability = 1% chance of occurring in any given year). Hydrologic analyses are used to estimate the discharge of a range of return internals, e.g. 1.5, 2, 5, and 10-year events. Field measurements are then taken to determine the cross-sectional dimension of the channel along with slope and roughness. Desktop tools and sometimes hydraulic models are used to determine the return interval of a discharge that just fills the channel. If that the channel-filling 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. A channel that only conveys discharges between a 1 and 2-year return interval is well-connected to the floodplain. More detail about the field and desktop methods are provided in the user manual.

This metric was included in the CSQT as an alternative to bank height ratio (BHR). Where flow alteration rather than incision has reduced floodplain connectivity, the return interval metric should be used.

Reference Curve Development:

Typical return intervals for bankfull discharge range from 1.01 to 2.0-year return intervals in perennial streams (Mulvihill and Baldigo 2012; Moody et al. 2003; Emmert 2004). Table 5-1 summarizes findings relating bankfull discharge to return intervals for both natural and regulated flows; the San Antonio sites include a lot of regulated watersheds.

Table 5-1: Published Return Intervals for Bankfull Features

Bankfull Regional Curve	Return Interval (Min)	Return Interval (Max)
Oklahoma/Texas (Dutnell 2000)	1.01	3.65
Arid Southwest (Moody et al. 2003)	1.01	1.80
Kansas (Emmert 2004)	1.06	1.77
WY (Foster 2012)	1.01	1.70
San Antonio, TX (San Antonio River Authority, unpublished report)	1.01	2.10
Average	1.01	2.20

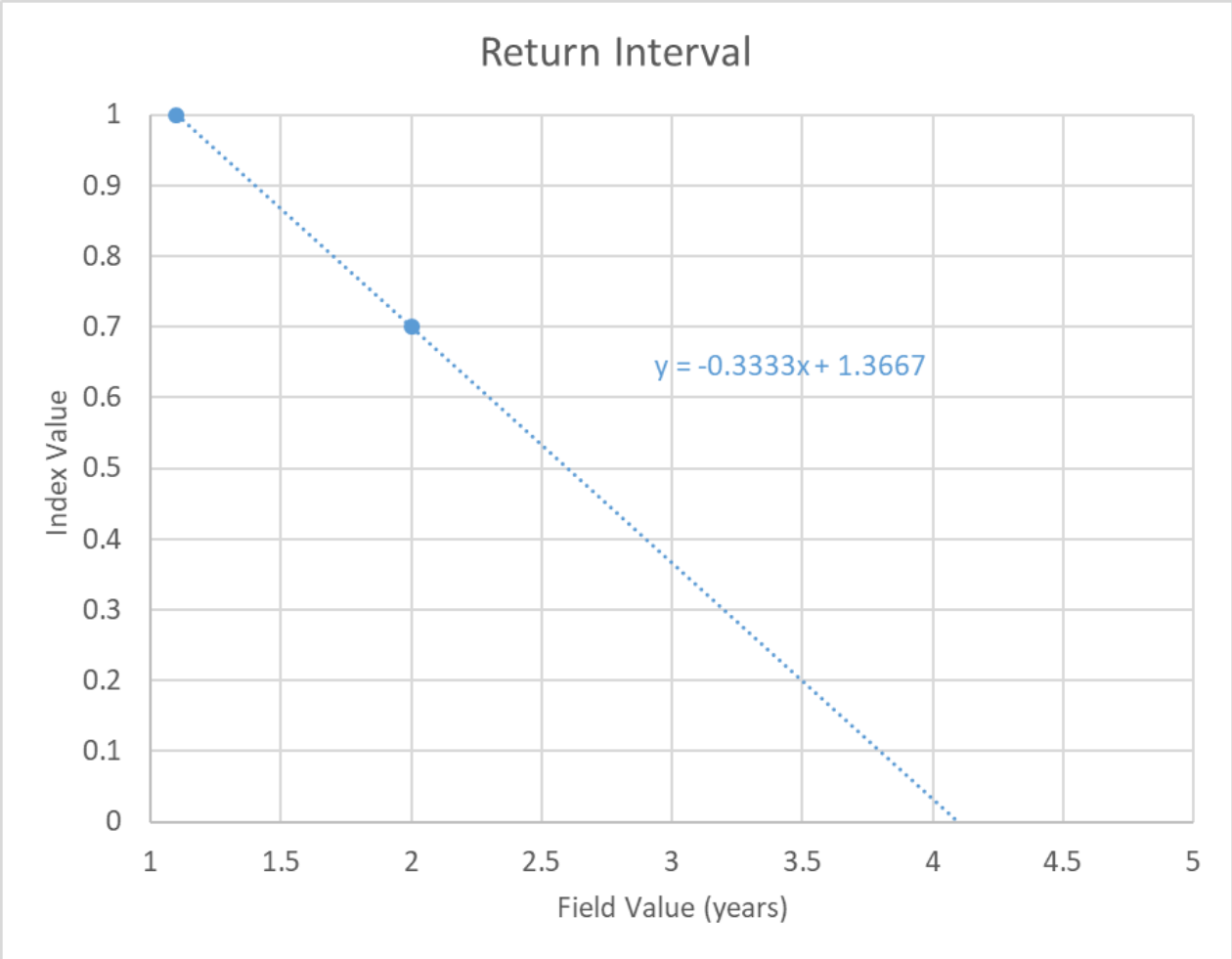
These data, and best professional judgement, were used to set the threshold values shown in Table 5-2. The national range for a bankfull return interval is generally accepted to be 1 to 2 years. A return interval of 2.0 was used to define the break between functioning and functioning-at-risk. This is supported by the studies in Table 5-1 where the average of the maximum return interval was 2.2. Only two of the reviewed studies documented return intervals that exceeded 2.0. One was a thesis, and the second is San Antonio where many of the gages have less than 20 years of data.

Anything less than or equal to a return interval of 1.1 was assigned an index value of 1.00. A return interval of 1.1 corresponds to an exceedance probability of 91%, meaning there is a 91% chance each year of this occurrence. It is possible to calculate return intervals down to 1.01, but the value cannot be less than 1.0 using a Log Pearson distribution. The team decided that trying to discern between a 1.01 and 1.1 required more accuracy than most estimating methods allowed. Therefore, all values less than 1.1 are assumed to be fully functioning/pristine with greater than a 91% chance of occurrence. A linear curve was fit to these values and extrapolated to generate functioning-at-risk and not-functioning values (Figure 5-1).

Table 5-2: Threshold Values for Return Interval

Index Value	Field Value
1.00	1.1
0.70	2.0

Figure 5-1: Return Interval Reference Curve



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

This metric is new for use in the CSQT, and a single reference curve has been developed for beta testing. Reference curves may benefit from additional stratification that accounts for natural variability in flow regimes and return intervals of overbank flows. The metric would benefit from additional validation, review and refinement as the tool is applied.

This metric is not intended to be used to influence or guide project design. A low field value like 1.1 may create a channel that is too small for systems with gravel/cobble sediment supply, and other systems where higher shear stress values are needed for sediment transport requirements. Designers should size the channel based on hydrology, hydraulic, and sediment transport requirements rather than using a particular return interval as a target.

Furthermore, all hydrologic and hydraulic models are not created equal. Some are simple estimates with large errors. Others are quite complex with moderate errors. Designers should have the expertise to select and operate the appropriate models and be able to analyze and defend the results.

5.2. Bank Height Ratio

Summary:

The bank height ratio (BHR) is a measure of channel incision and indicates whether a stream is or is not connected to an active floodplain or bankfull bench. The BHR is defined as 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 CSQT, BHR is measured at every riffle in the sampling reach, and a weighted BHR is then calculated from these measurements. The weighted bank height ratio is preferred over a simple average to increase repeatability. Methods for data collection and metric calculation are in the CSQT User Manual.

While non-incised channels dissipate high flows across the floodplain, incised channels within the same region lack floodplain connectivity and therefore contain flows of greater recurrence intervals (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 inundating 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 (refer to Section 5.1). 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, which decreases 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) and can be made in any stream with a bankfull indicator or regional curve.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in CO.

The BHR metric was developed by Rosgen (2009) as a measure of channel incision as shown in Table 5-3. 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 5-3).

Table 5-3: 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 using the WY geomorphic reference dataset described in Section 1.7. The WY geomorphic reference dataset consists of 61 sites that report BHR (Table 5-4). 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; Torizzo and Pitlick 2004). 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 5-4: Statistics for BHR from the WY 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 WY geomorphic reference dataset had BHR less than 1.2 (Table 5-4). Because the 75th percentile from the dataset aligned well with the criteria identified in Table 5-3, a BHR of less than 1.2 was used to define the lower threshold for the functioning range of index values (Table 5-5). 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.

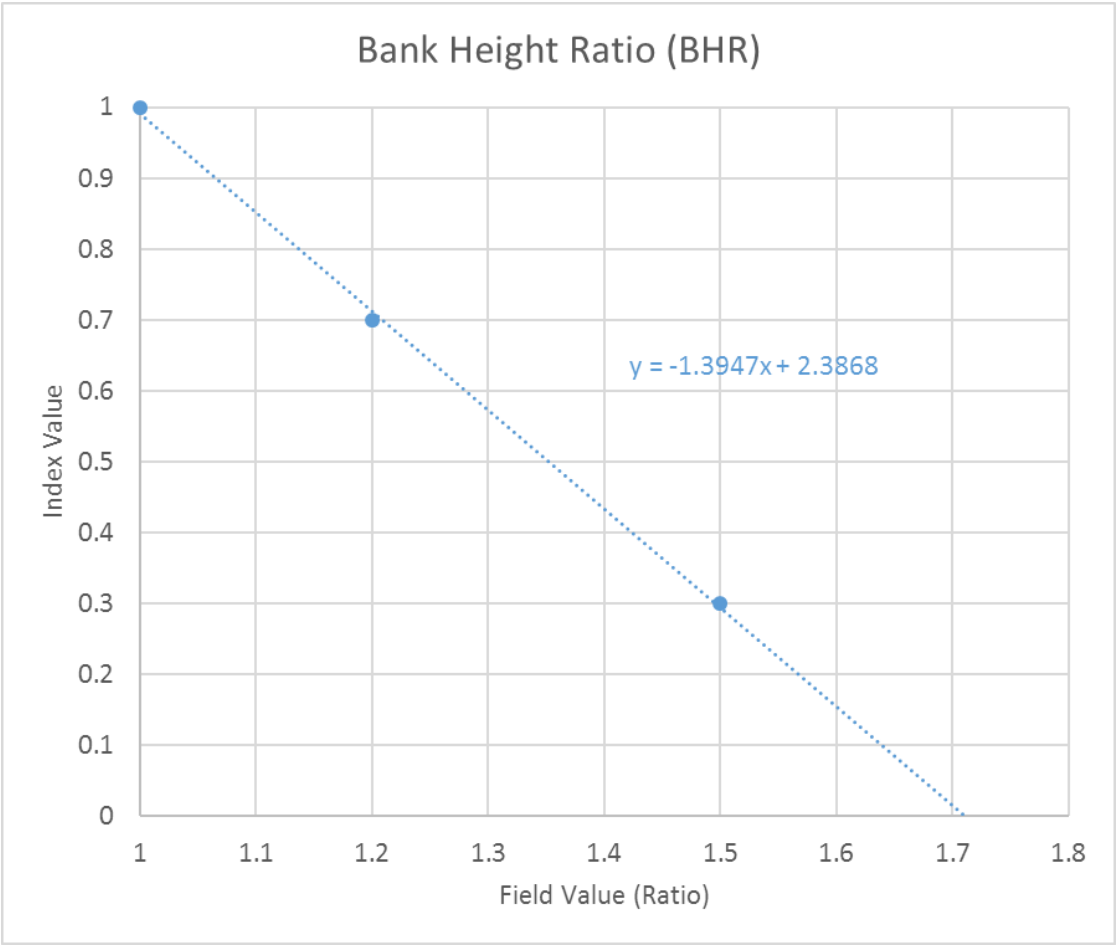
A threshold of 1.5 was used to differentiate index values within the functioning-at-risk and not-functioning ranges. BHRs of greater than 1.5 were considered not-functioning, consistent with the supporting literature classifying these as deeply incised channels with a greater likelihood of vertical instability (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 5-5 were plotted and a best-fit line was derived to provide a single equation to calculate index values from field values (Figure 5-2).

Table 5-5: Threshold Values for Bank Height Ratio

Index Value	Field Value
1.00	1.0
0.70	1.2
0.30	1.5

Figure 5-2: Bank Height Ratio Reference Curve



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

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. Information on verifying bankfull information is provided in the User Manual.

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, the return interval metric should be used.

5.3. 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). 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. Instructions for collecting and calculating the field value for this metric are provided in the User Manual.

ER estimates the lateral extent that floodwaters can spread across a valley. A stream is considered entrenched when flooding is horizontally confined, i.e. the flood prone width is small compared to the width of the channel. Large ERs are found in alluvial valleys where large flow events can spread out laterally. ER naturally varies by valley shape and is therefore used as a primary metric in differentiating stream types (Rosgen 1996). ER can also be a useful indicator of functional capacity as many anthropogenic alterations (e.g. levees, berms, and channelization) constrict the natural extent of floodplains and thereby decrease floodplain connectivity.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

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 5-6 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 5-6: Entrenchment Ratio Performance Standards from Harman et al. (2012)

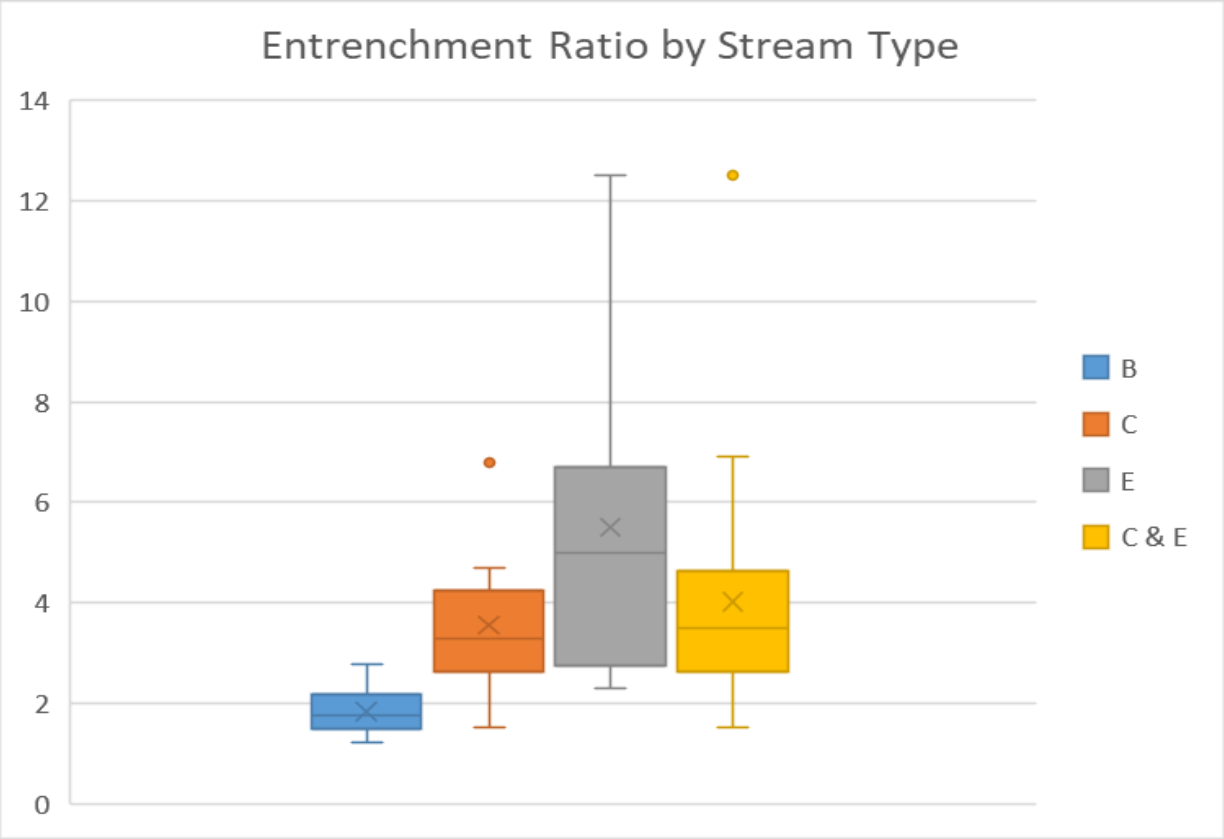
ER for C and E Stream Types	ER for B and Bc Stream Types	Functional Capacity
> 2.2	> 1.4	Functioning
2.0 – 2.2	1.2 – 1.4	Functioning-At-Risk
< 2.0	< 1.2	Not-Functioning

The criteria proposed by Harman et al. (2012) were evaluated using the WY geomorphic reference dataset described in Section 1.7 of this manual. The WY 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 5-7 and Figure 5-3.

Table 5-7: Statistics for ER from the WY 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 5-3: Box Plots for ER from the WY Geomorphic Reference Dataset



Bankfull width was used as a denominator of this metric, and thus stratification by stream size was not needed. Scaling 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 WY 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 performance standards from Table 5-6 were evaluated using the WY geomorphic reference dataset to develop the threshold values in Table 5-8.

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 plan form. 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 5-6). The WY 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 WY 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 5-6 was consistent with the minimum ER observed for B stream types in the WY 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 WY geomorphic reference dataset (1.5).
- The ER value that yields the maximum index value was set at 2.2, the 75th percentile value from the WY 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 are shown in Figure 5-4.

Table 5-8: Threshold Values for Entrenchment Ratio

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

Figure 5-4a: Entrenchment Ratio Reference Curves

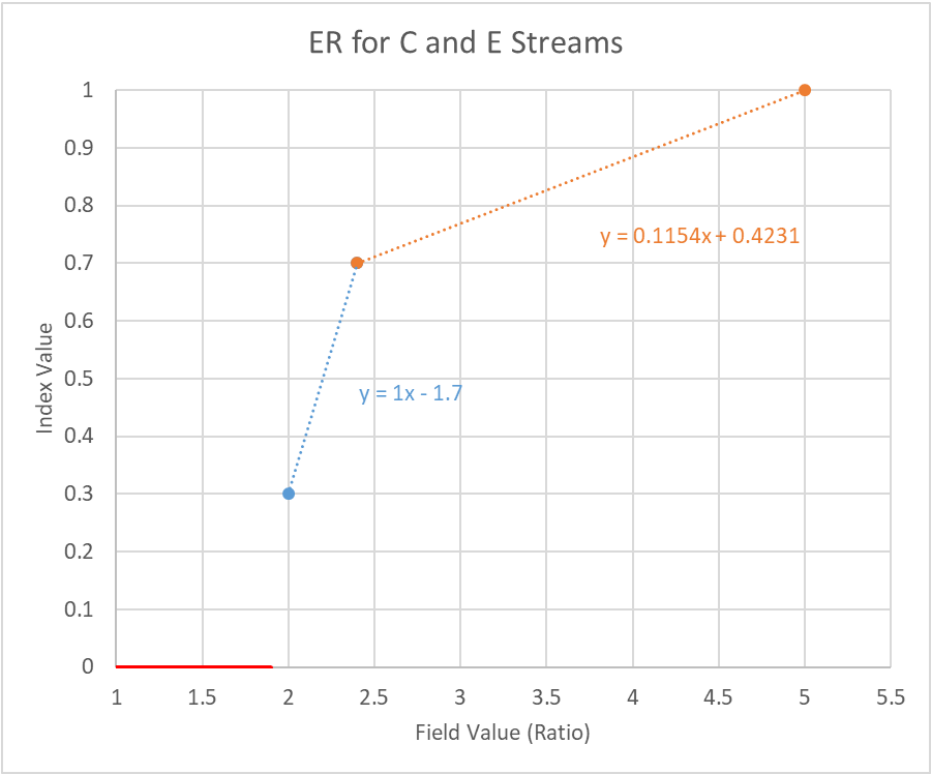
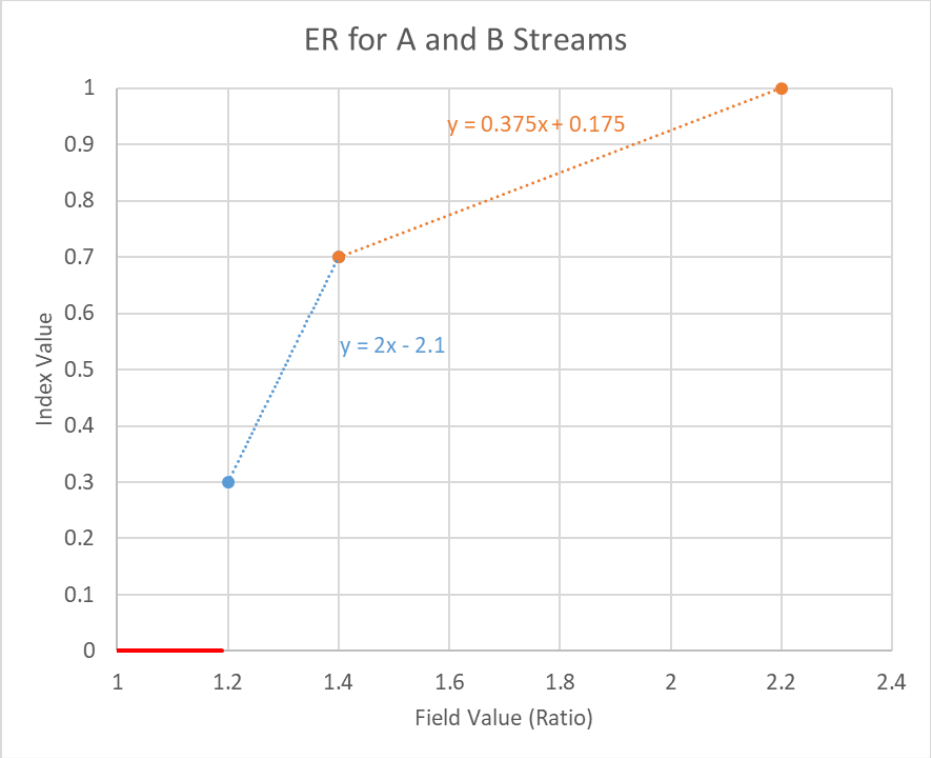


Figure 5-4b: Entrenchment Ratio Reference Curves



Limitations and Data Gaps:

If bankfull dimensions are not accurately determined for a site, then the entrenchment ratio will not accurately represent entrenchment processes. When possible, localized regional curves and flood frequency analysis should be used to verify the field indicators of bankfull. Information on verifying bankfull information is provided in the User Manual.

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 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. Guidance is provided in the User Manual to assist practitioners in identifying the reference stream type.

5.4. Percent Side Channels

Summary:

Side channels can provide thermal refugia (Fernald et al. 2006; Torgersen et al. 2012), habitat refugia during high flows, and may also be used as spawning habitat if they contain the ideal depths, velocities, and substrate size for targeted species (Pitlick and Steeter 1998). They may also provide critical juvenile rearing habitat for various fish species, as well as refuge from larger predatory fish (Brown and Hartman 1987; Fausch *et al*. 2002; Sommer *et al*. 2001; Angermeier and Schlosser 1995). In general, side channels increase hydraulic and geomorphologic habitat diversity and can create conditions that support a diverse assemblage of species during different life stages.

This metric estimates the percent of the project reach length that has side channels. Side channels were considered an important metric for inclusion in the Oregon Stream Function Assessment Method (SFAM; Nadeau et al. 2018), and their approach was used to inform this metric. Side channels include all open channels connected to the main channel of the project reach that carry water between baseflow and half-bankfull, even if it is only connected at one end, e.g. a slough. Floodplain channels that are not connected on either end to the main channel are not considered a side channel, an example being an oxbow that is filled on both ends (Landers et al. 2002). In addition, channels that are only inundated at bankfull and higher flows are not included.

Reference Curve Development:

Reference curves in the CSQT are adapted from the reference curves in the Oregon SFAM (Table 5-9; Nadeau et al. 2018). The SFAM is similar to the SQT in that it scores metrics on a 0.0 to 1.0 scale, with scores between 0.7 and 1.0 indicating a high functioning system. Nadeau et al. (2018) compiled data from multiple studies showing that increases in the area of side channel habitat leads to increases in coho smolt production. While Colorado does not support the same fish assemblages as Oregon, members of the CSQT SC recommended the metric and reference curves be tested for use in the CSQT Beta Version.

