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Stream Restoration as a BMP: Crediting Guidance

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THE Water Research FOUNDATION

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Abstract and Benefits

Abstract:

Stream restoration provides a multitude of benefits to ecosystems and communities. Stream restoration projects may also provide pollutant trading and mitigation opportunities under Clean Water Act section 402 and similar state and local laws. This crediting guidance provides a general technical framework for quantifying the water quality benefits of a specific suite of stream restoration practices, focusing on sediment, nutrients, and temperature. The four practices addressed in this guidance include stream stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection. The general technical considerations and challenges for developing stream restoration credits are discussed, along with guidance for credit development. Guidance for assigning credits for each of the four stream restoration practice groups includes background information, project information/data requirements, regional geomorphic considerations, longevity and response time, uncertainty and simplifying assumptions, and recommended crediting approach. Concepts such as applicable credit area, safety factors (i.e., credit multipliers), credit life, and tracking and accounting are also discussed. This guidance also provides information related to verification and monitoring of stream restoration projects.

Benefits:

- Establishes a framework for crediting water quality benefits of stream restoration projects.
- Provides a summary of the water quality benefits of various stream restoration practices based on an extensive literature review.
- Summarizes nutrient-, sediment-, and temperature-related benefits for stream stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection.
- Provides guidance for verifying and monitoring stream restoration projects.
- Provides recommended standardized reporting protocols for stream restoration studies.
- Identifies research needs and data gaps related to stream restoration practices in the context of water quality crediting.

Keywords: Stream restoration, water quality credit, pollutant trading, bed and bank stabilization, instream enhancement, hyporheic exchange, floodplain reconnection, riparian buffer.

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Acronyms and Abbreviations

BA	Before-after
BACI	Before-after-control-impact
BSTEM	Bank Stability and Toe Erosion Model
CEM	Channel Evolution Model
CONCEPTS	CONservational Channel Evolution and Pollutant Transport System
CSR	Capacity-supply ratio
CWA	Clean Water Act
EPA	United States Environmental Protection Agency
EPT	Extensive post treatment
HUC	Hydrologic Unit Code
IBI	Index of Biotic Integrity
IPT	Intensive post treatment
MODFLOW	USGS three-dimensional finite-difference groundwater model
NPDES	National Pollutant Discharge Elimination System
REM	River Erosion Model
REMM	Riparian Ecosystem Management Model
RGA	Rapid geomorphic assessment
RHA	River and Harbors Acts
RUSLE	Revised Universal Soil Loss Equation
SWAT	Soil and Water Assessment Tool
TMDL	Total maximum daily load
USACE	U.S. Army Corps of Engineers
WE&RF	Water Environment & Reuse Foundation
WEF	Water Environment Federation
WEPP	Water Erosion Prediction Project
WRF	The Water Research Foundation

Executive Summary

Stream restoration provides a multitude of benefits to ecosystems and communities. Stream restoration projects may also provide pollutant trading and mitigation opportunities under Clean Water Act section 402 and similar state and local laws. For example, as a part of water quality regulatory programs such as total maximum daily loads (TMDLs) or as part of overall watershed planning efforts to address multiple water quality and quantity issues. This crediting guidance provides a general technical framework for quantifying the water quality benefits of a specific suite of stream restoration practices, focusing on sediment and nutrients. The four practices addressed in this guidance include stream stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection.

The general technical considerations and challenges for developing stream restoration credits are discussed, along with guidance for credit development. Guidance for assigning credits for each of the four stream restoration practice groups includes background information, project information/data requirements, regional geomorphic considerations, longevity and response time, uncertainty and simplifying assumptions, and recommended crediting approach. Concepts such as applicable credit area, safety factors (i.e., credit multipliers), credit life, and tracking and accounting are also discussed. However, this guidance only provides a framework for crediting programs; therefore, many specifics, including trading ratios and project eligibility, are not prescribed and are instead left for individual programs to develop on their own.

This guidance also provides information related to verification and monitoring of stream restoration projects. Many reputable guidance documents for monitoring streams pre- and post-restoration have previously been developed. Several general approaches can be used to quantify and/or verify the benefits of stream restoration projects, including direct monitoring of water quality and stream geomorphology, functional assessment, modeling, and/or some combination of these approaches. Each of the verification approaches has strengths and weaknesses. The most useful approach is to incorporate aspects of each, depending on the project type and the goals of the monitoring. A new stream restoration database has been created in tandem with this guidance that can be used to report and track project monitoring and assessment data (WRF, n.d.).

Information in this guidance is appropriate for supporting the initial technical basis of water quality crediting programs for stream stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection as part of water quality trading and/or crediting programs. General conclusions and caveats that should be considered when incorporating stream restoration into these programs include:

- Stream restoration can provide nutrient removal benefits; however, the magnitude of water quality benefits is highly site-specific and variable, which leads to substantial uncertainty, especially with respect to denitrification processes and long-term pollutant retention and prevention.
- The empirical basis for stream restoration as a water quality best management practice (BMP) is improving, but additional research is needed, especially for regions and stream types that are poorly represented in the literature. Similarly, some practices have stronger empirical bases than others, and some practices have inherently higher functional capacity for nutrient and sediment removal and temperature mitigation than others. Currently, the relative magnitude of benefits is also more certain than the absolute magnitude of the benefits. To be scientifically defensible, stream restoration crediting schemes must acknowledge uncertainty through safety factors, updating assumptions and methods and ultimately water quality credit(s) or performance levels as the empirical basis for quantification improves over time.

- Direct measurement of the water quality benefits of stream restoration can be very challenging and expensive. For this reason, monitoring approaches that incorporate surrogate (proxy) measures are important aspects of evaluating the benefits of stream restoration practices. Functional assessment approaches developed in the Clean Water Act section 404 compensatory mitigation arena provide logical frameworks and principles that are believed to be transferable to crediting programs for stream restoration. For example, wetland mitigation protocols distinguish among hydrogeomorphic types and potential success of restoration and regional differences when quantifying credits. Developing rapid assessment indicators of stream restoration functions greatly simplifies monitoring and reduces costs once the empirical relationships between indicators and actual functions are established.
- This crediting guidance focuses on the science supporting crediting for stream restoration projects; however, there are many policy decisions that must be made based on local objectives and physical settings, which are not addressed in this report. Examples include trading ratios, incentives for project implementation, and methods for prioritizing watersheds and segments for projects that provide the greatest system-level benefits over the long term. Additionally, credit values should consider stream context, type, and regional/watershed setting (i.e., classification/stratification is important).
- Communities should consider these guiding principles in the implementation of stream restoration projects:
    - Restoration should be targeted where it is most needed. A watershed approach (e.g., eight-digit Hydrologic Unit Code [HUC] or smaller) should be applied and efforts prioritized to support broader water quality goals.
    - Restoration approaches should have a sound empirical basis and quantification of credits must be scientifically credible/defensible.
    - Restoration projects should consider and avoid adverse impacts onsite and offsite (e.g., downstream sediment starvation, impeded aquatic organism passage, increased flooding risk upstream or downstream, destruction of habitats, etc.).
    - The most beneficial stream restoration projects are those with multiple benefits that go beyond pollutant reductions and provide other functions such as enhanced aquatic habitat, increased floodplain connectivity, and balanced sediment fluxes. Ideally projects create self-sustaining stream ecosystems.
    - Ideally, restoration projects should improve watershed-scale continuity and connectivity, restoring long segments and reaches with poor biological and physical function between functioning segments.
    - Understanding long-term channel evolution is important to understanding what would occur in the absence of intervention (~10 to 30-year time scale). It is important to recognize that time lags between restoration and improvements in stream function are common and unavoidable.
    - Non-technical and regulatory requirements must be considered in restoration designs. Examples include regulations related to wetlands, floodplains, water quality, and threatened and endangered species. When evaluating the cost and feasibility of stream restoration projects, these regulatory requirements can be significant considerations affecting the feasibility of a project. Additionally, property ownership, access, stakeholder support, and adequate funding for capital improvements and long-term maintenance are important considerations, among others.
- For entities considering water quality crediting programs for stream restoration, performance assessment and accountability for the credit value over time are important. Evaluation of the performance of a restoration project is likely most cost effective if based on a functional assessment

approach. However, actual water quality monitoring in selected cases to build data sets for comparison with functional assessment results is critically important.
- Although the empirical evidence of stream restoration's potential to increase nutrient processing and retention has increased recently, targeted additional research may address uncertainties in estimates and safety factors applied to credits. Recommendations for targeted research are also provided in this guidance.

Update and Re-release

The original version of this crediting guidance was published in 2016. This re-release includes several updates:

- Guidance was added on the benefits of stream restoration for reducing water temperatures. A credit quantification section for temperature was added for riparian buffer restoration, following the temperature crediting work from the Oregon Department of Environmental Quality.
- Stack et al. (2018) conducted an independent review of the original release of this crediting guidance. Their review agreed with the original assessment (Bledsoe et al., 2016) that there is insufficient empirical evidence to include watershed processes and channel reconfiguration as stream restoration practices eligible for crediting. Other recommendations from the Stack et al. (2018) report, including updated references, have been incorporated into this update.
- Additional stream restoration guidance and studies published since the initial release of this guidance have been incorporated where appropriate.
- Two case studies have been incorporated, including one on temperature crediting in Oregon and another that applied the crediting procedures outlined in this guidance to a stream restoration project in Denver, CO (Earles et al., 2020). Providing example applications of crediting approaches was suggested by survey respondents in Stack et al. (2018).
- The functional assessment section has been expanded to include recommendations from a recent WRF-sponsored report (Bledsoe et al., 2019).
- Recent recommendations published for updating the bed and bank stabilization crediting approach (Wood 2020) and hyporheic zone and floodplain reconnection crediting approaches (Wood and Schueler, 2020) outlined in the original Chesapeake Bay crediting guidance (Schueler and Stack 2014) have been incorporated into this updated report. Additional information on verifying stream restoration credits has also been added (Schueler and Wood, 2019).

CHAPTER 1

Introduction

The benefits of stream restoration in urban, agricultural, silvicultural, and other areas are diverse and widely recognized. Representative benefits include improved aquatic life conditions and ecosystem services, protection of property and infrastructure, floodwater storage, reduced sediment and nutrient loading, reduce water temperature, and others. Stream restoration can help meet water quality goals such as complying with total maximum daily loads (TMDLs), source water protection, and habitat improvement for aquatic and terrestrial life. Stream restoration projects may provide pollutant trading and mitigation opportunities where water quality regulatory programs require pollutant reduction within a watershed. Stream restoration is often overlooked as a nutrient reduction strategy in areas where more traditional efforts (e.g., stormwater and agricultural best management practices) have failed to yield the desired water quality improvements.

The purpose of this crediting guidance is to provide technical support to entities that want to better recognize and quantify the water quality benefits of stream restoration practices as part of crediting (including trading) programs. There are many additional benefits of stream restoration projects, but this guidance focuses on water quality issues, particularly phosphorus, nitrogen, sediment, and temperature impairments. The focus of this guidance is the scientific basis of crediting procedures for four categories of stream restoration practices: bed and bank stabilization, riparian buffers, floodplain reconnection, and in-stream enhancement. The programmatic and market aspects of crediting programs are beyond the scope of this guidance.

Overview of Project 1T13: Stream Restoration as a BMP

This guidance document is a result of a broader Water Environment & Reuse Foundation (WE&RF) research effort that included these tasks:

- Conduct a national literature review of available data and existing crediting protocols to support this project.
- Identify stream restoration techniques that have reliable data to support pollutant reduction crediting.
  - Develop pollutant reduction crediting guidance for stream restoration projects that includes:
  - Identification of pollutant parameters of interest to be included in the analysis.
  - Identification of restoration techniques within the stream channel and riparian zone for which pollutant reduction credits should be developed.
  - Develop a protocol for calculating pollutant reduction credits based upon relevant and site-specific variables.
  - Establish a consistent methodology that can be used by regulatory agencies with a protocol for developing needed site-specific data to support use of the methodology.
- Support participating communities as they interface with their appropriate regulatory agencies in assessing the viability of a crediting approach.
- Develop sampling and data collection guidance and protocols for assessing performance.

Other project deliverables and a supporting Stream Restoration Database are available on the Stream Restoration Database website (WRF, n.d.).

CHAPTER 2

Crediting Concepts

Environmental crediting concepts have been developed and are continually evolving via state and federal water quality trading policies and programs and as part of stream and wetland related mitigation under Section 404 of the Clean Water Act. These two conceptual areas are important background because they represent well-vetted approaches accepted by regulatory agencies and others.

2.1 Water Quality Trading

Stream restoration crediting concepts are rooted in the concept of water quality trading, with early policy statements from the U.S. Environmental Protection Agency (EPA) issued over 20 years ago (EPA, 1996).

EPA's current water quality trading policy was released in 2003 and identifies these elements of a viable trading program:

- Clearly defined units of trade.
- Use of standardized protocols to quantify pollutant loads and reductions.
- Provisions to address the uncertainty of traded nonpoint source loads and reductions.
- Accountability mechanisms for all trades.
- Public participation and access to information.
- Monitoring and program evaluation.

EPA's 2003 policy also discusses Clean Water Act (CWA; EPA, 2003) requirements that are relevant to water quality trading including requirements to obtain permits, anti-backsliding provisions, development of water quality standards including antidegradation policy, National Pollutant Discharge Elimination System (NPDES) permit regulations, total maximum daily loads (TMDLs), and water quality management plans. Although water quality trading is encouraged, it is only allowed under specific conditions. For example, water quality trading is allowed to maintain compliance with water quality standards or to improve water quality where standards are not currently being met (e.g., as a part of a TMDL agreement). However, this type of trading is not allowed to meet technology-based standards under the NPDES program.

Water Quality Trading (as defined by EPA, 2003)

Water quality trading is a market-based approach to improve and preserve water quality. Trading can provide greater efficiency in achieving water quality goals in watersheds by allowing one source to meet its regulatory obligations by using pollutant reductions created by another source that has lower pollution control costs. EPA's policy endorses trading as an economic incentive for voluntary pollutant reductions from point and nonpoint sources of pollution and as a way to achieve ancillary environmental benefits such as creation of habitat.

EPA's policy supports trading of nutrients (e.g., total phosphorus, total nitrogen) and sediment load reductions. The policy recognizes the potential for environmental benefits from trading of pollutants other than nutrients and sediments but believes that these trades may warrant more scrutiny. The policy does not support any trading activity that would cause a toxic effect, exceed a human health criterion, or cause an impairment of water quality. EPA does not support trading of persistent bioaccumulative toxic pollutants at this time.

In 2019, the EPA released a memo reaffirming the agency's support for water quality trading programs and noting some confusion and barriers that have limited the development of these programs. This memo identifies six market-based principles that should underly water quality trading programs (EPA 2019):

- States, tribes, and stakeholders should consider implementing water quality trading and other market-based programs on a watershed scale, rather than being limited to jurisdictional boundaries.
- Adaptive management strategies should be used to implement market-based programs, allowing for programs to evolve over time and supporting data collection for credit verification.
- Water quality credits and offsets may be banked for future use to encourage early adoption and reduce risk.
- Simplicity and flexibility are encouraged when implementing baseline concepts to reduce barriers to entry and avoid uncertainty.
- A single project may generate credits for multiple markets (for example, water quality and habitat benefits can both be credited separately).
- Financing opportunities exist to assist with deployment of nonpoint land use practices (for example, bonds, Clean Water Act section 319 grants, and state revolving loan funds).

While water quality trading has focused on both point sources (e.g., wastewater treatment) and non-point sources (e.g., agriculture), increased nutrient loading and decreased nutrient removal function in degraded stream systems have not been recognized and included. Stream restoration is an increasingly common practice that can influence water quality in degraded systems. Water quality credits are the currency of water quality trading programs; this guidance outlines methods for quantifying these credits for stream restoration projects. This crediting guidance focuses on the technical underpinnings of crediting approaches for stream restoration projects, as opposed to programmatic or regulatory considerations related to crediting programs. Thus, the primary focus of this document includes protocols for quantifying pollutant loads and reductions from stream restoration practices (EPA policy element #2) and provisions to address uncertainty related to stream restoration credits (#3), with some supporting discussion for units of trade (#1) and monitoring (#6). For a contemporary discussion of the programmatic considerations related to water quality trading programs, readers are directed to the Water Environment Federation publication *Advances in Water Quality Trading as a Flexible Compliance Tool* (WEF, 2015).

Advances in Water Quality Trading as a Flexible Compliance Tool
(WEF, 2015)

In 2015, the Water Environment Federation completed a special publication that provides a comprehensive overview of water quality trading approaches and programmatic, policy, and legal considerations. Case studies of water quality trading programs are provided for the Chesapeake Bay states, Great Miami River Watershed (Ohio), Minnesota, California (Tahoe and Santa Rosa), North Carolina, Connecticut (Long Island Nitrogen Credit Exchange), Oregon, and Ontario (the South Nation River and Lake Simcoe). The case studies include information on the specifics of the trading program, the drivers, program development and implementation, and future directions.

2.2 Quantifying Ecosystem Restoration under Clean Water Act Section 404 and Rivers and Harbors Act Section 10

From a technical perspective, policies described in CWA Section 404 and River and Harbors Act (RHA) Section 10 are relevant to this discussion because they address quantifying benefits of ecosystem restoration. Both the scientific literature on stream systems and experience with the 404 permitting program indicate that a one-size-fits-all number or formula at a national or state level is scientifically indefensible. Instead, a regional stratification is necessary for a science-based assessment of benefits; for example, hydrogeomorphic type is commonly used for stratifying wetland systems and a variety of approaches are used for streams.

In 2008, the U.S. Army Corps of Engineers (USACE) and EPA jointly issued regulations clarifying compensation requirements for losses of aquatic resources (USACE and EPA, 2008). While compensation is certainly distinct from voluntary restoration, the recommendations in this rule for adopting a watershed approach for project site selection and design is particularly applicable to this guidance.

The watershed approach is an analytical process for making site selection and design decisions that support the sustainability or improvement of aquatic resources within a watershed and goes beyond evaluation of a single site. It involves consideration of watershed improvement needs, and how locations and types of restoration projects address those needs. This approach uses a landscape perspective (e.g., consideration of landscape scale impacts and benefits when evaluating pollutant contributions, source areas, and existing and future infrastructure and development) to identify the types and locations of projects that will benefit the watershed and restore aquatic resource functions and services.

Watershed Approach: A comprehensive planning approach for site selection and design that incorporates knowledge of the history, physical processes, and social context in a watershed to more effectively achieve restoration goals. Importantly, this approach considers a systems perspective rather than only addressing site-specific concerns.

The watershed approach involves consideration of historic and potential aquatic resource conditions, past and projected aquatic resource impacts in the watershed, and terrestrial connections between aquatic resources. Projects should be located where they are most likely to successfully restore lost functions and services, taking into account such watershed scale features as aquatic habitat diversity, habitat connectivity, relationships to hydrologic sources (including the availability of water rights), trends in land use, ecological benefits, and compatibility with adjacent land uses. Spatial considerations also require careful consideration in water quality crediting programs because environmental benefits may be realized at very different spatial scales. If the desired outcome is water quality improvement of the stream itself (as opposed to a downstream receiving water, such as a lake or reservoir), it is generally illogical and inadvisable to restore a stream in one watershed with the goal of offsetting pollutant loading in an adjacent watershed. However, watersheds may be defined at vastly different scales. Womble and Doyle (2012) indicate that potentially acceptable scales for mitigation are eight-digit HUC for urban watersheds and six-digit HUC for rural watersheds, but should be set based on the scope of the impacts being mitigated. For the purposes of this crediting guidance, smaller watersheds (12- or 14-digit HUC scale), is recommended.

Under the 2008 mitigation rule (USACE and EPA 2008), performance standards must be based on attributes that are objective and verifiable. Ecological performance standards must be based on the best available science that can be measured or assessed in a practicable manner. Performance standards may be based on variables or measures of functional capacity described in functional assessment methodologies, measurements of hydrology or other aquatic resource characteristics, and/or comparisons to reference aquatic resources of similar type and landscape position. Reference aquatic resources are often used to establish performance standards to ensure that those performance standards are reasonably achievable, as this approach can encompass the range of variability exhibited by the regional class of aquatic resources from both natural processes and anthropogenic disturbances.

Performance standards based on measurements of hydrology should take into consideration the hydrologic variability exhibited by reference aquatic resources. Where practicable, performance standards should take into account the expected stages of the aquatic resource development process, in order to allow early identification of potential problems and appropriate adaptive management. These performance standards recommendations are directly applicable to the development of a stream restoration crediting program.

Despite the 2008 mitigation rule, there remains substantial uncertainty in the efficacy of wetland and stream mitigation and restoration projects to meet ecological objectives (Bernhardt and Palmer 2011, Palmer et al., 2010). This may be partly due to a lack of effective monitoring and evaluation practices (Morgan and Hough, 2015), but also may be attributed to a failure of mitigation projects to meet even basic administrative and regulatory requirements (Palmer and Hondula, 2014). This uncertainty extends beyond mitigation projects to stream restoration in general. Restoration effectiveness varies between practice types and remains sensitive to project-specific definitions of success (Palmer et al., 2014). Thus, when developing a crediting program, uncertainty should be transparently acknowledged and incorporated via credit multipliers or other factors of safety that err on the side of conservatism.

Although the stream restoration projects addressed in this water quality crediting guidance are voluntary projects, as opposed to required mitigation projects under the CWA or RHA, many principles associated with these regulatory mitigation requirements are transferable to development of a water quality crediting framework.

2.3 Water Quality Trading and TMDLs

The EPA allows water quality trading programs as long as the achievement of water quality goals is not delayed and applicable TMDL load and wasteload allocations are met. In the Chesapeake Bay, for example, these requirements preclude point to non-point source trading, meaning only trading between like source types is allowed (Wiedeman and Trask, 2001). When meeting regulatory requirements, such as TMDL loading caps, scale issues become important. Watershed scale nutrient and sediment loadings are often estimated with coarse models that apportion loadings between sources (e.g., agriculture and channel erosion) and may incorporate delivery ratios and other factors to account for transport processes not explicitly modeled. Load reduction benefits of an individual restoration project are quantified at a site-scale, meaning these two estimates may not be directly comparable. A crediting program may have to account for this discrepancy and uncertainty using safety factors and/or trading ratios (see Section 7.2). Furthermore, the water quality goal targeted by a TMDL can influence the development of trading

Stream Restoration and MS4 Permits

Stream restoration has the potential to improve water quality and may help MS4 permit holders meet stormwater goals. However, there is significant regulatory confusion about if and how stream restoration projects can be "counted" as stormwater BMPs. Much of this uncertainty lies around the definition of jurisdictional waters – if a stream is a jurisdictional water, the stormwater must be treated *before* it reaches the channel, meaning treatment *within* the channel may not be allowed. However, these stream restoration projects can often provide greater water quality benefits than would be feasible (or cost-effective) with traditional stormwater designs (Earles et al., 2020). There is a desire among stormwater practitioners to use novel in-stream water quality treatment techniques, but regulatory hurdles are often perceived as too great for this to be practical (Herzog et al., 2019). However, there is precedent for off-site stormwater crediting (Parrish, 2018), and it is possible a similar regulatory approach could be applied to stream restoration projects.

programs. If the goal is improved water quality in the local watershed, direct credit quantification of an individual project is appropriate. If water quality of a large downstream receiving waterbody is the target (e.g., the Chesapeake Bay), more intensive modeling or credit discounting may be required to account for the impact of the individual project on nutrient or sediment delivery to the receiving water. This is not to discount the potential local benefits of stream restoration projects or their usefulness in attaining TMDL goals; however, considering the spatial scale of interest is essential when developing a crediting program.