Table 5-9: Threshold Values for Side Channels Metric from the Oregon SFAM (Nadeau et al. 2018)

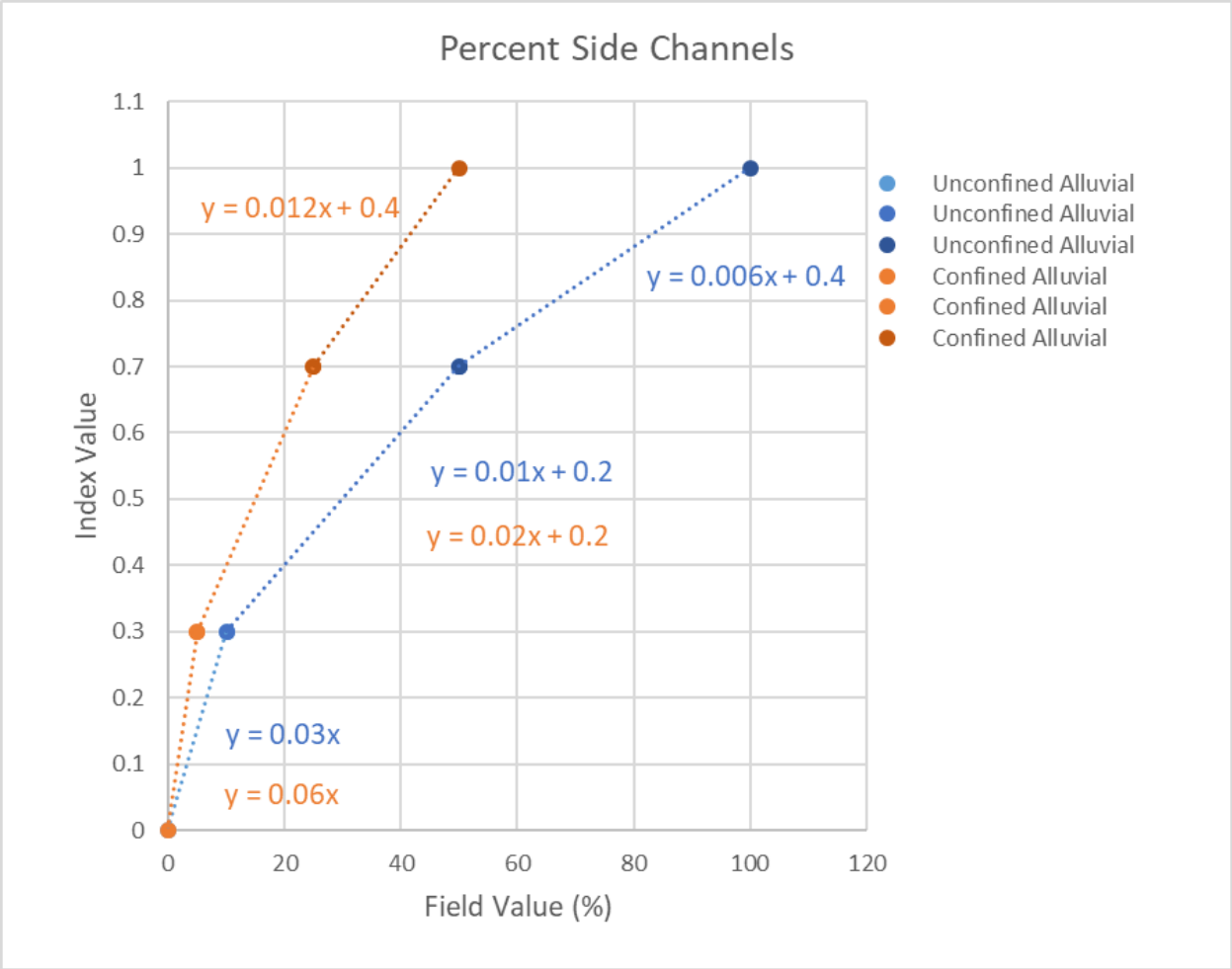
Index Value	Field Metric
1.0	100
0.7	50
0.3	10
0.0	0

While the side channels metric is not stratified in the SFAM, Nadeau et al. (2018) notes that side channels are features that are more common within alluvial valleys. The CSQT SC therefore decided to stratify the metric by valley type since impacts and restoration activities occur in both alluvial and colluvial valleys. The decision was made that the side channel metric would only be assessed in alluvial valleys. Additionally, two curves were developed for unconfined and confined alluvial valleys. The threshold values for perennial streams in unconfined alluvial valleys were aligned with the thresholds within the SFAM shown in Table 5-9. A second set of thresholds was developed for perennial streams in confined alluvial valleys (Table 5-10). The CSQT SC used best professional judgement to determine reference standards for the confined alluvial valleys. Confined alluvial valleys reference curves were based on those for alluvial valleys, but were then reduced to better represent that confined valley widths cannot support as much secondary channel length. Linear reference curves (Figure 5-5) were fit to the threshold values shown in Table 5-10.

Table 5-10: Threshold Values for Percent Side Channels

Index Value	Field Values by Valley Type	
	Unconfined Alluvial Valleys	Confined Alluvial Valleys
1.00	100	50
0.70	50	25
0.30	10	5
0.00	0	0

Figure 5-5: Percent Side Channels Reference Curves



Limitations and Data Gaps:

This metric was developed largely based on data from the Pacific Northwest and best professional judgment. While there are studies demonstrating the benfits of side channels in Colorado (Pitlick and Steeter 1998), this metric would benefit from additional validation, review and refinement as the tool is applied. In particular, the reference curves for streams in confined alluvial valleys would benefit from testing at field sites.

This metric is also limited in that it only measures the presence of side channels and not the quality of the side channels. The hydraulic conditions within both the side channel(s) and the main channel are not characterized by this metric and it is possible that suitable habitat conditions for target species are not met merely by the presence of side channels.

Chapter 6. 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 and jams) 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 two metrics from the WSQT v1.0 are also used in the CSQT Beta Version: 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)

6.1. 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

In developing the WSQT v1.0, data were collected at minimally disturbed reference standard sites primarily in the mountains of Wyoming, but a few sites were within the basin ecoregion of Wyoming. Table 6-1 shows the statistics for these data and for two additional sites, one reference site (West Plum) and one restored site (Fossil Creek), in Colorado. 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.

Table 6-1: Statistics for LWDI from Reference Sites. All values are per 328 feet (100 meters) of stream.

Statistic	LWDI Value	
	WY	CO
Number of Sites (n)	22	2
Average	689	292
Standard Deviation	416	11
Minimum	17	284
25th Percentile	433	-
Median	656	292
75th Percentile	948	-
Maximum	1583	300

The following threshold values were proposed based on the WY dataset (Table 6-2):

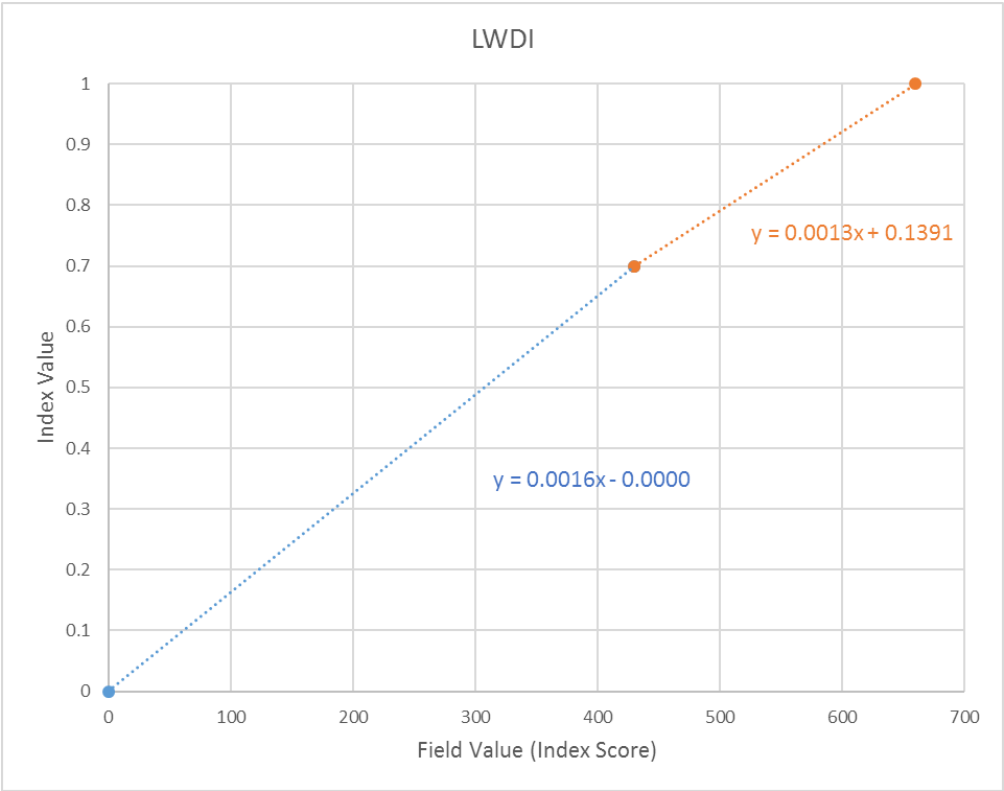
- The median of the reference dataset was used to determine the maximum index score (the median value of 656 was rounded up to 660). The median value was used instead of the 75th percentile to account for lower potential for LWD in plains and basins ecoregions. While there are sites from across the state of Wyoming 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 not-functioning index values. Index values within this range are interpolated from the reference curve.

Table 6-2: Threshold Values for the LWDI (per 328 feet or 100 meters)

Index Value	Field Value
1.00	660
0.70	430
0.00	0

These reference standards would indicate both the restored site and the reference site in CO score at the upper range of functioning-at-risk, which is consistent with the field observations of the CSQT SC. Note that while the reference site in Colorado, West Plum, had a large number of pieces, most of these pieces were relatively small in size and movable; therefore the LWD presence was not contributing as significantly to channel structure and roughness as reference sites observed in Wyoming. A reference curve (Figure 6-1) 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 6-1: LWDI Reference Curves



Limitations and Data Gaps:

The LWDI is a new metric for streams in Colorado and Wyoming and the reference curves have been developed from a relatively small dataset primarily from the mountains of Wyoming. As more data are collected, further refinement and stratification of these data and reference curves may be possible. Future 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. Note 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 do not qualify as LWD. Guidance is provided in the User Manual to address these situations.

6.2. 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 for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Reference curves were developed using the NRSA dataset, described in Section 1.7, 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 from pieces, the total number of pieces was estimated by assuming all dams only contained three pieces of LWD. Therefore, these estimated piece counts are likely lower than the actual number of pieces that would be collected using the CSQT 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 methods for SQTs. 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 CSQT 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 SQT 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 study and no single attribute was found 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.

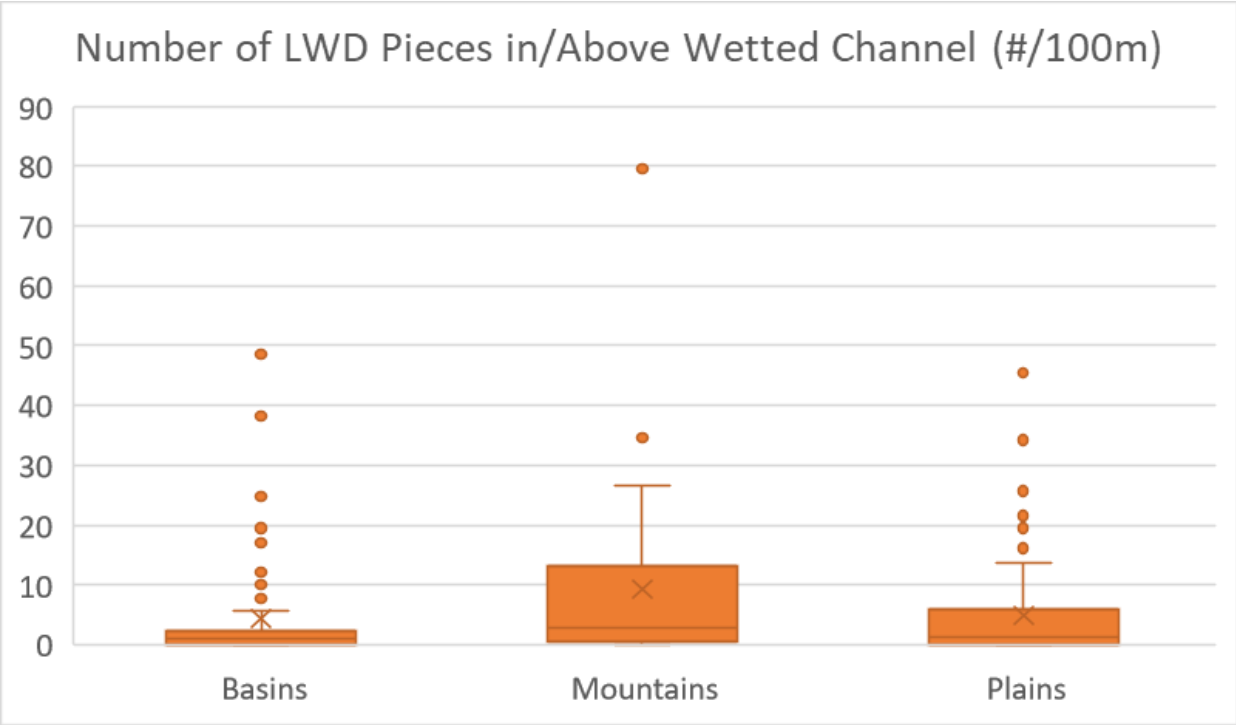
Stratification of the data by region was explored using the NRSA dataset. Stratification by bankfull width and dominant canopy type (coniferous, deciduous, mixed, or evergreen) were also considered. However, many of the NRSA sites listed the dominant canopy as absent, indicating that there was no canopy 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 6-3 and Figure 6-2. 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 6-3. 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, a single reference curve was applied to all ecoregions at sites occurring within naturally forested watersheds or riparian gallery forests.

Table 6-3: Statistics for Number of LWD Pieces from the NRSA Dataset

Statistic	Number of LWD pieces/100m by Ecoregion			Number of LWD pieces/100m All NRSA Data	LWDI Estimated Piece Counts
	Basins	Mountains	Plains		
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 6-2: 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 6-4 and Figure 6-3):

- 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 6-3).

Table 6-4: Threshold Values for the Number of LWD Pieces per 100 meters

Index Value	Field Value
1.00	28
0.70	13
0.00	0

Data from three reference quality sites within Colorado were scored to test this metric. The three reference quality sites had 13, 15, and 57 pieces of wood per 100m, which all score in the functioning range of index values.

Figure 6-3: Number of LWD Pieces Reference Curves

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

This metric is not applicable to streams without forested catchments, riparian gallery forests, or other streams that naturally have a limited supply of LWD. Note 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. Guidance is provided in the 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. Future stratification could consider the within-ecoregion differences associated with drainage area, forest age, valley type, canopy type, and other variables.

Chapter 7. Lateral Migration Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Lateral migration is a function-based parameter used to characterize streambank erosion rates. 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 four metrics from the WSQT v1.0 lateral migration parameter are also used in the CSQT Beta Version: 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. For systems with naturally high rates of bank erosion, the Greenline Stability Rating is a better metric than dominant BEHI/NBS and percent eroding streambank. The percent armoring metric should be used in stream segments where riprap or other hardened 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 (Section 1.2) and should be included in the design process.

Metrics:

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

7.1. 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 build/rebuild eroded portions of streambanks by filtering sediments (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 CSQT 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

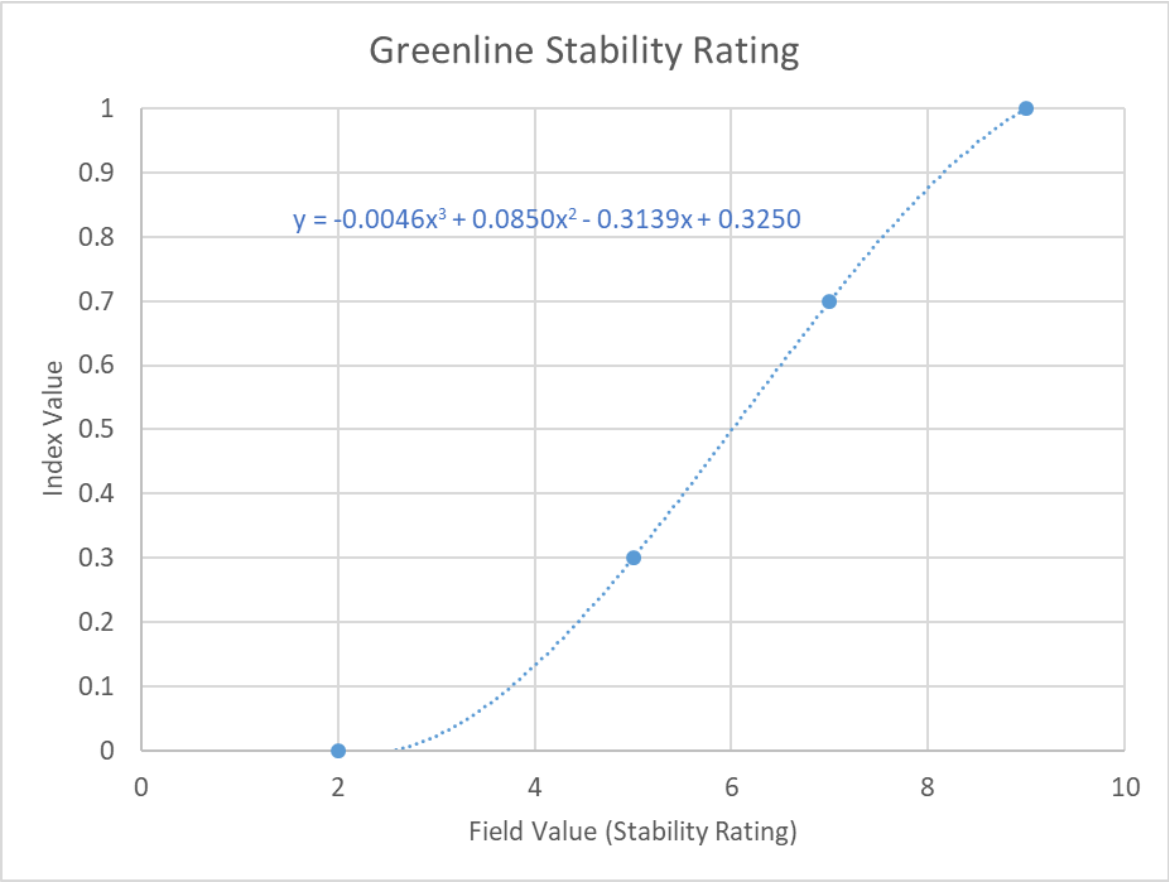
The threshold values and reference curve were constructed on the index rating classes established by Winward (2000) as shown in Table 7-1.

Table 7-1: 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 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 7-1. 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 in order to have a single equation to calculate index values (Figure 7-1).

Figure 7-1: Greenline Stability Rating Reference Curve



In August 2016, the Wyoming Stream Technical Team (WSTT) team visited several sites that were considered to represent minimally disturbed reference standard sites (Table 7-2). 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, the WSTT team concluded it was reasonable to characterize these sites as functioning or (high) functioning-at-risk. These sites have the potential to support aquatic ecosystem structure and function and were not in a clearly degraded state.

Table 7-2: 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

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 Multiple indicator monitoring Technical Reference (Table H1. on p. 136 of USDOI, 2011) 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).

7.2. 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 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). However, the use of BEHI/NBS 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 CSQT 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

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). These stability categories were converted into functional capacity ratings as follows: stable represents functioning, moderately unstable represents functioning-at-risk, and unstable and highly unstable represent not-functioning.

Table 7-3: 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. Additionally, not all BEHI/NBS combinations are included in the source reference and have been left out of this table.*

Because the metric relies on categorical data, reference curves were not developed. Instead, the ratings and categories from Table 7-3 were assigned index values based on relating the stability ratings to functional capacity as described below (and shown in Table 7-4).

- 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 conditions (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 ratings receiving lower scores.

Table 7-4: 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.

If bankfull dimensions are not accurately determined for a site, then the BEHI will not accurately represent erosion processes. When possible, localized regional curves and flood frequency analysis should be used to verify the field indicators of bankfull. Information on verifying bankfull information is provided in the User Manual.

7.3. 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. The BEHI/NBS ratings that represent non-eroding and actively eroding banks are listed in Table 7-5. 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 7-5: 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/VL, H/L, H/M, H/H, H/VH, H/Ex, VH/VL, VH/L, VH/M, VH/H, VH/VH, VH/Ex Ex/VL, Ex/L Ex/M, Ex/H, Ex/VH, Ex/Ex

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

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

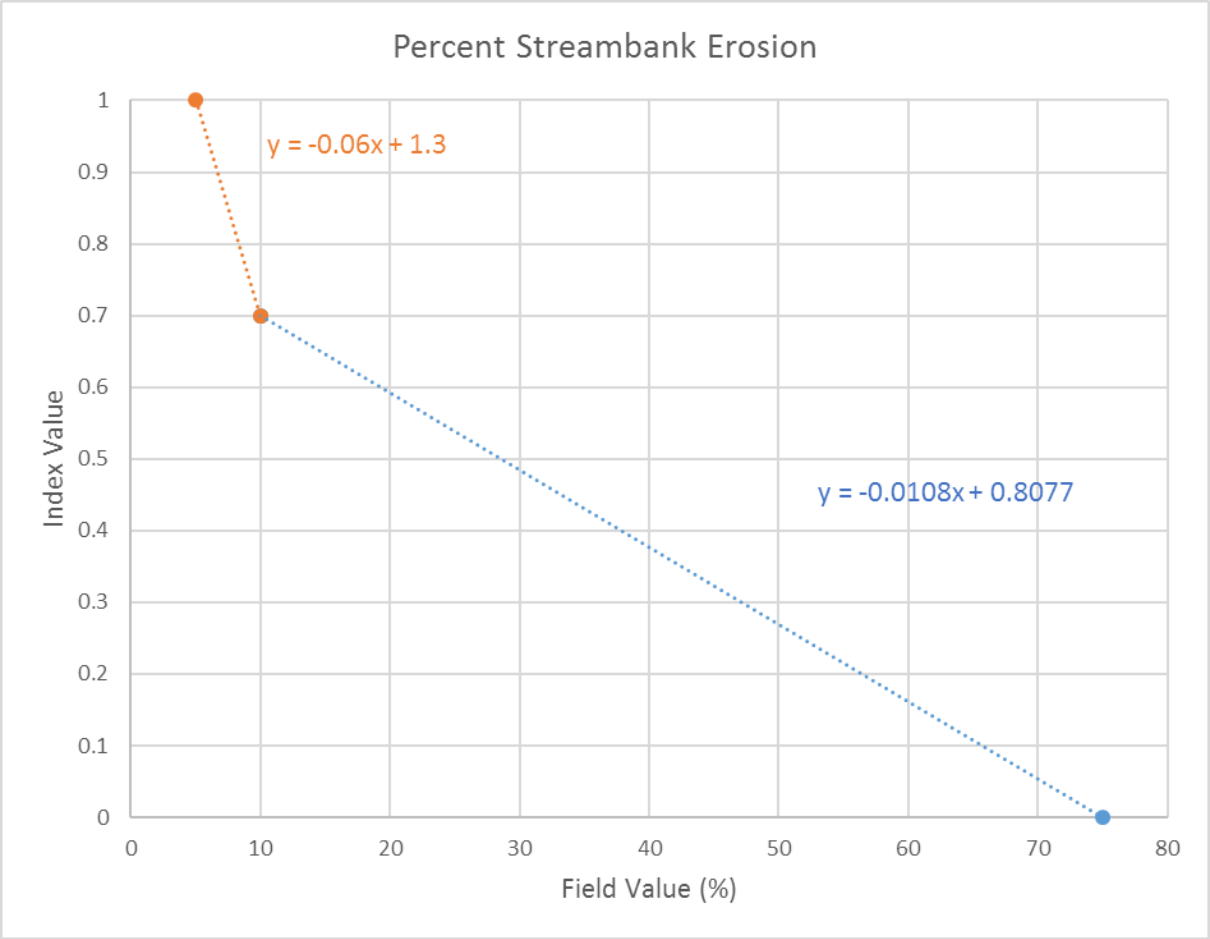
- 100% to 75% eroding banks are inadequate to support trout,
- 74% to 50% provide very limited potential,
- 49% to 25% provide limited potential,
- 24% to 10% provide moderate potential to support trout, and
- 9% to 0% eroding banks are completely adequate to support trout

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 7-6 were used to develop reference curves (Figure 7-2). 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 7-6: Threshold Values for Percent Streambank Erosion

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

Figure 7-2: Percent Streambank Erosion Reference Curves



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

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.

7.4. 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., riprap, gabion baskets, concrete or other engineered materials that prevent streams from meandering) along the bank edge. More natural approaches to reducing excessive bank erosion, like toe-wood and/or bioengineering, are not counted as armoring. 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. CPW monitoring experience in Colorado has not shown many beneficial effects of armoring on native species habitat except for the native transition species of stonecat. 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 CSQT includes 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Even though there are some benefits to armoring, the negative impacts to ecological function generally outweigh the positives, especially given the potential to affect erosion rates and sediment dynamics in adjacent reaches. Hard armoring does not support natural sediment processes and function, and the intent of stream restoration and mitigation is often to restore and enhance natural processes.

While research has shown a negative relationship between armoring and functional impairment in streams, there were no studies found that explicitly evaluated the relationship between the extent of armoring to functional impairment by stream length. 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 7-3). Setting the minimum index value at 30% armored stream length seemed reasonable, as it means that almost a third of the project reach is armored on both sides of the channel. At this level of armoring, the reach could be considered channelized

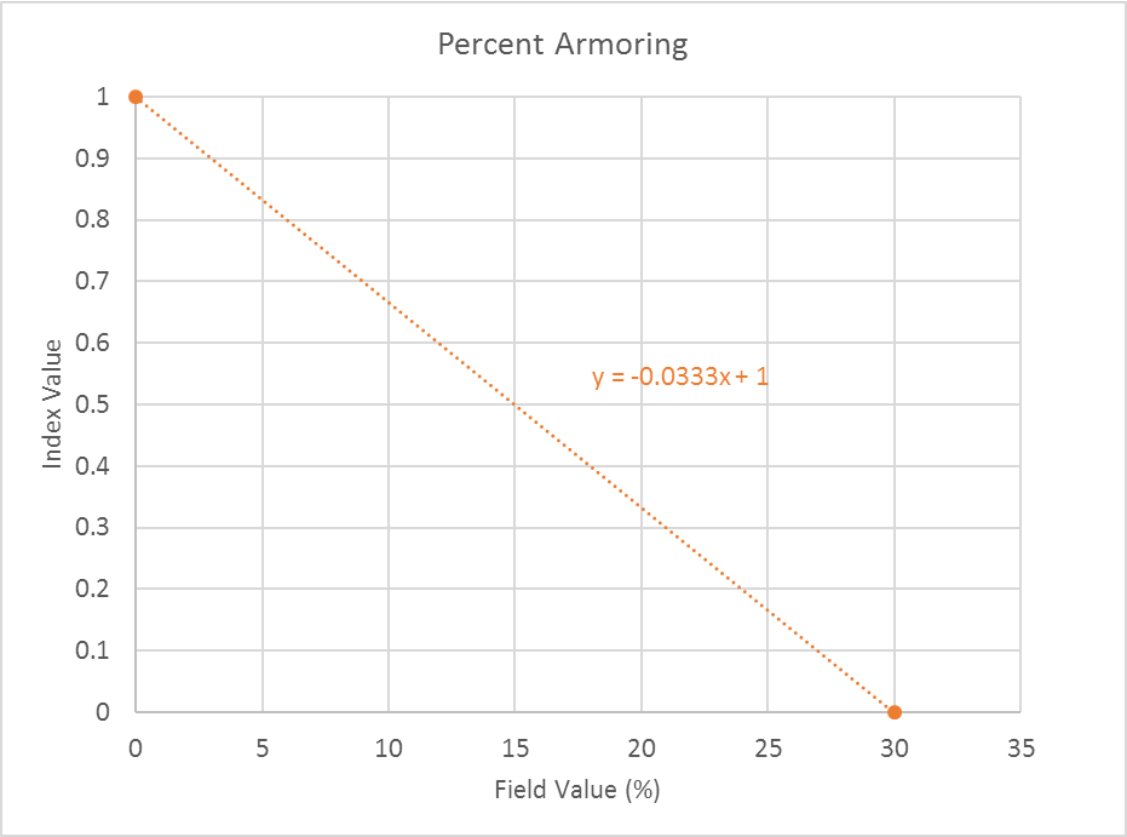
and functional loss of channel migration processes could be severe. Threshold values from the WSQT v1.0 are shown in Table 7-7.

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 7-7: Threshold Values for Percent Armoring

Index Value	Field Value (%)
1.00	0
0.00	30

Figure 7-3: 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.

Chapter 8. Bed Material Characterization Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

The ecological effects of fine-sediment accumulation are ubiquitous and wide-ranging (Wood and Armitage 1997). 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 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 CSQT 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 cumulative watershed effects of land management practices on changes to grain size distributions (Bevenger and King 1995). An embeddedness metric was considered for this parameter but existing metrics (e.g. Rosgen 2014 and USEPA 2016) are qualitative and it was decided to not include them in the Beta Version of the CSQT.

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, which have slopes ranging between 2 and 4% to show if upstream sediment supply is depositing on riffles. Results are placed into three bins that closely relate 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 CSQT Beta Version 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 CSQT.

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 CSQT 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)

8.1. 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 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 User Manual recommends 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

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.

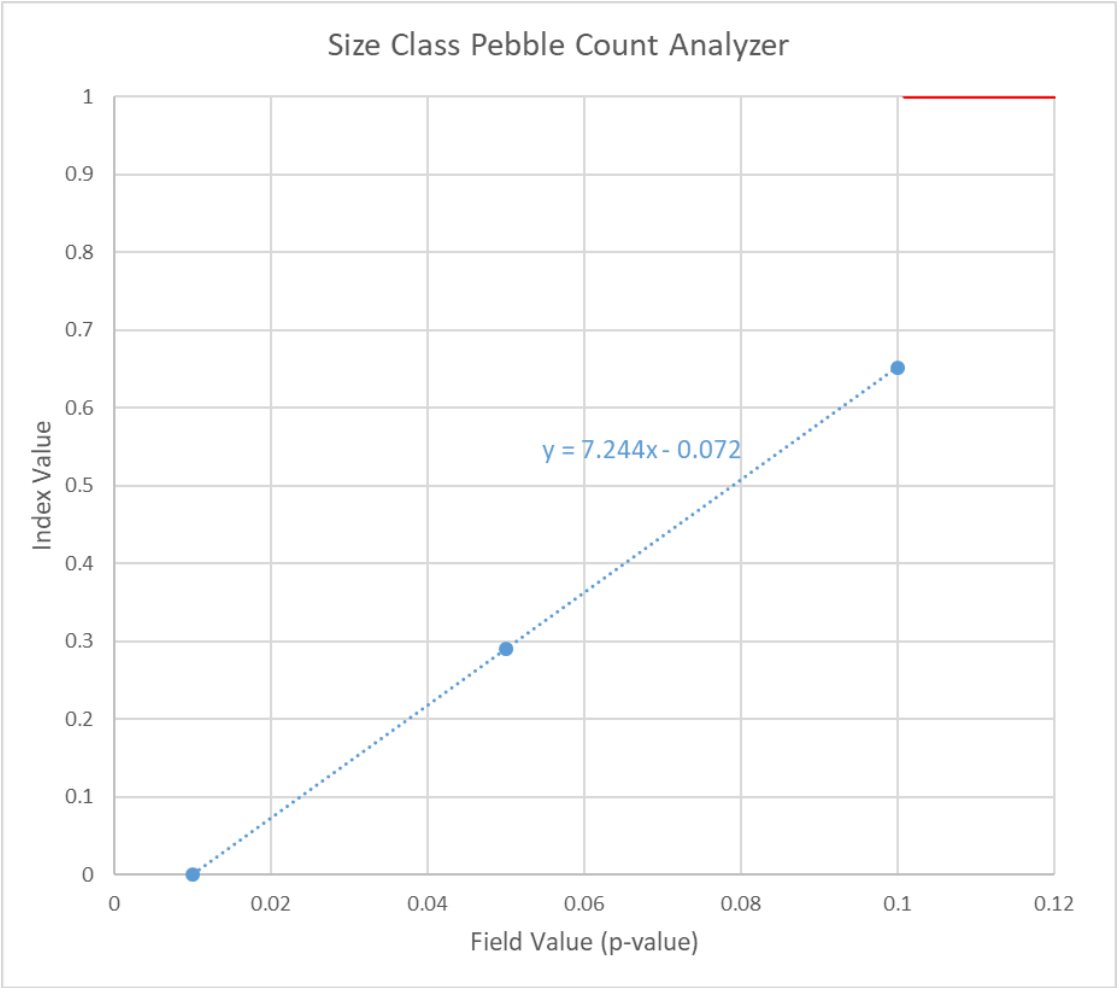
The case studies presented by Bevenger and King (1995) show that highly significant results (p-value < 0.001) were observed in the Shoshone National Forest due to fires, grazing and timber harvesting practices. No field data was collected to develop reference curves for this metric, instead typical statistical confidence intervals of 90%, 95% and 99%, corresponding to p-values of 0.10, 0.05 and 0.01, were used (Haldar and Mahadevan 1999). Typical p-values were used to assign threshold index values based on their degree of departure from the reference site as shown in Table 8-1.

Table 8-1: 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 8-1).

Figure 8-1: 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, field data was not collected to develop reference curves for this metric. Field data collection and analysis is still needed to test the sensitivity of this metric to stream restoration practices. In the future, the CSQT SC would also like to develop a percent fine sediments metric that does not rely on a reference site, but additional work 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 9. 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 (Rosgen 2014; Vermont Stream Geomorphic Assessment, Appendix M: Delineation of Stream Bed Features). 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 plane beds, 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, and 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 CSQT because quantitative measures are available and regularly used by practitioners.

The four metrics from the WSQT v1.0 bed form diversity parameter are also used in the CSQT Beta Version: 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, an example being a mid-channel bar within a riffle section.

Metrics:

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

9.1. 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 CSQT. The median is used instead of the mean because the sample size per reach tends to be small with a wide range of values and it was thought that the median provided a better estimate of central tendency than the mean. Field testing has also 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 CSQT SC 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 this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

The WY geomorphic reference dataset, described in Section 1.7, 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 Wyoming and Colorado SQT methodology based on maximum pool depth locations and station data from longitudinal profiles at each site.

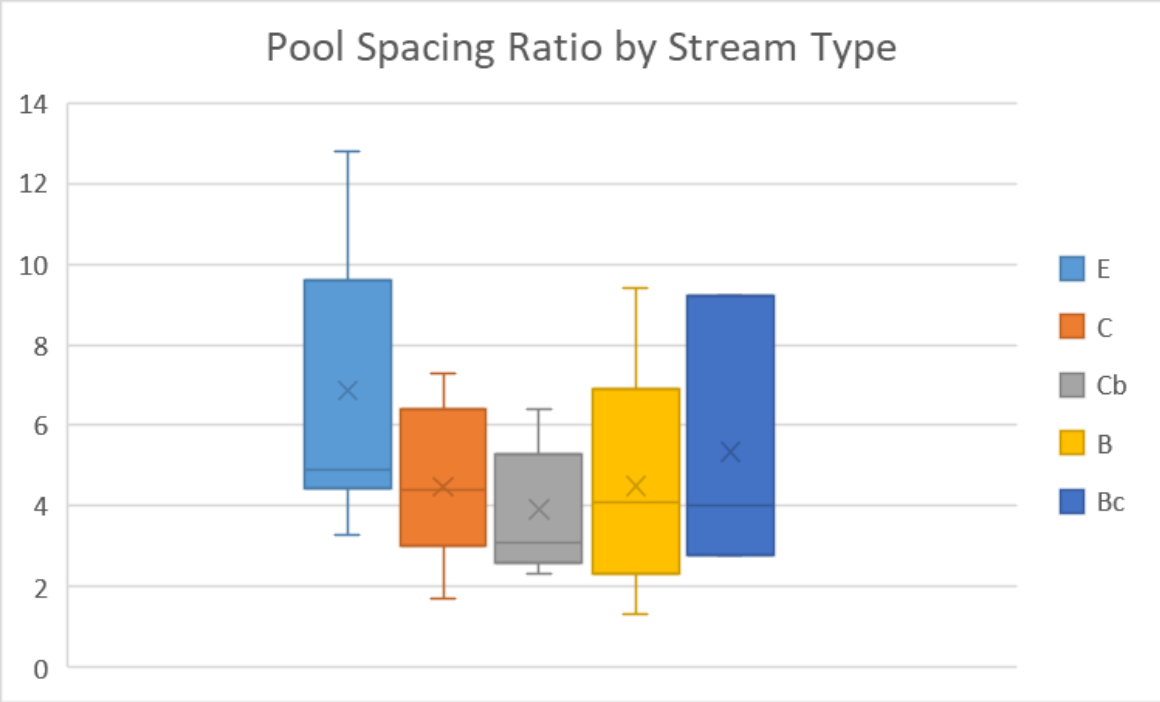
The metric accounts for differences in stream size by using bankfull width as the denominator and converting the value into a dimensionless ratio. The ratio accounts for the differences in stream size that may otherwise be relevant in determining pool spacing. Bankfull dimensions may vary based on differences 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 WY geomorphic reference dataset was assessed to determine whether stratifications based on drainage area or region were also appropriate (see discussion in Section 9.3). 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 9-1 and Figure 9-1. 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 9-1: Statistics for Pool Spacing Ratio from the WY 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 9-1: Box Plots for Pool Spacing Ratio from the WY Geomorphic Reference Dataset



Given that single-thread perennial streams exhibit a range of stable pool spacing, the data analysis shown in Table 9-1 and Figure 9-1 was combined with best professional judgement to derive the threshold values and reference curves shown in Table 9-2.

Table 9-2: 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	-

The 25th and 75th percentile values for each stream type were used to characterize the reference standard range of index scoring. Adjustments were considered 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.

Since the WY 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 9-2. 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. While the 25th percentile value was 3.2, a 4.0 was set as the 1.00 index value and 3.7 was set as the 0.70 index value to equal the low end of the reference condition. As ratios become less than 3.7, 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 WY geomorphic reference dataset. A pool spacing ratio of 7.0 was set at the 0.70 using best professional judgement. Values greater than 7.0 were not considered likely to 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 9-1, but it is common in E stream types with more variability in the belt width and meander wavelength. Figure 9-1 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 values shown in Table 9-2 are considered more conservative from a stream stability and habitat perspective.

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 editors 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.

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 9-2a and 9-2b. It is important to remember that the values in the CSQT are medians; therefore, a range of values can be used in the design process. Field testing of SQTs 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 9-2c and 9-2d do not show a loss of function with lower index values.

Figure 9-2a: Pool Spacing Ratio Reference Curves

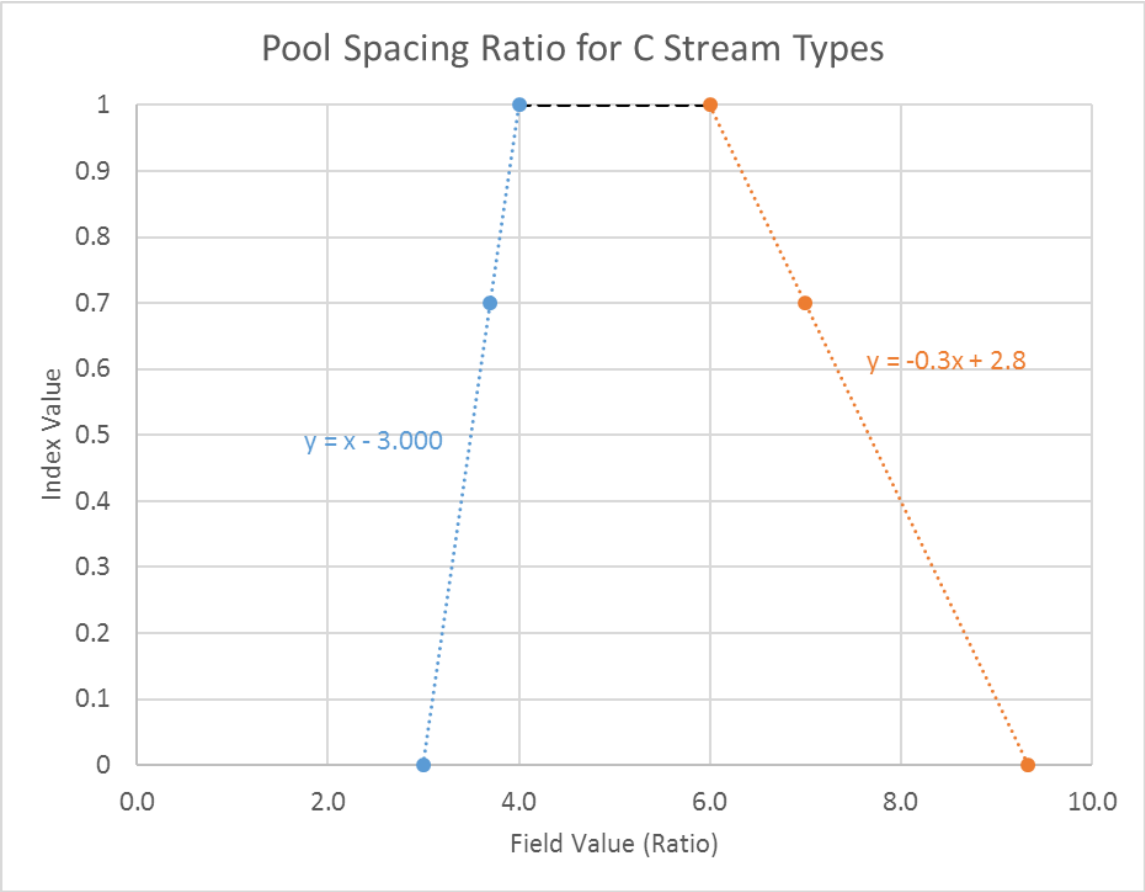


Figure 9-2b: Pool Spacing Ratio Reference Curves

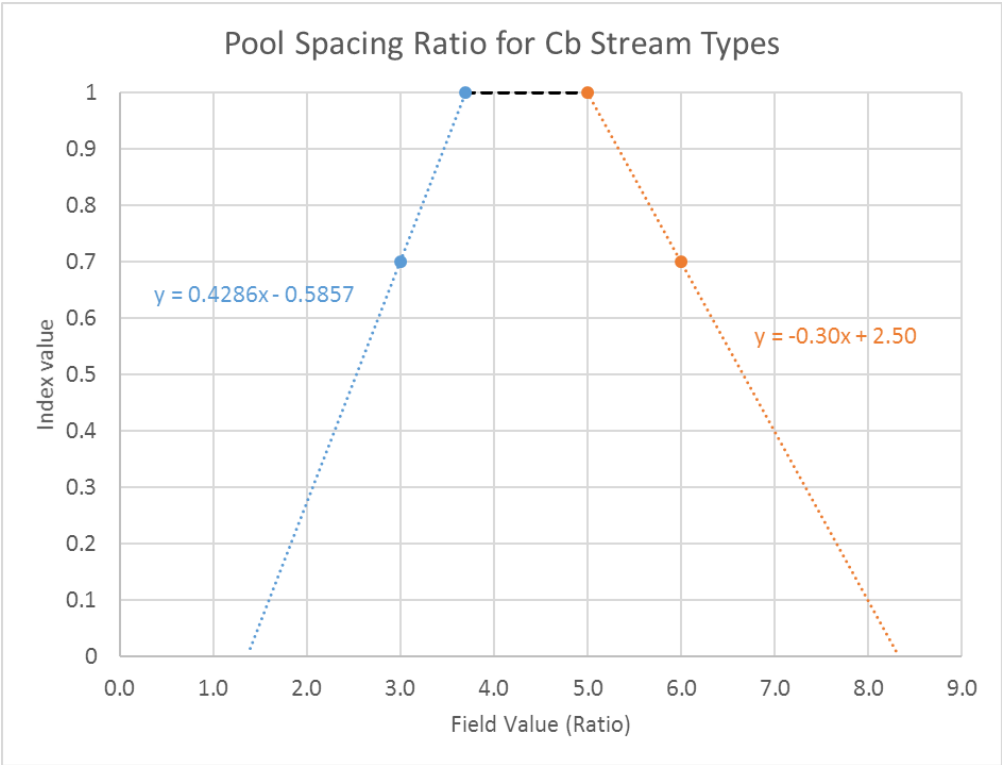


Figure 9-2c: Pool Spacing Ratio Reference Curves

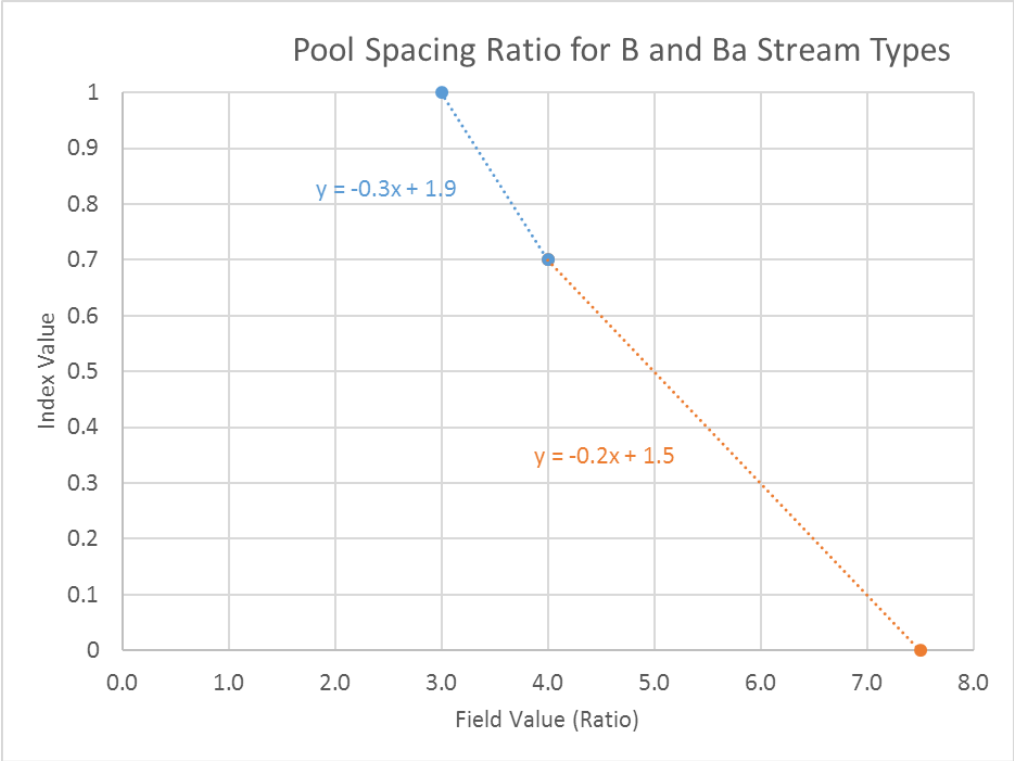


Figure 9-2d: Pool Spacing Ratio Reference Curves

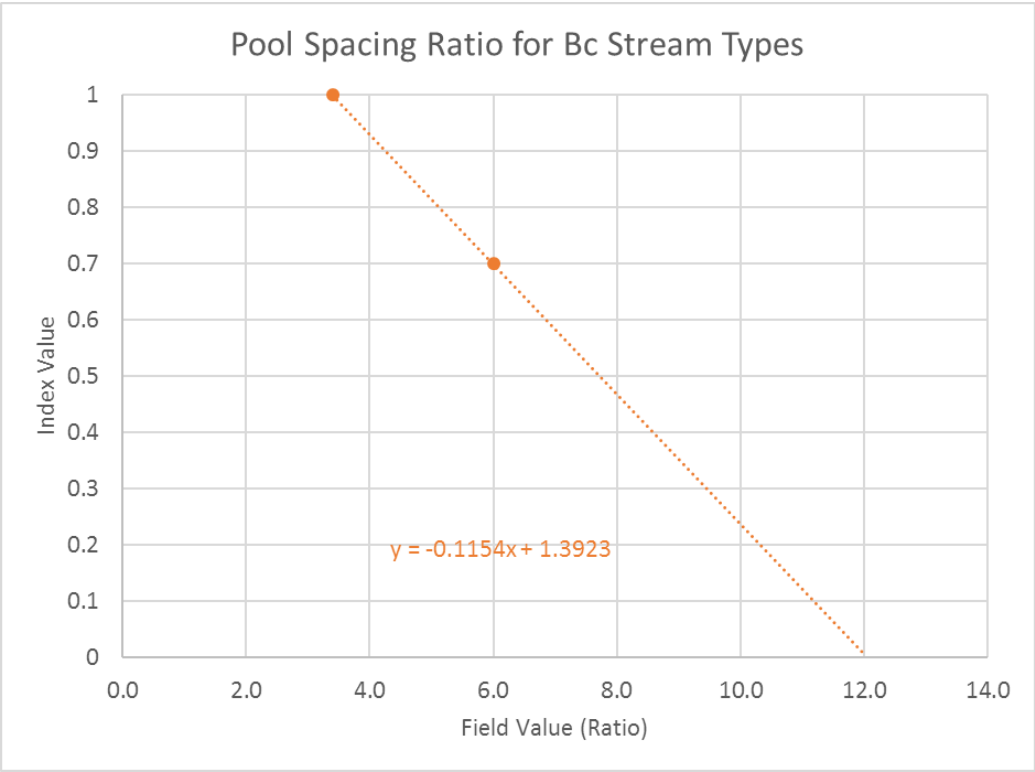
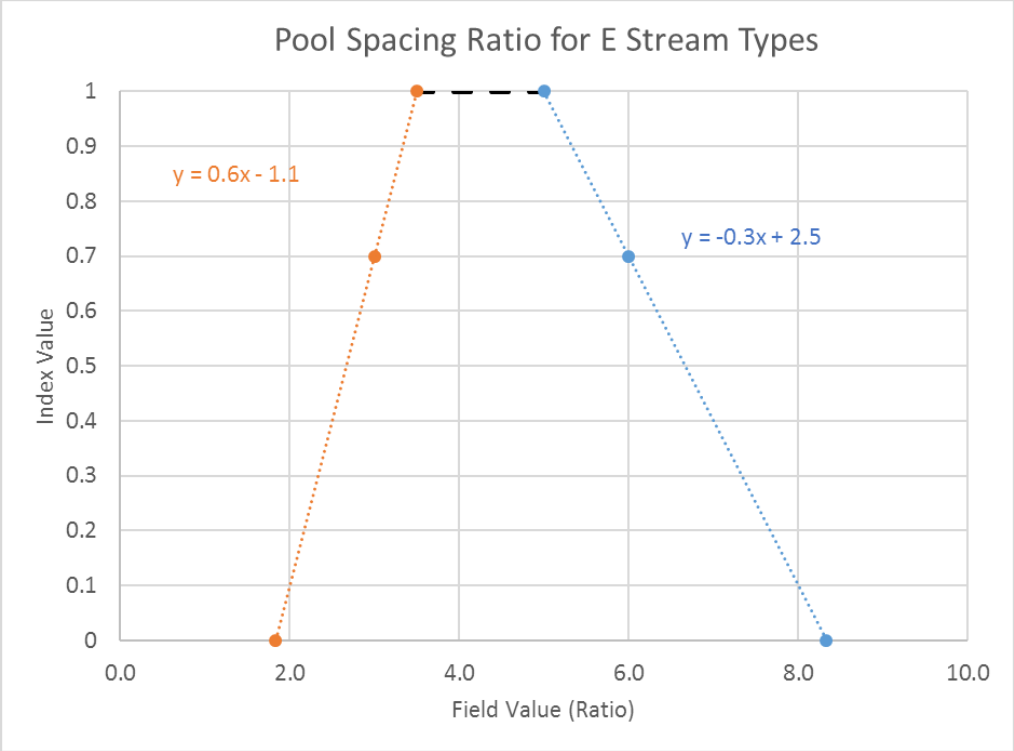


Figure 9-2e: Pool Spacing Ratio Reference Curves



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. 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 will not accurately represent bed forming processes. When possible, localized regional curves and flood frequency analysis should be used to verify the field indicators of bankfull. Information on verifying bankfull information is provided in the User Manual.