Fundamental Elements of a Restoration Plan
Under 2008 Mitigation Rule (USACE and EPA 2008)

1. Objectives: A description of the benefit(s) and amount that will be provided, the method of restoration, and how the anticipated functions of the project will address watershed needs.

2. Site Selection: A description of the factors considered during the site selection process. This should include consideration of watershed needs and the practicability of establishing an ecologically self-sustaining project site.

3. Site Protection: A description of the legal arrangements and documentation of site control or ownership, and demonstration of arrangements for the long-term protection of the project site.

4. Baseline Information: A description of the pre-project ecological characteristics of the proposed project site. This may include descriptions of historic and existing conditions.

5. Determination of Credits: A description of the number of credits to be provided including a brief explanation of the rationale for this determination.

6. Work Plan: Detailed written specifications and work descriptions for the project, including: construction methods, timing, and sequence; source(s) of water; methods for establishing the desired plant community; plans to control invasive plant species; proposed grading plan; soil management; and erosion control measures.

7. Maintenance Plan: A description and schedule of maintenance requirements to ensure the continued viability of the project site once initial construction is completed.

8. Performance Standards: Ecologically-based standards that will be used to determine whether the mitigation project is achieving its objectives. These are often tailored to the region or even the individual site.

9. Monitoring Requirements: A description of parameters monitored to determine whether the project is on track to meet performance standards, and if adaptive management is needed. A schedule for monitoring and reporting monitoring results must be included.

10. Long-term Maintenance Plan: A description of how the project will be managed after performance standards have been achieved to ensure the long-term sustainability of the site, including long-term financing mechanisms and identification of the party responsible for long-term management.

11. Adaptive Management Plan: A management strategy to address unforeseen changes in site conditions or other components of the project.

12. Financial Assurances: A description of financial assurances that will be provided, and how they are sufficient to ensure a high level of confidence that work on the project will be successfully completed in accordance with its performance standards.

Source: Data from USACE and EPA, 2008.

CHAPTER 3

Stream Restoration Practices for Consideration in Water Quality Crediting Programs

A wide range of stream restoration practices have been implemented to achieve multiple purposes; however, not all stream restoration practices are currently well suited for the purpose of establishing water quality credits. Characteristics of stream restoration approaches that are good candidates for crediting programs include:

- The approach has been successfully implemented across multiple regions and stream types.
- The nutrient or sediment removal or temperature reduction processes enhanced by the approach are well understood.
- Scientific literature on the approach provides an adequate and defensible empirical basis for quantitatively estimating reductions in nutrient or sediment loading or net improvements to in-stream water quality that result from the approach across regions and stream types.
- The approach is self-sustaining and generally requires limited long-term maintenance.
- The approach is not likely to cause adverse effects onsite or offsite (e.g., downstream sediment starvation, reduced fish passage, etc.).
- The approach provides additional functional benefits beyond nutrient removal or sediment load reduction.

What is Stream Restoration?

Stream restoration has many definitions, but may be best summarized as "assisting the establishment of improved hydrologic, geomorphic, and ecological processes in a degraded watershed system" (Wohl et al., 2005). Importantly, this definition focuses on restoring processes, rather than simply creating a desired river form. Other terms, including enhancement and rehabilitation, are used to describe stream improvement. The focus of this Guidance is practices aimed at addressing specific nutrient-related processes regardless of whether they conform to a specific definition of restoration.

Based on these criteria and a literature review of stream restoration studies nationally (Lammers, 2015), practices currently considered appropriate for use in water quality crediting programs include: bed and bank stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection. Table 3-1 provides an overview of these four practice areas, along with specific techniques, metrics of interest, scales of application, potential interactions, and suitability of these practices to different stream types. As research advances and more data become available for other restoration practices, this guidance may grow to include other practices.

Although the selected practices have the most empirical evidence of their ability to improve nutrient retention and processing, the success of stream restoration projects for improving water quality remains somewhat equivocal (Doyle and Shields, 2012; Palmer et al., 2014; Lammers, 2015). This guidance recognizes this uncertainty and much of the subsequent discussion focuses on quantifying the processes responsible for nutrient retention at each project, rather than assuming a functional uplift without site-specific monitoring. This contrasts with some mitigation programs that assume a 1:1 replacement length for restored versus impacted sites. Project size can be an important determinant of project success.

Larger projects may be less susceptible to watershed changes, increasing longevity, and certain restored functions (e.g., flood attenuation) may require greater restored length (Doyle and Shields, 2012). While longer restoration projects are likely to have a greater influence on water quality, the credit quantification approaches described here have no inherent size-dependence.

This crediting guidance document assumes that the reader is familiar with stream restoration terminology and concepts. Examples of references for additional information include Roni and Beechie (2013), NRCS (2007), FISRWG (1998), Soar and Thorne (2001), Wohl et al. (2005), Yochum (2018), and Harman et al. (2012), among others.

Table 3-1. Examples of Stream Restoration Practices Potentially Suitable for Water Quality Crediting.

	Practice Category			
	Bed and Bank Stabilization	**Riparian Buffers**	**In-Stream Enhancement**	**Floodplain Reconnection**
Specific Techniques	• Bioengineering • Vanes (partial or full channel span) • Drop structures and weirs (bendway or channel spanning) • Spur dikes • Toe wood • Rock walls • Riprap • Constructed riffle • Guidebanks	• Active planting (grass and/or trees) • Grazing management (livestock exclusion/fencing)	• Log jams • Beaver dams • See Bed and Bank Stabilization for others	• Breaching levees • Bank lowering • Raising stream bed • Floodplain (e.g., legacy) sediment removal
Metrics of Interest	• Erosion rates (mass or volume per time per stream length) • Soil nutrient content (mass nutrient per mass soil) • Soil bulk density (mass of soil per volume) • Total nutrient and sediment loading rates	• Groundwater inflow and outflow rate • Inflow and outflow sediment & nutrient concentrations • Inflow and outflow loads (product of above) • Nutrient uptake/ removal rates on per area basis (e.g., denitrification) • Canopy shading and solar radiation	• Nutrient uptake/removal rates on per area basis (e.g., denitrification) • Hyporheic flow rates/ percentage of baseflow • Volume of hyporheic zone or area of hyporheic exchange • Biochemical potential of subsurface (may be qualitative)	• Floodplain inundation frequency and duration • Typical nutrient removal rates/effluent nutrient concentrations • Sediment and nutrient deposition rates • Nutrient load to floodplain
Scales	Can be reach to small watershed scale. Larger scale implementation will result in more effective pollutant retention overall, but may be cost prohibitive	Can be reach to watershed scale. Larger scale implementation (i.e., fewer “gaps” in buffers) will lead to greater nutrient and sediment retention.	All spatial scales, although site to reach scale typical. Hyporheic exchange potential increases with number of structures	Typically reach scale but could be larger.
Interactions	Stabilization techniques will be more successful if root causes of degradation (e.g., altered hydrology, livestock trampling) are also addressed	Riparian buffers can increase bank stability (i.e., root reinforcement) and provide organic carbon and large wood to streams, which can increase nutrient processing	Many in-stream structures are installed to increase bed and bank stability but also encourage in-stream processing. Bed particle size, carbon content/carbon retention potential may influence nutrient removal rates	Can promote groundwater/hyporheic denitrification (similar to riparian buffers and in-stream enhancement) as well as overbank deposition and nutrient retention
Suitability	Appropriate where fine-grained banks are unstable and eroding. Not suitable where only outside of meander bends are subject to erosion in a naturally migrating stream	Suitable for areas with significant loading of sediment and/or nutrients from adjacent upland sources. Should have saturated root zones for denitrification/nitrogen removal crediting	Suitable for streams with high potential for hyporheic exchange (e.g., coarse bed material) where loss of bedforms or complexity has limited this natural process	Appropriate in systems with regular flood peaks and unconfined valleys. Less suitable where overbank flows are naturally infrequent (e.g., ephemeral streams)

References for More Information on Stream Restoration Terminology and Approaches

Many publicly accessible references are available to guide stream restoration practice, although they do not specifically address crediting concepts. A few key examples are listed below.

- Natural Resources Conservation: The NRCS is a key source of information on stream restoration practices, providing two National Engineering Handbooks, design guidance, and multiple case studies of stream restoration projects.
    - Stream Corridor Restoration (National Engineering Handbook 653; FISRWG, 1998): This manual is also known as the Federal Stream Corridor Restoration Handbook (NEH-653) and titled *Stream Corridor Restoration: Principles, Processes, and Practices.* This handbook was prepared in 1998 by the Federal Interagency Stream Restoration Working Group (FISRWG), including 15 federal agencies. This interagency document was intended to help plan stream corridor restoration projects; however, it does not provide specific design criteria for various practices.
    - Stream Restoration Design (National Engineering Handbook 654; NRCS, 2007): This manual was completed in 2007 to provide NRCS specialists and field personnel with design tools for use in designing stream restoration projects, essentially taking the 1998 manual to the next level. The primary emphasis of Handbook 654 is on "how-to" techniques. The manual was developed based on input from the NRCS, stream and aquatic ecology experts from a variety of federal, state, and local agencies, as well as private consultants and universities.
- U.S. Army Corps of Engineers:
    - Copeland, R. R., D. N. McComas, C. R. Thorne, P. J. Soar, M. M. Jonas, and J. B. Fripp. 2001. Hydraulic design of stream restoration projects. Vicksburg, MS: US Army Engineer Research and Development Center. Coastal and Hydraulics Laboratory.
    - Fischenich, J. C. 2006. Functional Objectives for Stream Restoration. Vicksburg, MS: US Army Engineer Research and Development Center.
    - Soar, P. J., and C. R. Thorne. 2001. Channel Restoration Design for Meandering Rivers. Vicksburg, MS: US Army Engineer Research and Development Center.
- Shields, F. D., R. R. Copeland, P. C. Klingeman, M. Doyle, and A. Simon 2003. Design for Stream Restoration. *Journal of Hydraulic Engineering* 129(8): 575-584.
- Wohl, E., Angermeier, P.L., Bledsoe, B.P., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., Tarboton, D. 2005. River restoration. Water Resources Research, 41: W10301. doi:10.1029/2005WR003985.
- U.S. Fish and Wildlife Service: multiple references, available through the Chesapeake Bay Field Office (USFWS, 2020)

3.1 Bed and Bank Stabilization

> **Bed and Bank Stabilization:** Any technique that prevents erosion of the channel boundaries, by either reducing the erosive power of the stream or increasing the erosion resistance of the bed or bank material.

Bed and bank stabilization includes direct channel modifications such as installation of grade control structures to prevent incision (and increase bank stability via toe protection) as well as direct bank stabilization. Bank stabilization practices may be resistive (increasing bank resistance to erosion) or redirective (reducing the erosive power of the stream by redirecting the flow). Resistive practices include various combinations of riprap, revetments, and bioengineering (vegetation). Redirective practices include vanes, j-hooks, spurs/groynes, bendway weirs, and guidebanks. Grade controls can include log and rock drop structures.

Regardless of the specific design, erosion protection techniques attempt to reduce erosion of the bed and/or banks, leading to more stable channel form and reduced sediment and nutrient loading. It is important to note that bank erosion is a beneficial natural process that allows channel migration and provides sediment to streams. Complete cessation of bank erosion is neither practical nor desired (Florsheim et al., 2008); however, changes in watershed hydrology or sediment supply can significantly destabilize channels, leading to accelerated bank erosion, increased sediment and nutrient loading, floodplain disconnection, and channel incision and widening (Booth, 1990). Preventing these adverse impacts early in the sequence of channel evolution should be the goal of bed and bank stabilization projects.

Allowing for natural channel mobility and adjustment is an ideal goal of stream restoration projects. However, in urban and other settings where streams are often highly constrained by infrastructure and adjacent land uses, some form of channel armoring may be the only practicable alternative. In such cases, it is important to recognize that although some measurable nutrient reductions may be achieved by armoring unstable channels in certain situations, these reductions may be accompanied by other undesirable side effects. Examples include habitat degradation and an increased risk of shifting the erosion elsewhere, either by redirecting erosive flows to another location or by inducing downstream instability because of decreased sediment supply. As a result, nutrient reductions upstream may be negated by downstream instability of channels that are adjusted to the prevailing upstream sediment load. It follows that different levels of crediting may be warranted for bank stabilization projects that create a static channel versus those that reestablish dynamic channel processes with an understanding of the response potential of downstream channels. This underscores the need to consider local actions in a broader context by assessing and monitoring downstream segments during planning and post-project evaluation.

In the context of bank stabilization, terms such as bioengineering refer to a broad range of practices ranging from exclusive reliance on plant materials to hybrid approaches that combine armoring and vegetation (Bentrup and Hoag, 1998). For example, a combination of longitudinal stone toe, soil lifts, an emplaced carbon source such as sawdust, and willow plantings will most likely result in a laterally armored channel that still has appreciably more riparian function than a channel with a riprap blanket. Thus, there is a spectrum of potential benefits for both bank stabilization and riparian functions that depend on the selected bioengineering approach.

Nutrient reduction quantification requires estimates of both bank and bed erosion rates (i.e., volume or mass of sediment eroded per length of channel per year) as well as the nutrient content of the sediment (e.g., mass phosphorus per mass of sediment). The product of these two values yields the total nutrient loading rate. This analysis can be used to predict avoided nutrient loading via bed and bank stabilization.

There is substantial uncertainty in predicting future erosion rates and determining the proportion of future erosion that may be avoided by channel stabilization. For example, the Channel Evolution Model (CEM) predicts a sequence of channel erosion from incision (bed lowering) and subsequent widening, followed by deposition and eventual stabilization (Figure 3-1). Arresting this process early in the sequence will likely provide greater avoided sediment and nutrient loading than installing erosion protection in the aggradation/deposition stage. Variations of this CEM have also been proposed, recognizing that other channel forms such as braiding can occur and that channel evolution is largely dependent on site-specific factors such as hydrology, relative resistance of bed and banks, and vegetation (Booth and Fischenich, 2015; Cluer and Thorne, 2014; Hawley et al., 2012).

In addition to considering potential for future channel evolution, incorporating knowledge about the current watershed hydrology is essential. Bank stabilization structures may be more prone to failure in urban watersheds that have a flashy hydrologic regime and ineffective stormwater controls. Similarly, bioengineering bank stabilization projects that rely exclusively on plant materials may have a higher risk of failure in the first few years as vegetation establishes. Furthermore, stabilization structures are prone to being “flanked” by the flow, which renders them ineffective and may actually contribute to accelerated erosion (Miller and Kochel, 2013). Erosion protection in one reach could “starve” a downstream reach of sediment and initiate instability (Kondolf, 1997). Focusing solely on increasing erosion resistance of the bed and banks may simply move the erosion problem elsewhere; reducing the erosive power of flow (e.g., stormwater controls or changing channel slope and cross section) may be more effective. These complex factors make quantification of avoided nutrient and sediment loading difficult and this uncertainty should be explicitly accounted for in crediting procedures. For example, this uncertainty may be incorporated by taking a risk-based or probabilistic approach to modeling bank erosion and channel evolution.

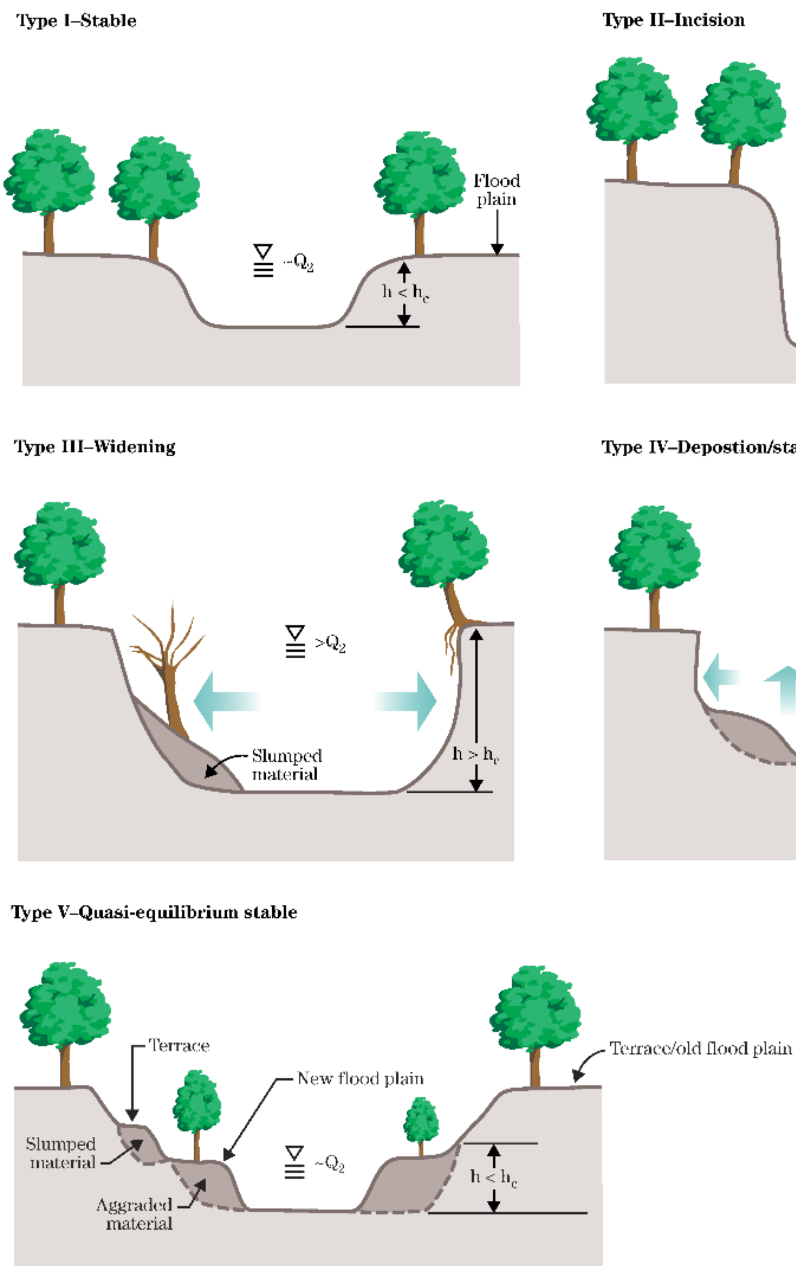

Figure 3-1. Incised Channel Evolution Model.

Note how installing channel control structures in Stage II can prevent significant sediment loading and widening predicted in Stage III but later intervention may result in less avoided sediment and nutrient loading. [Q_2 is the instantaneous peak flow with a two-year recurrence interval; h is the bank height; h_c is the critical bank height for failure; $\underset{\equiv}{\nabla}$ is the water surface elevation.

Source: NRCS 2007, based on Schumm et al. 1984.

3.2 Riparian Buffers

Riparian Buffers: Protected or replanted vegetated areas adjacent to stream channels that can intercept pollutants in surface and subsurface flow. Buffers can also shade the stream and reduce water temperatures.

Preservation or restoration of minimally disturbed vegetated areas adjacent to streams is a common strategy aimed at reducing pollutant discharges to streams by retaining and processing pollutants in both the surface and subsurface. Surface removal occurs primarily through deposition of sediment-bound nitrogen and phosphorus (and other pollutants). Subsurface removal can occur via plant uptake and microbial metabolism. However, in some cases, reducing conditions (which may be beneficial for subsurface nitrate removal) can cause phosphorus dissolution and increase groundwater phosphorus concentrations (Newbold et al., 2010). Additional benefits of riparian buffers include increased bank stability, stream shading to reduce water temperatures, and supply of in-stream wood and organic carbon to streams. Buffers may consist of predominately grasses, trees, or some mixture of both. Establishment and protection of intact riparian buffers may require some combination of direct planting and grazing management/livestock exclusion.

Buffer effectiveness is site-specific and depends on multiple factors. Nutrient removal in buffers tends to increase with increasing buffer width, decreasing slope, increased vegetation density and maturity, increased soil permeability, and other factors. For example, buffers wider than 50 m removed more nitrate than 0-25 m wide buffers (Mayer et al., 2007), although buffers as narrow as 30 m have shown very high removal efficiency (Pinay et al., 1993). There is evidence that nitrate removal in grass or forested buffers is not significantly different (Mayer et al., 2007). Likely more important is the interaction between the organic carbon-rich soil within the rooting zone and the nitrate-laden groundwater. Furthermore, denitrification (microbial conversion of nitrate [NO_3^-] to nitrogen gas [N_2]) often occurs at discrete points in space ("hot spots") and in time ("hot moments") when conditions are favorable (i.e., available nitrate, carbon, and anoxia). In agricultural areas, tile-drains may route groundwater through riparian buffers with little or no nutrient removal, limiting the effectiveness of these buffers for subsurface nutrient removal.

Buffers can decrease stream water temperatures by increasing shade, thereby reducing solar radiation inputs. Large trees provide more shade than shrubs and grasses, and smaller streams can be shaded more effectively than large rivers. Other factors also influence the temperature reduction benefits of buffers. For example, shading will be greatest on sections of streams oriented north-south, and on areas with slow moving water that would be especially susceptible to warming in full sun (Jackson et al., 2017). Dense stands of conifers can reduce water temperature more than open, deciduous woodlands (Dugdale et al., 2018), although there may be tradeoffs between shade and other benefits (e.g., habitat).

Quantification of nutrient retention capabilities of riparian buffers requires an assessment of both the volume of groundwater and surface runoff flowing through the buffer and the nutrient concentration at both the upslope and stream edge sides. Groundwater flow rates and nutrient concentrations are temporally and spatially variable which complicates accurate quantification of nutrient retention rates. Actual rates of nutrient removal may also be used to estimate total nutrient load reductions. Nutrient removal is likely to occur through plant uptake, microbial assimilation, denitrification, and other processes, all of which are temporally and spatially variable.

Riparian buffers also allow for sedimentation, filtration, and adsorption. Additionally, these processes may have been occurring to some extent in the degraded riparian area; therefore, post-restoration removal rates must be compared to this pre-restoration condition. Furthermore, buffers may take time to establish, and substantial nutrient removal will likely not be observed immediately.

Riparian buffers may be less effective at reducing in-stream nutrient concentrations and temperatures if they are installed in a patchwork throughout a watershed. (Realistically, some breaks in buffer continuity are likely to occur.) Intact buffers (i.e., no or few breaks along a stream length) provide the greatest potential for improved water quality, even if total buffer length is equal (Figure 3-2). Gaps in non-continuous buffers allow unimpeded nutrient loading which reduce the benefits of established buffers (e.g., Collins et al., 2013), assuming all other characteristics are equal. Similarly, buffer gaps cause stream warming, and these higher water temperatures may persist even as the stream flows through intact buffers downstream (Coats and Jackson, 2020). Additionally, the vast majority of stream length in a watershed is in small, headwater streams that naturally have a greater connection to the uplands and therefore a greater potential benefit from riparian buffers. These headwater streams may also see a greater in-stream water quality improvement due to their smaller size relative to the buffers. There are also significant interactions between riparian buffers and bank stabilization. On one hand, buffers increase bank stability through root reinforcement. Conversely, unstable banks and incising channels may result in undercutting of the riparian zone, which lowers groundwater tables and reduces nutrient removal because of a disconnection with the all-important rooting zone.

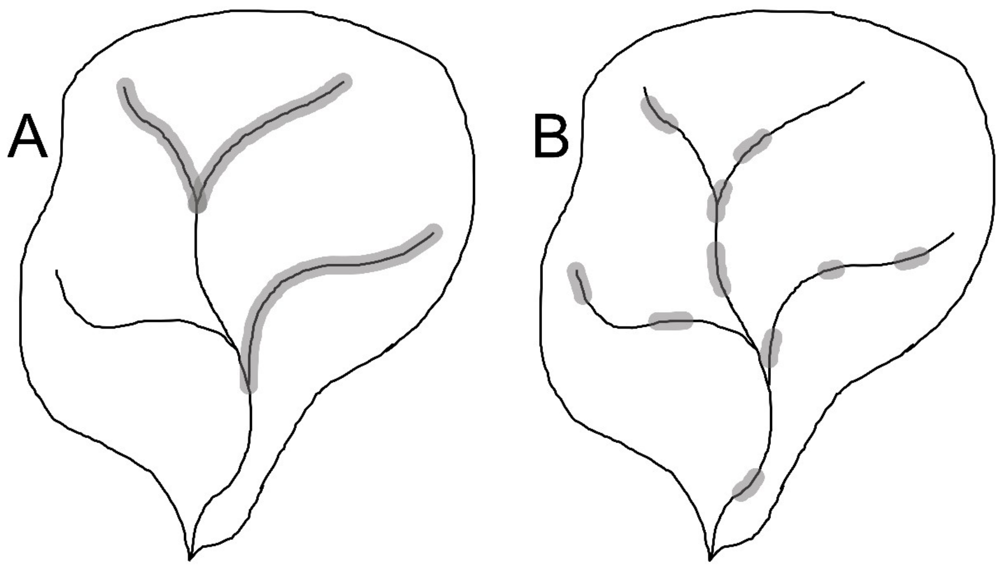

Figure 3-2. Riparian Buffer Distribution Scenarios.
Conceptualization of riparian buffers distributed throughout a watershed. Continuous buffers (A) result in greater water quality improvements and would receive more nutrient reduction credits than discontinuous buffers (B), even if the total length of buffer is the same, assuming all other characteristics are equal.

3.3 In-Stream Enhancement

In-Stream Enhancement: Modifications within the stream channel that increase geomorphic complexity, encourage hyporheic exchange, and/or enhance aquatic habitat.

A common symptom of stream degradation is simplification of the channel[1] and associated loss of habitat. Primarily with a focus on improving fish and benthic invertebrate habitat, many stream restoration projects attempt to increase geomorphic complexity via installation of structures. Structures vary widely and can include j-hooks, cross-vanes,

[1] Channel simplification refers to the loss of complex channel features, such as pools and riffles, in-stream wood, and bars that drive hyporheic exchange processes and provide important habitat for fish and macroinvertebrates. Causes of channel simplification may be direct (e.g., dredging or channelization) or indirect (e.g., altered sediment or flow regime which causes channel erosion or deposition).

constructed riffles, and log jams. Beaver reintroduction is also a potential restoration strategy and has the added advantage of being self-sustaining as beavers reconstruct and repair dams following damaging flood events. These structures can increase habitat heterogeneity and may increase nutrient retention and cycling by increasing hydraulic retention time and encouraging hyporheic exchange (Figure 3-3). Simply adding large wood to streams, either directly or indirectly through establishment of riparian buffers, can also increase channel complexity and hyporheic flow (Roberts et al., 2007). There is also recent work to design sub-grade structures to specifically encourage hyporheic exchange and associated biochemical processing (Herzog et al., 2015). Hyporheic flow potential at individual structures can be assessed by considering the difference in hydraulic head across the structure, along with knowledge about the hydraulic conductivity of the bed material. To quantify nutrient removal and retention, it is also important to consider the availability of organic carbon and oxygen in the subsurface.

The percentage of total flow that moves through the hyporheic zone at an individual structure is typically small (1% or less; Azinheira et al., 2014; Gordon et al., 2013) but the cumulative effect of multiple structures could be significant and result in particulate retention and denitrification. Monitoring using tracers, temperature measurements, or more detailed hydraulic modeling can be used to quantify hyporheic exchange and biochemical transformations at the reach scale. Hyporheic exchange and nutrient retention will likely be highest during baseflow whereas structures likely provide little or no nutrient retention benefits during storm events. Organic carbon availability is also an important control on in-stream metabolism and nutrient cycling. At small scales, organic carbon can be artificially added to streams and hyporheic zones to jump start denitrification (e.g., Robertson and Merkley, 2009). At larger scales, restoring natural organic carbon fluxes may be an important restoration strategy. This would primarily be accomplished through riparian buffer restoration, which would restore both short- term (i.e., leaf litter) and long-term (i.e., large wood) carbon inputs to streams (Stanley et al., 2012). However, there is insufficient research on restoring carbon inputs for this component of restoration to be considered for quantification and crediting.

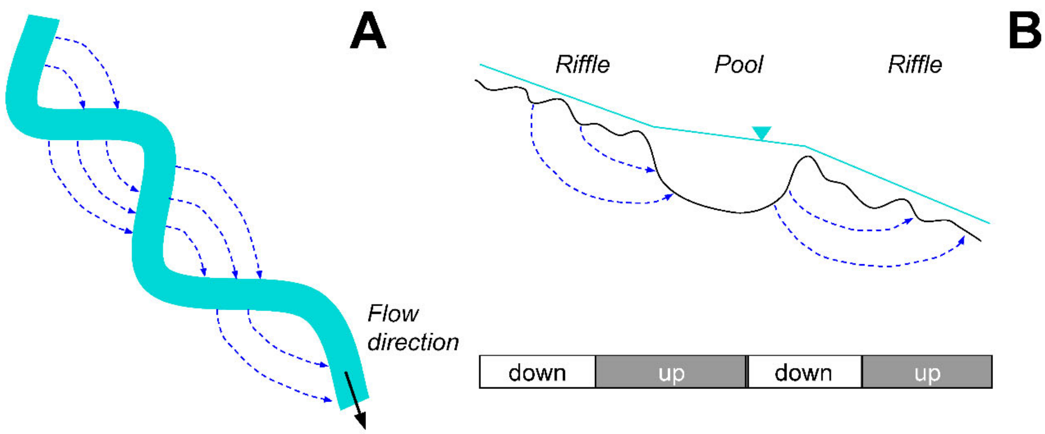

Figure 3-3. Conceptual Diagram of Hyporheic Flow in Natural Channels.

Hyporheic flow paths indicated with dashed lines. Installed structures can induce hyporheic exchange similar to what is observed in natural riffles.

Source: Adapted from Hester and Gooseff 2010. Reprinted with permission. Copyright American Chemical Society.

3.4 Floodplain Reconnection

> **Floodplain Reconnection:** Restoration of the hydrologic connection between the channel and floodplain, allowing for overbank inundation and raised riparian groundwater tables.

Natural floodplains can be important nutrient sinks (e.g., Forshay and Stanley, 2005), a function which is often lost as floodplain-channel connections are severed via channelization, incision, or levee construction (Loos and Shader, 2016). Hydrologic reconnection of floodplains and streams can be achieved by excavating floodplain sediment to lower elevations, removing or breaching levees, raising the channel bed, wood recruitment and log-jam formation, removing infrastructure from floodplains, and complete channel reconstruction.[2] Whatever the method, floodplain reconnection tends to bring the riparian groundwater table closer to the surface and allows for more frequent overbank flows. This increases sediment and nutrient retention and processing while providing ancillary benefits such as enhanced aquatic and riparian habitat, reduced in-channel erosive power and downstream flooding, and increased aquifer recharge. Floodplain reconnection can increase nutrient retention through overbank deposition as well as increased groundwater denitrification in the riparian zone (Fink and Mitsch, 2007; Kronvang et al., 2007; Roley et al., 2012a, 2012b). Construction of a two-stage ditch, a specific form of floodplain reconnection typically associated with agricultural channels, can increase denitrification and phosphorus retention in these nutrient-laden waterways (Davis et al., 2015; Mahl et al., 2015). Increasing stream-groundwater connections (either through floodplain reconnection or in-stream enhancement) can also increase baseflow and lower stream temperatures (e.g., Ponce and Lindquist, 1990); however, these benefits vary by site and cannot be easily quantified.

A more intensive version of floodplain reconnection has recently been applied in some locations – the complete removal of “legacy” sediment in the valley bottom, restoring single-thread meandering channels to their pre-disturbance condition of low-energy wetland complexes. These projects may be cost-effective approaches for reducing sediment and nutrient loading in some situations (Fleming et al., 2019). A recent report from the Chesapeake Bay Stormwater Network (Wood and Schueler, 2020) modified crediting procedures to explicitly account for legacy sediment removal projects for floodplain reconnection and hyporheic zone restoration.

Deposition of sediment and sediment-bound nutrients can be estimated based on inundation frequency and floodplain area. Increased denitrification is more difficult to quantify but should include some assessment of increases in groundwater tables post-restoration and the potential for nitrate-laden groundwater to interact with organic carbon available in near surface soils. Data on the effectiveness of wetlands for nutrient removal may also be used to quantify the benefits of reconnected floodplains (see Wood and Schueler, 2020 and Section 5.5). There have only been limited studies on the impacts of floodplain reconnection on nutrient dynamics, despite the fact that this is a relatively common restoration practice (Bernhardt et al., 2005). Future studies that focus specifically on the impacts of riparian groundwater tables and subsurface nutrient processing post-restoration would be especially useful. Quantification and crediting of nutrient removal via overbank deposition is feasible, but understanding the subsurface nutrient dynamics may be more difficult and require site-specific monitoring.

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[2] Regulatory floodplain considerations are often an important consideration for design and implementation of this practice and should be evaluated accordingly, as is the case for other types of stream restoration practices.

3.5 Practices Not Suitable for Direct Crediting

Dam removal, channel reconfiguration, and watershed process are not currently included in this guidance, primarily because there is insufficient scientific evidence for quantifying their potential nutrient reduction benefits. An independent review of an initial version of this guidance agreed that these practices are not currently suitable for direct crediting (Stack et al., 2018).

3.5.1 Dam Removal

> **Dam Removal:**
> Partial breaching or complete removal of a dam, often with the goal of restoring a more natural flow regime and allowing for unimpeded movement of sediment and aquatic organisms.

Dam removal is an increasingly common restoration approach to improve fish passage and enhance water quality. However, dam removal can have the unintended consequence of increasing nutrient loading in watersheds (e.g., Doyle et al., 2003). Impounded areas are relatively effective at nutrient and sediment retention and dam removal can mobilize this stored material and transport it to downstream waterbodies. It is possible to manage this potential sediment and nutrient source by modifying the dam removal process; for example, by dredging accumulated sediment to use for amending soils elsewhere. Dam removal plans should account for this and other consequences beyond increased longitudinal connectivity. Dam removal and modification is an effective tool for restoring biological integrity of rivers, but its effects on nutrient and sediment pollution are not well enough understood to be considered for crediting.

3.5.2 Channel Reconfiguration

> **Channel Reconfiguration:**
> Significant realignment or complete channel reconstruction to provide a more appropriate channel geometry and planform.

One of the more intensive and expensive stream restoration strategies, channel reconfiguration may entail reconnection of a historically abandoned channel, partial channel realignment, or complete construction of a new channel. This technique is especially valuable in areas with a significantly altered channel geometry and planform (e.g., from channelization) where more passive approaches aimed at restoring natural erosion and depositional processes may not be viable at required time scales. Channel reconfiguration can decrease velocities by reducing slope and increasing sinuosity and is typically accompanied by other stream restoration strategies such as erosion protection and installation of in-stream structures. Reducing in-stream velocities may increase hydraulic residence time and nutrient uptake. Channel reconfiguration can also help balance sediment transport to avoid channel aggradation or degradation or loss of flood capacity.

Since channel reconfiguration is often completed in conjunction with other stream restoration techniques, it is difficult to quantify the pollutant removal benefits of channel changes alone. However, change in hydraulic residence time pre- and post-construction may be a useful surrogate for increased nutrient retention. Changes in hydraulic residence time would require pre and post tracer studies or detailed hydraulic modeling. Translating hydraulic residence time to nutrient retention would then require consideration of the specific biochemical processes occurring in individual systems and is laden with uncertainty. Therefore, only limited nutrient reduction crediting and quantification for channel reconfiguration are recommended as it relates to increased hydraulic retention time. Instead, it is proposed that the primary focus be on other stream restoration techniques that have a more direct and quantifiable impact on nutrient uptake and sequestration.

Channel reconfiguration may reduce downstream sediment delivery by balancing sediment capacity and supply and therefore preventing channel erosion in the restored reach. The benefits of this sediment reduction may be quantified by comparing sediment transport capacities pre- and post-restoration with sediment supplies. However, there is little empirical data supporting this potential benefit of channel reconfiguration and therefore this is not incorporated within this guidance.

The four practices identified for crediting are often associated with some form of channel reconfiguration; however, channel reconfiguration is a somewhat vague term used to describe a broad spectrum of activities that can influence many geomorphic characteristics, but it does not necessarily result in pollutant reductions. Accordingly, this guidance focuses on crediting for more specific characteristics linked to nutrient reductions that may or may not arise as a consequence of channel reconfiguration. Assigning nutrient reduction credits for both channel reconfiguration and characteristics that follow from it also runs the risk of double counting nutrient reduction credits.

3.5.3 Watershed Processes

Watershed Processes: Actions taken throughout a watershed but outside of the stream corridor itself to mitigate the damaging effects of land use change or other disturbances.

Streams are integrators of their environment, and stream restoration is unlikely to be successful without an assessment of changes in the watershed that are contributing to channel degradation (Roni and Beechie, 2013). Alterations to hydrology and sediment and nutrient fluxes are often primary causes of water quality degradation. Addressing these causes, rather than only addressing the symptoms in the channel, may be the most successful restoration strategy to achieve measurable improvements in stream ecosystem function. Combinations of all the above techniques, including addressing watershed processes, is likely to be the most successful restoration approach. Although typically not considered stream restoration, stormwater controls in urban watersheds that attempt to mimic pre-development hydrology, when practical, can significantly improve stream channel conditions and prevent the mobilization of substantial quantities of sediment and nutrients (Lammers et al., 2019). These management practices are commonly referred to as *Low Impact Development* or *Green Infrastructure* approaches and include a variety of techniques such as disconnecting impervious areas, bioretention, grass swales, permeable pavement, and others. These practices may also be beneficially implemented in combination with traditional stormwater and flood control practices such as retention ponds, wetland basins, and regional detention basins.

These urban stormwater practices as well as agricultural BMPs can retain and remove nutrients before they reach the stream and, in certain situations, may be more cost effective than active stream restoration. While these watershed restoration projects may seem independent of in-stream enhancement, they can affect other stream restoration techniques. For example, in-stream nutrient uptake may be highest at low flows so stormwater controls that reduce peak flows can enhance overall nutrient retention. Restoration credits may not be given directly for watershed improvements; however, incorporating watershed-scale restoration techniques (e.g., stormwater controls) may result in conditions that increase the longevity and performance of practices such as bank and bed stabilization by attenuating flashy flow events. Implementation of stormwater controls in the watershed could be a consideration for reducing safety factors applied to other stream restoration credits. For example, a greater percentage of avoided bank erosion could be credited for a bank stabilization project if complementary stormwater controls are also installed in the watershed.

Stormwater controls have the greatest potential benefit to complement restoration projects that aim to create a dynamically stable channel, reducing sediment and nutrient loading downstream. Reversing the negative effects of hydromodification is an ongoing challenge, but there has been significant progress on defining relevant erosion thresholds in streams (Bledsoe et al., 2012; Hawley & Vietz, 2016; Russell et al., 2020) and designing stormwater controls to keep stream flows below these thresholds (Tillinghast et al., 2011). Properly implemented, stormwater controls may have greater potential to reduce channel erosion and pollutant loading than channel stabilization alone (Lammers et al., 2019). Even small retrofits to existing stormwater infrastructure can lead to rapid channel stabilization (Hawley et al., 2020). There is significant potential for well-coordinated stormwater control and channel restoration to improve the ecological function of streams and water quality, but there is currently insufficient quantitative evidence for this technique to be considered in this crediting guidance. These stormwater controls would have to go beyond current practice (e.g., reducing peak flows of the ~2-year, 10-year, and 25-year storms). Additional analysis would be needed to identify the erosion threshold of a stream (Hawley & Vietz, 2016) and specifically design stormwater controls to keep flow rates below this threshold.

CHAPTER 4

Pollutants for Consideration in Crediting Programs for Stream Restoration

Stream restoration can provide a multitude of water quality benefits, as well as broader ecosystem services that are increasingly recognized by communities and researchers. The focus of this crediting guidance is limited to nutrients, sediment, and temperature as discussed below; however, crediting programs could be expanded to include other pollutants (e.g., metals). Additional discussion of nutrient processing in streams is provided since this is an important component of assessing performance of stream restoration projects.

4.1 Overview of Pollutants

Pollutants for potential consideration in crediting programs currently include phosphorus, nitrogen and sediment. Limited discussion of other pollutants that could be considered in the future is also provided.

4.1.1 Phosphorus

Phosphorus is naturally found in soils worldwide, although the abundance and chemical composition are associated with a number of factors including soil texture, pH, metals concentrations, and the geology of the soil parent material (Brady and Weil, 2002). Total phosphorus content of streambanks and riparian soils is also correlated with these factors (Palmer-Felgate et al., 2009). However, the silt-clay content may be the largest driver in some catchments (Agudelo et al., 2011; Bledsoe et al., 2000; Cooper and Gilliam, 1987; Young et al., 2013, 2012), but not others (Hongthanat, 2010; Howe et al., 2011; Kerr et al., 2011; Schilling et al., 2009; Veihe et al., 2011). Streambank phosphorus concentrations may also be higher in intensively farmed catchments (Palmer-Felgate et al., 2009) or in deforested areas (Haggard et al., 2007), although others have shown little correlation to land use (Nellesen et al., 2011; Tufekcioglu, 2010; Zaimes et al., 2008a). Anthropogenic impacts have altered the global phosphorus cycle, primarily through the mining of phosphorus-bearing rock to meet the increasing demand for agricultural fertilizer. Cropland fertilizer application has in some cases led to the ongoing accumulation of phosphorus in soils, where it can become a potential source of water pollution (Carpenter et al., 1998; Smith et al., 1999).

Sources of phosphorus are generally similar to nitrogen, although atmospheric deposition is not as significant. Numerous efforts have been made to identify and quantify the various sources of phosphorus pollution in watersheds (e.g., DeWolfe et al., 2004; Kronvang et al., 1997; Sharpley and Syers, 1979). Recent evidence has made it increasingly clear that bank and bed erosion may be a significant source of particulate phosphorus loading to streams, accounting for between 10% (Sekely et al., 2002) and 40% (Howe et al., 2011) of the total phosphorus load in an individual watershed. However, sediment and phosphorus loading is only part of the picture. In-channel and overbank storage of eroded material can be an important control on downstream transport and the ecological effect of the introduced nutrients (Kronvang et al., 2013, 2007). Additionally, geomorphic complexity influences nutrient transport and cycling, primarily by affecting residence time and transient storage, which has important implications for biochemical transformation and uptake (Ensign and Doyle, 2006).

The chemical partitioning of phosphorus is also important to understanding its transport. Phosphorus species are relatively insoluble and are typically found adsorbed to soil particles. They have a high affinity for the larger specific surface area of clay and silt particles and are commonly found bound in various metal oxyhydroxides including Fe-OH, Al-OH, and Ca-OH (Brady and Weil, 2002). Phosphorus may be found in inorganic (typically phosphate, PO_4^{-3}) or organic form. The partitioning of phosphorus among its various states determines its bioavailability for uptake by organisms, which is directly tied to its importance as a limiting nutrient. The relative abundance of bioavailable phosphorus in sediment has been shown to vary markedly within single study sites (1-55%; Veihe et al., 2011) and between study areas (averaging 0.5-22% of total phosphorus; Nellesen et al., 2011; Howe et al., 2011; Hubbard et al., 2003; McDowell and Sharpley, 2001; McDowell and Wilcock, 2007; Thompson and McFarland, 2007).

Particulate phosphorus eroded from streambanks may not be immediately bioavailable, but this can change during downstream transport. For example, if iron-bound phosphorus is placed in a reducing environment (such as a lake bottom with low oxygen levels), the iron may be reduced from Fe(III) to Fe(II), causing it to solubilize and releasing its stored phosphorus (Weitzman, 2008). Because of this, there may be a delay between erosion of phosphorus from streambanks and when the effects of this loading are manifested (Meals et al., 2010). Additionally, sediment may serve as either a sink or a source of phosphorus depending on the difference between the sediment sorptive capacity and the in-stream dissolved phosphorus concentrations (e.g., Hoffman et al., 2009; McDaniel et al., 2009). The bioavailability of phosphorus has important implications for its effects on water quality. However, because of the difficulty in both measuring bioavailable phosphorus and predicting how phosphorus speciation changes over time, most water quality monitoring programs focus only on total phosphorus. Unlike denitrification of nitrate, there is no natural biotic or abiotic process that effectively removes phosphorus from an ecosystem. Therefore, phosphorus “removal” is likely only temporary biotic uptake, although burial and storage in floodplain or lacustrine sediment may be a more long-term removal mechanism.

4.1.2 Nitrogen

Nitrogen, like phosphorus, can be a limiting nutrient in aquatic ecosystems and is therefore a major concern for nutrient managers. Sources of nitrogen include urban wastewater effluent, septic systems, agricultural, industrial and urban stormwater runoff, natural sources (e.g., organic material decomposition), and atmospheric deposition. Most notably, combustion of fossil fuels and production and application of fertilizer have greatly increased the amount and mobility of nitrogen worldwide (Vitousek et al., 1997). Nitrogen occurs in organic form and in various inorganic forms including ammonium (NH_4^+) and nitrate (NO_3^-). In natural systems, nitrogen cycles between these dominant forms through plant uptake, organic material decomposition, mineralization, and microbially-mediated nitrification (formation of nitrate from ammonia) and denitrification (conversion of nitrate into nitrogen gas). Nitrate is the most soluble form of nitrogen and is therefore commonly found in groundwater and stream water. Ammonium is also often present in streams, although it tends to have a higher adsorption potential and often experiences rapid uptake or is quickly nitrified into nitrate (Peterson et al., 2001). Although these inorganic forms of nitrogen are often the focus of nutrient studies (likely because they are the most soluble and bioavailable), organic nitrogen may be dominant in many streams (Scott et al., 2007).

One of the primary processes of nitrogen removal from a system is denitrification. Denitrification is the anaerobic reduction of nitrate by heterotrophic bacteria under anoxic conditions, leading to the production of N_2O or N_2 gas (Hill, 1996), which can then be released to the atmosphere. Denitrification thus completely removes nitrate from a system, whereas biotic assimilation merely changes the availability of the nitrogen and it may be released to the system later. Rates of denitrification are highest under saturated conditions (providing both a source of nitrate and anoxic conditions) and when organic carbon availability is high (to serve as an energy source). However, denitrification is rarely a constant process. High rates of denitrification may occur in discrete locations (“hot spots”) and at discrete points in time (“hot moments”) when conditions are right (McClain et al., 2003).

Nitrate may also be reduced to ammonium (dissimilatory nitrate reduction to ammonium, DNRA; Tiedje, 1988); this process may be more common than previously recognized in streams (Burgin and Hamilton, 2007). DNRA is a fermentative process that occurs under similar conditions as denitrification (anoxia, availability of nitrate and organic carbon), but DNRA may be favored in high carbon, nitrate-limited environments while denitrification occurs under carbon-limited conditions (Burgin and Hamilton, 2007). Nitrification is the biotic oxidation of ammonium to nitrate (via the intermediary nitrite, NO_2^-). This process requires oxygen; therefore, it typically does not occur in the same locations as denitrification (Ward, 2003). However, small-scale variation in oxygen availability is possible, allowing nitrification and denitrification to occur nearly simultaneously (e.g., Zarnetske et al., 2011). Nitrogen cycling and transformation is a complex issue with numerous biotic and abiotic controls. Most studies focus on net nitrogen uptake or removal from the water column (but not necessarily from the stream system), although some more sophisticated isotope tracer experiments have successfully quantified total nitrate loss via denitrification (e.g., Mulholland et al., 2008).