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 Kirkby 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 2013; Rosgen 2014) and described in the 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, guidance is included in the 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 Colorado.

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 Colorado. Additional data collection and analyses would be required to adapt the tool and reference curves for use with other classification approaches.

9.2. 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 CSQT. 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, as 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).

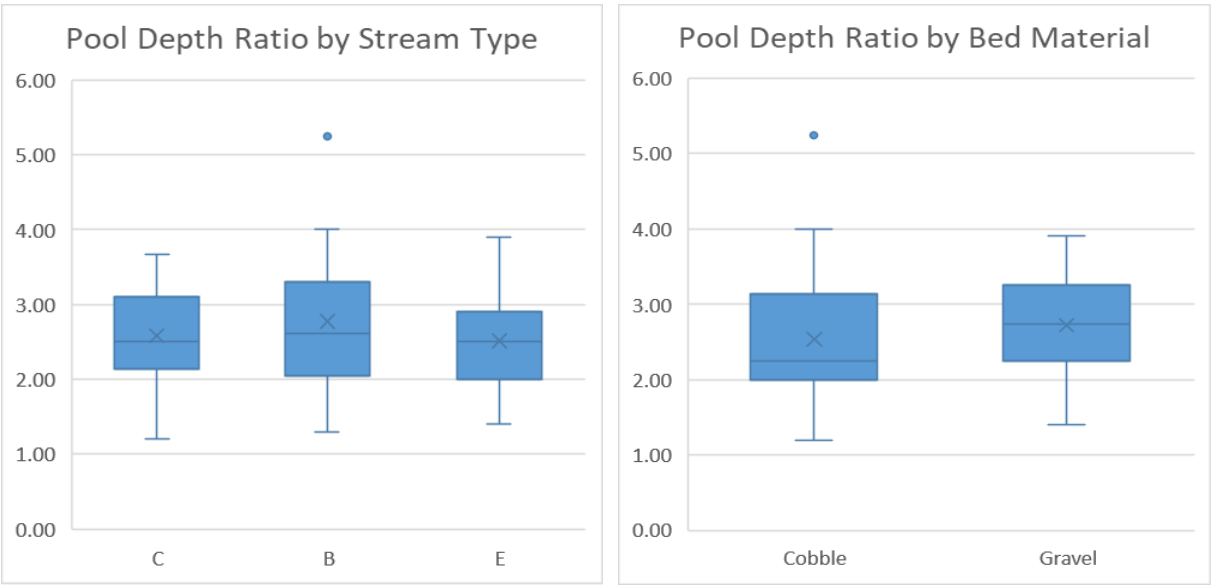
Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Reference curves were based on analysis of the WY geomorphic reference dataset described in Section 1.7. The WY 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 9.3) 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 9-3). 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 9-3: Box Plots for Pool Depth Ratio from the WY 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 WY geomorphic reference dataset are provided in Table 9-3.

Table 9-3: Statistics for Pool Depth Ratio from the WY Geomorphic Reference Dataset

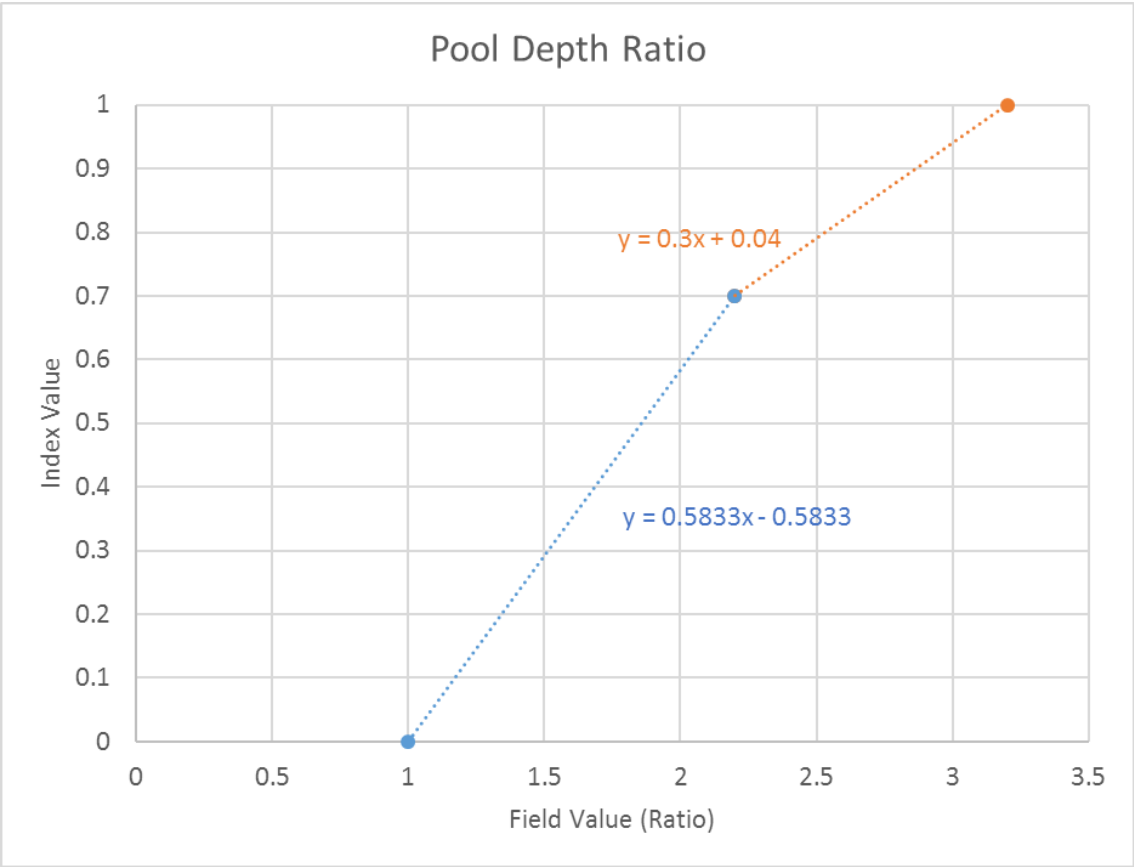
Statistic	Pool Depth Ratio
Number of Sites (n)	54
Average	2.7
Standard Deviation	0.8
Minimum	1.2
25th Percentile	2.2
Median	2.5
75th Percentile	3.2
Maximum	5.3

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 9-3 came from reference standard reaches, no threshold value was selected for the functioning-at-risk and not-functioning ranges. Threshold values are shown in Table 9-4. A broken linear relationship was fit to the identified threshold values to develop the reference curve (Figure 9-4).

Table 9-4: Threshold Values for Pool Depth Ratio

Index Value	Field Value
1.00	3.2
0.70	2.2
0.00	1.0

Figure 9-4: Pool Depth Ratio Reference Curves



Limitations and Data Gaps:

If bankfull dimensions are not accurately determined for a site, then the pool depth ratio will not accurately represent bed forming processes. When possible, localized regional curves and flood frequency analysis should be used to verify the field indicators of bankfull. Information on verifying bankfull information is provided in the User Manual.

The WY 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 Colorado, particularly in intermittent, ephemeral and braided systems. Sand bed streams may have lower pool depth ratios but should be evaluated using the CSQT recognizing that the current reference curves may not accurately characterize the level of functioning in these systems.

9.3. 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 this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Stratification was modified for the CSQT, removing a unique bioregion stratification that was applicable to Wyoming. Reference curves are based on analysis of the WY geomorphic reference dataset described in Section 1.7. 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 WY geomorphic reference dataset was assessed using various possible stratifications including bioregion, stream type, drainage area, slope, and bed material. Trends in percent riffle based on stream type, bed material, and drainage area were not observed in the data, though differences in percent riffle were observed in streams of different slope. A 3% slope break matches well with other literature showing that mountain streams with slopes greater than 3% often have a stair-like appearance (Chin 1989; Abrahams et al. 1995) and are riffle dominated. These trends matched professional experience of the WSTT and the results from the WY geomorphic reference dataset are shown in Table 9-5.

Note, the WSQT v1.0 included unique reference curves for the volcanic mountains and valleys bioregion of Wyoming. Streams in this bioregion had higher percent riffle values than the rest of the data. Based on this observation, the decision was made to develop specific reference curves for this bioregion in the WSQT. Because this ecoregion does not occur within Colorado, this reference curve and stratification were not included in the CSQT.

Table 9-5: Statistics for Percent Riffle from the WY Geomorphic Reference Dataset

Statistic	Percent Riffle (%)	
	Slope < 3%	Slope ≥ 3%
Number of Sites (n)	20	6
Geomean	52	72
Average	55	73
Standard Deviation	18	8
Minimum	28	60
25th Percentile	39	68
Median	57	74
75th Percentile	69	78
Maximum	88	83

Threshold values presented in Table 9-6 were developed using the stratification and data outlined above:

- For streams with low slope (< 3%), the functioning range of scoring was set equal to the interquartile range observed in the WY geomorphic reference 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 best professional judgement was used 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 WY geomorphic reference dataset.
- Since the WY 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 9-5).

Table 9-6: Threshold Values for Percent Riffle

Index Value	Field Value (%)	
	Slope < 3%	Slope ≥ 3%
1.00	50 – 60	68 – 78
0.70	39, 69	60, 83

Figure 9-5a: Percent Riffle Reference Curves

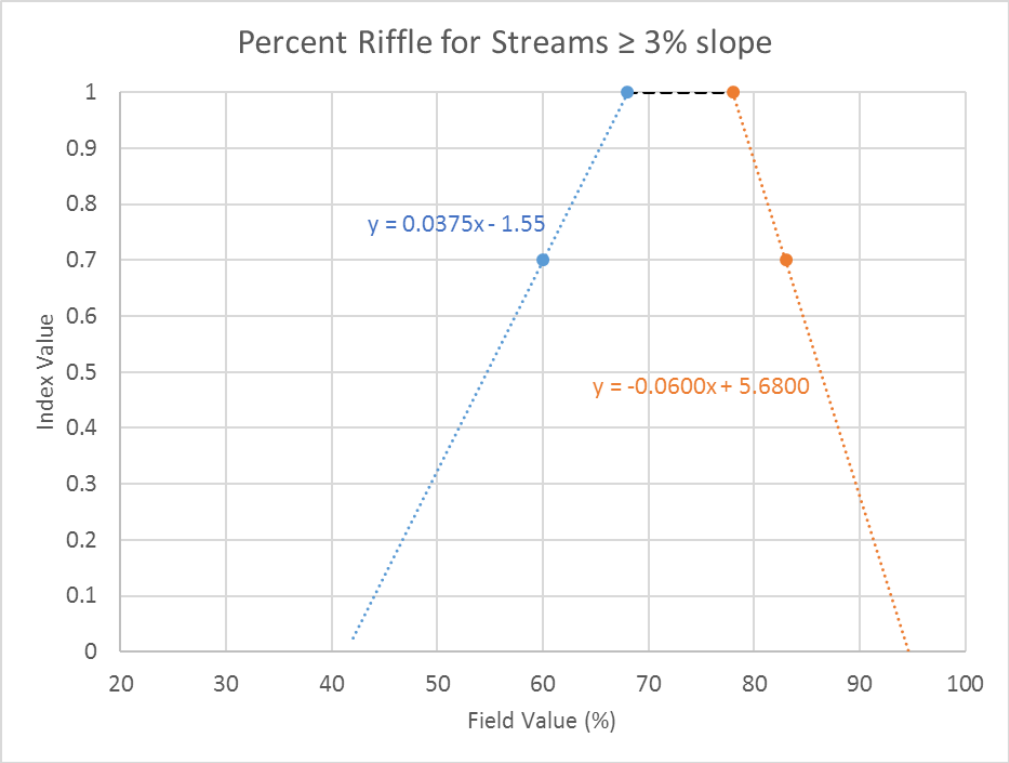
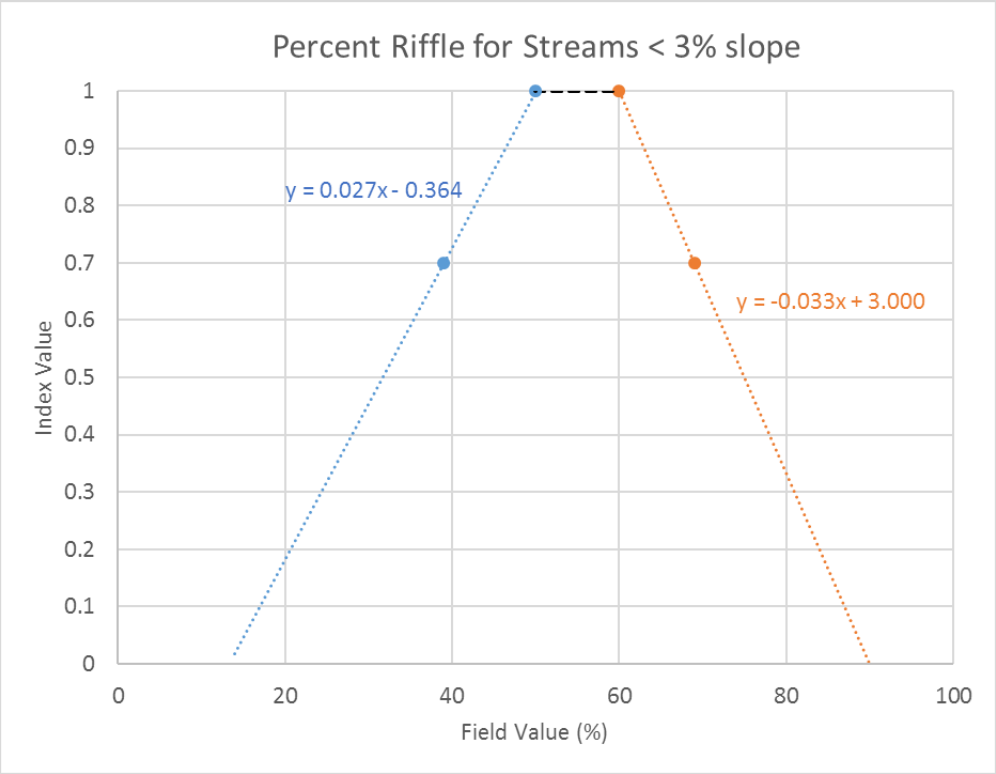


Figure 9-5b: Percent Riffle Reference Curves



Limitations and Data Gaps:

The WY 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 and further stratification will be necessary for Colorado.

9.4. Aggradation Ratio

Summary:

The aggradation ratio is an application of the riffle width depth ratio (WDR) compared to a reference WDR to assess the degree of aggradation in the project reach. The WDR is the bankfull riffle width divided by the mean depth (Rosgen 2014). Mean depth is the riffle bankfull cross sectional area divided by the riffle bankfull width. Within the assessment segment, each riffle exhibiting signs of excessive deposition should be surveyed and the WDR calculated. The aggradation ratio is the WDR measured at the widest riffle in the sampling reach.

Deposition of sediments within a channel is a natural fluvial process, but excessive aggradation can be an indicator of sediment imbalance, where sediment supply exceeds the stream's transport capacity. Accumulation of sediments in pools would result in a lower pool depth ratio, which is captured in the bedform diversity parameter. Similarly, accumulations of sediment in a riffle (e.g. forming a mid-channel bar) would yield a higher WDR than would be expected from a stable riffle.

The aggradation ratio was developed based on the Width Depth Ratio State (WDRS) described by Rosgen (2014) to assess departure from a reference condition caused by streambank erosion, excessive deposition, or direct mechanical impacts that lead to an over-wide channel. The WDRS method assesses increases and decreases in WDR to show departure from reference. Increasing WDRs represent aggradation risk and decreasing WDRs represent degradation risk. Degradation risk is assessed in the CSQT using the BHR. Since this metric is meant to only assess aggradation, only the increasing WDR method is used.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Reference curves were developed using the stability ratings provided in WDRS, which show the ratio of an observed WDR over an expected WDR. The expected WDR should come from reference reach streams of the same stream type as the proposed or design stream type. In addition, hydraulic and sediment transport models, such as Torizzo an Pitlick (2004), may be used to select a channel dimension (along with slope) that yields a stable WDR. The channel stability descriptions presented by Rosgen (2014) in the WDRS are provided in Table 9-7. The stability ratings are calculated by dividing the observed WDR by the reference WDR. The stable range is 1.0 to 1.2 meaning that observed WDRs are 100% to 120% of reference. As the ratio increases, the risk of aggradation increases. When the value exceeds 140% of reference condition, the channel is likely to be unstable due to aggradation. Based on these values, thresholds for the aggradation ratio were developed and are shown in Table 9-8 and the curve is shown in Figure 9-6.

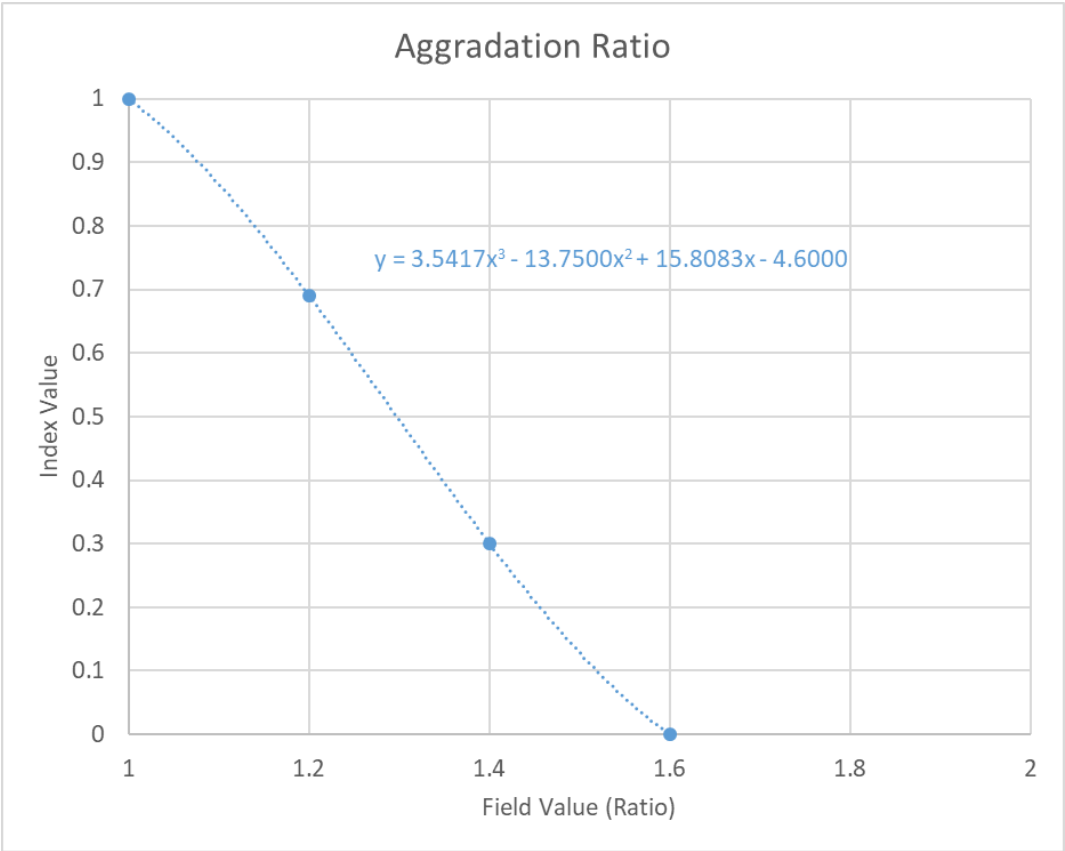
Table 9-7: Width Depth Ratio State Categories (Rosgen 2014)

Width Depth Ratio State	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 9-8: 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

Figure 9-6: Aggradation Ratio Reference Curve



Limitations and Data Gaps:

If bankfull dimensions are not accurately determined for a site, then the aggradation ratio will not accurately represent bed forming processes. When possible, localized regional curves and flood frequency analysis should be used to verify the field indicators of bankfull. Information on verifying bankfull information is provided in the User Manual.

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, 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.

Chapter 10. Plan Form Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Channel pattern or plan form 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 plan form 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 affecting 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. For the CSQT, sinuosity is measured for each project reach as the entire reach length divided by the valley length. Sinuosity can be measured in each of the classifications listed above but is mostly used in single-thread meandering streams.

Metric:

- Sinuosity

10.1. 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 plan form 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 this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

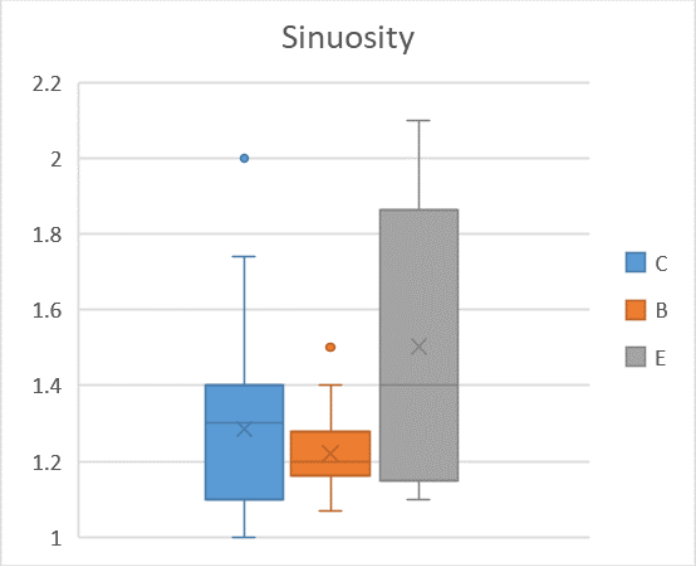
Reference curves were based on analysis of the WY geomorphic reference dataset and NRSA dataset described in Section 1.7. These datasets were evaluated for potential stratification by stream type and valley type, as the sinuosity of a stream varies by the shape and slope of the valley.

The WY geomorphic reference dataset consists of 56 sites that report sinuosity. This dataset was stratified by Rosgen stream type (Table 10-1 and Figure 10-1). 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 10-1: Statistics for Sinuosity from the WY 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 10-1: Box Plots for Sinuosity from the WY Geomorphic Reference Dataset

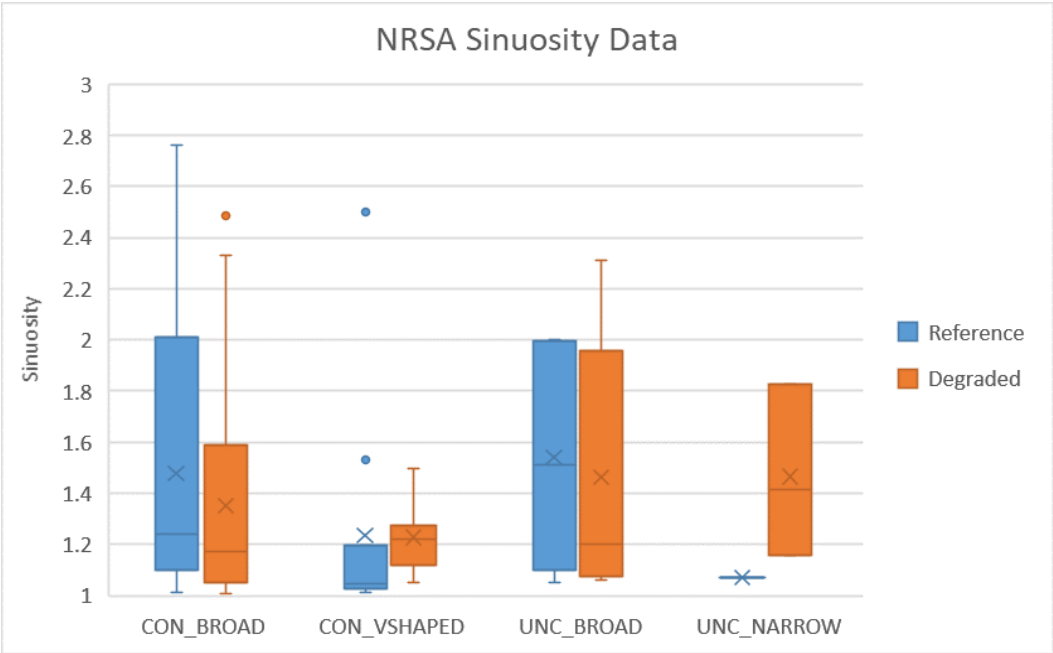


Analysis using the NRSA dataset was performed on 156 sites from Colorado, Wyoming, and the surrounding states. Sites with a single thread channel pattern (attribute CONPATTERN) were selected and identified as reference or degraded (attribute RT_NRSA). The dataset identifies the channel constraint type (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 10-2 and Figure 10-2. Reference designations were made according to the RT_NRSA attribute. Sites with sinuosity values greater than 3 were considered outliers and removed from the dataset.

Table 10-2: Statistics for Sinuosity from the NRSA Dataset

Statistic	Sinuosity by Valley Constraint and Condition							
	CON_BROAD		CON_VSHAPED		UNC_BROAD		UNC_NARROW	
	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 10-2: Box Plots for Sinuosity from the NRSA Dataset



The valley types described in the NRSA dataset were related to the SQT 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.