4.1.3 Sediment

Sediment can range in size from fine clay to large boulders; however, finer sediments (sand, silt, and clay) are generally of greatest concern from a water quality standpoint. In many river basins worldwide, natural sediment regimes have been significantly altered through land use change and impoundments (Wohl et al., 2015). Conversion of land for agriculture, forestry, or construction can significantly increase fine sediment loading to streams. Conversely, mature urban systems and dams tend to prevent sediment from reaching streams and trap the majority of the sediment load of impounded rivers (Owens et al., 2005). Each of these alterations can impact water quality. For example, fine sediment can increase turbidity, reduce light penetration, impair fish spawning habitat, and blanket lakebeds. Reduced sediment delivery can create a “hungry water” affect where the stream erodes its own channel to make up for the sediment capacity-supply deficit (Kondolf 1997). Essentially, the stream has energy to move more sediment than is available, so the flow erodes the channel until the sediment capacity of the stream is satisfied. This erosion may trigger widespread loading from channel downcutting and bank failures that overwhelms the sediment capacity of the stream. Fine sediment can have indirect water quality impacts as well. These fine particles can carry a variety of pollutants, including nutrients, metals, and other toxic compounds, which have their own detrimental effects. As noted previously, phosphorus especially is commonly found adsorbed to fine sediment and therefore processes of sediment and phosphorus delivery to streams are often linked.

Unlike nitrogen and phosphorus, there is no biochemical demand for inorganic sediment and therefore no processes that change the form or quantity of sediment within stream systems. Sediment transport is instead dominated by purely physical processes, notably erosion and deposition (chemical weathering can also act on sediment but this is generally at longer time scales and of smaller practical significance). Management of sediment issues therefore necessarily focuses on preventing sediment from reaching streams, or allowing for unimpeded downstream sediment delivery, as in the case of dams (Owens et al., 2005). For example, reforestation or the effective use of sediment containment measures may reduce fine sediment delivery to streams. Modified dam management may allow for sediment passage, although in many cases physical dredging is the only available option. From a stream restoration perspective, increasing floodplain connectivity can encourage overbank sediment deposition. Bank and bed stabilization is an obvious approach to reduce sediment loading from in-channel sources. However, reducing this sediment delivery may result in a “hungry water” scenario downstream, initiating new channel erosion. Because there are few processes that the stream restoration techniques included in this document can directly address to improve sediment-related water quality issues, the majority of this crediting document focuses on issues related to nitrogen and phosphorus. However, because sediment and nutrient issues are linked, appropriate application of stream restoration techniques may result in improvements to sediment-related outcomes as well.

4.1.4 Temperature

Water temperature is a critical physical characteristic of waterbodies for supporting appropriate ecological communities. Rivers and stream experience a “thermal regime,” with water temperatures controlled by incoming solar radiation, stream size and shape, groundwater inflow, and shading from riparian areas (Caissie, 2006). The natural temperature regimes of rivers have been altered by a variety of human activities, such as clearing riparian vegetation, climate change, and direct discharge of warm effluent from wastewater treatment facilities and power plants. Elevated water temperatures have a number of negative effects on fish and other biota, including direct mortality. Salmon and trout – important commercial and recreational fish species – are especially sensitive to water temperature for survival and reproduction. Given the economic importance of these and other fisheries, temperature Total Maximum Daily Loads (TMDLs) have been developed across the country. Reducing effluent water temperature from point sources is possible, but may be expensive and energy intensive (e.g., ODEQ, 2007). Other activities can effectively address elevated water temperatures and may be suitable for water quality trading. For example, riparian restoration increases stream shading and reduces temperatures. Managed flow releases from reservoirs can supply colder water to streams. Increasing hyporheic flow through in-stream restoration can also reduce water temperatures.

4.1.5 Other Pollutants

Other pollutants could potentially be considered for inclusion in future updates to this crediting guidance, but have not been included at this time due to their additional biogeochemical and regulatory complexity. Notable examples include metals (e.g., arsenic, iron, and selenium) and bacteria. From a regulatory perspective, bioaccumulative/toxic metals would likely be more challenging, despite the fact that stream restoration practices can effectively reduce loading of a variety of naturally occurring metals.

4.2 Stream and Floodplain Nutrient Processing

In-stream nutrient processing is a complex and important process that has received increasing attention in recent years (see the review of Ensign and Doyle, 2006). The Lotic Intersite Nitrogen Experiments (LINX and LINX-II), for example, have greatly improved understanding of ammonium and nitrate dynamics in both developed and undeveloped watersheds across the U.S. (Mulholland et al., 2008; Webster et al., 2003). However, only limited research has focused specifically on the impact of stream restoration on in-stream nutrient processing. Urban and other degraded streams tend to incise and erode due to high water velocities and shear stresses (Booth, 1990) and have lower geomorphic complexity (i.e., bedforms, sinuosity, and in-stream wood) than reference streams (Jacobson et al., 2001). Therefore, restoration typically focuses on reducing velocities and increasing channel complexity by increasing channel sinuosity, installing in-stream structures, and/or reconnecting floodplains. These altered water velocities and associated hydraulic retention times can have significant impacts on in-stream nutrient processing (Bukaveckas, 2007; Ensign and Doyle, 2005; Kasahara and Hill, 2006a; Klocker et al., 2009; Kuenzler et al., 1977; Roberts et al., 2007), regardless of whether this is a specified objective of the restoration project.

Increased channel complexity and in-stream structure installation can also encourage hyporheic exchange (e.g., Crispell and Endreny, 2009; Gordon et al., 2013). The hyporheic zone is an aquifer beneath and adjacent to the stream where surface water and groundwater mixing occurs. This mixing creates physical and chemical gradients that make the hyporheic zone an important area for biogeochemical cycling (Brunke and Gonser, 1997; Hester and Gooseff, 2010). Around constructed or natural geomorphic features, there are often distinct zones of upwelling (flow from the hyporheos to the stream) and downwelling (flow from the stream to the hyporheos) (Crispell and Endreny, 2009; Kasahara and Hill, 2006b). These exchanges can be important for nutrient retention and processing (e.g., Kasahara and Hill, 2006a).

Although the body of knowledge related to instream nutrient processing is increasing, the high variability in nutrient uptake across sites makes generalizations difficult (Ensign and Doyle, 2006). Floods can also significantly alter nutrient uptake dynamics in a single reach, although response is variable depending on the initial geomorphic condition (Mueller Price et al., 2015b). In addition, most studies on in-stream nutrient processing are at a single, baseflow discharge (e.g., Mulholland et al., 2008) which makes extrapolation of results to other flows challenging. In some cases, nutrient retention may be most effective at these low discharges because the higher surface area to volume ratio and lower velocities relative to high discharges encourage biogeochemical processing (Doyle, 2005). This means that these findings may not be representative of other flow conditions. Assuming constant uptake rates across a range of natural flow variability may therefore lead to an overestimate of in-stream nutrient processing potential. However, nutrient removal may also be significant during high flows.

Flooding is an important component of natural hydrologic regimes and floodplain access can encourage deposition of adsorbed nutrients and increase biological processing. Nutrient retention, especially of nitrogen, can be significant in both natural (Forshay and Stanley, 2005) and restored floodplains (Fink and Mitsch, 2007). Floodplain restoration typically involves either lowering floodplain elevations or raising the stream bottom to restored hydraulic connectivity between these two systems. This increases the frequency and duration of floodplain inundation and reduces the depth to groundwater in riparian areas, both of which lead to nutrient retention and removal. Increasing soil saturation and hydraulic residence times encourage denitrification (Kaushal et al., 2008), while slow floodplain flows encourage sediment and nutrient deposition and storage (McMillan and Noe, 2017). Reconnecting floodplains to restore floodplain wetlands can be especially effective for nutrient removal (Filoso and Palmer, 2011).

CHAPTER 5

Technical Procedure and Considerations for Developing Credits

In this section, the general technical considerations and challenges for developing stream restoration credits are discussed, along with providing guidance for credit development for these four specific practice categories: bed and bank stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection. Guidance for assigning credits for each of the four stream restoration practice groups includes background information, project information/data requirements, regional geomorphic considerations, longevity and response time, uncertainty and simplifying assumptions, and recommended crediting approach. For each of the four practices below, the credit (unit of trade) is defined as pounds (or kilograms) of pollutant per year (nutrients and sediment) or kilocalories (or kilowatt hours) of solar energy reduction per year (temperature). Although equations are provided for each of the four categories, some of the important input variables are generally difficult to quantify given current knowledge and data availability. For example, estimating “avoided channel erosion” requires knowledge of how channels would evolve over time in a particular setting with and without implementation of stream restoration practices. Similarly, estimating net changes in denitrification potential and hyporheic exchange associated with various restoration practices, regions, and stream types remains challenging. Nevertheless, the principles and relationships described below for each practice provide a foundation for conceptualizing potential crediting approaches as the empirical basis for quantification improves over time.

Because of the difficulty and uncertainty in direct quantification, recommendations for a functional assessment type approach are also discussed in Chapter 6, as part of the monitoring guidance discussion. This approach relies on indicators of the presence or absence of functions important for nutrient removal and retention. Generally, this requires a less intensive assessment than direct quantification of potential benefits; however, it requires adequate evaluation of function in regional reference streams for comparison. Functional assessment frameworks also provide an example of how projects can be assessed or deemed to qualify for crediting consideration and/or be checked/validated over time in the context of the overall system and watershed needs and/or environmental drivers. For example, functional assessments could be used within a crediting program to assign trading ratios or safety factors.

Table 5-1 provides a summary and comparison of the four practice categories considered in this guidance. Of the four practice categories, bed and bank stabilization and riparian buffers have better developed empirical data sets than in-stream enhancement and floodplain reconnection. Additionally, bed and bank stabilization and riparian buffers can be viewed as source controls, whereas the latter two practices may help to reduce pollutant loading that has already been introduced into the system. The crediting procedure for the first two practice categories is more refined. Stream restoration projects may incorporate more than one these four practice categories. Although it is important not to double count credits from these practices, water quality benefits can in fact be additive and projects could be eligible for multiple credits. This may increase the cost effectiveness of restoration projects that provide these multiple benefits.

Table 5-1. Summary and Comparison of Stream Restoration Practice Categories.

	Practice Category			
	Bed and Bank Stabilization	**Riparian Buffers**	**In-Stream Enhancement**	**Floodplain Reconnection**
Project Cost[1]	Moderate - High (~ $200/ft)	Low (~$5–$15/ft)	Low - Moderate (~$150/ft)	Moderate - High (~ $120/ft)
Response Time (years)[2]	1-5	5-20; >50	1-5	1-5
Longevity (years)[2]	10-50	>50	10-50	>50
Maintenance[2]	Low	Moderate	Low to Moderate (dependent on specific technique)	Moderate
Level of Empirical Evidence	Moderate: erosion-loading process understood but few studies monitor in-stream nutrient concentrations post-stabilization. Also, N content of bed and bank soils have not been studied as extensively as P.	High: many studies on individual buffers, particularly in agricultural settings, but fewer focus on watershed-scale water quality.	Low: evidence of enhanced nutrient processing at individual structures but less certainty about reach-scale removal, especially at flows other than baseflows.	Low: physical and biogeochemical processes understood but few empirical studies that focus on water quality benefits. Data from wetlands may be used, assuming similar functions in reconnected floodplains.
Relative Pollutant Removal Potential[3]	P: High N: Low Sed: High Temp: NA	P: Moderate to High N: High Sed: High Temp: High	P: NA N: Moderate Sed: NA Temp: Low	P: Low N: Low Sed: High Temp: NA
Prerequisite Considerations for Credits	Applicable in alluvial/adjustable channels, not bedrock or armored channels. Benefits depend on evolution of channel and extent of potential impact (e.g., potential depth of incision and movement of headcuts).	N credits only appropriate if denitrification factors present (e.g., receive groundwater with elevated NO_3^-, adequate carbon source present, water table intersects rooting zone, and wet enough for low oxidation potential). N credits more appropriate for land uses such as agriculture.	Applicable to alluvial channels because little or no hyporheic exchange is expected in a bedrock or clay-lined channel.	A long-term hydrologic connection must be established between the channel and the floodplain where such a connection did not exist before.

[1]Relative cost ranges based on Bernhardt et al. (2005); cost per linear foot from Hassett et al. (2005).
[2]Response time, longevity, and maintenance fields were adapted from Roni and Beechie (2013).
[3]Pollutant removal potential is highly site- and design-specific and depends on a number of geophysical and biological factors.

5.1 Fundamental Technical Considerations Related to Water Quality Credits for Stream Restoration Projects

The following section discusses general considerations, challenges, and technical constraints for assigning credits to stream restoration projects, and the benefits of stream restoration projects.

Technical Considerations for Determining Whether a Stream Should Be Prioritized for Restoration

Candidates for stream restoration will typically have some of the following characteristics (note that existing tools, including EPA's Recovery Potential Screening Tool (EPA, 2018), may also be useful for identifying candidate restoration sites):

- Nutrient loading from stream segment results in downstream water quality degradation.
- Active incision (e.g., CEM stages II-IV and no effective grade control, CEM stage II is high priority to avoid bank failure).
- Active bank failures and mass wasting of banks (not just in bends).
- Steep, high, unvegetated banks.
- High P content in unstable or failing banks.
- Entrenched channel disconnected from former broad floodplain. Flows greater than median annual flood (Q2) are contained in banks.
- Streamflow amplified above threshold for erosion of boundary material.
- Channel crossed planform threshold as a result of disturbance (e.g., meandering to straight or braided).
- Homogeneous/prismatic channel lacking topographic complexity and bedforms.
- Former alluvial channel scoured to impermeable layer.
- Water table no longer intersects riparian root zone/carbon source due to incision or enlargement.
- Surface runoff not dispersed and infiltrated through riparian zone.
- Narrow or unvegetated buffer.
- Existing functional stream reaches in the watershed are fragmented due to intervening non-functional reaches.
- Potential for propagation of incision/headcuts and increased nutrient loading. The potential for propagation depends on several factors including grade control, bed armoring potential (e.g., cobble vs. sand), bank consolidation, and depth to a resistant layer in channels without armoring potential. Potential for propagation can generally be categorized as (Bledsoe et al., 2010):
    - Low: Very limited or no spatial propagation (approximately 10 m).
    - Medium: Local spatial propagation, contained within approximately 100 m.
    - High: Potential for propagation of headcutting/base-level change upstream and/or downstream but contained within approximately 100 to 1,000 m domain of control. Relatively long relaxation time given magnitude and spatial extent of change.
    - Very High: Potential for widespread spatial propagation – headcutting/base-level change upstream and downstream uncontained within approximately 1,000 m domain of control. Relatively long relaxation time given magnitude and spatial extent of change.

5.1.1 General Considerations

To develop credits for stream restoration practices, these basic questions should be addressed:

- Restoration Implementation
  - Is the stream a priority for restoration? (See box "Technical Considerations for Determining Whether a Stream Should Be Prioritized for Restoration" for representative factors to consider.)
  - What is the baseline pollutant loading contributed by a degraded or unstable stream? Are any adjustments to the baseline needed due to "worse than baseline" conditions?[3]
  - What is the potential for achieving the desired function(s) (e.g., nutrient removal) within constraints imposed by the stream type and its valley and watershed setting? (See discussion of functional assessment procedures in Section 6.3.)
  - What is the prevented pollutant load/year as a result of the restoration practice?
  - Does the practice have unintended consequences or stream adjustments upstream or downstream of the project area? These consequences could also include legal considerations if the floodplain/ flood risks change.
  - Do landowners and other stakeholders support the project?
  - Is adequate funding available to support proper design and implementation of the capital improvements and future operation and maintenance requirements?
- Crediting and Trading
  - What is the appropriate duration of the credit? A viable credit must have a clearly defined duration as well as a clearly defined water quality benefit (WEF 2015).
  - Does the project have sufficient quality and duration to result in quantifiable water quality improvements (WEF 2015)? The design basis of the improvements is an important factor to ensure that the improvements will be resilient during large storm events.
  - What is the response time and longevity of the project? Response time and longevity of stream restoration practices are important technical considerations when establishing water quality-related credits. Roni and Beechie (2013) estimated response times for a variety of stream restoration practices, some of which are summarized in the callout box below.
  - How is "reasonable assurance" provided that the expected load reductions are being and/or will be achieved?
  - What is the appropriate trading ratio to account for uncertainty for nonpoint-source load reductions?
  - How is credit verification conducted for the project?
  - How is double-counting (improper accumulation of nutrient credits for same project) avoided?
  - Are regulatory requirements for the project met (e.g., 404 permit, 401 consultation, floodplain, threatened and endangered species, other permits)?

In addition to technical considerations related to water quality, communities may also want to consider and recognize other benefits related to stream restoration projects in terms of ecosystem services, "triple bottom line," community health (e.g., recreational and aesthetic benefits), and other sustainability concepts. For example, urban river parkways have been characterized as an "essential tool for public health" providing benefits such as exercise, active commuting, children's mental health, environmental education, heat island reduction, and other benefits (Jackson et al., 2014).

Credit quantification and equivalency considerations identified by EPA (2003) also include:

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[3] As an example for agricultural lands in the Chesapeake Bay Watershed, farmers are expected to implement basic practices that are not eligible for credits. Credits can be generated for projects above and beyond these basic practices (WEF 2015).

- **Environmental Equivalence:** Trading programs should be designed to ensure environmental equivalence of traded pollutants. For example, benefits that occur because of a trade should be equivalent to or better than conditions that would have occurred in a watershed without the trade.
- **Reliable Estimation and Trading Ratios**: Nonpoint source pollutant reductions must be reliably estimated. Due to challenges with estimated nonpoint sources of pollution, trading ratios are often incorporated (e.g., 2:1 nonpoint source to point source). Conversely, trading ratios should not be so conservative that they inhibit trading (WEF 2015). Specifying trading ratios is ultimately a policy-related decision that is beyond the scope of this guidance; however, precedents for trading ratios have been developed in many existing crediting programs. Trading ratios can be modified depending on the method used for quantifying pollutant credits. For example, a credit based on site-specific data would receive a more favorable trading ratio (i.e., more credit earned) than a credit estimated from generic literature values which introduce greater uncertainty.

Trading Ratios vs. Safety Factors

This Guidance discusses both trading ratios and safety factors as methods to account for uncertainty in the actual water quality benefit of a stream restoration project. Safety factors are used to discount credit value based on uncertainty in the credit quantification process. Trading ratios address different sources of uncertainty, including future performance and persistence of the restoration project, potential for additional environmental degradation, and actual transferred benefit between the project site and waterbody of interest (Section 7.2). While trading ratios and safety factors address different issues, it is important that they are not duplicative and do not inappropriately reduce credit value, thereby making trading less attractive.

5.1.2 Challenges and Technical Constraints for Assigning Credits to Stream Restoration Projects

Some of the significant challenges associated with developing and assigning credits to stream restoration projects include:

- Practical constraints related to cost and duration of monitoring for stream restoration projects, both before and after projects are implemented.
- Limited empirical data for some stream restoration practice types. For example, Bernhardt et al. (2005) developed a database of metadata for stream restoration projects but did not compile and evaluate the water quality-related performance data associated with these efforts. This was in part due to poorly defined and measurable objectives, inadequate assessment metrics, and inadequate pre- and post-construction monitoring.
- Although theoretical equations to quantify the water quality load reduction benefits of stream restoration practices exist and/or can be developed, availability of sufficient data to populate these equations is often lacking.
- Evaluation of monitoring data can be challenging because stream restoration results and benefits are not “steady state” conditions.

Many restoration projects utilize more than one of the practices discussed below. Credit quantification for these practices is designed to be independent, allowing multiple practice credits to be obtained by a single restoration project. The benefits of each practice are quantified separately, but may be applied together to yield a cumulative nutrient reduction credit. However, since the longevity and performance of individual practices may vary, these credits should not be lumped but instead be subject to separate monitoring and validation requirements to ensure continued function. In these cases, it is also important to ensure that credit quantification for one practice does not inadvertently account for nutrient removal from another. This is primarily of concern where in-stream enhancement may enhance hyporheic exchange into the riparian zone (see Section 5.4.6).

5.1.3 Additional Considerations and Benefits of Stream Restoration Projects

In addition to the technical considerations and constraints above, stream restoration projects often provide intangible and/or difficult to quantify benefits. These benefits crossover into the realm of policy-related considerations; however, these benefits should be taken into account when considering trading ratios. Some of the reasons that stream restoration projects are valuable to communities and ecosystems include:

- Improvements in degraded habitat to a more natural condition benefiting aquatic and terrestrial life.
- Protection of adjoining and downstream structures, property, and utilities, as well as increasing the useful life of culvert and bridge crossings.
- Reconnecting streams with adjoining floodplains, which can enhance water quality via flows that interact with riparian vegetation and organic matter (Craig et. al., 2008).
- Improved interaction between the riparian zone and the channel.
- Floodwater storage.
- Biogeochemical cycling.
- Recreational benefits, both in-stream and on trails adjacent to streams (with associated human health benefits).
- Enhanced aesthetic condition of waterways, providing both economic and health benefits.
- Source-water protection.
- Increased water storage, temperature reduction, and enhanced habitat, especially in urban systems.
- When integrated into watershed-scale approaches to reduce overall pollutant loading in the context of pollutant trading programs, costly wastewater and/or stormwater treatment requirements that provide marginal benefits at high cost can be minimized, postponed or avoided. Advanced wastewater treatment can have environmental impacts such as a significant carbon footprint (energy requirements), chemical usage, treatment residual disposal issues, and other impacts, in addition to financial burden for local communities.

5.2 Bed and Bank Stabilization

This section provides an overview of bed and bank stabilization.

5.2.1 Background

Literature reviews have suggested that sediment and phosphorus loading rates from bank erosion can span several orders of magnitude: 0.1-17,600 tonne/km-yr of sediment and 0.1-4,100 kg/km-yr of phosphorus (Lammers and Bledsoe, 2017) or 7-92% (sediment) and 6-93% (phosphorus) of the total watershed load (Fox et al., 2016). This wide range is due to both variability in bank erosion rates as well as the bank phosphorus content and introduces large uncertainty into any quantification scheme (see Section 5.2.4 below). Other empirical results suggest that bank stabilization (in conjunction with riparian buffer establishment) can reduce both phosphorus and sediment loading (Carline and Walsh, 2007; Meals, 2001), although this benefit may not be seen in all cases (Selvakumar et al., 2010).

A bed and bank stabilization crediting approach in the Chesapeake Bay watershed is based on a three-step process (Wood, 2020): 1) estimate existing annual stream sediment loading rates; 2) convert erosion rates to nitrogen and phosphorus loadings; and 3) estimate reduction attributed to restoration. In this program, erosion rates may be quantified via monitoring (preferred) or by empirical (e.g., BANCS method) or mechanistic (e.g., BSTEM) models. This crediting approach *requires* site-specific data on soil bulk density and soil nutrient concentrations. To account for the fact that no stream restoration strategy can be 100% effective in limiting bank erosion, Wood (2020) only gives 50% of the calculated reduction as credits for individual projects (e.g., essentially a 2:1 credit ratio); however, this is a conservative approach, and the crediting procedure allows for greater than 50% credit if at least three years of post-restoration monitoring can demonstrate sufficiently lower erosion rates.

Response Times and Longevity of Practices in Crediting Programs

One of the sources of uncertainty in stream restoration projects is lag time between restoration project implementation and in-stream response and longevity due to events that "reset" the stream system, such as major floods. The manner in which crediting programs address lag times and longevity is partly a policy decision. The initial step to reduce this uncertainty is to ensure that the key functional aspects of the restoration design are properly carried out during construction.