Except for highly sinuous channels (E stream types), the WSQT v1.0 and CSQT Beta Version stratify sinuosity reference curves by valley type. Further 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 stratifying factor, but there were not enough data points to determine meaningful differences when data were stratified by valley type and ecoregion.

The threshold values are shown in Table 10-3. 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, a 1.00 was assigned 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 not-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 10-2). Since natural valley constraints can limit sinuosity in these settings, the reference curve is one sided rather than bell shaped.
- Sinuosity values greater than or equal to 1.2 were 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 WY 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 10-3 to create the reference curves shown in Figure 10-3.

Table 10-3: 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
0.69	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 10-3a: Sinuosity Reference Curves

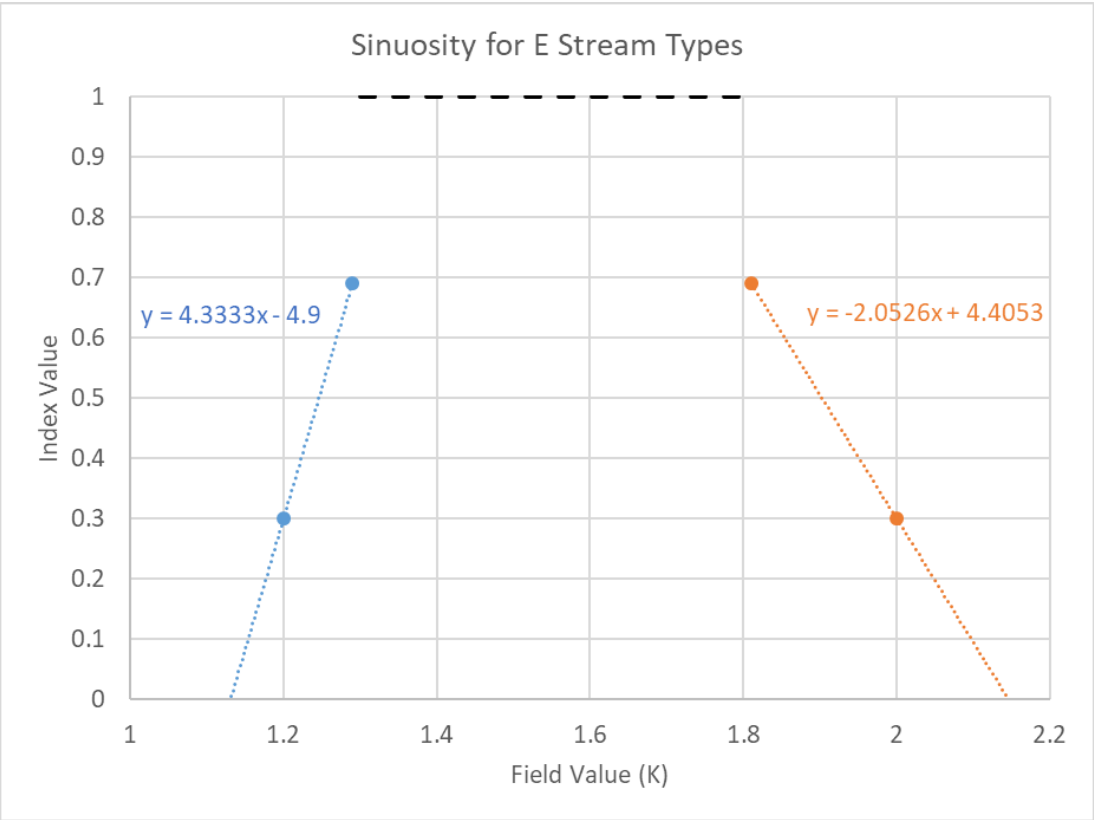


Figure 10-3b: Sinuosity Reference Curves

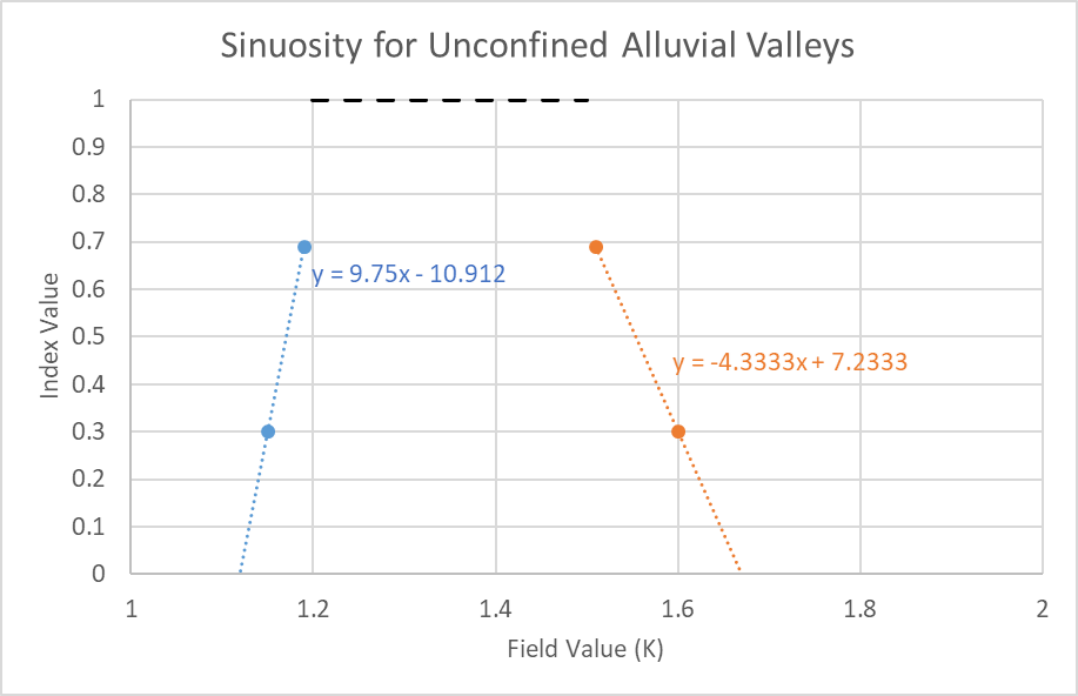


Figure 10-3c: Sinuosity Reference Curves

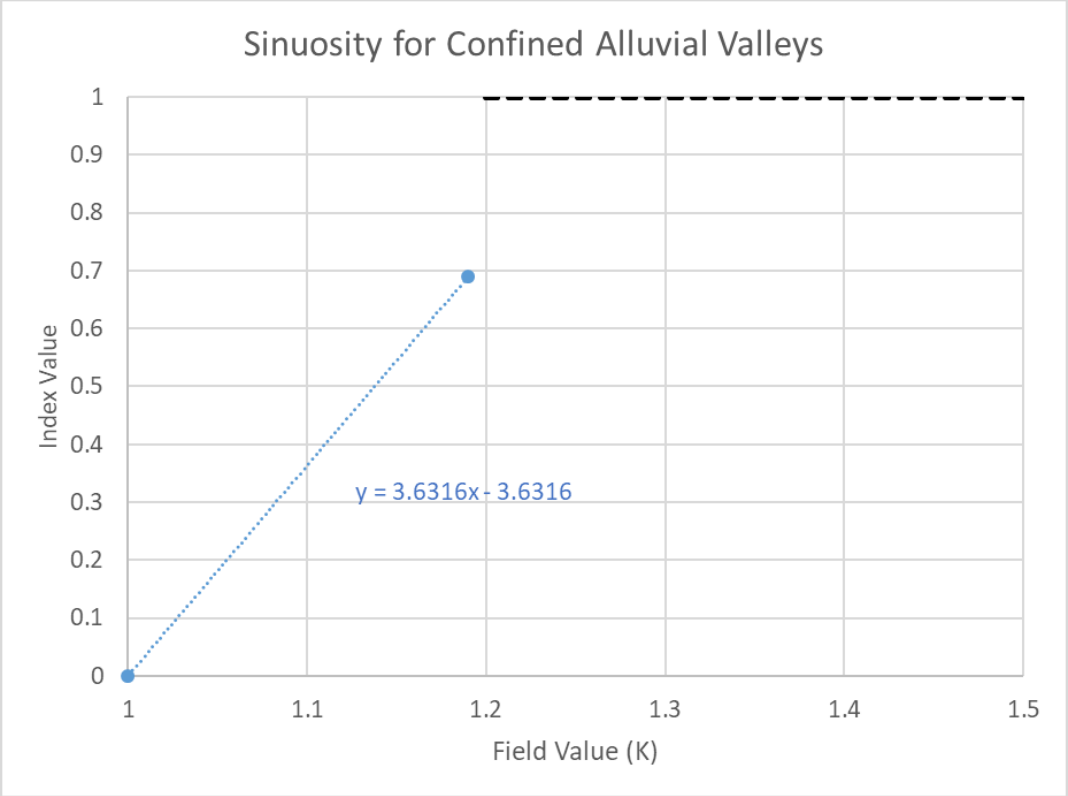
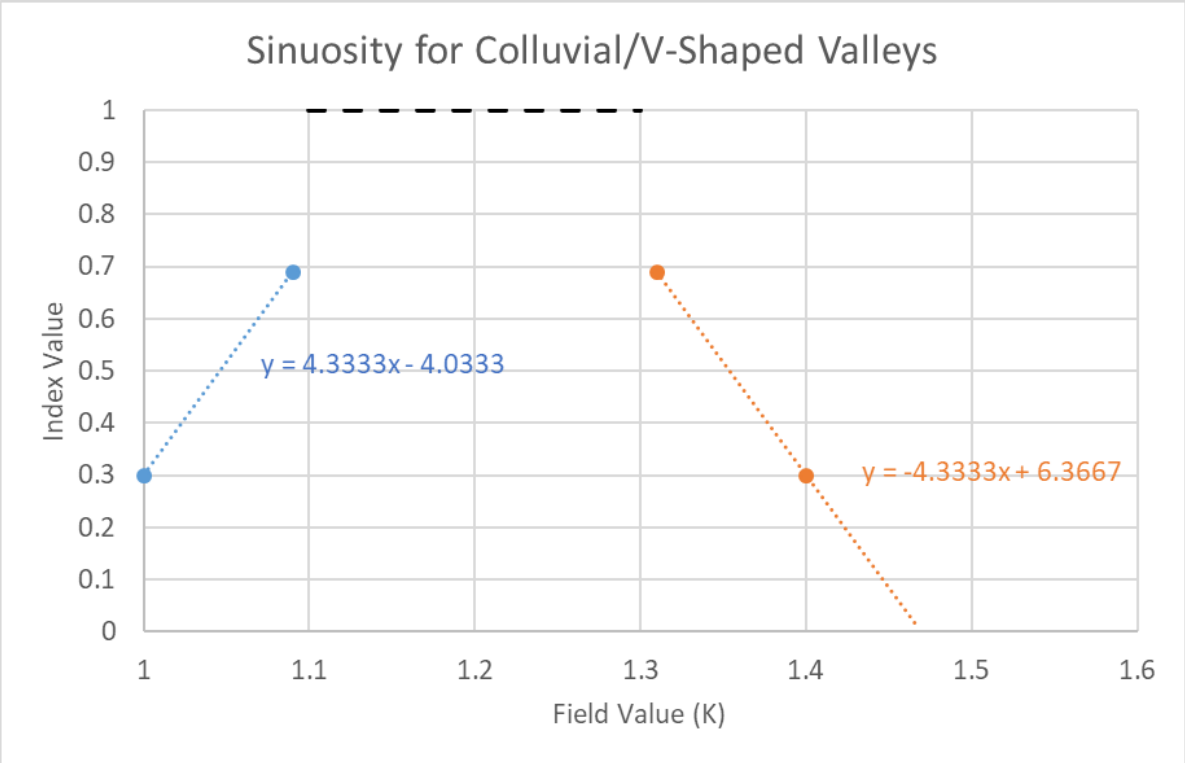


Figure 10-3d: Sinuosity Reference Curves



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

In this methodology, 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 these reference curves should be used. See Section 1.7 and Section 9.1 for additional discussion.

To improve repeatability and better align the sinuosity metric with restoration activities, sinuosity should be measured for the length of the project reach. Typical sinuosity recommendations, and other SQT versions, recommend 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. Instructions on how to measure sinuosity are provided in the User Manual.

Plan form metrics have yet to be developed for braided and anastomosing systems, 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 11. Riparian Vegetation Parameter

Functional Category: Geomorphology

Function-based Parameter Summary:

Riparian vegetation is a critical component of stream ecosystem structure and function. 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, as well as 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

The four metrics from the WSQT v1.0 riparian vegetation parameter are also used in the CSQT Beta Version: 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 CSQT field value per metric, which is standard among other methods.

Absolute woody vegetation cover is an important metric as woody vegetation is a primary component of western riparian systems that provides important structural and biophysical functions. Stem density is an additional woody vegetation metric that is not included in the CSQT Beta Version but 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.

The riparian width metric evaluates the extent of hydrophytic species and/or more vigorous vegetation growth and is an indicator of the extent of hydrologic connectivity. Hydrophytic vegetation may be considered as its own metric in future versions of the tool, because 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 are being considered. Hydrophytic vegetation data can be obtained via the data collection methods in the CSQT Beta Version, allowing the CSQT SC to evaluate how this metric could be developed and applied in the future.

Availability of species-level data from the CNHP dataset (Kittel et al. 1999; Section 1.7) provided the opportunity to align methods with existing protocols required by the Corps for wetland delineations (USACE 1987, USACE 2008, USACE 2010a, USACE 2010b). 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 and requires plant identification expertise. Corps field staff and many practitioners are already familiar with these methods, and wetland delineations will likely be conducted where stream projects also contain wetlands in the project area. Data from the CNHP dataset was used to evaluate the cover metrics and develop reference curves for the WSQT v1.0 and CSQT Beta Version.

Metrics:

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

11.1. Riparian Width

Summary:

The riparian width metric, developed specifically for the Wyoming and Colorado SQTs, 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 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. 2005). 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 the following process domains: 1) steeper gradients yield narrower riparian areas, while lower gradients yield broader floodplains; 2) riparian areas in confined valleys are constrained by hillslope processes while riparian zones in unconfined valleys are defined more by valley processes (e.g., microtopography, groundwater movements, etc.); and 3) 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 western states, 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 recommends 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 meander width ratio is used to determine expected riparian width.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

This metric was developed by the Wyoming Stream Technical Team specifically to replace fixed buffer width approaches included in other SQTs (e.g., Harman and Jones 2017, TDEC 2018). Limited data and peer reviewed literature are 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. The reference curves and thresholds are intended to encourage and incentivize restoration activities that restore floodplain connectivity or remove stressors and human land uses from the riparian zone.

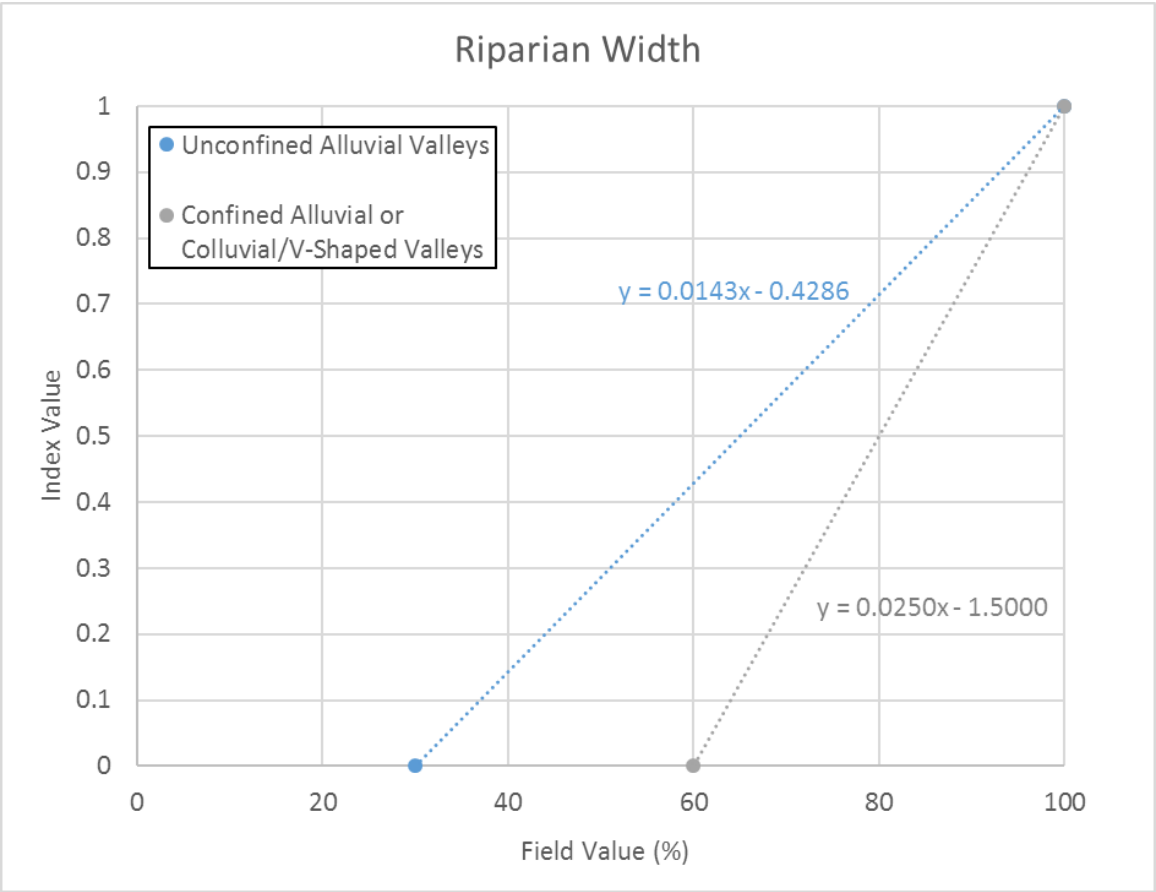
Stratification of reference curves took into consideration how hydrologic and geomorphic processes drive riparian zone development. Merritt et al. (2017) recommends stratifying riparian areas by valley type using a Hydrogeomorphic Valley Classification framework, which identifies nine valley types, but 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 type, recognizing the differences in hillslope and valley bottom processes that influence riparian extent in confined and unconfined valleys (Table 11-1).

Once stratified into valley types, the reference curves take into account the influence of potential stressors in the floodplain or adjacent stream area and changes to the hydrologic regime on 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 availability (Fischer and Fischenich 2000; Sweeney 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 11-1: 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 11-1: Riparian Width Reference Curves



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

Because this is a new metric 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. This metric would benefit from additional validation, review and refinement as the tool is applied.

Beta testing in Wyoming 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. The 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.

11.2. Woody Vegetation Cover

Summary:

Riparian areas in the western U.S. 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, riparian communities are often comprised of a combination of hydrophytic and upland species. Kittel et al. (1999) describe nearly 120 riparian woody plant assemblages in Colorado, including coniferous, deciduous and mixed forests, and shrubland 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 CSQT. 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. The field value for this metric reflects the sum of the absolute cover of all woody species and can be greater than 100% cover. Methods are outlined in the User Manual.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Data collection methods for this metric 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. The reference curves were developed using the CNHP dataset and a small data collection effort in Wyoming.

Colorado Natural Heritage Program: Woody vegetation cover values were calculated for woody sites in the CNHP dataset, described in Section 1.7. 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 11-2). Sample sizes were limited, particularly for degraded sites and for all sites within the plains and tablelands (plains) ecoregions.

Table 11-2: Statistics for Woody Vegetation Cover from the CNHP Dataset

Statistic	Woody Vegetation Cover (%) by Ecoregion and Condition						
	Mountains		Basins & Plateaus		Mountains, Basins & Plateaus	Plains & Tablelands	
	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 basin and plateau ecoregions (basins), the CNHP cover values were substantially higher than the other ecoregions. Cover values may be higher because the data collection within the xeric ecoregions (basins) 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 on the west slope of Colorado. 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 11-3. Note, these data reflect cover values by lifeform, and thus are lower than absolute cover value by species.

Table 11-3: 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 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 were 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. Where sufficient data were not available, this threshold could not be identified and values within these index ranges were 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 combined mountain, basins, and plateaus CNHP ecoregion dataset, the 75th percentile value (122% cover) 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 this dataset was 69%, and was used as the threshold value between functioning and functioning-at-risk index values.

We did not identify the break between not-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, due to the natural variability in woody riparian ecosystems we 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 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 microtopography 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 aquatic ecosystem structure and function 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 these ecoregions.

The combined CNHP dataset for the plains and tablelands 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, shrubs that tolerate a broader range of environmental factors and land uses, and/or thick stands of non-native tamarisk (Kittel et al. 1999; 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, a two-sided reference curve is used to capture 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 the 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 (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 not-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 11-4: 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 11-2a: Woody Vegetation Cover Reference Curves

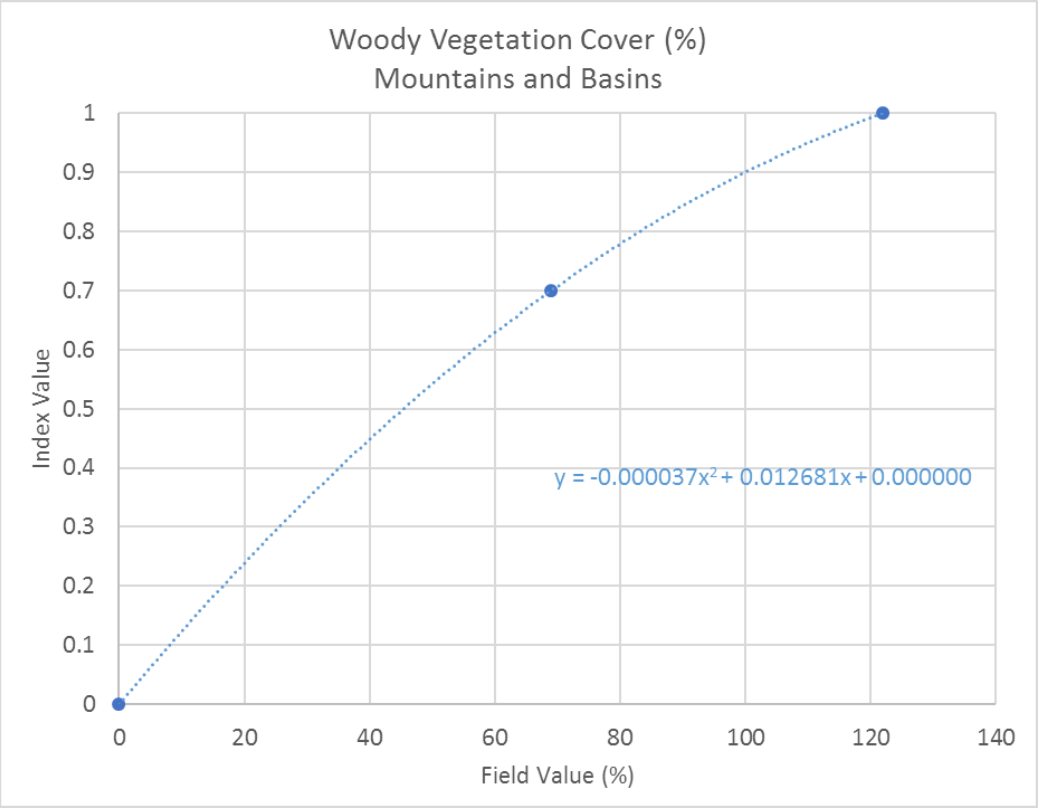
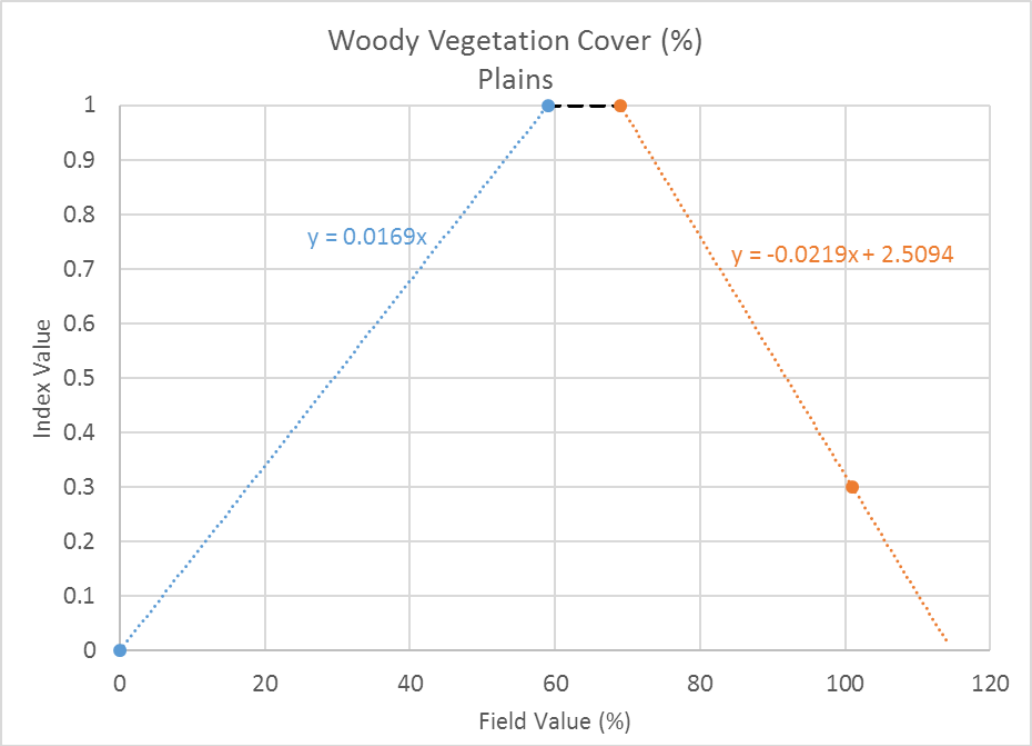


Figure 11-2b: Woody Vegetation Cover Reference Curves



Limitations and Data Gaps:

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 would benefit from additional field data to validate the criteria and curves identified above. Additional data would also allow for us to consider whether additional stratification or refinement beyond the three ecoregions could occur. For example, there was a broad range in cover values across reference standard conditions. 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.