Although sophisticated approaches to quantifying functional lags have been developed, they are generally not very practical to apply from a programmatic perspective. Pragmatically, a reasonable approach to account for lag times is to design functional assessment variables at various points in time. Using bed and bank stabilization and riparian buffers as examples, the key time lags will be vegetation development on streambanks and in buffers, and carbon accumulation unless the system is intentionally spiked with an appropriate carbon source.

In terms of longevity, functions provided by stream restoration projects have the potential to largely stop after "resetting" events. Depending on the channel design and magnitude of the event, streams may recover over some time lag or may need to be restored if the stream cannot heal itself. Generally, the less constrained the channel and the more sediment and stream power the stream has to work with, the more self-healing potential there is. For example, see "Setting Goals in River Restoration: When and Where Can the River 'Heal Itself'?" (Kondolf, 2011).

This guidance generally follows the approach developed by Wood (2020), with some modification. The original crediting approach (Schueler and Stack, 2014) and updated guidance (Wood, 2020) include nitrogen in bank stabilization crediting; however, nitrogen is not a focus of this crediting guidance for bank stabilization because of the minimal research available for assessing nitrogen content of stream banks. Generally, the focus is on phosphorus, which is more commonly found adsorbed to sediment. Nitrate is highly soluble and unlikely to be present in significant concentrations in streambanks. Organic nitrogen and a small fraction of adsorbed NH_4^+ could be present and may contribute to total nitrogen loads to receiving waters. However, organic nitrogen is not biologically available and would require adequate residence time in the waterbody for decomposition to occur. Therefore, nitrogen credits for bed and bank stabilization may only be appropriate for projects intended to reduce total nitrogen loading to slow moving waters, such as low gradient rivers, lakes, reservoirs, and estuaries.

5.2.2 Information/Data Requirements

To qualify for nutrient reduction credits for bank and bed stabilization, the following conditions must be met:

- The restored stream must consist of erodible material that is eroding or has the potential to erode (e.g., bedrock channels are not eligible).
- The reach in question must not already be protected by a downstream grade control that is expected to prevent incision from migrating upstream and limit the erosion potential of the study reach.

- The stream should be prone to future erosion if restoration is not completed. That is, the stream should not have reached a new stable state where future channel change is unlikely (e.g., Stage V of CEM; Figure 3-1).

The general quantification approach used by Wood (2020) is similar to what is recommended below. This approach requires knowledge of bank erosion rates that can be converted to sediment loads and then to associated phosphorus (and potentially nitrogen) loads. Bank erosion rates are ideally quantified via pre- and post-restoration monitoring. However, this may be unrealistic in some cases and therefore modeling may be an appropriate substitute. Empirical and mechanistic approaches of varying complexity exist for quantifying bank erosion rates. Two of the better known are the Bank Assessment for Non-point Source Consequences of Sediment (BANCS; Rosgen, 2001a) and the Bank Stability and Toe Erosion Model (BSTEM; Simon et al., 2000). Other mechanistic models include the Channel Evolution and Pollutant Transport System (CONCEPTS; Langendoen, 2000) and River Erosion Model (REM; Lammers & Bledsoe, 2018).

Erosion rates must be converted to sediment erosion volumes by multiplying by the length of eroding banks as well as bank heights. Soil bulk density and nutrient concentrations are required to convert this volumetric sediment loading rate to a nutrient loading rate. Soil bulk densities may be measured in the field or obtained from the U.S. Department of Agriculture Soil Survey. Bank nutrient concentrations are ideally obtained from field samples; however, cost may make this approach infeasible. Data collected from the literature may be used in lieu of site-specific data. It is important to recognize that different types of phosphorus concentrations are often reported (e.g., total, Mehlich-3, oxalate-extractable). It is therefore important to use the most representative value based on local objectives. Additionally, relative bank phosphorus content may be predicted based on nearby upland soil phosphorus concentrations, although there is significant regional variability (Lammers and Bledsoe, 2017). For example, streambank nutrient concentrations in North Carolina are much lower than average values from the Chesapeake Bay Region (Doll et al., 2018). Streambank nitrogen concentrations are published much less frequently than phosphorus; however, some local data are available (e.g., Walter et al., 2007; Earles et al., 2020).

5.2.3 Uncertainty and Simplifying Assumptions

There is significant uncertainty in quantifying current or historic levels of nutrient loading from bank erosion due to variability in both bank erosion rates, soil bulk density, and bank nutrient concentrations. This uncertainty is compounded when trying to estimate future avoided nutrient loading from bank erosion as historic erosion rates are not always representative of future channel change potential, especially if incision triggers widespread mass wasting. Uncertainty may be decreased by using data from the project in question, rather than selecting erosion rates or bank nutrient content data from the literature. In fact, the Chesapeake Bay crediting approach requires that soil samples be collected and analyzed for bulk density and nutrient content for each restoration project. While modeling may be used to estimate restoration benefits, the uncertainty of the model results may be significant and should be quantified, where possible, by running the model iteratively across a range of plausible input parameters.

5.2.4 Regional Geomorphologic Differences

Differences in hydrologic regime, soil type, riparian vegetation, and land use impact both erosion rates and bank soil nutrient content. Furthermore, channel response to disturbance may be vastly different depending on initial boundary conditions. For example, channels in southern California have been observed to transition from single thread to braiding following incision (Hawley et al., 2012), while sand bed streams with cohesive banks in many regions tend to follow the classic, single-thread channel evolution model (Figure 3-1; Schumm et al., 1984; Simon, 1989). Thus, contextual differences in channel evolution processes determine the net the benefits of stream restoration by controlling the extent of erosion that would have occurred over decadal time scales in the absence of stream restoration interventions. Bank phosphorus content varies by soil type and silt-clay content and may be correlated with nearby upland soil phosphorus concentrations. Nitrogen content is often strongly correlated with organic carbon content (Avramidis et al., 2015) and both carbon and nitrogen content are lower in riparian forests than agricultural streambanks (Willett et al., 2012). If using data from the literature, it is important to note these trends and use data from similar soils and region as the study site.

5.2.5 Response Time and Longevity

Both response time and longevity vary depending on the specific bank stabilization technique used; however, it is reasonable to expect relatively short response time (1-5 years) and high longevity (10-50 years) assuming proper construction and no extreme event flows (Roni and Beechie, 2013). In the absence of suitable hydrologic controls, both urbanization and climate change are expected to increase the magnitude of high flow events in the future (AECOM, 2013; Hollis, 1975); it may therefore be necessary to design stabilization structures to withstand these higher flows, or accept higher risk and lower longevity associated with these projects. Response time for sediment-related load reductions may be more rapid for hard bank stabilization measures (e.g., riprap, gabions) than for softer, bioengineering approaches (e.g., live plantings, fascines). However, these softer approaches may have greater longevity given their ability to regenerate over time and also provide ecological co-benefits that hard engineering (armoring-only) approaches lack. In practice, effective bioengineering designs often integrate a combination of “hard” (e.g., hidden armoring) and “soft” approaches. The Chesapeake Bay crediting program recognizes these differences, and only allows full crediting of bioengineering bank stabilization approaches that provide functional uplift. Hard armoring approaches can only receive partial credits (up to 30% of a project length), while engineered stabilization structures installed specifically to protect infrastructure are not eligible for any credits (Wood, 2020). The longevity of any bank stabilization structure is often dependent on the magnitude of the flow it is designed to resist, among other factors. This may be especially true for techniques that depend on vegetation as large flows during plant establishment may compromise bank integrity. Because of this potential for large flow events to destabilize projects, any crediting program should include monitoring and evaluation to ensure the project functions as expected through time.

5.2.6 Crediting Approach

The direct quantification crediting procedure for phosphorus uses a general equation of the form below. Estimating nitrogen credits would be identical. Equation 5-1 shows a sample calculation of avoided phosphorus loading from bank stabilization. Note that complete cessation of bank erosion is impractical and undesirable; therefore, this calculation must incorporate only that bank erosion which is actually avoided.

$$Avoided\ bank\ erosion\ \left[\frac{m^3}{km*yr}\right] * Soil\ density \left[\frac{kg}{m^3}\right] * P\ concentration \left[\frac{mg\ P}{kg\ soil}\right] * \\ Restored\ stream\ length\ [km] = \ Avoided\ P\ loading \left[\frac{mg\ P}{yr}\right] \tag{5-1}$$

Quantification of the necessary data inputs requires the following four steps.

5.2.6.1 Step 1: Quantify Bank Erosion Rates [Data: Site Specific]

Quantifying expected bank and bed erosion in the absence of any restoration is essential in determining avoided sediment and phosphorus loading, and subsequent credit value. Bank erosion rates can be quantified using three basic approaches: direct monitoring, time-series aerial photography or similar analysis, or modeling (for a somewhat dated review on the benefits and drawbacks of different bank erosion measurement methods, see Lawler, 1993). Direct monitoring involves the use of erosion pins, repeated cross section surveys, terrestrial LiDAR scanners, or similar methods to directly measure bank erosion rates in the study reach. Erosion pins consist of simple rebar hammered into an eroding streambank. The change in the length of the exposed bars is measured regularly to quantify erosion rates. Repeated cross section surveys and terrestrial LiDAR scanners are similarly used to assess bank erosion (or aggradation) through time, with varying spatial resolution. Alternatively, time series of aerial photographs or airborne LiDAR data may be used to estimate channel change over time. This technique requires two or more images or data sets collected over time. The channel boundaries in each image are delineated and overlain, allowing the lateral migration (and area of eroded bank) to be calculated. Care should be taken when extrapolating erosion rates from a given time period because a high magnitude, low frequency event may have caused significant channel change that is not necessarily representative of longer-term erosion rates.

> **A Note on Equations**
>
> Although conceptual equations have been developed in this Guidance to calculate nutrient and sediment reduction benefits of many stream restoration strategies, the data necessary for these calculations require site-specific monitoring or use of regional studies, which may not be available. Regional data are expected to become more available over time and the complementary Stream Restoration Database (WRF, n.d.) would ideally be used as the primary repository of these data.

In each of these cases, linear retreat rates must be converted to volumetric erosion rates by multiplying by bank heights, which can be measured directly in the field. High resolution airborne LiDAR has the additional advantage of providing bank height information (at least from the water surface to the top of the bank), eliminating the need for field data collection (see Rhoades et al., 2009). These quantification approaches can be used to measure erosion rates both pre- and post-restoration, allowing for the change in loading to be assessed. However, the quantified historic erosion rates may not accurately represent future channel changes (e.g., an incision threshold may have been crossed in the future, leading to much larger bank failure and loading rates than what were observed historically). Modeling can be used to predict how the channel may have evolved in the absence of bed and bank stabilization; but this is subject to significant uncertainty, given the disparate mechanisms of bank retreat (e.g., mass wasting versus fluvial detachment, and the potential for non-linear episodic inputs). Regardless of the method used, monitoring is required to ensure continued channel stability and avoided sediment and nutrient loading. If a sediment, rather than phosphorus, credit is desired, the calculated volumetric sediment loading rate can be converted into a mass loading rate by multiplying by the bank soil bulk density and restored project length (Equation 5-1), skipping Steps 2 and 3. Soil bulk density can be measured for the site of interest, or may be estimated from the USDA Web Soil Survey. Site-specific spatial variability in bulk density and uncertainty in soil bulk density estimates can significantly affect calculated sediment mass loading rates from bank erosion, although this uncertainty is generally lower than for bank nutrient concentrations (e.g., Purvis et al., 2016).

5.2.6.2 Step 2: Quantify Bank Phosphorus Content [Data: Site Specific; Literature]

Bank phosphorus content can be measured directly by obtaining soil samples and performing necessary laboratory analysis. There are several common methods for determining soil phosphorus content that target different forms of phosphorus (e.g., orthophosphate, organic phosphorus, iron-bound phosphorus). These different methods can result in significantly different concentrations, and it is important to use the extraction method most applicable to the interests of the specific crediting program. For example, if only the labile forms of phosphorus are of concern, using an extraction that quantifies total phosphorus content may not be appropriate. Common methods include the EPA Methods 3050a or 3051 + 6010 for total phosphorus and the Mehlich-3 extraction and water extraction method for an estimate of “bioavailable” phosphorus. In lieu of direct measurement, bank phosphorus content may be assumed equal to values published in the literature, ideally for locations in close proximity to the study site, since bank phosphorus concentrations exhibit considerable regional variation (Doll et al., 2018; Lammers & Bledsoe, 2017).

5.2.6.3 Step 3: Calculate Bank Erosion Phosphorus Loading

Bank erosion rates and bank phosphorus content can be combined to yield an estimate of phosphorus loading from bank erosion (Equation 5-1).

5.2.6.4 Step 4: Apply a Safety Factor to Yield the Final Credit

Once the phosphorus or sediment loading rate from bank erosion is calculated, a safety factor should be applied to account for discrepancies between this calculated value and the actual avoided loading expected from bank and bed stabilization. This safety factor will be unique to each crediting program (and site-specific factors) but should take into account the following uncertainties:

- Historic erosion rates are not representative of future channel evolution. Depending on the specific stream, including its current stage in the channel evolution model (Figure 3-1), channel erosion rates may be expected to increase or decrease in the future. To account for this, calculated erosion and loading rates may have to be modified over time.
- No bank or bed stabilization can completely eliminate bank erosion in a stream. For example, the Chesapeake Bay crediting program assumes that restoration prevents 50% of bank erosion (Wood, 2020), although this assumption can be updated with sufficient post-restoration monitoring of project performance. Other assumptions regarding avoided bank erosion may be supported, depending on site-specific conditions and objectives of the local crediting program.
- The quantification method proposed here has significant uncertainties. For example, uncertainty in bank phosphorus content and bulk density is higher if literature values are used rather than direct measurement in the study reach. Safety factors should be adjusted to reflect this higher uncertainty, giving a lower total credit for literature-based quantification than for study-site specific analysis.

Within this crediting procedure, it is also essential to account for potential future channel evolution that is avoided due to restoration (see Section 6.3 and Table 6-4 for a functional assessment approach to this issue). While conceptual models of channel evolution exist (e.g., Figure 3-1), quantification of channel evolution through time is difficult and is a function of hydrologic regime, sediment supply, and erosion resistance of the channel bed and banks. Multiple, sometimes linked, adjustment mechanisms occur simultaneously and in sequence: bed incision, fluvial bank erosion, mass failure, and subaerial erosion processes (e.g., freeze/thaw cycles). Accounting for all of this complexity in models is difficult and therefore most models make simplifications, which may limit the accuracy of their results. Examples of potential approaches for quantifying channel evolution potential, in order of increasing complexity, include:

- Width-Depth Ratio: A simple rule-of-thumb approach for estimating channel evolution is scaling the current width-depth ratio of the stream. Empirical results indicate that as streams evolve from Stage I-V of the CEM, their width-depth ratio may increase by a factor of 3-5 and even up to 10 times the pre-disturbance geometry. This is a very coarse estimate based on limited data in one region of the U.S., and it may not be widely applicable.

- Capacity-Supply Ratio: The CSR approach (Soar and Thorne, 2001) quantifies the balance between reach sediment transport capacity and an upstream supply reach across an entire flow duration curve. This approach may be used to estimate an equilibrium slope in the restored reach to which the channel would adjust in the absence of intervention. This may provide a rough estimate of channel response potential, but it is important to note that the upstream supply reach may be evolving as well; thus, the inflowing sediment load may change over time. This method also does not directly model bank erosion. The CSR approach has been integrated into a spreadsheet tool (NAS, 2017).
- SWAT-DEG: This is a module built into the Soil and Water Assessment Tool (SWAT) that simulates channel evolution over time based on a given flow regime and known critical shear stresses and erodibilities for the bed material. Simplifications of this method include an *a priori* assumed stable channel slope and constant width-depth ratio (i.e., no mechanistic bank erosion model) (Allen et al., 2008).
- BSTEM: The Bank Stability and Toe Erosion Model mechanistically simulates bank erosion from fluvial entrainment and mass failure. This model neglects erosion of the channel bed and is meant to be applied at small spatial scales (Simon et al., 2000).
- REM: The River Erosion Model is a watershed-scale mechanistic model that simulated both channel bed incision and aggradation and channel widening. REM can be applied to full river networks including both alluvial and cohesive channels (Lammers and Bledsoe, 2018; Lammers and Bledsoe, 2019). Recent updates also allow for some bank and bed stabilization measures to be simulated (Lammers, 2018).
- CONCEPTS: The CONservational Channel Evolution and Pollutant Transport System is a reach-scale, mechanistic model that simulates both channel incision and widening due to bank erosion. CONCEPTS can accurately model channel response over time but requires a large amount of input data and may not be feasible to run for all restoration projects (Langendoen, 2000; Langendoen and Alonso, 2008; Langendoen and Simon, 2008).

5.3 Riparian Buffers

This section provides an overview of riparian buffers.

5.3.1 Background

Riparian buffers have many well-documented, multi-purpose benefits related to water quality, aquatic life, terrestrial wildlife, and aesthetics, among others. From the perspective of water quality, riparian buffers have received considerable attention due to their ability to remove pollutants in both surface and subsurface flow. Much of the literature focuses on nitrate removal, as this mobile nitrogen species is often present in high concentrations in groundwater discharging to streams. A meta-analysis by Mayer et al. (2005) suggests average nitrate removal effectiveness of ~75%. Buffer effectiveness for nitrate removal often increases with width and age, but vegetation type has little or no impact on nitrate removal (King et al., 2016). Although buffer effectiveness has been demonstrated at the pilot scale, empirical evidence has not yet demonstrated that buffer restoration results in significantly lower in-stream nutrient concentrations (Sutton et al., 2010; Collins et al., 2013).

Buffers are effective in removing particulate and dissolved phosphorus in surface runoff. However, groundwater concentrations may increase due to reducing conditions in saturated soils that mobilize adsorbed phosphorus. It is important to recognize the complex nutrient cycling that can occur when quantifying potential nutrient removal credits. A variety of empirical and mechanistic models have been developed to estimate nutrient and sediment removal potential in riparian buffers. These typically account for buffer width, slope, surface roughness, and soil parameters such as hydraulic conductivity or infiltration rate (e.g., Nieswand et al., 1990; Mander et al., 1997). The Riparian Ecosystem Management Model (REMM; Lowrance et al., 2000) is a very detailed model that simulates water, sediment and nutrient processing in riparian buffers at a daily time step. While REMM can provide accurate results of buffer performance, its high data requirements may make it impractical for use in crediting programs. Riparian denitrification processes have also been incorporated into the updated Soil and Water Assessment Tool + (SWAT+) (Bieger et al., 2017).

5.3.2 Information/Data Requirements

Projects restoring riparian buffers for their nutrient and sediment removal capabilities or temperature reduction benefits should meet the following qualifying criteria before credits are assigned:

- For nitrogen-reduction credits, the stream should receive groundwater inflow. Losing streams (streams where water flows from the channel into the subsurface) should not be eligible for nitrogen credits. In addition, the riparian buffer should not be bypassed by tile-drains.
- For nitrogen-reduction credits, conditions for denitrification should be met. These include saturated soils, nitrate-rich groundwater, and organic carbon availability. This requires high groundwater tables, meaning riparian buffers next to incised streams may not be eligible for crediting. If nitrogen-reduction credits are given for riparian buffer restoration, they should not also be provided for hyporheic zone denitrification in the floodplain to avoid double counting (see Section 5.4).
- For sediment and/or phosphorus reduction credits, the buffer should intercept overland flow from a land use identified as a source loading area to allow for sediment and phosphorus capture.
- For stream temperature reduction credits, the buffer should increase stream shading either now or in the future (as trees mature). There should also be reasonable assurance that the buffer will be protected in the future. Crediting programs may want to specify a minimum duration of protection (e.g., 30 years) and create legal mechanisms to guarantee this is achieved (e.g., through conservation easements).

Quantification of riparian buffer benefits for nitrogen in this guidance focuses primarily on subsurface nitrate removal via denitrification because this has been recognized as the dominant removal pathway in most cases (Hill, 1996). There are two primary approaches for quantifying nitrate removal in riparian buffers. The first approach is to use denitrification rates, along with a computed area (or volume) of active denitrification, to calculate total nitrate removal rates. Denitrification rates can be measured directly in restored riparian areas. However, due to the high cost of this approach, published literature values may be used. Denitrification rates are often reported on either a per area or per soil mass basis. Depending on the units used, the crediting calculation will differ (Equation 5-2). The second is a mass balance approach that computes the difference between the inflowing and outflowing nitrate load. In this case, monitoring of groundwater flow rates and groundwater nitrate concentrations are required both pre- and post-restoration to account for baseline (e.g., pre-restoration) nutrient removal.

For sediment phosphorus removal, quantification focuses on sediment and particulate phosphorus retention during overland flow. These calculations require a known buffer width and slope as well as quantification of the inflowing sediment load along with the sediment phosphorus concentration.

For water temperature reduction, the quantification procedure requires comparing solar inputs with and without the restored buffer vegetation. These calculations require both pre-project and post-project data on stream and buffer size, as well as vegetation type and local climate.

5.3.3 Uncertainty and Simplifying Assumptions

Published riparian denitrification rates span several orders of magnitude, making selection of a representative rate difficult. In addition, the area or volume where active denitrification is expected to occur varies in time and space and may be difficult to delineate. In general, denitrification is expected in groundwater-saturated soil with sufficient organic carbon availability and where sufficient cycling between aerobic and anoxic zones exists. Organic carbon availability is often assumed to be sufficient within the rooting zone of riparian vegetation, although root depths are often variable. It is therefore important to estimate both the rooting depth and the time period that groundwater levels intersect this soil layer to calculate removal via denitrification only when this process would actually occur. This may involve groundwater-modeling efforts that, like bank erosion modeling, introduces uncertainty both in the context of input parameters and inherent model uncertainty.

Sediment and phosphorus removal calculations are dependent on an empirical equation of sediment trap efficiency based on buffer width and slope. This simplified equation is subject to significant uncertainty as it is based off a single data set and excludes other variables (e.g., surface roughness and vegetation density) that influence sediment-trapping capacity. In addition, accurately assessing incoming sediment load and sediment phosphorus concentrations is difficult and introduces more uncertainty into the calculation.

The primary uncertainty in calculating temperature reduction credits is in predicting future canopy cover from newly planted riparian areas. Plant survival rates are often low, and repeated monitoring and assessment are needed to ensure the riparian area is functioning as expected.

5.3.4 Regional Geomorphologic Differences

Denitrification rates vary based on soil type, nitrate concentrations, and organic matter availability. Recent analysis also suggests that climate may play a factor, although there were no observed differences among vegetation types (Lammers and Bledsoe, 2017). Groundwater flow is also highly dependent on soil type, land use, and regional hydrology. This may be the largest source of regional differences as the timing and extent of groundwater-root zone interactions is essential in computing nitrogen removal via denitrification. The sediment and phosphorus load supplied to the buffer via surface runoff varies significantly based on topography, adjacent land use, and the underlying geology.

Temperature benefits are influenced most by solar radiation, which varies significantly by region. Additionally, streams in different geomorphic settings have different susceptibility to warming and also potential for temperature reduction. For example, steep mountain streams are likely to be home to temperature-sensitive biota, but also less susceptible to warming than a larger, slow-moving plains river. Furthermore, vegetation types vary regionally. There is minimal research on the effects of different types of trees on riparian shading (but see Dugdale et al., 2018), but generally large woody vegetation will provide more shade benefits than smaller shrubs or wetland plants.