This 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 Colorado. 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 regimes. 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.

11.3. Herbaceous Vegetation Cover

Summary:

While riparian areas in Colorado are predominately characterized by a woody canopy (as noted above), a ground layer of herbaceous vegetation is often also present. Herbaceous species are an important component of the riparian community, as they often provide 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. Kittel et al. (1999) describes over 30 riparian plant assemblages in Colorado that are predominantly herbaceous vegetation.

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 et al. 2017). Riparian areas dominated by these species can perpetuate degraded conditions.

It is important to include herbaceous vegetation in the CSQT 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 a 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 User Manual.

Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

Data collection methods for this metric align 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. The reference curves were developed using the CNHP dataset, the Northern Rocky Mountain Hydrogeomorphic Manual (Hauer et al. 2002), and a small data collection effort in Wyoming.

Colorado Natural Heritage Program: Herbaceous vegetation cover values were calculated for sites in the CNHP dataset, described in Section 1.7. 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, 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 11-5).

Table 11-5: Statistics for Herbaceous Vegetation Cover from the CNHP Dataset

Statistic	Herbaceous Vegetation Cover (%) by Reference Community Type and Condition			
	Forested or Scrub-Shrub		Herbaceous	
	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 11-6. 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 reference curves in the HGM manual were all linear.

Table 11-6: 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 11-7. Note, these data reflect cover values by lifeform, and thus are lower than absolute cover value by species.

Table 11-7: Herbaceous Vegetation Cover at Sites Visited by the WSTT

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

Analysis: Best professional judgement was used to interpret the datasets to develop threshold values for this metric (Table 11-8). 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 were used to determine the break between functioning and the functioning-at-risk condition.
- Due to small sample sizes, the threshold between functioning-at-risk and not-functioning was not identified a priori and values within these index ranges could be determined from the reference curve. Therefore, 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 not-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, more conservative reference curves were developed to consider 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 11-8

are consistent with the 1.0 variable sub-index scores identified by Hauer (2002) for woody reference community types (Table 11-6).

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.

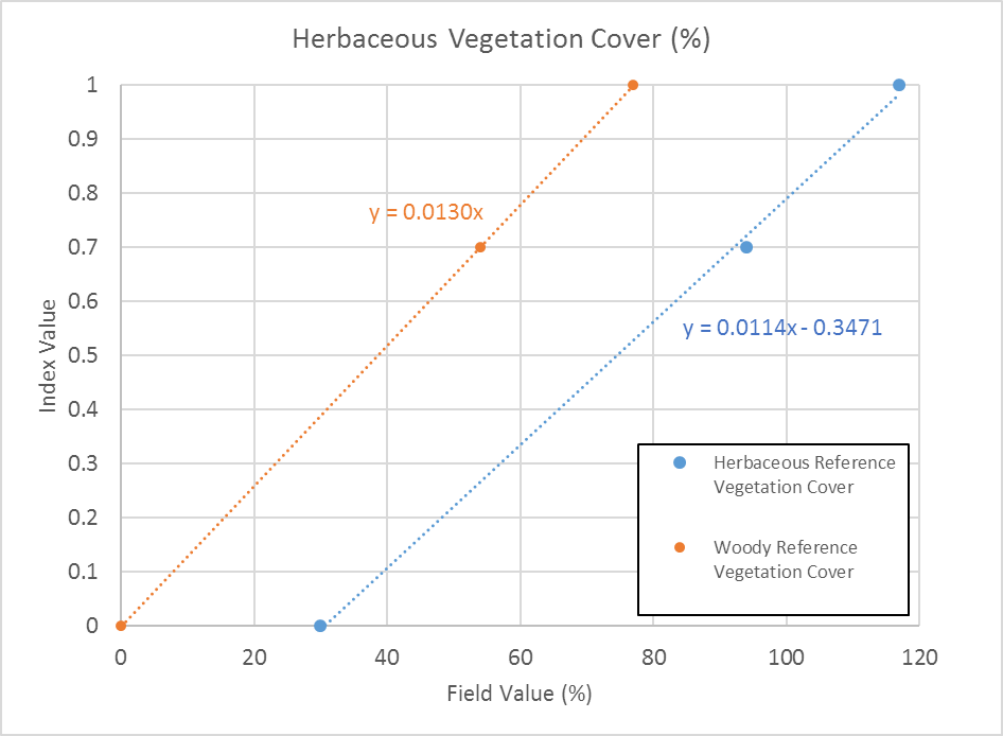
For herbaceous vegetation in herbaceous community types, the 75th percentile of the reference data (117%) was used to determine the maximum index value. The median of the reference data (94%) was used to determine the break between functioning and functioning-at-risk. We did not identify the break between not-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 (2002) 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 sub-index scores.”

Table 11-8: Threshold Values for Herbaceous Vegetation Cover

Index Value	Field Value (%)	
	Forested or Scrub-Shrub Reference Vegetation Cover Type	Herbaceous Reference Vegetation Cover Type
1.00	≥ 77	≥ 117
0.70	54	94
0.00	0	30

Figure 11-3: Herbaceous Vegetation Cover Reference Curves



Limitations and Data Gaps:

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 would benefit from additional field 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 occurs across stream sizes, elevations, soil types or between different target herbaceous community compositions.

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 regime. We believe this metric should still be applied in ephemeral stream systems but would expect they may have greater natural variability in what is considered reference standard condition and/or may score lower than their perennial counterparts.

11.4. 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in Colorado.

The data collection methods for this metric 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. The reference curves were developed using the CNHP dataset and a small data collection effort in Wyoming.

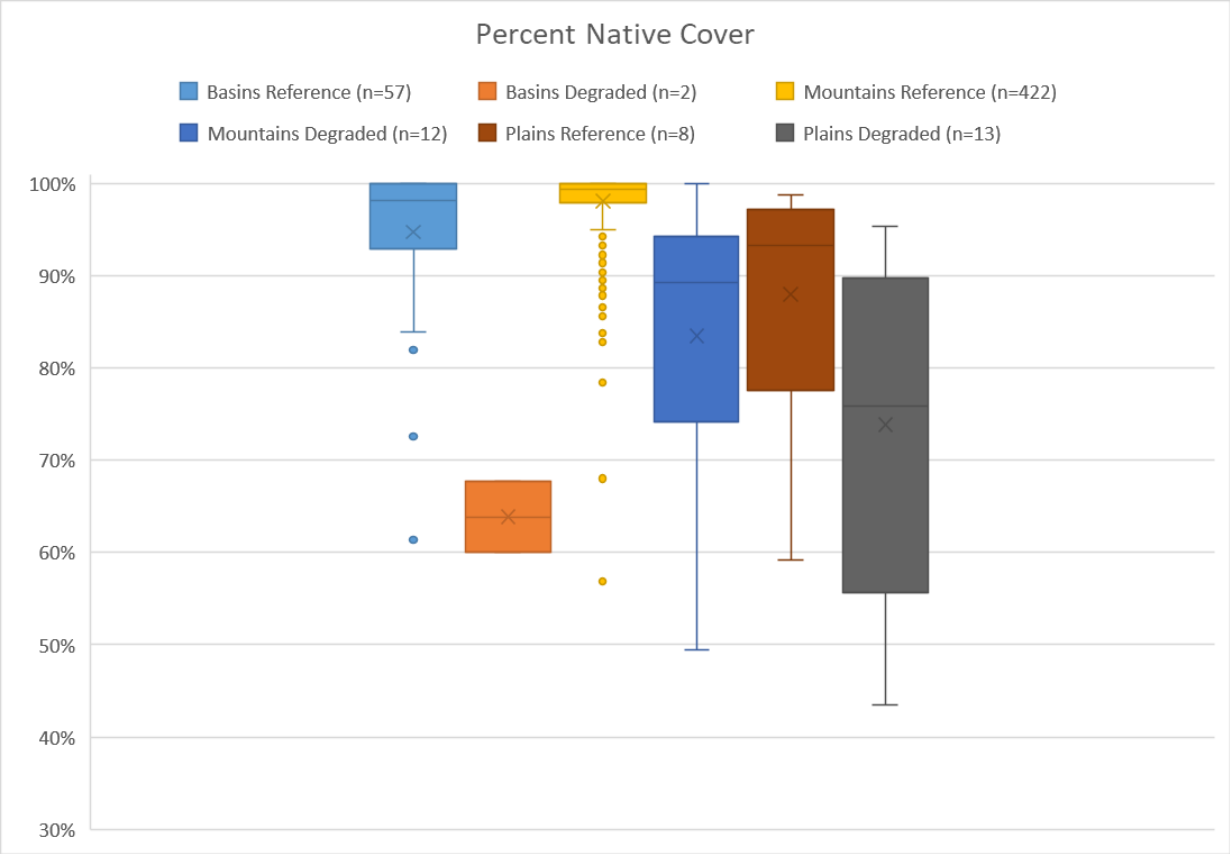
Colorado Natural Heritage Program: Percent native cover values were calculated from sites in the CNHP dataset, described in Section 1.7. Percent native cover was calculated by summing the absolute cover values for all native species and dividing by the total absolute cover value for a site. Statistics were derived from the CNHP dataset for the reference standard and degraded sites (Table 11-9). Sample sizes were limited, particularly for degraded sites and for all sites within the plains and tablelands (plains) ecoregions.

Table 11-9: 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 11-4), 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 of 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, this metric is not stratified by ecoregion or reference community type.

Figure 11-4: 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 11-10.

Table 11-10: 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 11-11.

- The 75th percentile of reference standard sites was used to determine the maximum index value of 1.00.
- The 75th percentile values from degraded sites were used to determine the threshold between the 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.

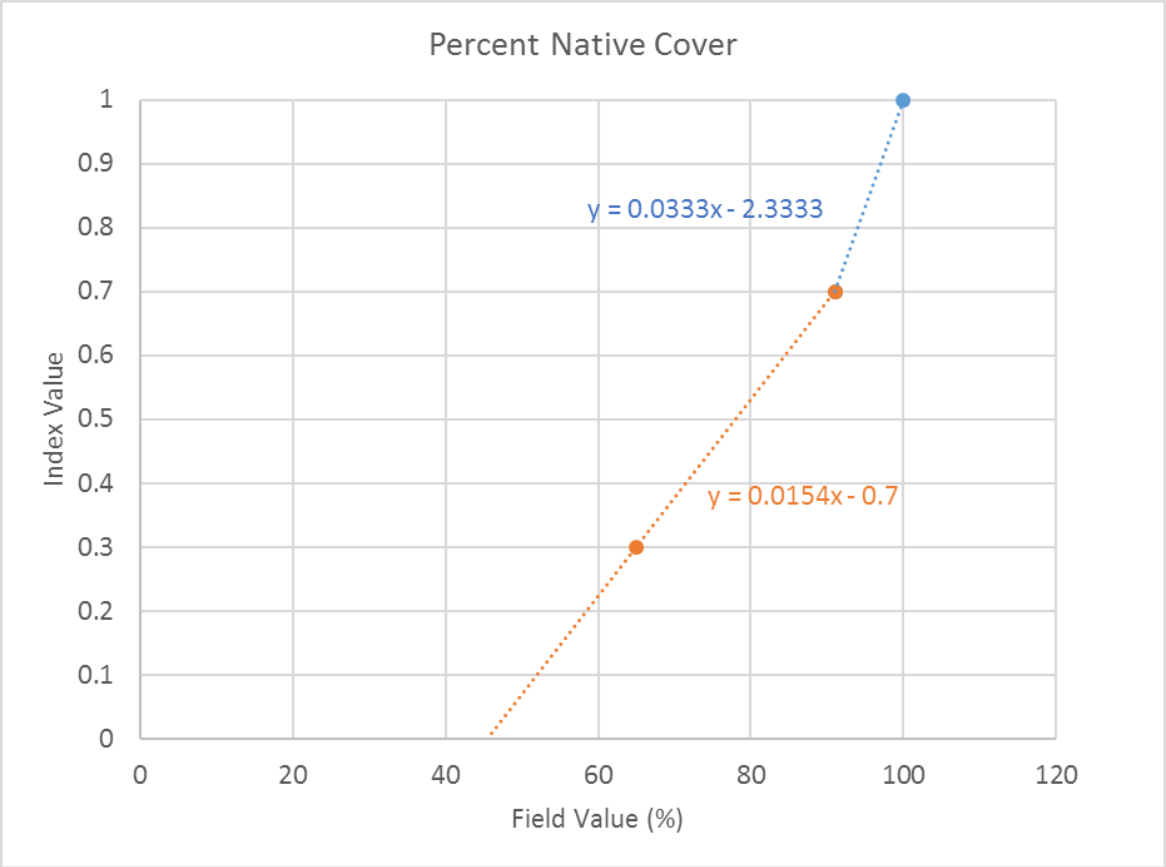
A broken linear curve was used to fit the threshold values. The minimum index value extrapolated from the curve was 46% native cover, which aligns with the 5th percentile from the degraded dataset (48%) and is thus a reasonable minimum value.

Data collected 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 it was considered reasonable to characterize these sites as functioning or (high) functioning-at-risk.

Table 11-11: Threshold Values for Percent Native Cover

Index value	Field Value
1.00	100
0.70	91
0.30	65

Figure 11-5: Percent Native Cover Reference Curves



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

The reference curve development for the CSQT would benefit from additional field 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 Colorado. 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 regime. 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.

Chapter 12. 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).

The metrics included in the CSQT Beta Version are the summer maximum weekly average temperature (MWAT) and daily maximum (DM) temperature. A winter MWAT was also considered to include in the CSQT in order to protect egg incubation in the river over-winter and ovary development for spring spawners and may be included in future versions of the tool.

Metrics:

- Daily Maximum Temperature (°C)
- Maximum Weekly Average Temperature (MWAT) (°C)

12.1. Daily Maximum Temperature

Summary:

As defined by state regulation 31 (5 CCR 1002-31) the daily maximum (DM) temperature is the highest two-hour average water temperature recorded during a given 24-hour period. The daily maximum temperature is determined based on monitoring throughout the summer months of July and August with a maximum sampling interval of 30-minutes. 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' (USEPA 2014) or USFS's 'Measuring Stream Temperature with Digital Data Loggers: A User's Field Guide' (Dunham et al. 2005).

The DM is the acute temperature criterion for waterbodies in Colorado implemented to help prevent temperature changes that are deleterious to the resident aquatic life. Temperatures that exceed this threshold may be lethal to aquatic species (CDPHE 2011).

Reference Curve Development:

The threshold values for the DM metric is based on the acute temperature criterion defined in state regulation 31 (5 CCR 1002-31) which is derived from laboratory experiments determining lethal temperatures for fish acclimated to average summer temperatures. The DM temperature standards are provided in Table 12-1, stratified by the regulatory temperature tiers. Temperature tiers are first identified as cold stream (CS) or warm stream (WS) then numbered by increasing temperature. There are two CS-I tiers, with more protective standards set for streams where mountain whitefish (MWF) spawn.

Table 12-1: DM Standards for Colorado Streams

Temperature Tier Code	MWAT (°C)
CS-I $_{MWF}$	21.2
CS-I	21.7
CS-II	23.9
WS-I	29.0
WS-II	28.6
WS-III	31.8

The index value for DM is binary and defined using the DM temperature standards shown in Table 12-1. DM temperatures that exceed the acute criteria are considered not-functioning and score a 0.00 in the CSQT Beta Version while DM temperatures that are less than the acute criteria score a 1.00. As long as temperatures do not exceed the DM, average temperatures are better at detecting thermal impacts to fish, which is captured by the other temperature metric in the CSQT Beta Version, the MWAT (Imholt et al. 2011). No reference curve was developed since the scoring is binary.

Table 12-2: Threshold Values for DM

Index Value	Field Values (^{o}C) by Temperature Tier Code					
	CS-I $_{MWF}$	CS-I	CS-II	WS-I	WS-II	WS-III
1.00	< 21.2	< 21.7	< 23.9	< 29.0	< 28.6	< 31.8
0.00	≥ 21.2	≥ 21.7	≥ 23.9	≥ 29.0	≥ 28.6	≥ 31.8

Limitations and Data Gaps:

The DM metric relies on the user to select an appropriate temperature tier. All streams in Colorado have assigned temperature tiers (see Regulations 32 – 38) but the assigned tier may not be appropriate for a particular project reach. In selecting the appropriate temperature tier, users should also consider the most thermally-sensitive species expected to occupy the reach in summer once restoration is complete.

12.2. Maximum Weekly Average Temperature (MWAT)

Summary:

As defined by state regulation 31 (5 CCR 1002-31) the Maximum Weekly Average Temperature (MWAT) is the largest weekly average temperature in the period of interest. The temperature is monitored throughout the summer months of July and August and the weekly average temperature is the average of daily average temperatures over a seven-day consecutive period.[3] 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' (USEPA 2014) or USFS's 'Measuring Stream Temperature with Digital Data Loggers: A User's Field Guide' (Dunham et al. 2005).

The MWAT is the chronic temperature criterion for waterbodies in Colorado implemented to help prevent temperature changes that are deleterious to the resident aquatic life. 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.

Reference Curve Development:

Reference curves were derived from the chronic temperature criterion defined in state regulation 31 (5 CCR 1002-31) and the optimum temperatures for growth and swim speed of representative species within each temperature tier. The MWAT temperature standards for the summer season are provided in Table 12-3, stratified by the regulatory temperature tiers. Temperature tiers are first identified as cold stream (CS) or warm stream (WS) then numbered by increasing temperature (ex. CS-II). There are two CS-I tiers, with more protective standards set for streams where mountain whitefish (MWF) spawn.

[3] The sampling interval for the MWAT requires a minimum of three data points spaced equally through each day but the sampling interval for the daily maximum temperature metric is 30-minutes.

Table 12-3: MWAT Standards for Colorado Streams

Temperature Tier Code	MWAT (°C)
CS-I $_{MWF}$	16.9
CS-I	17.0
CS-II	18.3
WS-I	24.2
WS-II	27.5
WS-III	28.7

The optimum temperatures for growth and swim speed were determined by CPW for representative species within each temperature tier. Species with similar thermal requirements are grouped into tiers shown in Table 12-4. The optimum temperatures were selected as either the median value for the target species of the tier or the lowest temperature for a sensitive species was selected to ensure protection. The optimum temperatures for each temperature tier are shown in Table 12-5.

Table 12-4: Representative Species within each Temperature Tier. The representative species used to determine the optimum temperature for the tier are bolded.

Tier	Species Expected to be Present
CS-I $_{MWF}$	**Mountain whitefish** early life stages (applied to spawning grounds only)
CS-I	Brook trout and **cutthroat trout**
CS-II	**Brown trout**, **rainbow trout,** mottled sculpin, mountain whitefish, and longnose sucker
WS-I	**Common shiner, johnny darter, orangethroat darter**, and stonecat
WS-II	Brook stickleback, central stoneroller, creek chub, longnose dace, northern redbelly dace, finescale dace, **razorback sucker**, white sucker, and mountain sucker.
WS-III	Other Warmwater Species (Arkansas darter, bigmouth shiner, black bullhead, bluegill, bluehead sucker, **bonytail,** brassy minnow, brown bullhead, channel catfish, Colorado pikeminnow, common carp, fathead minnow, flannelmouth sucker, flathead catfish, freshwater drum, green sunfish, horneyhead chub, Iowa darter, plains killifish, plains minnow, plains topminnow, orangespotted sunfish, pumpkinseed sunfish, quillback, red shiner, Rio Grande chub, Rio Grande sucker, river carpsucker, roundtail chub, sand shiner, smallmouth bass, smallmouth buffalo, southern redbelly dace, speckled dace, spotail shiner, western mosquitofish, yellow bullhead)

Table 12-5: Optimum temperature based on species within each temperature tier for Colorado streams

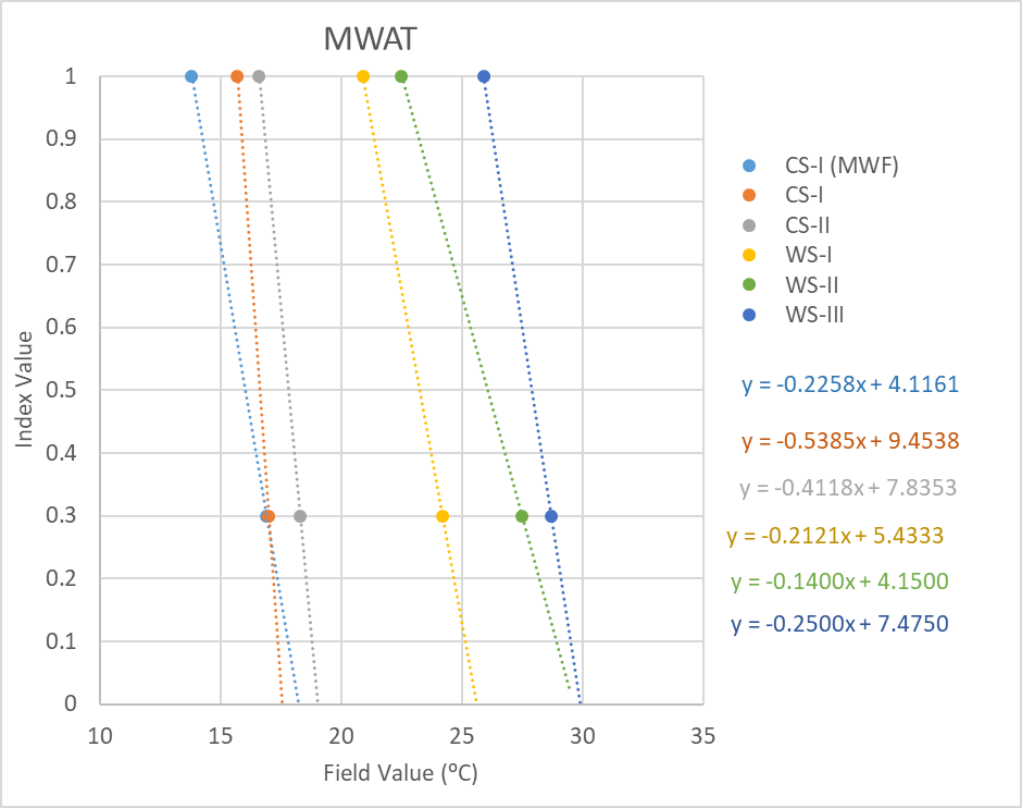
Temperature Tier Code	Optimum Temperature (°C)
CS-I $_{MWF}$	13.8
CS-I	15.7
CS-II	16.6
WS-I	20.9
WS-II	22.5
WS-III	25.9

The maximum index value (1.00) for each temperature tier was defined using the optimum temperatures shown in Table 12-5. The MWAT chronic temperature standards from state regulation 31 (5 CCR 1002-31) shown in Table 12-3 were used to define the 0.30 threshold for each temperature tier, as they represent the state's threshold for impairment. Linear curves were fit to the values in Table 12-6. The reference curves used in the CSQT Beta Version to calculate index values are shown in Figure 12-1.

Table 12-6: Threshold Values for MWAT

Index Value	Field Values (^{o}C) by Temperature Tier Code					
	CS-I $_{MWF}$	CS-I	CS-II	WS-I	WS-II	WS-III
1.00	13.8	15.7	16.6	20.9	22.5	25.9
0.30	16.9	17.0	18.3	24.2	27.5	28.7

Figure 12-1: Maximum Weekly Average Temperature (°C) Reference Curves



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

The MWAT metric relies on the user to select an appropriate temperature tier. All streams in Colorado have assigned temperature tiers (see Regulations 32 – 38) but the assigned tier may not be appropriate for a project reach. In selecting the appropriate temperature tier, users should also consider the most thermally-sensitive species expected to occupy the reach in summer once restoration is complete.

This metric considers colder summer MWATs to represent reference standard condition 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 functional capacity due to these changes.

Chapter 13. Dissolved Oxygen Parameter

Functional Category: Physicochemical

Parameter Summary:

Dissolved oxygen plays a key role in biological functioning and freshwater aquatic life requires adequate amounts of dissolved oxygen (DO) to survive. The amount of DO in a stream affects biological respiration rates and the solubility of chemical constituents such as inorganic nutrients (Harman et al. 2012).

Including this parameter in the CSQT Beta Version incentivizes in-stream structures and channel geometry that improves oxygenation where DO may be limiting biological and physicochemical function. DO can be improved through stream restoration practices that create eddies and induce mixing such as drop structures, habitat boulders, constructed riffles, and large woody debris. Permitted impacts that remove these types of features are likely to result in functional loss.

The SFPF (Harman et al. 2012) describes two measurement methods for DO: concentration and percent saturation. The CSQT SC considered including both but ultimately the reference standards for DO concentration were more directly linked to biological functioning than percent saturation, which varies considerably with temperature and elevation (USEPA 1986). Percent saturation was also considered indicative of nutrient cycling which is assessed in a separate parameter within the CSQT Beta Version.

Metric:

- Dissolved Oxygen Concentration

13.1. Dissolved Oxygen Concentration

Summary:

This metric is a direct measure of the concentration of dissolved oxygen (mg/L) in the project reach collected according to procedures outlined in WDEQ/WQD (2017).[4] DO sampling for use in the CSQT requires that a DO logger be deployed for at least one week during the summer months of July or August and record daily measurements between one and three in the afternoon.