5.3.5 Response Time and Longevity

Riparian buffer restoration has both a long response time and high longevity (>50 years; Roni and Beechie, 2013). The primary benefit of plant establishment for nitrate removal is the addition of organic carbon to the subsurface, which requires sufficient time for plant maturation. Plant establishment may also influence groundwater dynamics (e.g., greater water retention and interception). Response times for denitrification may be shorter if an organic carbon source is added to the soil during restoration in order to “jumpstart” this nutrient removal process. Mature plants and surficial soil litter are also important in trapping inflowing sediment and particulate phosphorus. Furthermore, larger trees provide more shade than saplings. There may be a significant lag time between planting a riparian zone and when maximum shade benefits are realized. Once established, riparian buffers can last for a long time, provided they are protected from disturbance.

5.3.6 Crediting Approach

5.3.6.1 Nitrogen

There are two general approaches to direct quantification of nitrogen removal in riparian buffers: a removal rate approach and a mass balance approach.

Approach #1: Denitrification Rate

Equation 5-2 shows an example calculation of groundwater nitrogen removal via denitrification in a restored riparian buffer given denitrification rates as either per area (A) or per soil mass basis (B). Note that pre-restoration nutrient removal (if any) must be subtracted from this post-restoration value to yield the net increase in nitrogen removal following restoration.

$$
\begin{aligned}
&(\boldsymbol{A}) Denitrification\ rate \left[\frac{mg\ N}{m^2\ hr}\right] * Area\ of\ active\ denitrification\ [m^2] \\
&\quad * Time\ groundwater\ intersects\ root\ zone\ \left[\frac{hr}{yr}\right] = N\ removal\ \left[\frac{mg\ N}{yr}\right] \\
&(\boldsymbol{B})\ Denitrification\ rate \left[\frac{mg\ N}{kg\ soil\ hr}\right] * Area\ of\ active\ denitrification\ [m^2] \\
&\quad * Depth\ of\ acive\ denitification\ [m] * Soil\ bulk\ density\ \left[\frac{kg}{m^3}\right] \\
&\quad * Time\ groundwater\ intersects\ root\ zone\ \left[\frac{hr}{yr}\right] = N\ removal\ \left[\frac{mg\ N}{yr}\right]
\end{aligned}
\tag{5-2}
$$

The denitrification rate approach requires a value for denitrification within the restored buffer, along with the area over which denitrification is expected to occur. If denitrification rates are measured on a per soil mass basis, the depth of denitrification, as well as the soil bulk density, are also required.

Step 1: Quantify Denitrification Rates [Data: Site Specific; Literature]

Denitrification rates can be measured directly in restored and unrestored buffers using a variety of techniques (Groffman et al., 2006). However, if this is cost-prohibitive, representative denitrification rates may be obtained from the literature (e.g., Lammers and Bledsoe, 2017) or potentially from the Stream Restoration Database (Section 6.5) as additional studies become populated with such information. Denitrification rates increase with nitrate concentration (Mulholland et al., 2008) so it is important to select data from sites where similar nitrate loading is expected.

Step 2: Determine Area of Active Denitrification [Data: Site Specific]

The area of active denitrification may be assumed to equal the area of the restored buffer. However, most denitrification within riparian buffers may occur in relatively small "hot spots" so it may be appropriate to consider only a portion of the total buffer area that is active.

Step 3: Quantify Root Zone Depth [Data: Site Specific; Literature]

The root zone depth may be one surrogate variable for the soil depth where sufficient organic carbon is present for denitrification to occur. Generally, root depths are assumed to be approximately one meter for most mature vegetation, but the root depth and density at depth can vary significantly depending on the region and vegetation type (Wynn and Mostaghimi 2006).

Step 4: Determine the Duration of Groundwater-Root Zone Interactions [Data: Site Specific]
Denitrification will only occur when nitrate-laden, anaerobic groundwater intersects the organic carbon rich root zone. This intersection may happen only sporadically depending on local hydrology and terrain.[4] Direct groundwater monitoring or modeling may be used to quantify the groundwater-root zone interaction variable. Alternatively, the groundwater level can be assumed to equal baseflow stream stage and these values can be compared to root zone depth.

Step 5: Calculation of Annual Nitrate Removal Rates and Application of Safety Factor
Using the variables described above, annual nitrate removal rates can be computed using Equation 5-2A or 5-2B. It is important to take into account any nitrate removal that was occurring prior to riparian buffer restoration. Restoration may have increased nitrate removal by increasing denitrification rates, area of active denitrification, root depth, duration of groundwater-root zone interactions, or some combination of these variables. Accounting for only the increase in nitrate removal post-restoration, rather than just the gross nitrate removal rate, is essential for determining the final credit value. A safety factor may be applied that takes into account the uncertainty in all these variables (most notably the denitrification rate).

Approach #2: Mass Balance

Equation 5-3 provides an example calculation of groundwater nitrogen removal in a restored riparian buffer. Note that pre-restoration nutrient removal (if any) must be subtracted from this post-restoration value to yield the net increase in nitrogen removal following restoration.

$$GW\ inflow\left[\frac{L}{yr}\right] * Inflow\ N\ concentration\left[\frac{mg\ N}{L}\right] - GW\ outflow\left[\frac{L}{yr}\right] * Outflow\ N\ concentration\left[\frac{mg\ N}{L}\right] = N\ removal\ \left[\frac{mg\ N}{yr}\right] \tag{5-3}$$

The mass balance approach requires quantification of the inflowing and outflowing nitrogen load to determine nitrogen retention within the buffer. This requires monitoring of both groundwater flow rates and groundwater nitrogen concentrations.

Step 1: Quantify Groundwater Flow Rates [Data: Site Specific]
Groundwater flow rates can be quantified based on groundwater elevation monitoring in wells up-gradient and down-gradient from the buffer. These groundwater elevations, along with knowledge of the hydraulic conductivity of the soil, can be used to estimate a groundwater flux through the buffer. In lieu of continuous measurement, modeling may also be used to determine groundwater fluxes.

Step 2: Quantify Groundwater Nitrate Concentrations [Data: Site Specific]
Up-gradient and down-gradient groundwater nitrate concentrations can be determined from repeated sampling of riparian groundwater. Because nitrate concentrations are likely highly variable through time and space, adequate sampling must be conducted to capture this heterogeneity.

Step 3: Quantify Load Reduction and Apply a Safety Factor
Using the groundwater flux and groundwater nitrate concentrations, mass removal rates can be computed (Equation 5-3). Some nitrate removal may have been occurring in the riparian buffer prior to

[4] Essentially, these denitrification conditions occur on a "patchy" basis, both spatially and temporally. Vidon et al. (2010) describe this phenomenon as "hot spots" and "hot moments," when the combination of conditions are conducive to denitrification.

restoration. If this is the case, this pre-restoration nitrate removal rate should be subtracted from the post-restoration value to credit only the nitrate removal enhanced as a result of restoration. If pre-restoration removal rates are not available, comparison to an applicable reference site characterized by the same overall conditions (pre-restoration project) can be made. A safety factor should be applied to account for the uncertainty in this quantification approach. The value of this safety factor could be calculated based on specific uncertainty in the nitrogen concentration values (e.g., standard deviation) and the groundwater flow rates.

5.3.6.2 Sediment and Phosphorus

Phosphorus removal via sediment deposition in riparian buffers can be computed as shown in Equation 5-4. The equation provides a conceptual approach to calculate phosphorus load reduction in a riparian buffer based on sediment trapping efficiency, sediment P concentration, and length of restored buffer (Wenger, 1999).

$$\frac{Buffer\ width\ [m]}{(30.5 + 0.61 * \%\ slope)\ [m]} * Incoming\ sediment\ load\ \left[\frac{kg}{yr - m}\right] * Sediment\ P\ concentation\ \left[\frac{mg}{kg}\right] * Length\ of\ restored\ buffer\ [m] = P\ removal\ \left[\frac{mg\ P}{yr}\right] \tag{5-4}$$

Step 1: Calculate Sediment and Phosphorus Removal Efficiency

Sediment removal efficiency generally increases with buffer width and decreases with buffer slope. The empirical relationship used in Equation 5-4 (Wenger, 1999) first calculates the sediment trap efficiency of the restored buffer and then the total phosphorus load removed based on the trapping efficiency, incoming sediment load, sediment phosphorus concentration and length of restored buffer. (Neglect the phosphorus concentration factor to calculate sediment load only.) The sediment trap efficiency is estimated based on an assumption that a buffer with a width of 30.5 m (100 ft) plus 0.61 m (2 ft) per each 1% of slope is sufficient to trap 100% of incoming suspended sediment (Wenger, 1999). Any areas with impervious surfaces or slopes over 25% are not counted in the buffer width calculation.

Step 2: Quantify Incoming Sediment Load [Data: Site Specific]

The incoming sediment load in surface runoff (per length of restored buffer) can be estimated based on long-term monitoring, back-calculated from sediment accumulation rates, or determined using modeling. Examples of models include the Revised Universal Soil Loss Equation (RUSLE), Soil and Water Assessment Tool (SWAT), and the Water Erosion Prediction Project (WEPP). It is important to account for the sediment generated throughout the watershed that will actually intersect the restored buffer. Depending on local conditions, a field-scale model may be more appropriate to quantify the sediment load actually entering the buffer.

Step 3: Quantify Sediment Phosphorus Concentration [Data: Site Specific; Literature]

Direct quantification of sediment phosphorus concentration is ideal but may be impractical. Assuming representative values based on adjacent land use and soil type is an alternate approach. For example, if adjacent land has a history of agriculture, phosphorus concentrations would likely be higher than in a forested setting. Note that incoming sediment and phosphorus loads are from upland surface soil erosion and are therefore expected to be significantly higher than bank phosphorus concentrations discussed in Section 5.2. Phosphorus concentrations are also typically higher in sediment than the underlying soil due to enrichment.

Step 4: Determine Sediment and/or Phosphorus Removal Rate and Apply a Safety Factor
The parameters defined above can be combined via Equation 5-4 to compute a sediment and/or phosphorus load reduction rate. A safety factor should be applied to this value accounting for uncertainty in sediment phosphorus content as well as the modeled or measured incoming sediment load.

5.3.6.3 Temperature

The heat load offset by riparian shading can be estimated as shown in Equation 5-5, the conceptual approach to quantify heat load reduction due to riparian shading (ODEQ, 2007):

$$Heat\ load\ offset\ \left[\frac{kwh}{day}\right] = Stream\ width\ [m] * Stream\ length\ [m] * Increase\ in\ shade\ density\ [\%] * Solar\ insolation\ rate\ \left[\frac{kwh}{m^2 * day}\right] \tag{5-5}$$

This approach can be applied manually – estimating the increase in shade density after riparian planting using a solar pathfinder or densiometer and using published data on solar insulation rates around the country. However, more detailed modeling of heat load reduction benefits can be quantified using the Shade-a-lator tool developed specifically for this purpose by the Oregon Department of Environmental Quality (DEQ). This tool accounts for seasonal changes in solar inputs, as well as shading potential of different vegetation types. The Shade-a-lator tool is incorporated into the Heat Source software (ODEQ, n.d.). Other documentation of the model is available in Willamette Partnership (2014). Using this existing tool is preferred, but details on applying Equation 5-5 manually are provided below:

Step 1: Calculate Stream Width and Length
Stream length is the total length of stream along which the buffer will be restored. Stream width should be the baseflow wetted channel width, as streams are most often at baseflow and most susceptible to warming under these conditions.

Step 2: Estimate Increase in Shade Density
The increase in shade density must be calculated as the difference in shading after the buffer is restored compared to the current, degraded condition. Common tools for direct shade measurement are a densiometer and solar pathfinder. A forestry densiometer is specifically designed to estimate forest canopy density and can be easily applied to riparian areas. A solar pathfinder uses a reflective dome and known solar trajectories at different latitudes to estimate total shade. However, quantifying shade density with these tools can be complicated due to factors such as overlapping tree canopy and seasonal changes in the sun's position. The Shade-a-lator tool avoids some of these complexities by using a standard GIS-based method for quantifying shade benefits, including projecting future shade as vegetation matures. Other models estimate riparian shading based on stream location and stream and vegetation characteristics (e.g., stream width, canopy height, canopy overhang) (Li et al., 2012).

Step 3: Estimate Solar Insolation
Solar insolation is the solar radiation reaching the earth surface, or the direct solar energy responsible for warming stream water. Seasonal and annual estimates of solar insolation are provided by the National Renewable Energy Laboratory (NREL, n.d.). There are obviously large seasonal differences in solar radiation. Crediting quantification should focus on the time of year of most concern biologically – usually later summer when stream temperatures are highest.

Step 4: Calculate Total Heat Load Offset

The parameters listed above can be combined to calculate total heat load offset using Equation 5-5. An appropriate safety factor (or trading ratio) should be applied. This is especially important for riparian re-vegetation projects where the shade benefits may not be realized for 5-20 years, depending on the type of vegetation. Oregon DEQ uses a 2:1 trading ratio to account for this effect.

Case Study: Temperature Crediting from Stream Restoration in the Tualatin River Watershed

Clean Water Services (CWS) is a water resources utility that manages stormwater and wastewater for 600,000 residents in the Tualatin River watershed in northwest Oregon.

Challenge: In 2001, a Total Maximum Daily Load (TMDL) was developed in the watershed for temperature. Per the TMDL, CWS is required to reduce thermal loading from four wastewater treatment facilities (WWTFs) in the watershed. Directly reducing this thermal loading would require the installation of refrigeration equipment to cool the effluent – an expensive and energy-intensive approach.

Solution: CWS and the Oregon Department of Environmental Quality (DEQ) developed a water quality trading program in the watershed. CWS is taking a two-pronged approach to protecting the water quality of the Tualatin River: 1) thermal load reduction and 2) thermal load trading.

(1) Thermal load reduction: A number of steps have been taken to reduce thermal loading from CWS' WWTFs. A recycled water program reuses WWTF effluent, limiting the volume of water discharged to the Tualatin River. The Forest Grove Natural Treatment System is a wetland system that provides additional treatment of effluent from a conventional WWTF. Updates to another WWTF include a cogeneration facility with air-cooled radiators that cool the treated effluent before discharge to the river. Finally, several industrial dischargers implemented cooling systems that reduce the thermal load reaching CWS' WWTFs.

(2) Thermal load trading: In addition to improved management at the WWTFs, CWS has undertaken an ambitious plan to offset thermal loads through a combination of reservoir management and stream restoration. Cool water from two reservoirs in the watershed is released at strategic times to maintain stream baseflows and reduce water temperatures. Riparian planting projects occur on public lands where large-scale stream restoration opportunities are available and multiple water quality and ecological benefits can be achieved.

Thermal load credits are calculated for both reservoir releases and riparian planting projects to offset thermal loads from the WWTFs. Flow enhancement credits are estimated based on predicted temperature benefits of the flow releases downstream of each reservoir. This depends on the average flow rate in the river and the average release rate from the reservoirs. Thermal load credits from riparian planting are quantified using the "Shade-a-Lator" tool that is integrated into the Oregon DEQ's Heat Source temperature model (see crediting guidance above for more details). The additional thermal load blocked by the restored riparian area (i.e., restored shading minus baseline shading) is the effective thermal load reduction. The crediting program specifies a 2:1 trading ratio for shade credits (i.e., credit is only given for 50% of the calculated benefit). Thermal load reductions must be met for July and August.

Results: Through a combination of direct planting and landowner incentives, 161 riparian planting projects were completed from 2004-2019, totaling 73 stream miles. These riparian planting projects also decrease bank erosion, reducing sediment and nutrient loading to the watershed. In 2019, 938 million kcal/day of excess thermal loading was more than offset by 1,047 million kcal/day in credits from flow augmentation and 481 million kcal/day in riparian planting. A combination of thermal load reductions and thermal load trading is allowing CWS to effectively mitigate temperature impacts on the Tualatin River. Riparian planting programs are providing a significant portion of these benefits more cost effectively than could be achieved with technological improvements alone.

For more information, see Clean Water Services' 2019 Water Quality Credit Trading 2019 Report (CWS, n.d.).

5.4 In-Stream Enhancement

This section provides an overview of in-stream enhancement.

5.4.1 Background

Although research to quantify in-stream nutrient processing is limited, the science continues to improve with significant new published literature and data-driven nutrient studies since the 1990s (Newcomer Johnson et al., 2016). Considering the complexity of subsurface flow and geochemical interactions and the often prohibitive costs for more accurate estimation of performance metrics, developing a representative "credit" or quantification of nutrient removals resulting from in-stream enhancements is difficult and may not be technically appropriate at this time. However, to begin development of such frameworks and considering the increase in applicable nutrient performance studies, two initial approaches for the crediting of in-stream enhancement projects are suggested. Both include a theoretical safety factor to account for variability and lack of site-specific data. As noted in Section 4.2, the bulk of existing literature and data studies focus on the baseflow condition where the majority of hyporheic nitrogen removal is likely to occur. Little or no sediment removal is expected from in-stream enhancement since this restoration technique impacts biogeochemical processes that do not affect sediment dynamics.

In theory, credit for the removal of nutrients and primarily nitrogen (e.g., denitrification) via in-stream enhancement can be conservatively estimated as the result of increased hyporheic exchange between the floodplain and stream channel and within the streambed itself, the latter being representative of in-stream enhancement projects. When this exchange can be estimated and quantified in combination with monitored water quality data and where sufficient conditions exist, a safety factor may be applied to address uncertainties in the estimate of benefits. Although increasing surface-subsurface flow interaction is relatively simple in theory, the design professional should be careful not to inadvertently target in-stream enhancement at the cost of other design objectives. Examples to avoid could include over-widening or raising of the channel bed to qualify for this crediting approach, which could negatively impact other stream functions.

Hyporheic exchange also represents an important process for the potential removal of nutrients in projects such as riparian buffers and floodplain reconnection. A crediting approach for projects that increase wetland and floodplain connections during stormflow conditions is addressed separately in Section 5.5. Given that stream restoration projects often involve more than a single design component and/or objective it is important not to double-count nutrient removals from this underlying process. Projects under consideration for in-stream enhancement credits should clearly identify any spatial limits used to define in-stream hyporheic exchange and floodplain/riparian hyporheic exchange. Hyporheic nutrient removal benefits may be credited for the whole stream-floodplain system under this procedure; however, the project would then be ineligible for nitrogen removal credits under the riparian buffer crediting approach.

5.4.2 Information/Data Requirements

To "qualify" for the development of in-stream enhancement credits, the project in question should first be technically appropriate (i.e., there is a need for improvement) and feasible given the physical/environmental processes at work, the watershed inputs, site-specific conditions, overall drivers, and constraints (see below). Functional assessment approaches offer a method for evaluating how a stream restoration project fits within the "needs" of the stream and watershed system. Furthermore, functional assessments also provide an often cost-effective approach to qualitatively monitor conditions over time and assess various aspects of project performance, which may be used to refine or adjust credits over time.

Hyporheic vs. Surface Transient Storage

When discussing in-stream nutrient removal capacity, it is important to distinguish between hyporheic and surface transient storage (HTS vs. STS). Transient storage is the short-term retention of water within a fixed area, which allows for biogeochemical processing to occur. HTS occurs under the channel bed and is driven by a variety of factors including variable bed topography and bed material conductivity. STS occurs within the stream channel in pools, eddies and other regions where separation from the main downstream flow direction occurs. STS may be more significant than HTS in some streams (Baker et al., 2011; Ensign and Doyle, 2005), but HTS can be responsible for the majority of nutrient uptake (Johnson et al., 2014). The hyporheic zone is more likely to have conditions conducive to denitrification (stable substrata for microbial films, low oxygen water, organic carbon) and therefore has greater potential for permanent nitrogen removal. Differentiating between HTS and STS in the field and in modeling is difficult but important for accurately representing water and nutrient dynamics in stream systems (Boano et al., 2014).

Examples of characteristics that may potentially qualify a project for initial consideration of credit development for in-stream enhancement include:

- Demonstrable surface and subsurface water interaction.
- Available source of carbon.
- High in-stream nitrogen concentrations.
- Periods of saturation from subsurface flow.

Recognizing that quantification of hyporheic nutrient load reductions associated with in-stream enhancement is technically difficult, two potential approaches for consideration have been outlined in the following sections.

5.4.2.1 Approach #1

This first approach is based on the removal of nitrogen through post-restoration hyporheic flow based on a method original developed by the Chesapeake Bay Expert Panel (Schueler and Stack, 2014), but which was recently updated and expanded (Wood and Schueler, 2020). To support this approach, the following data are required:

- Literature denitrification rates for pre- and post-restoration conditions. A default value for the Chesapeake Bay region is given by Wood and Schueler (2020) but may not be appropriate everywhere.
- The area of the Effective Hyporheic Zone (EHZ) where surface and groundwater exchange and denitrification potential area high.
- Basic understanding of baseflow frequency, restored channel-floodplain geometry, and aquifer material

5.4.2.2 Approach #2

The second approach utilizes the fundamental theories included in approach #1 but extends the concepts to a more detailed site-specific framework. To support this approach, the following data are required:

- Hyporheic flux and residence time. These can be calculated using various methods including Darcy's law, with vector direction determined via Freeze and Cherry (1979), numerical modeling, tracer studies, discharge measurements, etc.

- Monitored water quality data within the hyporheic zone of influence or applicable denitrification rates from the literature for similar systems.
- Determination of baseflow characteristics.

For both approaches, the noted information and site-specific data would ideally be collected pre- and post-project implementation to verify estimated removal rates/loads and any credits developed. The reality is, however, that these efforts come at a financial cost and involve multiple forms of uncertainty beyond application to any crediting framework (e.g., method and lab variability, sampling error, representative data sets, and accounting for system heterogeneity). Without long-term or at least post-project monitoring (staged over the course of the respective response time and/or channel forming/bankfull events), a higher factor of safety should be considered when estimating the credit to be granted for nutrient removals. Modeling efforts can also be useful in the development of in-stream enhancement credits as they allow for estimation of the hydraulic components needed to quantify nutrient removals and hyporheic characteristics based on limited data. For example, MODFLOW can be used to estimate both hyporheic flux and residence time based on limited site information and data. Although modeling can produce usable results with limited site-specific inputs, results tend only to be as reliable as the inputs and should be used with caution. With additional uncertainty inherent in the modeling of complex systems, simplifying assumptions are often needed.

More Complicated Hyporheic Quantification Approaches

The approaches described here are relatively simple – with the goal of being easily applied in a variety of situations. Other approaches are more complex but may provide more realistic estimates of pollutant removal in hyporheic zones. One example is comparing the residence time in the hyporheic zone and the first order reaction coefficient for the pollutant of interest. The product of these values is the Damkhöler number (Harvey et al., 2013). If the Damkhöler number is near one, there is sufficient residence time for pollutant removal. If the Damkhöler number is much less than one, the water is not in the hyporheic zone long enough for significant pollutant removal to occur (Herzog et al., 2015). As an example, if the first order decay coefficient for nitrate is 0.038 hr^{-1}, the hyporheic residence time should be at least 26 hours (1/0.038) for significant pollutant removal to occur (Herzog et al., 2015). First order decay coefficients are published in the literature. Residence times at individual in-stream structures can be measured using tracers, modeled numerically, or estimated based on bed material porosity and hydraulic head.

5.4.3 Uncertainty and Simplifying Assumptions

Given the complexity of stream and groundwater flow interactions and geochemical processing, it is necessary to make several assumptions for the quantification of nitrogen reduction benefits for in-stream enhancement. These assumptions include identification of where the actual denitrification is likely to occur, at what conditions, and how restoration projects affect these characteristics. The original Chesapeake Bay procedure defined the theoretical area in which the enhanced denitrification occurs as the "hyporheic box" (Schueler and Stack, 2014). This theoretical box represents the physical space in which the geochemical processes for the denitrification are likely to occur and generally can be assumed to extend some distance in either direction from the stream channel along the length of the respective project.