Reference Curve Development:

Dissolved oxygen concentrations were obtained from the WQCD dataset, described in Section 1.7. In order to maintain consistency with the sampling methodology for the CSQT SC, only sampling events that occurred in summer months (June to September) and after noon on the sampling day were included in the analysis. Values within the dataset that were less than 0 mg/L or greater than 16 mg/L were removed from the analysis. Sites within the dataset were

[4] The CSQT SC reviewed the WDEQ/WQD methods and compared them to the CO SOPs (CDPHE 2016) and found the Wyoming methods were comparable and provided more detail.

stratified based on whether they were cold stream (CS) or warm stream (WS) temperature habitats (refer to regulatory temperature tiers identified in Chapter 12). Sites that were not identified as CS or WS were also removed from the analysis.

Sites within the dataset were identified as reference, other, or stressed (Table 13-1) based on an assessment of the upstream watershed using GIS and aerial imagery (CDPHE 2017). Impacts considered in the analysis included land use, point sources, water diversion, road density, abandoned mines, oil and gas facilities, and concentrated animal feeding operations. For sites that were not identified as stressed, the data show that the CS had higher DO values than the WS, as expected. However, the stressed sites demonstrate higher DO values than the rest of the sites and the WS stressed sites have the highest 75th percentile and the second highest average. The observed trends are likely a result of stressed sites with excessive algae that leads to high DO during the day and really low DO at night. The 25th percentile values follow the expected trend of higher DO in the sites identified as reference.

Table 13-1: Statistics for Dissolved Oxygen Concentrations (mg/L) from WQCD Dataset by Stream Condition

	Cold Stream (CS)			Warm Stream (WS)		
Statistic	**Reference**	**Other**	**Stressed**	**Reference**	**Other**	**Stressed**
Count (n)	70	511	43	17	134	29
Average	8.4	8.2	8.7	8.1	8.0	8.6
Standard Deviation	1.1	1.3	1.9	1.4	1.6	1.8
Minimum	6.8	1.8	6.3	6.1	3.1	5.5
25th Percentile	7.8	7.5	7.5	7.3	7.1	7.1
Median	8.3	8.0	8.3	7.9	7.7	8.2
75th Percentile	8.8	8.7	9.3	8.5	8.8	10.2
Maximum	13.1	14.4	15.4	11.4	13.0	13.0

The dissolved oxygen concentrations from the WQCD were compared to the 1986 USEPA report on dissolved oxygen concentrations that support aquatic life, summarized in Table 13-2.

Table 13-2: Dissolved Oxygen Concentrations (mg/L) that Support Aquatic Life (USEPA 1986)

Description	Salmonids		Non-Salmonids		Invertebrates
	Embryo and Larval Stages	Other Life Stages	Early Life Stages	Other Life Stages	
No production impairment	11	8	6.5	6	8
Slight production impairment	9	6	5.5	5	-
Moderate production impairment	8	5	5	4	-
Severe production impairment	7	4	4.5	3.5	-
Limit to avoid acute mortality	6	3	4	3	4

Even though salmonids are not species expected to be present in WS, the water quality criteria in Table 13-2 shows that the salmonid criteria are attainable in WS streams and the CSQT SC decided not to stratify the reference standards by stream temperature class (Table 13-3). The following criteria were used to set reference standards for DO concentration in the CSQT Beta Version:

- A DO concentration of 6 mg/L would be the minimum to avoid acute mortality of salmonid embryos and larvae and is likely to result in a slight production impairment for salmonids during other life stages. Therefore, 6 mg/L was set as the minimum index value (0.00). The data in Table 13-1 shows that all of the reference quality WS and stressed CS meet this criterion.
- A DO concentration of 9 mg/L was determined reasonable as the break between functioning-at-risk and functioning (index value of 0.70) since it would be protective of adult salmonids while providing protection for salmonid embryos and larvae. Additionally, this value was consistent with the 75th percentile for all sites in the WQCD dataset (Table 13-3).
- The CSQT SC considered a bell shaped curve and the declining function associated with high DO concentrations, but due to the difficulty in assigning a value at which function is lost based on the WQCD and the criteria laid out in Table 13-2 this was not incorporated into the CSQT Beta Version. High DO is indicative of excessive algae which is measured by the nutrients parameter and further research is needed to determine threshold values for the ‘falling limb’ of a reference curve.

Table 13-3: Statistics for Dissolved Oxygen Concentrations (mg/L) from WQCD Dataset

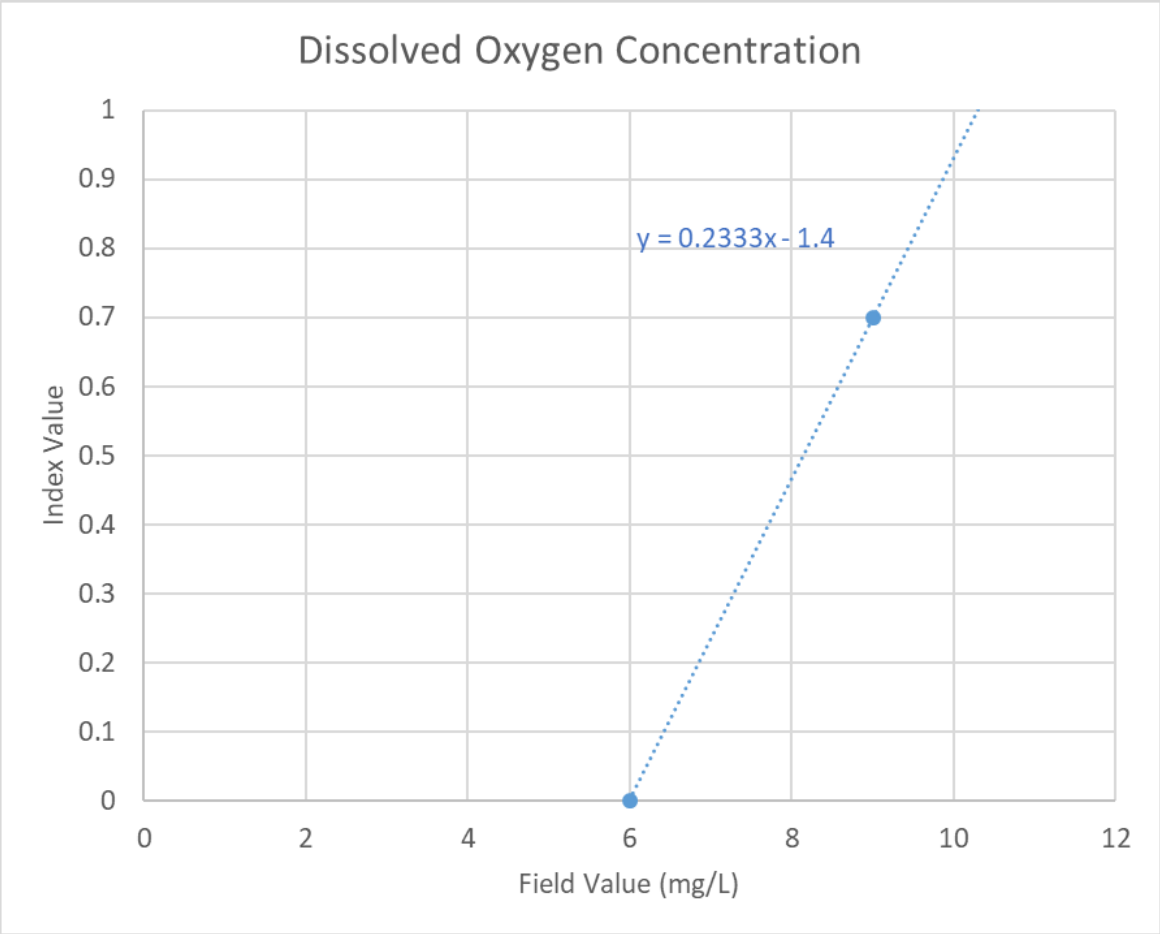
Statistic	CS	WS	All Sites
Count (n)	624	180	804
Average	8.3	8.1	8.2
Standard Deviation	1.3	1.6	1.4
Minimum	1.8	3.1	1.8
25th Percentile	7.5	7.1	7.4
Median	8.1	7.8	8.0
75th Percentile	8.8	9.2	8.8
Maximum	15.4	13.0	15.4

Table 13-4: Threshold Values (mg/L) for Dissolved Oxygen Concentration

Index Value	Field Value
0.70	9.0
0.00	6.0

These reference standards result in a DO concentration of 7.3 mg/L as the break between functioning-at-risk and not-functioning, which aligns with a moderate to severe production impairment in salmonids.

Figure 13-1: Dissolved Oxygen Concentration Reference Curve



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

Annual monitoring of DO may not be sufficient to show functional lift or loss. While continuous meters are available to track DO throughout a monitoring period, they are expensive.

Dissolved oxygen has diel fluctuations as primary producers generate oxygen during the day through photosynthesis and consume oxygen at night through respiration. Dissolved oxygen is generally lowest just before sunrise. Measuring dissolved oxygen in the afternoon does not capture the time period when dissolved oxygen is most likely to be at its lowest point, and therefore most stressful for fish and invertebrates. Since monitoring at night is impractical, measurements in the late afternoon will capture the diel fluctuation of water temperature and therefore a local minimum in the diel fluctuation of DO.

Chapter 14. Nutrients Parameter

Functional Category: Physicochemical

Parameter Summary:

Nutrients in stream ecosystems are, by definition, 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 greatly exceed optimum values which can lead to excess algae growth and result in degraded aquatic habitat and physicochemical conditions, altered fish and invertebrate communities, occasional fish kills, and aesthetic degradation.

Several nutrient-related metrics were considered including total nitrogen (TN), total phosphorus (TP), nitrate-nitrite, ammonia, ash free dry mass (AFDM), chlorophyll a, and cell counts with biovolume. Chlorophyll a is the predominant type of chlorophyll found in green plants and algae and concentrations are directly affected by the amount of nitrogen and phosphorus in the stream (Dodds and Smith 2016). This metric is preferred to direct sampling of nitrogen and/or phosphorus since water column nitrogen and phosphorus concentrations can be misleading when these nutrients are largely assimilated by excess algae and plant growth. This metric is preferred over AFDM which is unable to distinguish algal biomass from other periphyton components like dead algae, fungi, bacteria, protists, and detritus (Steinman et al. 2017). Cell count with biovolume is the most direct measurement of algal biomass but was not selected for use in the CSQT Beta Version due to a lack of data to develop reference standards.

Metric:

- Chlorophyll a

14.1. Chlorophyll a

Summary:

This metric is a direct measure of the concentration of chlorophyll a (mg/m^2) collected according to procedures outlined in CDPHE (2015). Chlorophyll a 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 CSQT Beta Version as a surrogate for nitrogen and phosphorus. Sampling should target maximum periphyton biomass, which occurs in summer; avoid sampling after high flows have scoured periphyton.

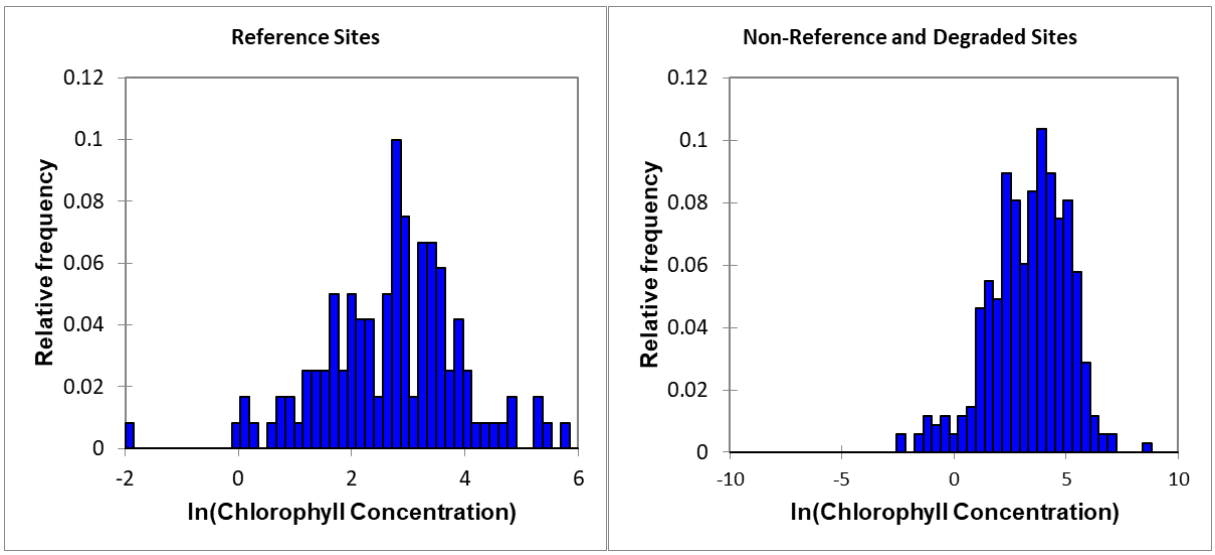
Reference Curve Development:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in CO. The analysis performed for the WSQT v1.0 to develop reference curves is presented first, followed by the WQCD dataset results from CO.

WSQT v1.0 (USACE 2018b): 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 and 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 visual assessment of the datasets indicated that the data were not normally distributed (Figure 14-1). 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 14-1: Histogram of Chlorophyll a Concentrations (mg/m^2) for Reference and Non-Reference Datasets from WY



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 to address geographic variability. Statistics for each dataset, stratified by ecoregion, are provided in Table 14-1.

Table 14-1: Statistics for Chlorophyll a Concentrations from the WDEQ Dataset

Statistic	Chlorophyll a 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 summary statistics for the two datasets were evaluated and the following decisions were made using best professional judgement to determine the threshold values shown in Table 14-2.

- 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 and basins was set to 150 mg/m2, 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. Note that the index values calculated by the WSQT v1.0 differ from the threshold values identified above, 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 v1.0.

CDPHE WQCD: The CDPHE collects nutrient, benthic algae, and chlorophyll data in streams throughout Colorado as part of their efforts to address nutrient pollution. There are currently no numeric chlorophyll a criteria for Colorado's streams for the protection of aquatic life, as Colorado has a chlorophyll a standard for recreation rather than biological health (Suplee et al. 2009). Chlorophyll a concentrations were obtained from the WQCD dataset (described in Section 1.7) to compare to the WSQT v1.0 reference curves and update the stratification.[5] The WQCD dataset is stratified by biotype instead of ecoregion because the periphyton data collection by CDPHE support efforts to develop biological indicators. While biotypes and ecoregions are similar (CDPHE (2017) and Section 1.7), further data analysis is needed to determine an whether biotype or ecoregion would be a better stratification for chlorophyll a.

Sites within the dataset were identified as reference, other, or stressed based on an assessment of the upstream watershed using GIS and aerial imagery (CDPHE 2017). Impacts considered in the analysis included land use, point sources, diversion, road density, abandoned mines, oil and gas facility, and concentration animal feeding operations. Due to the small number of stressed sites, these data are not presented but the reference sites are shown in addition to the results from all the sites within each biotype. The statistics for the WQCD dataset are provided in Table 14-2.

Table 14-2: Statistics for Chlorophyll a Concentrations from the WQCD dataset

	Chlorophyll a Concentrations (mg/m^2) by Biotype and Condition					
	1 - Transition		2 - Mountains		3 - Plains & Xeric	
Statistic	All Sites	Reference	All Sites	Reference	All Sites	Reference
Number of Sites (n)	149	19	86	24	29	4
Geometric Mean	14	14	9	14	56	113
Average	32	21	23	26	104	150
Standard Deviation	93	20	36	25	105	96
Minimum	0	2	0	1	0	26
25th Percentile	7	8	4	7	30	102
Median	15	14	9	18	74	169
75th Percentile	37	27	25	41	127	218
Maximum	1,114	70	214	82	460	238

[5] https://www.waterqualitydata.us/portal/

The statistics from the WQCD dataset were evaluated against the criteria used to develop reference standards for the WSQT v1.0, and found they generally corresponded well, including:

- The geometric mean of 12 mg/m2 from the reference standard dataset for the Wyoming mountains, which was used to define an index value of 1.00 in the WSQT v1.0 corresponds well with the geometric mean of 14 mg/m2 observed at the reference sites within the Colorado transition and mountains biotypes.
- The 75th percentile value of 27 mg/m2 from the reference standard datasets for the Wyoming mountains, which was used to define an index value of 0.70 in the WSQT v1.0 corresponds well with the 27 mg/m2 observed at the reference sites within the Colorado transition biotype and the 75th percentile from all the sites in the mountains biotype (25 mg/m2).
- The 75th percentile value of 117 mg/m2 from the non-reference and degraded sites for the Wyoming basins and plains, which was used to define an index value of 0.30 in the WSQT v1.0 is near the 75th percentile value of 127 mg/m2 observed in the Colorado plains and xeric biotype.

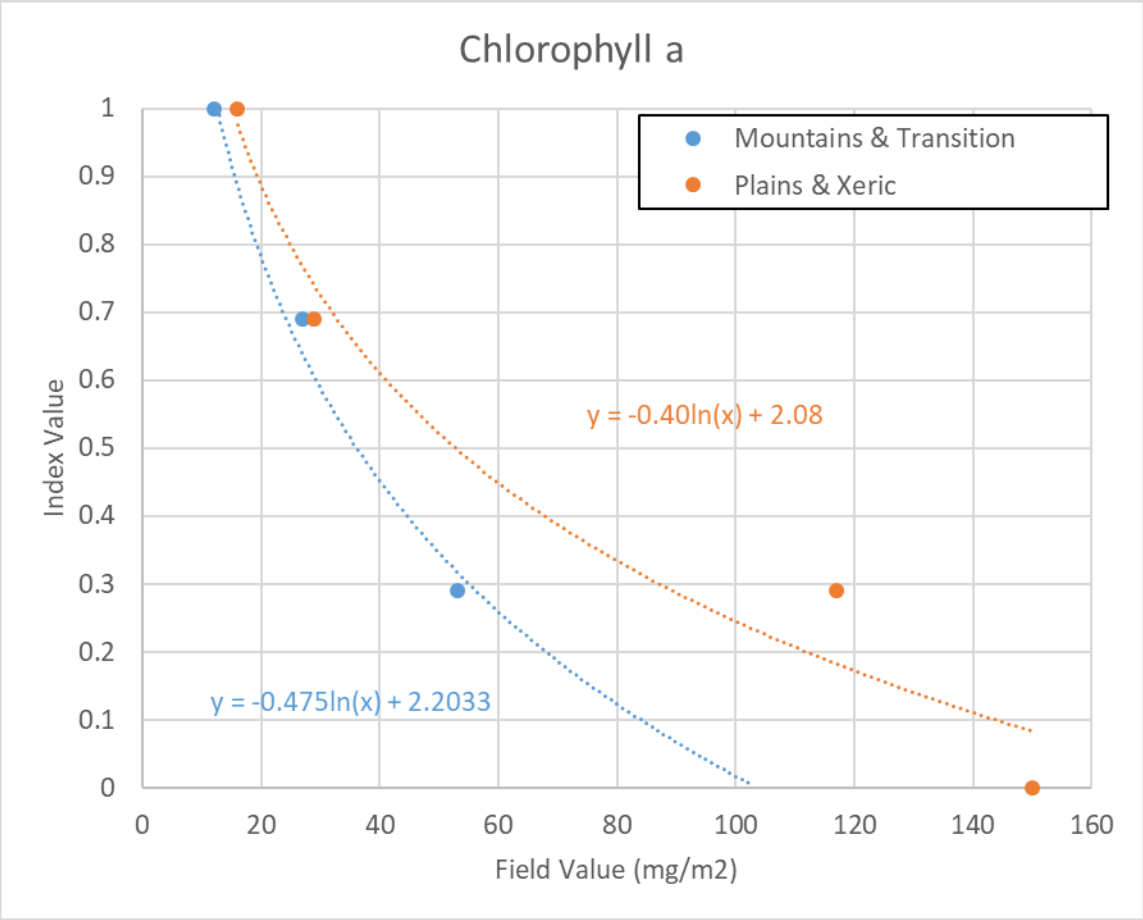
The CSQT SC also took into account the following observations regarding the CO dataset: in the mountain region the reference sites exhibit higher concentrations of chlorophyll a than the overall mountain dataset; this trend is also apparent for the plains and xeric region where there are only 4 reference sites. The CSQT SC is not sure that reference standard conditions are captured in the dataset for these biotypes.

Reference curves for this metric are shown in Table 14-3 and Figure 14-2 and have been adopted from the WSQT v1.0 for use and testing in CO. Further testing and analysis are needed to refine Colorado-specific reference curves.

Table 14-3: Threshold Values (mg/m^2) for Chlorophyll a by Biotype

Index Value	Field Value	
	1 – Transition 2 – Mountains	3 – Plains & Xeric
1.00	12	16
0.70	27	29
0.30	53	117
0.00	-	150

Figure 14-2: Chlorophyll a Reference Curves



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

Data from Wyoming were used to derive the reference curves for this metric. Testing and further analysis are desirable to determine whether additional or modified reference curves are needed for Colorado.

The reference curves are based on biotype groupings and some sites may be unfairly assessed due to natural variations within biotypes. This limitation can be addressed as more data are gathered and further statistical analyses are 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 as the CSQT Beta Version is applied.

Use of the Chlorophyll a 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, scouring flows, water temperature, and grazing by fish and invertebrates also affect benthic algae biomass, thus also chlorophyll a concentrations, but are not directly accounted for by this metric. Site specific conditions need to be taken into account when applying this metric.

Chapter 15. Macroinvertebrates Parameter

Functional Category: Biology

Function-based Parameter Summary:

Benthic macroinvertebrates are commonly used as indicators of stream ecosystem structure and function and were 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:

- Colorado Multi-metric Index (CO MMI)

15.1. Colorado Multi-metric Index (CO MMI)

Summary:

The Colorado Multi-Metric Index (CO MMI) is a set of macroinvertebrate-based indices calibrated to respond to stressors affecting aquatic communities within streams across three biotypes (CDPHE 2017). According to CDPHE (2017), “[w]ithin the benthic macroinvertebrate assemblage, metrics are selected that represent some measurable aspect of the community structure and function. These measurements are grouped into five metric categories: taxa richness, composition, pollution tolerance, functional feeding groups, and habit (mode of locomotion). Combining metrics from these categories into a multi-metric index transforms taxonomic identifications and individual counts into a unitless score that ranges from 0-100.”

Information on data collection, sample preservation, identification and enumeration and calculation of the MMI can be found in the appendices of CDPHE (2017), as well as the User Manual.

Reference Curve Development:

Reference curves for this metric were developed using the numeric attainment and impairment thresholds identified in the *Methodology to Determine Use Attainment for Rivers and Streams. Policy Statement 10-1* (CDPHE 2017). Attainment and impairment values were developed for the three biotypes in Colorado: mountains, transition and plains & xeric (Table 15-1). CDPHE uses reference or expected condition as the basis for characterizing attainment, and a significant departure from reference or expected condition as the basis for characterizing impairment. CDPHE also considers a “gray zone” that represents “a biological condition is neither impaired nor in attainment of the use based on Interval and Equivalence tests in which neither hypothesis is rejected.” Through consultation with CDPHE, the threshold between functioning and functioning-at-risk was equated with the attainment values identified in Table

15-1. Similarly, the impairment values in Table 15-1 were equated to the threshold between the functioning-at-risk and not-functioning condition. With this approach, the gray zone described by CDPHE aligns with the functioning-at-risk range of index scores in the CSQT Beta Version. This approach is consistent with the functional capacity definitions outlined in Table 1-1. The maximum index score (1.00) in the CSQT was set equal to the MMI score for high scoring waters identified in CDPHE (2017). The minimum index score (0.00) was set equal to an MMI score of 0. The CO MMI index value thresholds and reference curves are shown in Table 15-2 and Figure 15-1.

A broken linear relationship was used to extrapolate values between these points for the reference curves (Figure 15-1).