Although the hyporheic box can be generalized in theory, it may not adequately represent where hyporheic exchange and denitrification are occurring. Recognizing this, Wood and Schueler

(2020) updated this approach using the concept of the Effective Hyporheic Zone (EHZ) – the area of the restored channel *and floodplain* eligible for crediting (Figure 5-1). This includes all portions of the floodplain within 18 inches of the channel bed (or low water level) as this zone is expected to be regularly saturated and have substantial root biomass – both important for denitrification. The extent of the EHZ should be verified in the field using elevation data and soil cores/borings to confirm the presence of saturated soils and porous aquifer materials (e.g., sand and gravel).

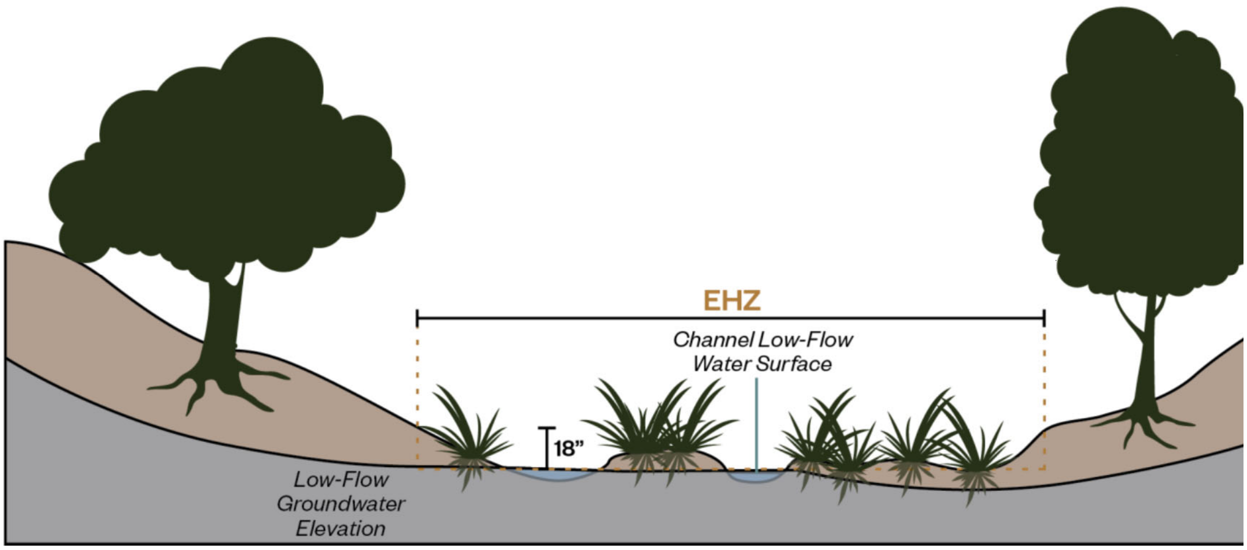

Figure 5-1. Conceptual Depiction of the Effective Hyporheic Zone (EHZ).
Source: Courtesy of Hazen and Sawyer.

The results of in-stream enhancements for denitrification also vary with the flow regime and nutrient loading upstream of the project. This crediting approach assumes that the removal or processing of nitrogen via nitrification/denitrification is most effective under baseflow conditions when hyporheic exchange is most significant. In the absence of long-term seasonal monitoring, modeling could theoretically be used to develop estimates beyond baseflow conditions, provided data are available to calibrate the model.

Crediting for in-stream enhancement (hyporheic exchange) also assumes that organic carbon is available and exists for the duration of the credit. Organic carbon sources may include established vegetation, wood, debris dams, or some combination of these.

Additional simplifying assumptions and technical considerations include:

- The hyporheic zone acts as a steady-state, completely-mixed reactor with first-order denitrification kinetics, constant flow, and sufficient nutrient input to drive reactions.
- The effective hyporheic zone is assumed to be a single discrete area. In reality, the actual zone would likely change with flow and stage. This is particularly relevant when considering the floodplain, which may have chemical processes similar to in-channel baseflow denitrification outside of the baseflow condition (i.e., high flows may extend into the floodplain and return to the stream while the stream itself is outside of the baseflow window).

5.4.4 Regional Geomorphologic Differences

Regional differences related to the denitrification chemistry of in-stream enhancements are important considerations, including factors such as:

- Soil types and their characteristics, including sorption, permittivity, carbon content, and density.
- Channel and floodplain geology.
- Nutrient loading.
- Potential channel evolution.

The original Chesapeake Bay Expert Panel (Schueler and Stack, 2014) recommends a crediting "cap," recognizing the existence of an upper threshold for nitrogen removal by in-stream enhancement that should be determined on a watershed-specific basis to move beyond percent removal metrics. An upper removal threshold could also be developed by identifying an irreducible concentration of nitrate given certain conditions or residence times. While the updated Chesapeake Bay crediting approach does not include such a cap (Wood and Schueler, 2020), this is an area of needed research and is an important consideration when developing regional crediting programs.

5.4.5 Response Time and Longevity

Response time for instream enhancement is expected to be relatively short (on the order of a year or less at the lower end). Some denitrification is likely to be realized shortly after project completion, provided that there is hydraulic connectivity and sufficient environmental conditions to drive the unit processes. Response time is likely to vary by project and location depending on the interactions between in-stream enhancement and other restoration features such as vegetation establishment, floodplain reconnections, rates of organic carbon regeneration, and other features.

The longevity of in-stream enhancement is likely to vary widely based on site-specific conditions and regional differences, as noted above. Conditions potentially reducing longevity include:

- Increased sediment load and grain size distributions that reduce hyporheic flow. These loads can originate in-channel or from the floodplain.
- Unsuccessful vegetation establishment and/or low carbon availability.
- Geomorphic alterations that affect the extents of hydraulic interactions with the hyporheic zone.

Post-project monitoring and/or the use of functional assessments should be considered where feasible to verify response time and longevity.

5.4.6 Crediting Approach

One of the two approaches outlined below should be considered when calculating water quality credits for in-stream enhancement projects that promote hyporheic exchange. The actual credit granted should be program-specific and must consider several programmatic and policy elements such as those introduced in Chapter 7.

5.4.6.1 Approach #1

Building off studies conducted by Kaushal et al. (2008) and Striz and Mayer (2008), the Chesapeake Bay Expert Panel (Schueler and Stack, 2014) developed an initial framework for the crediting of in-stream enhancement. This was subsequently updated by Wood and Schueler (2020) to more accurately quantify the area of hyporheic exchange and use more recent data on representative denitrification rates.

Step 1: Determine "Qualifying" Project Length [Data: Site Specific]
The first step in the approach is to determine how much of the restored project length meets the general requirements (i.e., qualifies) for consideration of the credit. Depending on design, in-stream enhancement projects may increase hyporheic exchange below the channel bed, laterally into the channel banks/riparian zone, or both. The qualifying project length will vary based on the specific processes being targeted. The qualifying project length can be quantified based on the length of stream with increased bedforms, complexity, or structure installation. Creation of meanders and/or bars may also qualify portions of the channel for this credit. Raising the channel bed or lowering floodplain elevations can also increase hyporheic exchange both below the channel and in the adjacent floodplain. Alternative thresholds or metrics for determining the length of project qualifying for credits could also be considered.

Step 2: Calculate the Effective Hyporheic Zone (EHZ) [Data: Site Specific]
The EHZ includes the restored channel area and any floodplain that are less than 18 inches above the channel bed or low water level. These areas are assumed to have sufficient surface water-groundwater exchange and organic carbon to promote denitrification. Dimensions should be determined based on site investigations including detailed topographic data and soil cores/borings to determine the extent of hydric soils, organic carbon, and/or permeable substrate (e.g., sand and gravel) that would promote hyporheic exchange and denitrification.

Step 3: Multiply the EHZ Area by a Reference Denitrification Rate [Data: Site Specific; Literature]
Based on studies of denitrification rates in both restored and unrestored streams, the Chesapeake Bay crediting approach recommends a rate of 2.69×10^{-3} lbs NO_3/ft^2/year of additional denitrification after restoration. This suggested denitrification rate may not be applicable to other areas; it is necessary for others to establish representative rates for their own regions. In general, use of a single denitrification rate neglects the significant variability and uncertainty associated with this process. Accounting for this uncertainty through region-specific monitoring of different stream types would yield more scientifically defensible nitrogen removal estimates. A summary of in-stream denitrification rates published in the literature is available in Lammers & Bledsoe 2017.

Step 4: Adjust Nitrogen Removal Based on Site-Specific Factors [Data: Site Specific]
The updated Chesapeake Bay crediting approach (Wood and Schueler, 2020) includes adjustment factors for nitrogen removal in the hyporheic zone based on three site-specific factors. The first is baseflow frequency, with only seasonal or infrequent baseflow presence leading to a reduction in total nitrogen removal. The second is based on the height of the restored floodplain above the channel bed or low water elevation. Shorter floodplains result in greater nitrogen removal than taller floodplains. The final factor is based the hydraulic conductivity of the underlying aquifer. Gravel and sand have conductivity and therefore result in more hyporheic exchange and more nitrogen removal credit than silt or clay soils. All factors range from 0-1 and are simple multiplied by the annual nitrate removal calculated in Step 3 (see Wood and Schueler (2020) for more details).

Three published studies have applied the original "hyporheic box" approach (Schueler and Stack, 2014) recommended by the Chesapeake Bay Expert Panel (Hester et al., 2016; Doll et al., 2018; Earles et al., 2020). Several important considerations were made that are still relevant, even with the updated procedures:

- The spatial extent of hyporheic exchange is site-dependent and depends on many factors. For example, confining layers below the channel bed may limit the depth of hyporheic exchange (Doll et al., 2018) and hyporheic exchange may only be occurring near installed structures (Hester et al., 2016; Earles et al., 2020).

- Pre-restoration denitrification rates should be accounted for (as they are in the updated protocol). Earles et al. (2020) recognized that denitrification was occurring prior to restoration and used published literature rates on pre- and post-restoration denitrification in their calculations.
- The proposed approach assumes sufficient hydraulic conductivity of the bed sediment for hyporheic flow, and that denitrification is constant regardless of nitrate concentration. This could cause the protocol to overestimate nitrate removal at sites with low hydraulic conductivity and over- or underestimate removal depending on in-stream nitrate concentrations (Hester et al., 2016).

Note that using this approach accounts for denitrification in reconnected floodplains. Projects using this quantification approach should not also receive credit for riparian buffer restoration, which would result in double counting of nitrogen reduction credits.

5.4.6.2 Approach #2

The second approach outlines a similar framework utilizing more site-specific information for projects where such data collection efforts are warranted. This more detailed approach may be necessary in streams where complex hyporheic flow paths and uncertainty regarding the extent and rate of hyporheic denitrification precludes accurate credit quantification via Approach #1. Defining the acceptable level of uncertainty in using the simpler Approach #1 is left to the individual regulatory or crediting agency.

Step 1: Estimate Hyporheic Flux under Baseflow Conditions [Data: Site Specific]

Following the simplifying assumption that the primary nutrient removals associated with hyporheic exchange occur during baseflow conditions, it is first necessary to estimate the portion of the annual flow volume passed as baseflow and the associated flux within the hyporheic zone under such conditions. Several methods, including water level data and system hydraulic properties, numerical modeling, direct discharge measurements, and tracer studies, can be used to estimate the hyporheic flux under baseflow conditions.

Similar to the flow-duration curve concept described in Section 5.5 (floodplain reconnection), time series of stream flow or stage can be related to estimates of hyporheic flux for given ranges. In-situ water level data and an understanding of hydraulic conductivities are required in both cases, regardless of whether modeling is used.

Step 2: Estimate Water Quality Metrics above/below the Restored Hyporheic Zone (or for Control/ Treatment Sites) [Data: Site Specific]

Ideally, these results should be compared to a control section above the restored project area or from similar pre-project conditions at other control sites to quantify how much additional hyporheic exchange and associated load removal is occurring relative to pre-restoration conditions. It may be difficult to isolate the impact of a single restoration project using bulk water quality data. The presence of more than one restoration technique may also complicate quantification. This is especially true if in-stream enhancement and riparian buffers are applied to the same reach for nitrogen removal. In this case, in-stream water quality results would capture the effects of both practices, resulting in double counting of nutrient credits. Either this single credit should be applied for the restoration project (i.e., accounting for both practices), or a credit should be quantified individually for the buffer and subtracted from the in-stream monitoring credit, allowing both practices to be quantified separately.

Step 3: Calculate Annual Load Removal and Apply Factor of Safety

Using the results from Steps 1 and 2, the arithmetic difference between average (or median) water quality concentrations between the control site(s) and the hyporheic zone enhancement can be multiplied by the annual volume of baseflow. The results of this calculation represent an annual estimate of load removed during baseflow conditions. Additionally, if sufficient water quality data are collected and analyzed on a project basis, nutrient spiraling[5] metrics and/or denitrification rates could be developed and multiplied by the respective spatial area of enhanced hyporheic exchange using Equation 5-6. Denitrification is often significant in degraded streams; therefore, only additional area of hyporheic exchange post-restoration should be included in this calculation.

$$Denitrification\ rate\ \left[\frac{mg\ N}{m^2 * yr}\right] * Area\ of\ hyporheic\ exchange\ [m^2] = Nitrogen\ removal\ \left[\frac{mg\ N}{yr}\right] \tag{5-6}$$

Ideally, denitrification rates and/or uptake metrics would be included with submittal to the Stream Restoration Database (Section 6.5). Together with metadata and functional assessment information, the data could potentially be applied to other similar systems.

5.5 Floodplain Reconnection

This section provides an overview of floodplain reconnection.

5.5.1 Background

Floodplain reconnection has the potential to provide water quality improvement via sedimentation, vegetative filtration, infiltration, and nutrient cycling and uptake. Additional improvement may be realized by the transient surface storage provided by the floodplain, which has the potential to reduce stream energy and erosion potential downstream of the connected floodplain. Thus, floodplain reconnection upstream of bed and bank stabilization may greatly improve the overall performance of the restoration effort. The potential level of water quality improvement (as well as other benefits) depends largely on the frequency at which the floodplain is accessed during discharge events (function of connectivity and flow variability) and the available storage or inundation area of the floodplain (spatial scale) relative to the discharge volume (Loos and Shader, 2016).

The Expert Panel of the Chesapeake Bay Program (Schueler and Stack, 2014) developed recommendations for water quality crediting of floodplain reconnection that were updated by Wood and Schueler (2020). This approach is based on estimated annual loads delivered to the stream segment of the project, the percentage of annual flows accessing the floodplain, and the pollutant removal rates documented in literature for wetlands (Schueler and Stack, 2014). The connection volume to receive credit is restricted to the first foot of floodplain inundation to ensure adequate contact with floodplain soils and plants. This one-foot limit can be relaxed if hydraulic modeling demonstrates that floodplain flow velocities are below 2 ft/s (at up to 3 feet of depth). The approach relies on analysis of USGS gage data and hydraulic modeling to estimate the connection volume.

As described below, a similar approach is proposed here with some recommended refinements for estimating load reduction credits based on site-specific monitoring data, more detailed hydrologic/hydraulic modeling, and using effluent concentrations from the International Stormwater BMP Database for wetland systems. Since the assumption that floodplains function similarly to wetlands may not be

[5] Nutrient spiraling describes nutrient processing and transport within stream ecosystems. Generally, a single nutrient molecule may be transported in dissolved form, be ingested by a microorganism (or adsorbed to a sediment particle) and retained for some period of time before being re-released into the streamflow. This cycle of transport, storage, and release gave rise to the term "spiraling."

appropriate for some projects or regions, alternative effluent concentrations could be used such as from detention systems or from monitoring data for comparable floodplain reconnection projects.

5.5.2 Information/Data Requirements

To qualify for credits for nutrient retention and removal, floodplain reconnection projects should have the following characteristics:

- Long-term hydrologic connection between the stream and floodplain is established.
- Organic carbon sources are present within the floodplain.
- Floodplain is deposition-dominated, leading to storage of sediments and nutrients.

To support the proposed approach, the following data are required:

- Average in-stream water quality concentrations immediately upstream of the floodplain reconnection project and preferably at the flow rates or stage at which the stream accesses the floodplain.
- Stage-storage curve for restored floodplain area.
- Flow-stage rating curves for stream segment.
- Flow-duration curves based on hydrologic simulation or long-term flow monitoring.
- Estimated treated concentrations or monitored return flow concentrations during shallow flooding events.

Treated concentrations may be estimated using median effluent concentrations for wetland systems (retention ponds and wetland basins) from the International Stormwater BMP Database. The use of effluent concentrations instead of percent removal is recommended to ensure that load reductions are not over-estimated when in-stream concentrations are low. A summary of effluent concentrations for total suspended solids, phosphorus, and nitrogen are provided in Table 5-2. Another source of is the Non-Tidal Wetlands Expert Panel, which summarizes nutrient removal in restored wetlands (NTW EP, 2019). These data are incorporated in the Chesapeake Bay crediting approach for floodplain restoration (Wood and Schueler, 2020). Restored floodplains may not remove nutrients as effectively as restored wetland systems. Alternative data (for example from detention basins) may need to be used to avoid over-estimating the removal potential of restored floodplains.

Table 5-2. Median Effluent Concentrations for Wetland Systems from the International Stormwater BMP Database.

Constituent	Median Effluent Concentration (mg/L)
Total Suspended Solids	12
Total Phosphorus	0.10
Orthophosphate as Phosphorus	0.03
Dissolved Phosphorus	0.05
Total Nitrogen	1.31
Total Kjeldahl Nitrogen	0.97
Nitrate+Nitrite as Nitrogen	0.20

Source: Clary et al. 2017a.

5.5.3 Uncertainty and Simplifying Assumptions

Similar to in-stream enhancements, the benefits of floodplain reconnection to retain sediment and uptake and/or transform nutrients are highly uncertain due to the complexity of the spatial and temporal interactions between the stream and the floodplain relative to the physical and biogeochemical processes that occur. The hydrodynamics of the stream system may vary significantly at different flow regimes, and it is extremely difficult to predict the quantity of flow and pollutant load that may reach the floodplain. A major simplifying assumption in the approach outlined above is that the storage made available by reconnecting the floodplain would be accessible and that the system would behave similarly to a constructed wetland in terms of water quality performance. During flood stages and for certain designs, there likely would not be clearly defined inlets and outlets to and from the floodplain system and the average residence time and flow path may vary substantially. A two-dimensional hydrodynamic model (e.g., FLO-2D or HEC-RAS 2D) is needed to quantify these flow paths; however, for simplicity, a stage-duration approach is recommended.

5.5.4 Regional Geomorphologic Differences

The functional capacity of floodplains for nutrient retention can vary considerably based on geomorphic region. This primarily arises from differences in floodplain size and inundation frequency, as well as temperature-related and seasonal differences. For example, steeper more confined channels likely have less floodplain area and less frequency of inundation compared to an unconfined lowland (flatter topography) river. Native vegetation and soil differences also affect floodplain deposition processes and organic carbon availability, which are important for denitrification.

5.5.5 Response Time and Longevity

Floodplain reconnection can have a relatively rapid response time (1-5 years), at least in terms of hydrologic connectivity. Vegetation-dependent processes, including deposition, uptake, and denitrification, can take longer to establish depending on the pre-restoration vegetation community. This restoration technique can also have high longevity (>50 years), as long as adequate hydrologic connectivity is maintained (e.g., the channel does not incise).

5.5.6 Crediting Approach

Floodplain reconnection should be considered when a stream is entrenched and disconnected from its active floodplain. The recommended approach for estimating the potential load reduction for floodplain reconnection follows a five-step process as described below.

5.5.6.1 Step 1: Develop Stage-Duration Curves Based on Flow-Duration Curve and Rating Curve [Data: Site Specific]

A flow-duration curve can be developed by summarizing a long-term flow data set (e.g., USGS gage data) or by continuous simulation hydrologic-hydraulic model results. Instantaneous (e.g., 15-minute) data are preferred, but daily data may suffice if more frequent data are unavailable. Higher resolution flow data are generally needed to capture the shorter flooding durations for smaller, flashier streams. At least a 10-year period of record is needed to adequately capture less frequent events. The procedure involves simply calculating the percent of time where various flow rates are equal to or exceeded and then plotting these flow rates versus the daily percent exceedances. This plot can then be coupled with a stage-discharge rating curve for the stream segment where the stream is being reconnected to the floodplain to produce a stage-duration curve. Figure 5-2 illustrates how a flow-duration and stream-rating curve can be coupled to generate a stage-duration curve. If a stage-discharge rating curve is not available, hydraulic modeling can be used to determine the discharge at which the floodplain is inundated.

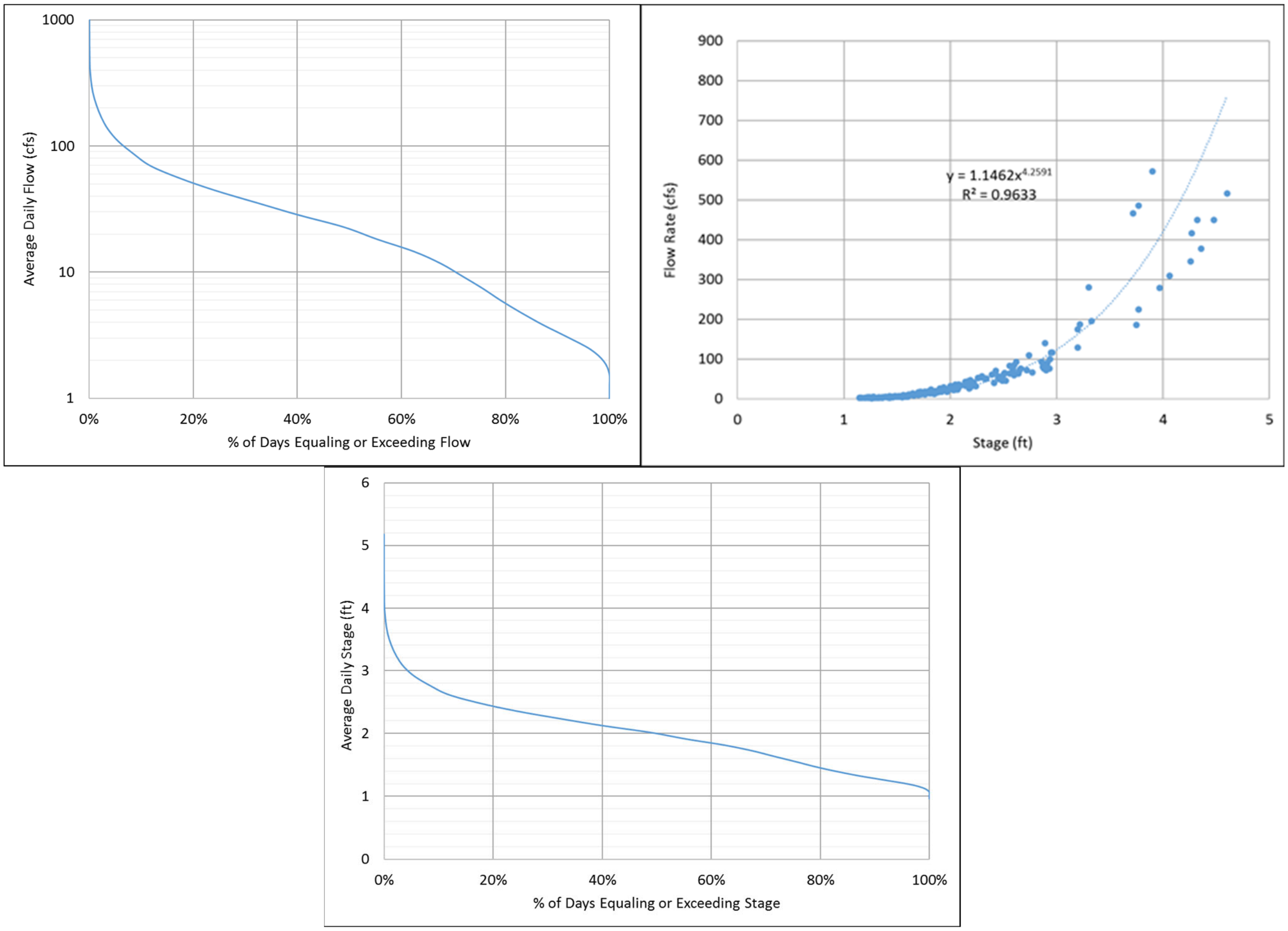

Figure 5-2. Example Integration of Flow-Duration Curve (Top Left) and Rating Curve (Top Right) to Produce Stage-Duration Curve (Bottom).