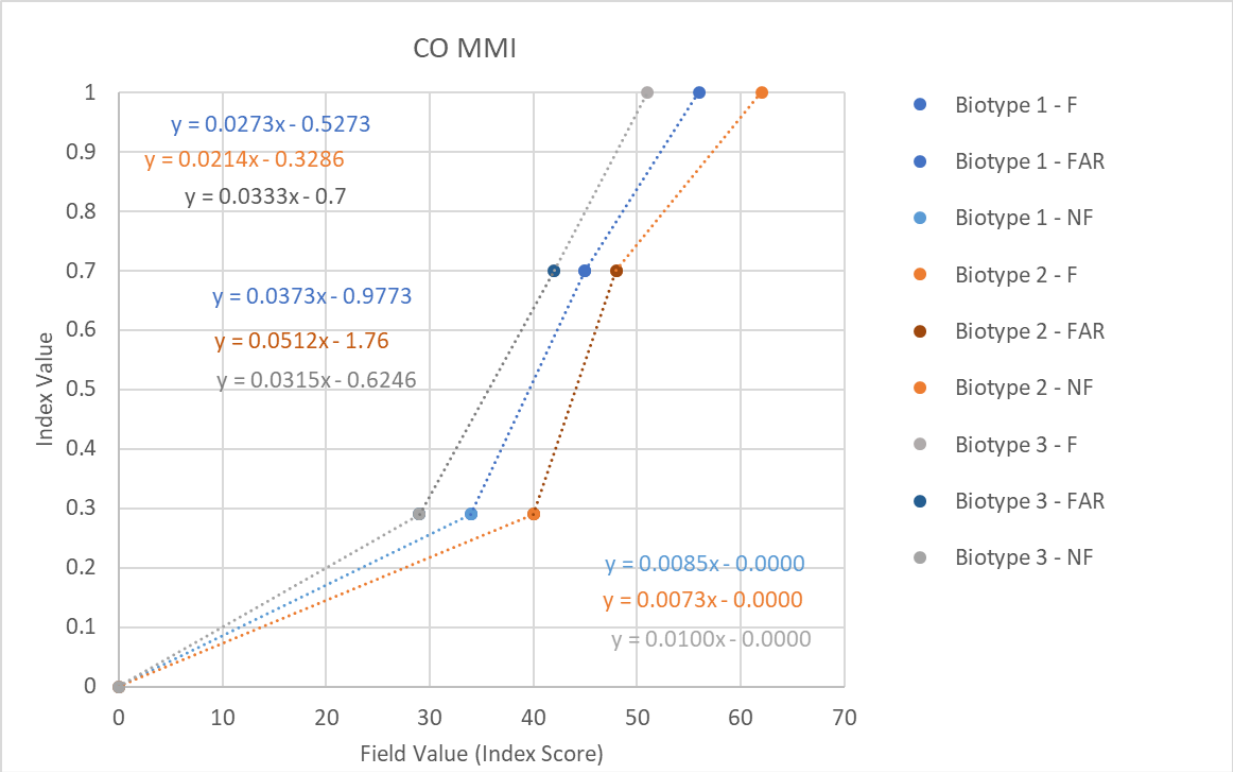
Table 15-1: CO MMI Aquatic Life Use Thresholds for each Biotype (CDPHE 2017)

Biotype	CO MMI Value		
	Impairment Threshold	Attainment Threshold	High Scoring Waters
1 - Transition	34	45	56
2 - Mountains	40	48	62
3 - Plains & Xeric	29	42	51

Table 15-2: Threshold Values for CO MMI Scores by Biotype

Index Value	Field Value		
	1 – Transition	2 – Mountains	3 – Plains & Xeric
1.00	56	62	51
0.70	45	48	42
0.29	34	40	29
0.00	0	0	0

Figure 15-1: CO MMI Reference Curves



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

According to CDPHE (2017), the metrics included in the CO MMI were selected based on their "ability to discriminate between reference and stressed sites, represent multiple metric categories, are ecologically meaningful, and are not redundant with other metrics in the index… Because of this metric selection process, the metrics used in the MMI vary in their ability to detect or diagnose specific types of stress. The MMI is designed to detect environmental stresses that result in alteration of the biological community. No specific stressors are identified because the intent is to have a generalized tool that responds to a wide range of potential stressors. In other words, the MMI tool cannot determine if the stressor is a specific pollutant, pollution or habitat limitation (including flow)." A complete description of CO MMI limitations can be found in CDPHE (2017).

The CSQT Beta Version does not include the auxiliary metrics defined in Policy Statement 10-1 (CDPHE 2017) that are used to determine attainment of aquatic life uses in Class 1 waters whose MMI scored between the attainment and impairment thresholds. These auxiliary metrics may be added to the CSQT as metrics for the macroinvertebrates parameter as the CSQT Beta Version is tested and applied.

CDPHE (2017) evaluated the applicability of the CO MMI to larger rivers (exceeding 2,700 mi2 watershed area), and concluded that the thresholds could be applied to rivers exceeding 2,700 mi^2 watershed area in the Yampa River, White River, Colorado River, Gunnison River, Dolores River, and San Juan River basins, but not in larger rivers in the South Platte River, Arkansas

River, Purgatoire River, and Rio Grande River basins. As such, the reference curves provided here may not be representative of the MMI values within the larger rivers in these basins.

While the CDPHE (2017) approach is intended for use in perennial streams, macroinvertebrate sample collection could also be completed in non-perennial streams when standing or flowing water is present. However, it is important to note that spatial and interannual variability may be greater within these systems, and sampling may have more limited repeatability.

Chapter 16. 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 Colorado. 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 and they serve as important indicators of ecological health.

This parameter is intended to document several aspects of Colorado 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 wild trout populations.

Since there are no existing statewide biological indices used for fish in Colorado, metrics and reference curves for fish were adapted from the fish metrics in the WSQT v1.0 following consultation with Colorado Parks and Wildlife. 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 wild trout biomass metric is also included to capture post-project increases in naturally-reproducing trout population biomass following restoration projects. This metric is only recommended at restoration sites where CPW management objectives relate to native or non-native wild trout species.

Reference standards for native fish species and SGCN are based on a departure from the expected species assemblages in Colorado (CSQT Beta Version; Appendix C). These reference standards are informed by a comparison to expected species assemblages, which have been identified by CPW in Appendix C and the Statewide Wildlife Action Plan (CPW 2015). 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, introduction of non-native species, 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 CSQT SC recommends project-specific coordination with CPW area 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
- Wild Trout Biomass (% increase)

16.1. 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, foraging or refugia habitats (Angermeier and Schlosser 1995). Reference standards for native fish species are derived from the expected species assemblages and are described in the Parameter Summary above.

Reference Curve Development:

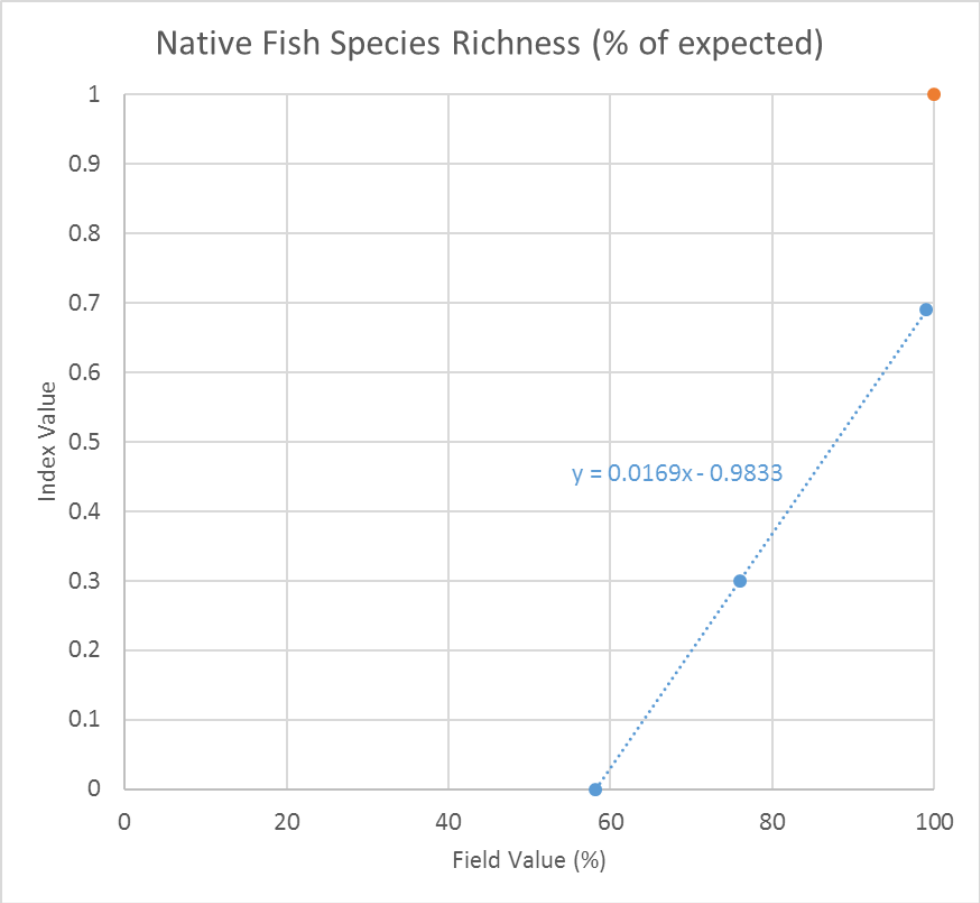
Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in CO. Reference curves were developed based on the best professional judgement of regional fisheries biologists. Achieving 100% native species richness was considered to represent a pristine reference standard 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 not-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 not-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 16-1 and Figure 16-1.

Because the total number of native species varies across basins, as well as between the thermal regimes within a basin, this metric is normalized by the expected number of species. 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 16-1: Threshold Values for Native Fish Species Richness

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

Figure 16-1: 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 Appendix C of the User Manual provide a starting point for the potential maximum number of species at a location within a given river basin. Consideration should also be given to whether species would naturally be present within the stream temperature/gradient class. However due to variability in sub-basin geology, flow regime and other natural factors, these lists require coordination with the CPW prior to finalizing an expected number of native species at any given site.

16.2. 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 (CPW 2015) as those species whose conservation status warrants increased management attention and funding, as well as consideration in conservation, and land use and development planning in Colorado. For any project where this metric is used, the practitioner should consult with the area fisheries biologist and Native Aquatic Species Biologist at CPW 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 two tiers where Tier 1 species have the highest conservation priority. The number of species with natural potential to occur at the site in each tier is used to calculate the field value for the CSQT Beta Version. 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 twice as high as Tier 2 species (Table 16-2). Note that if there are no species in a tier expected at a site then there are no species absent for that tier.

Table 16-2: How to Determine the Field Value for SGCN Absent Score

SGCN Species (A)	Multiplier (B)	Equation
# Tier 1 Species Absent	2	$\boldsymbol{C}_1 = \boldsymbol{A}_1 * \boldsymbol{B}_1$
# Tier 2 Species Absent	1	$\boldsymbol{C}_2 = \boldsymbol{A}_2 * \boldsymbol{B}_2$
Field Value for the CSQT =		$\boldsymbol{C}_1 + \boldsymbol{C}_2$

This weighted approach reflects the relative importance of species within the tiers and to remain consistent with the management goals and approaches for SGCN by the State of Colorado. 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 a Tier 1 species from a site should be considered a significant functional loss.

The reference criteria for this metric are the same as with those derived for the WSQT v1.0. The reference criteria are categorical, and each category was assigned a specific index value score based on best professional judgement after consultation with CPW area fisheries biologists. No reference curve was developed. If all expected 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 represents sites that have the potential to support SGCN. A site that lacks one Tier 2 species but all expected Tier 1 species are present will score an index value of 0.69. While a site that is missing either one Tier

1 species or two Tier 2 species will score an index value of 0.30. Sites with more than 1 SGCN absent were assigned an index value of zero, and were assumed to not support SGCN.

Table 16-3: 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 Appendix C of the User Manual provide a starting point for the potential maximum number of species at a location within a given river basin. Consideration should also be given to whether species would naturally be present within the stream temperature/gradient class. However, due to variability in sub-basin geology, flow regime and other natural factors, these lists require coordination with the CPW prior to finalizing an expected number of native species at any given site.

16.3. Wild Trout Biomass

Summary:

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

This metric focuses on the productivity of wild trout game fish where CPW management objectives relate to native or non-native wild trout species. Wild trout include naturally reproducing populations of native and non-native trout; trout of potential hatchery origin should not be included in this metric, and users should consider whether this metric should be applied based on the potential for nearby stocked populations to influence biomass numbers within a project reach. Measurements of biomass can be used to infer whether there have been gains in wild trout 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 wild trout 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 have a similar elevation and geomorphic setting as the project reach and should be of reference quality (to the extent practicable). 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:

Reference curves for this metric have been adopted from the WSQT v1.0 for use and testing in CO. The WSQT v1.0 stratified by productivity classifications identified by the State of Wyoming, and the CSQT modified the productivity classification, as described below, but maintained the same reference curves for each productivity class.

This metric focuses on the increase in wild trout biomass following a restoration project, and index values and reference curves are associated with the magnitude of change in biomass (pounds/acre) 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 productivity ranges, recognizing that streams with an already productive fishery may be less likely to see large additional increases in productivity following a restoration project. The productivity classes for this metric are modified from the WGFD classifications related to measured fish biomass (Annear et al. 2006) that were used to stratify the metric in WSQT v1.0. Modifications included revising the units from lbs/mile to lbs/acre to align with the biomass units used by CPW. The high productivity class includes streams where current biomass is equal to or greater than 60 pounds per acre, which is the biomass criteria for a Gold Medal fishery in Colorado. The moderate productivity class ranges from 30-60 pounds per acre and the low productivity class includes streams that have less than 30 pounds per acre.

A driving assumption in this stratification is that streams identified as high productivity 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 these streams. Conversely, the low productivity streams have the lowest level of productivity and are more likely to be below their biologic potential. The assumption is that there is greater potential to improve biomass in a low productivity stream. Reference curves were developed to reflect these assumptions, and therefore require less biomass improvement in a stream that is already highly productive than a low productivity 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 lower productivity fisheries yielded greater increases in biomass than in higher productivity fisheries (Table 16-4). However, the streams with low productivity, identified as green ribbon streams by WGFD, did not show greater increases in productivity than yellow ribbon streams. Binns (1999) inferred that yellow ribbon stream systems may be limited by watershed-scale issues that reduce the potential for greater increases in biomass. Based on these results, the WSQT v1.0 did not stratify between yellow and green ribbon streams but proposed the same reference curve for both. This approach was carried forward to the CSQT Beta Version to in define low productivity streams.

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. High productivity 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 productivity 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 high productivity stream. In moderate productivity streams, we determined that a 50% increase in biomass could reasonably represent a substantial lift. WGFD habitat improvement projects have exceeded this value in moderate productivity (red ribbon streams; Table 16-4), with an average increase of 104% pounds/acre for wild trout fisheries. Given the assumption that low productivity 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 a 200% increase in wild trout biomass in yellow and green ribbon streams.

Threshold values and reference curves are shown in Table 16-5 and Figure 16-2.

Table 16-4: Mean empirical values for trout biomass averaged over habitat improvement projects sorted for WGFD stream class. Note Wyoming stream classes are defined using lbs/mile, which is different than the productivity classes for Colorado that are defined using lbs/acre. Adapted from Binns (1999).

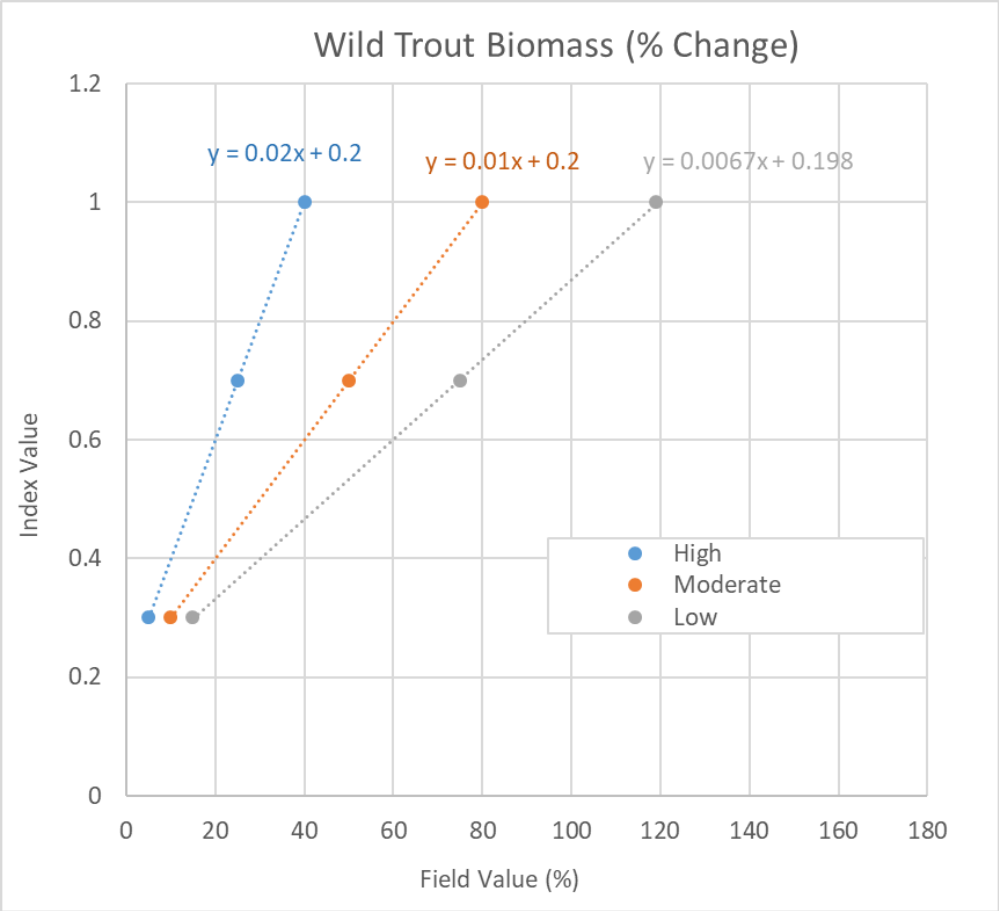
Stream Class	Number of projects with measurements	Reference (lbs./acre)	Treatment (lbs./acre)	Mean % Change
Blue Ribbon (>600 lbs/mile)	0	-	-	-
Red Ribbon (300-600 lbs/mile)	Wild Trout = 3 *Mixed Pop. = 5	64 52	122 106	104 115
Yellow Ribbon (50-300 lbs/mile)	Wild Trout = 15 *Mixed Pop. = 23	42 43	78 87	316 303
Green Ribbon (<50 lbs/mile)	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 16-5: Threshold Values for Wild Trout Biomass

Stream Productivity 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
High (>60 lbs/acre)	< 5	5	25	≥ 40
Moderate (30-60 lbs/acre)	< 10	10	50	≥ 80
Low (<30 lbs/acre)	< 15	15	75	≥ 119

Figure 16-2: Wild Trout Biomass Reference Curves



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

The percentage increases in Table 16-5 forming the basis for the reference curves are based on best professional judgement and supported by previous evaluations of restoration projects in Wyoming (Binns 1999). 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. Examples exist (Whiteway et al. 2010, Pierce et al. 2013), although research also suggests that fish populations can take five or more years to respond to restoration (Binns 1994; Hunt 1976). This limitation should be taken into consideration when developing monitoring plans.

An improvement in non-native wild trout 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 CPW management objectives relate to native or non-native wild trout species. Consultation with area fish biologists is necessary before selecting and using this metric in the tool. This consultation should inform metric selection and project design, reducing the potential for these types of trade-offs between native and non-native species.

Results from this metric could be influenced by stocking or modifications to fishing regulations. Consultation with the area fish biologists is important to determine whether certain species or age classes should be excluded from biomass estimates because of stocking efforts within the watershed. Also, if changes in regulations are implemented concurrently with a project, this should be considered when interpreting results from this metric, as changes in regulations may affect biomass estimates within a project site.

Chapter 17. References Cited

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Appendix A: CSQT 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		
Hydrology	Flow Alteration Module-Separate Routine outisde of the reach condition assessment	Mean Annual Q - O/E	-		0.0	*	*	0.9 - 1.1	# Literature values from Richter et al.(2012), WSTT and CSQT SC.	Applicable in all streams.
		Mean Sept Q- O/E			≥2.0	*	*			Applicable in perennial streams.
		Mean Aug Q- O/E								
		Mean Jan Q- O/E								
		Mean annual peak daily Q- O/E								Applicable in all streams
		7-Day Minimum - O/E								
Reach Hydrology and Hydraulics	Reach Runoff ●	Land Use Coefficient	-		80 ≥	*	68	≤ 40	# Literature values from NRCS, 1986.	Applicable in all streams, recommended for primarily agricultural watersheds. Additional testing is needed in all stream types.
		Impervious Cover			*	*	10	0	Literature values from Scheuler et al. (2009).	Applicable in all streams, recommended for primarily urban watersheds. Additional testing is needed in all stream types.
		Concentrated Flow Points			*	*	1.0	0.0	# Developed by WSTT and Stream Mechanics.	Applicable in all streams. Additional testing is needed in all stream types.
		Water Quality Capture Volume			0.0	*	*	1.0	Developed by Tennessee Department of Environment and Conservation (TDEC, 2018).	Applicable in all streams. Additional testing is needed in all stream types.
	Base Flow Dynamics	Average Velocity (fps)	-		NA				Developed using literature values from Hickman and Raleigh 1982, Raleigh 1982, Raleigh et al. 1984, Raleigh et al. 1986, Wesche et al. 1987, Shuler and Nehring 1993.	Reference curves are applicable in coldwater streams only; reference curves for warmwater streams have not been developed yet.
		Average Depth (ft)	Stream Temperature & Proposed Bankfull Width	CS-I MWF & CS-I; W < 20ft	0.2			1		
				CS-I MWF & CS-I; W > 20ft	0.4			1.5		
			Stream Temperature	CS-II	0.6			2.3		
	Floodplain Connectivity ●	Return Interval	-		*	*	2.0	≤ 1.1	Literature values from Moody et al. (2003), Emmert (2004), and Foster (2012).	Applicable in all streams.
		Bank Height Ratio (ft/ft)			*	1.5	1.2	1	# Literature values from Rosgen (2008), Harman et al. (2012) and data from the Wyoming 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 or E	*	2.0	2.4	≥ 5		Not applicable in naturally occuring canyon systems (e.g., F type streams) or braided (D) stream types.
				A or B	*	1.2	1.4	≥ 2.2		
		Percent Side Channels (%)	-						Literature values from Nadeua et al. (2018) and Stream Mechanics.	Applicable to all streams where side channels are a desirable end state for the project reach.
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 and 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	*	> 0.1	# 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	≤ 3.0		3.7	4.0 - 6.0	# Data from the Wyoming 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.
					≥ 9.3	*	7.0			
				Cb		*	3.0	3.7 - 5.0		
							6.0			
				B & Ba	≥ 7.5	*	4.0	≤ 3.0		
				Bc		*	6.0	≤ 3.4		
				E	1.8	*	3.0	3.5 - 5		
					≥8.3		6.0			
		Pool Depth Ratio ●	-		1	*	2.20	≥ 3.2		
		Percent Riffle (%) ●	Slope	S < 3%	*	*	39	50 - 60		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.
							69			
				S ≥ 3%	*	*	60	68 - 78		
							83			
		Aggradation Ratio	-		≥ 1.60	1.40	1.19	1.00	# Literature values from Rosgen (2014) and data from the Wyoming Geomorphic Reference Dataset.	Not applicable to braided (D) stream types.
	Plan Form	Sinuosity (k)	Valley Type	Unconfined Alluvial		1.15	1.19	1.20 - 1.50	# Literature values from Rosgen (2014) and modified using regional data from NRSA and the Wyoming Geomorphic Reference Dataset.	Only applicable in single-thread systems; reference curves have not been developed for natural F and G channels.
					*	1.60	1.51			
				Confined Alluvial	1.00	*	1.19	≥ 1.20		
				Colluvial/V-Shaped		1	1.09	1.1 - 1.3		
					*	1.4	1.31			
			Reference Stream Type	E		1.2	1.29	1.3 - 1.8		
					*	2	1.81			
	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	*	*	59-69		
					*	101	*			
		Herbaceous Vegetation Cover (%) ●	Cover Type	Herbaceous	30	*	94	117		
				Forested, Scrub-Shrub	0	*	54	77		
		Percent Native Cover (%) ●	-		*	65	91	100		
Physicochemical	Temperature	Daily Maximum (°C)	Steam Temperature	CS-I (MWF)	≥ 21.20	-	-	< 21.20	Developed using data and guidance from CPW and CDPHE.	Applicable in all streams with baseflow extending through August.
				CS-I	≥ 21.70	-	-	< 21.70		
				CS-II	≥ 23.90	-	-	< 23.90		
				WS-I	≥ 29.00	-	-	< 29.00		
				WS-II	≥ 28.60	-	-	< 28.60		
				WS-III	≥ 31.80	-	-	< 31.80		
		Mean Weekly Average Temp (MWAT) (°C)	Steam Temperature	CS-I (MWF)	*	≥ 16.90	*	≤ 13.80		
				CS-I	*	≥ 17.00	*	≤ 15.72		
				CS-II	*	≥ 18.30	*	≤ 16.56		
				WS-I	*	≥ 24.20	*	≤ 20.85		
				WS-II	*	≥ 27.50	*	≤ 22.50		
				WS-III	*	≥ 28.70	*	≤ 25.90		
	Dissolved Oxygen	DO Concentration (mg/L)	-		6.0	*	9.0	*		
	Nutrients	Chlorophyll a (mg/m2)	Biotype	1, 2	*	53	27	< 13	# Developed by WSTT.	
				3	≥ 150	117	29	≤ 16		
Biology	Macroinvertebrates	CO MMI	Biotype	1	0	35	45	≥ 57	Literature values from CDPHE (2017).	Applicable in all wadeable streams when standing or flowing water is present during the inde period. In non-perennial streams, spatial and interannual variability may be greater, and sampling may have more limited repeatability. the thresholds may not be representative in rivers with watershed areas >2,700 mi2 in the South Platte River, Arkansas River, Purgatoire River, and Rio Grande River basins.
				2	0	41	48	≥ 63		
				3	0	30	42	≥ 52		
	Fish	Number Native Fish Species (% of expected)	-		≤ 58	75	99	100	# Developed by Wyoming Game and Fish, metrics were modified for Colorado with input from CPW.	Applicable in all streams statewide, except naturally fishless streams.
		SGCN Absent Score	-		≥ 3	2	1	0		
		Wild Trout Biomass (% Increase)	Stream Productivity Class	High	< 5	5	25	≥ 40	# Developed by Wyoming Game and Fish and WSTT, stratification was modified for Colorado with input from CPW; and evaluated using data from Binns (1999)	Applicable in coldwater streams. Certain species or age classes should be excluded from biomass estimates where there are stocking efforts within the watershed.
				Moderate	< 10	10	50.0	≥ 80		
				Low	< 15	15	75.0	≥ 119		

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

Reference curves are same as WSQT v1.01 and are proposed for use and testing in Colorado

● Basic Assessment recommended elements