USGS 14138870 Fir Creek Near Brightwood, OR

Source: Data from USGS, n.d.

5.5.6.2 Step 2: Estimate Percent of Annual Flows That Access the Floodplain to Estimate Connection Volume on Average Annual Basis [Data: Site Specific]

An estimated storage curve or stage-versus-area table for the reconnected floodplain area must first be developed based on the restoration design. Inundation depth and area can be computed based on the known channel geometry and the topography of the surrounding landscape using GIS or a similar tool. Using this information along with the stage-duration curve developed in Step 1, the average volume per year accessing the floodplain can be estimated. Table 5-3 includes a hypothetical storage curve (first three columns) and the calculated average days per year and volume per year that the stream accesses the floodplain based on the curve shown in Figure 5-2. As shown in the table, the hypothetical storage curve indicates that the floodplain is accessed at flow stages of 2 to 3 feet. The actual depth at which the floodplain is accessed and the stage-storage relationship within the floodplain should be estimated using the post-construction channel geometry. Similar to the guidance for the Chesapeake Bay, only the first foot of floodplain inundation should be considered for water quality crediting unless it can be demonstrated that floodplain velocities are low (<2 ft/s) for larger depths (Wood and Schueler, 2020).

Table 5-3. Example Computation of the Average Volume Accessing a Reconnected Floodplain.

Stage (ft)	Floodplain Area Accessed (acre)	Incremental Storage Volume (ac-ft)	% of Days Equaling or Exceeding Stage	% of Days within Stage Increment	No. of Days per Year that the Stream Accesses the Incremental Floodplain Volume	Estimated Volume per Year Accessing Floodplain (ac-ft)
2.00	2.0	0	50%			
2.25	2.5	0.5625	30%	20%	73	41.1
2.50	3.5	0.75	17%	13%	47.45	35.6
2.75	5.0	1.0625	8%	9%	32.85	34.9
3.00	8.0	1.625	4%	4%	14.6	23.7
Total Volume Accessing Floodplain per Year (ac-ft):						135.3

5.5.6.3 Step 3: Estimate Connected Influent Load [Data: Site Specific]

The total volume accessing the floodplain calculated in Step 2 and the in-stream concentrations obtained from monitoring can then be used to estimate the average influent load to the floodplain area according to Equation 5-7.

$$Annual\ inflow\ volume\left[\frac{L}{yr}\right] * Average\ influent\ concentration\left[\frac{mg}{L}\right] = Influent\ load\left[\frac{mg}{yr}\right] \tag{5-7}$$

If a sediment or nutrient rating curve is available, the influent load could also be integrated across the various stream stages that reach the floodplain.

5.5.6.4 Step 4: Estimate Load Removed

The load removed can be estimated using the influent load calculated above and the characteristic effluent concentrations provided in Table 5-2 or other applicable effluent concentration (i.e., from more representative systems or from post-restoration monitoring of similar floodplain reconnection projects). Unless data support significant infiltration within the floodplain during periods of flooding, it is recommended that volume losses be neglected. This recommendation is based on the assumption that floodplain soils likely become saturated during these periods either from rainfall or rising groundwater before the stream accesses them. Considering these assumptions, Equation 5-8 can be used to estimate the load reduction.

$$\begin{aligned} Influent\ load\left[\frac{mg}{yr}\right] &- Annual\ floodplain\ volume\left[\frac{L}{yr}\right] \\ &* \min\left(Outflow\ concentration\left[\frac{mg}{L}\right], Inflow\ concentration\left[\frac{mg}{L}\right]\right) \\ &= Load\ removed\left[\frac{mg}{yr}\right] \end{aligned} \tag{5-8}$$

As noted above, if the volume reductions are neglected, then the annual floodplain volume equals the influent volume. Otherwise, the annual floodplain volume can be estimated via monitoring or modeling.

5.5.6.5 Step 5: Discount Load Removed Based on Drainage Area Ratio [Data: Site Specific]

The final step in this process is to discount the load removed based on a safety factor or regionally calibrated metric that accounts for floodplain connectivity and the expected effectiveness of floodplain reconnection. Originally, the Chesapeake Bay program suggested scaling the credit based on the ratio of the floodplain area to watershed area, with the expectation that this relates to hydraulic retention and the efficacy of nutrient uptake (Schueler and Stack, 2014); however, this is no longer recommended (Wood and Schueler, 2020). Some discounting based on this floodplain:watershed area ratio may be justified, since floodplains with larger drainage areas generally tend to be inundated for greater periods of time than floodplains with smaller drainage areas (Dodov and Foufoula-Georgiou, 2005). However, the topographic extent of floodplain-channel connection is a more direct measure of the likely benefits of reestablishing floodplain connectivity. Floodplain extent depends on stream geomorphology and the geologic setting of the watershed, among other factors. For example, low gradient streams in unconfined valleys have larger floodplains while steeper streams in confined valleys have less potential floodplain area (Church, 2002). Others have quantified the degree of floodplain-channel coupling using GIS data (e.g., Clarke et al., 2008; Carlson, 2009). These GIS methods use topographic data to directly estimate geomorphic floodplain extent and other metrics based on inundation by a representative flood (i.e., "100-year" discharge) and breaks in slope in the surrounding valley. In lieu of this process-based approach, crediting programs can select their own safety factors incorporating calculation uncertainty and their level of risk aversion.

The benefits of floodplain connection are significant, but nutrient credits calculated using this approach may be relatively small if large floodplain areas are not restored (Hester et al., 2016). Additionally, the crediting procedure can be time-intensive, and it may be more cost-effective to focus on crediting other restoration practices (e.g., bank stabilization) (Doll et al., 2018). It should be noted, however, that floodplain reconnection projects may have synergistic benefits for reducing bed and bank erosion and encouraging hyporheic flow (in-stream enhancement) and may be eligible for credits under these categories.

Case Study: Estimating nutrient reduction benefits from restoration of Cherry Creek, Denver, CO

In the Denver metro area, local governments and planning agencies have significant interest in the water quality benefits of stream restoration, but only a few quantitative evaluations from a water quality perspective have been completed. To further explore the benefits of stream restoration in a highly urbanized area with limited space for upland stormwater control measures, Earles et al. (2020) applied the credit quantification procedures outlined in the original 2016 release of this Guidance to a planned urban stream restoration project in Denver to estimate the water quality benefits of the project. A 1-mile reach of Cherry Creek is planned to be restored with the goals of creating a stable channel and floodplain in a degraded section with significant bank failure and channel incision. The restoration plan consists of a series of riffle grade control structures to minimize bed erosion, reconnect the channel with floodplain terraces, provide a stable low flow channel, and provide mildly sloped vegetated banks. Potential nutrient reduction benefits from this project were quantified for Bank Stabilization and In-Stream Enhancement. Riparian Buffer and Floodplain Reconnection benefits described in this Guidance were expected to be minimal and were not estimated.

Bank Stabilization: Historic erosion rates on this section of Cherry Creek were quantified using topographic data from 2004 (2-foot contours) and 2014 (1-foot contours). Comparing elevation contours from these two years allowed for the estimation of average annual sediment export during this time period. It was assumed that the proposed restoration project would achieve a channel in dynamic equilibrium, eliminating net erosion. Therefore, these historic erosion rates were assumed equivalent to the annual sediment load reductions for the restoration project. Six streambank soil samples were obtained and analyzed for total nitrogen (mean = 213 mg/kg) and total phosphorus (mean = 102 mg/kg) content. These average nutrient concentrations were multiplied by annual sediment load reductions to give annual nutrient load reductions.

In-stream Enhancement: Nitrogen removal in the hyporheic zone was estimated at each proposed constructed riffle using the hyporheic "box" approach from the 2016 Guidance. The size of the hyporheic "box" was estimated for each riffle, then multiplied by estimated soil bulk density (NRCS, 2008) and pre- and post-restoration denitrification rates from the literature (Kaushal et al., 2008). Nitrogen removal was calculated as the difference in post- and pre-restoration hyporheic removal.

Estimated average annual sediment, TN, and TP load reductions. Values in brackets represent minimum and maximum nutrient contents values from soil samples.

Project Feature	Sediment (tonnes/year)	TN (kg/year)	TP (kg/year)
Channel stabilization	1,150	245 [22-611]	118 [65-185]
In-stream Enhancement	--	394	--
TOTAL	**1,150**	**640**	**118**

Based on this exercise, Earles et al. (2020) concluded that the restoration project would result in significant reductions in suspended sediment, nitrogen, and phosphorus loads. For comparative purposes, the authors also calculated potential sediment load reductions from extended detention basins and determined that 100 regional basins (or 1,000 sub-regional basins) would be needed to achieve the same annual sediment load reductions as this restoration project. Important lessons for quantifying nutrient and sediment reduction benefits include:

- Obtain site-specific bank nutrient concentrations. Measured phosphorus concentrations in this stream are lower than many other areas (Lammers & Bledsoe, 2017). Using literature values would have resulted in over-estimating phosphorus reduction.
- Only account for *additional* hyporheic denitrification. The authors calculated the volume of the hyporheic zone only near individual structures and accounted for likely denitrification that was occurring prior to restoration.
- If this project were being used in a formal pollutant trading program, additional site-specific measurements for parameters such as soil bulk density would be important.

CHAPTER 6

Monitoring and Reporting for Credit Verification

Many reputable guidance documents for monitoring streams pre- and post-restoration have been developed in various parts of the country, as summarized in Table 6-1. Generally, stream restoration crediting programs should integrate locally vetted monitoring guidance when it is available. Appropriate restoration monitoring techniques may vary depending on regional geomorphic and climatic conditions, as well as the specific objectives of a local crediting program. While this guidance document provides information on how to calculate credits, steps for quantifying the necessary parameters may differ among regions. Four general approaches can be used to quantify and/or verify the benefits of stream restoration projects including direct monitoring of water quality and stream geomorphology, functional assessment, modeling, and/or some combination of these approaches. Each of the verification approaches described above has strengths and weaknesses. The most practical approach is likely to incorporate aspects of all three approaches. Each of these approaches are discussed further in the sections below, followed by an overview of how the recently developed Stream Restoration Database can be used to report and track project monitoring and assessment data.

Monitoring in-stream water quality can be especially useful in assessing the cumulative benefit of a number of restoration practices throughout a watershed. Depending on the size of an individual restoration project, it can be difficult to detect nutrient-related benefits among the noise of temporal in-stream water quality data. Quantification of the benefits of individual projects or practices may require a more intensive monitoring approach. As outlined in Chapter 5, direct monitoring of nutrient fluxes into and out of a restored floodplain, stream reach, or riparian buffer, or nutrient processing rates, may be the most robust approach. Direct monitoring of these attributes can be difficult and costly. These difficulties reinforce the utility of a functional assessment approach that only requires this detailed sampling and monitoring at a relatively small number of regional reference sites to which restoration projects can be compared using simple indicators. Detailed discussion on specific sampling plans or sampling density is not included in this chapter. Various monitoring guidance documents (Table 6-1) consider these issues and individual crediting programs should develop their own specific monitoring requirements.

While monitoring requires significant investment, it is critical to improve our understanding of the benefits of stream restoration and provide an empirical basis for quantifying water quality benefits. Crediting programs could incentivize this additional effort by allowing projects to receive higher credits than what are calculated using the crediting procedures outlined here if monitoring data show greater nutrient, sediment, or temperature reduction benefits. If less benefits are observed, the credit calculated using standard procedures should be given to the project. This avoids penalizing projects where this valuable monitoring data are collected (S. Herzog, personal communication September 8, 2020).

6.1 Monitoring: Direct Measurement of Water Chemistry

Monitoring and evaluation provides important information on restoration project effectiveness and is an essential aspect of crediting programs. Yochum (2018), Roni and Beechie (2013), Palmer et al. (2005) and others report that periodic literature reviews continue to show that only a small fraction of restoration projects internationally are adequately monitored, although monitoring may be becoming more common, at least in the Chesapeake Bay region (see Stack et al., 2018 for examples). Roni and

Beechie (2013) identify multiple explanations for inadequate monitoring that range from inadequate funding to technical and non-technical issues. These issues include a lack of clearly defined questions, improper study design, inadequate spatial and temporal replication, insensitive monitoring parameters, poor project implementation or management, and lack of period analysis and publication of results (citing Reid, 2001; Roni et al., 2005; and Roni et al., 2008). Ideally, monitoring and evaluation should be part of the project design and occur well before actions are implemented on the ground (Roni and Beechie, 2013). Figure 6-1 illustrates the general steps that should be considered in development of a monitoring program (adapted from MacDonald et al., 1991).

Monitoring programs for purposes of crediting typically require some type of baseline monitoring to document existing conditions, as well as some type of implementation and effectiveness monitoring. At a minimum, all projects should have some type of before and after assessment. Many references are available regarding experimental designs for stream restoration monitoring (Table 6-1). The most common approaches include the use of a before-after or a before-after control-impact (BACI) design (or some variation of these approaches). The three general types of stream conditions that are often monitored include:

- Treatment: the treatment is the restoration action, which focuses on the "impact" of the project.
- Control: A control site has characteristics that are very similar to the treatment site; however, no treatment is applied.
- Reference: A reference site represents the desired or targeted condition. An example would be a stream with similar characteristics to the treatment and control sites, but in a relatively natural condition.

Control, reference, and treatment sites should be similar in drainage area, stream flow, geology land use, gradient, vegetation, and potentially other factors (Roni and Beechie, 2013).

The three basic monitoring designs (which can be enhanced through the use of multiple replications) include:

- Before-After (BA): This approach involves monitoring the treated site before and after restoration. This is the simplest monitoring approach, but interpretation can be affected by natural trends or conditions at the time of monitoring.
- Before-After-Control-Impact (BACI): Building upon the before-after study design, a control site can be added to reduce the possibility of interpreting a natural trend as a treatment effect and to reduce the effect of temporal variability. In this case, the control and the treatment are both monitored before and after the restoration is implemented.
- Post-Treatment Designs: In cases where "before" data are not available for comparison due to planning and funding constraints, post-treatment monitoring designs rely on a comparison of treatment and appropriate control reaches. This design assumes that the control reach was similar to the treatment reach before restoration was implemented. There are two general types of post-treatment monitoring designs: the intensive post treatment (IPT) and extensive post treatment (EPT) design. The IPT approach focuses on long-term monitoring for a few pairs, whereas the EPT approach focuses on monitoring more pairs of locations spatially. (See Roni and Beechie [2013] for additional discussion.)

For more advanced guidance on monitoring design, see the references listed in Table 6-1.

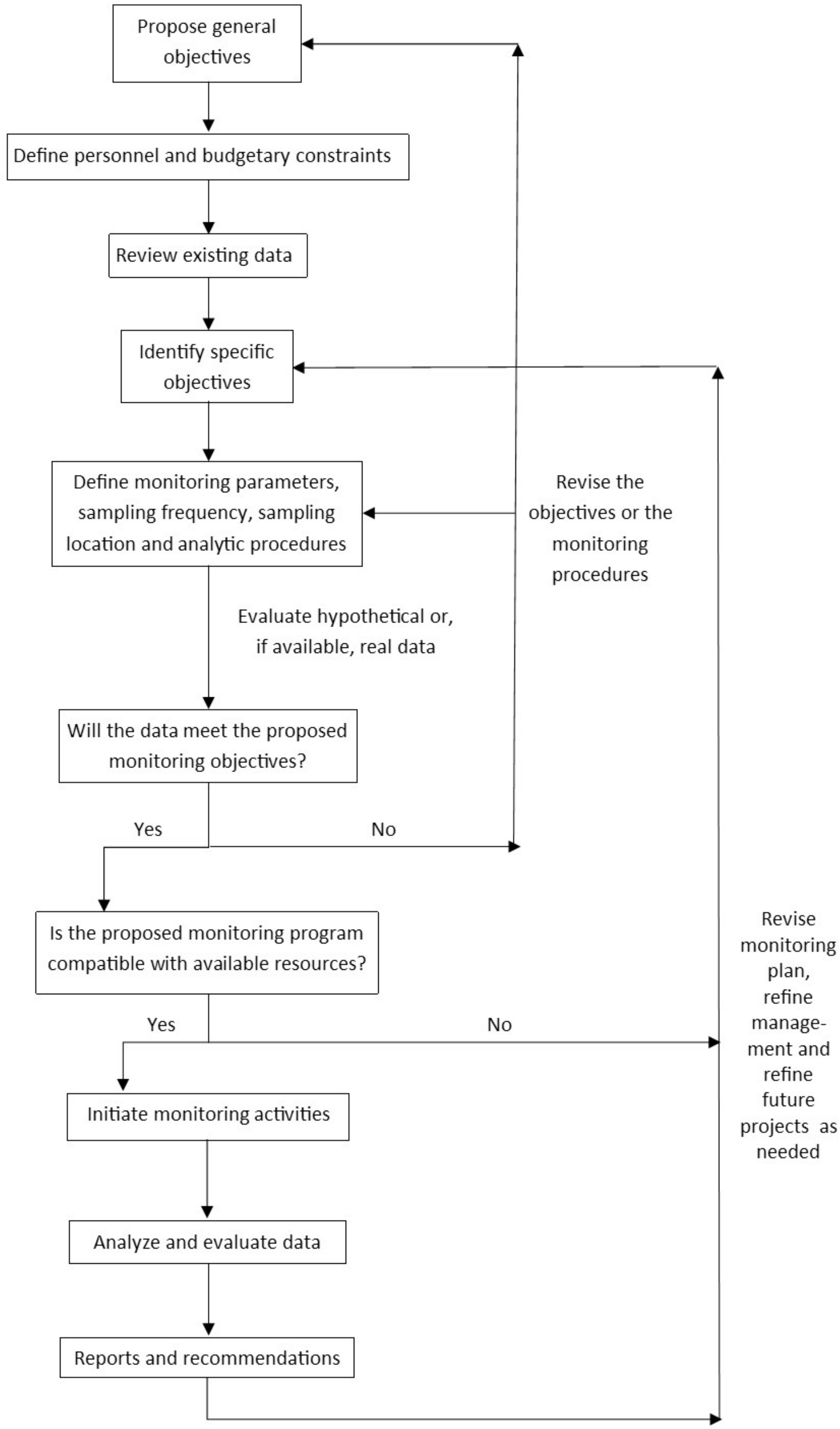

Figure 6-1. Steps for Designing a Stream Restoration Monitoring Program.
Source: Adapted from MacDonald et al., 1991.

Table 6-1. Representative References for Monitoring Stream Restoration Projects.

Berkowitz and White, 2013. Linking Wetland Functional Rapid Assessment Models with Quantitative Hydrological and Biogeochemical Measurements across a Restoration Chronosequence. *Soil Science Society of America Journal*, 77 (4): 1442-1451.
Bonfantine et al., 2011 Guidelines and Protocols for Monitoring Riparian Forest Restoration Projects.
Brinson et al., 1995. A guidebook for application of hydrogeomorphic assessments to riverine wetlands. WRP-DE-11. Vicksburg, MS: U.S. Army Engineer Waterways Experiment Station.
Burton et al., 2011 Multiple Indicator Monitoring of Stream Channels and Streamside Vegetation.
Davis, S., Starr, R., and Eng, C., 2014. Rapid Stream Restoration Monitoring Protocol. Coastal Program, Stream Habitat Assessment and Restoration, U.S. Fish and Wildlife Service Chesapeake Bay Field Office. CBFO-S14-01.
Doyle, et al., 2007. Developing Monitoring Plans for Structure Placement in the Aquatic Environment.
Guilfoyle and Fischer, 2006. Guidelines for Establishing Monitoring Programs to Assess the Success of Riparian Restoration Efforts in Arid and Semi-Arid Landscapes.
Natural Resources Conservation Service, 2007. Ch11 Rosgen Geomorphic Channel Design and Ch16 Maintenance and Monitoring.
Roni and Beechie, 2013. Stream and Watershed Restoration: A guide to Restoring Riverine Processes and Habitats, Chapter 8 Monitoring and Evaluation of Restoration Actions.
Starr, R., Harmann, W., and Davis, S., 2015. Final Draft Function-Based Rapid Stream Assessment Methodology. Habitat Restoration Division, Chesapeake Bay Field Office U.S. Fish and Wildlife Service. CAFE – S15 – 06. May.
Thom and Wellman, 1996. *Planning Aquatic Ecosystem Restoration Monitoring Programs.*
U.S. Environmental Protection Agency, 1998. EPA Guidance for Quality Assurance Project Plans. EPA QA/G-5. Office of Research and Development.
U.S. Fish and Wildlife Service, Chesapeake Bay Field Office Stream Protocols/Tools (website with multiple references pertaining to monitoring and functional assessment) accessible at: https://www.fws.gov/ChesapeakeBay/restoring-habitat/stream-restoration/stream-protocols.html.
U.S. Geological Survey. 2015. National Field Manual for the Collection of Water-Quality Data Techniques of Water-Resources Investigations Book 9, Handbooks for Water-Resources Investigations.
Wilder, T.C., Rheinhardt, R.D., and Noble, C.V., 2013. "A regional guidebook for applying the hydrogeomorphic approach to assessing wetland functions of forested wetlands in alluvial valleys of the coastal plain of the southeastern United States," ERDC/EL TR-13-1, U.S. Army Engineer Research and Development Center, Vicksburg, MS.
Yochum, S.E., 2018. Guidance for Stream Restoration. Technical Note TN-102.4 U.S. Department of Agriculture National Forest Service Stream & Aquatic Ecology Center. Fort Collins, CO.

Some of the challenges related to developing monitoring programs to verify the benefits of stream restoration projects include:

- Long-term monitoring to verify "success" – how long should monitoring be conducted?

- Minimum monitoring requirements – how many samples should be collected and for which pollutants?
- Timing of sample collection and performance assessment intervals – under what hydrologic conditions should samples be collected?

There is not a one-size-fits-all answer to these questions and the ideal monitoring design is often not met due to practical and funding constraints. Ideally, multiple years of pre-project monitoring data would be available for BA and BACI designs. The numbers of samples required are affected by the variability of the monitoring parameters selected. In some cases, data may already exist for non-project related reasons and can be used for pre-project characterization, provided that important factors such as collection methods are considered. From a practical perspective, most monitoring programs will also be limited in breadth of the monitoring that can conducted. Generally, it is preferable to monitor fewer parameters more robustly, than to monitor many parameters infrequently. Roni and Beechie (2013) provide this general guidance for selection of monitoring parameters, which should be:

- Tied to the objectives of the project
- Relevant to the monitoring questions or hypotheses
- Sensitive or responsive to the restoration action
- Efficient to measure
- Limited variability

While the focus of this guidance is primarily on water quality, it is important to recognize that stream restoration projects may be implemented for many reasons, with water quality being an ancillary or secondary objective in some cases. Regardless of whether the project is designed specifically for water quality or for other objectives, the minimum field and laboratory parameters in Table 6-2 are recommended for monitoring if the stream restoration project has sediment or nutrient related monitoring objectives.

Monitoring methods are important for obtaining representative samples. Significant guidance already exists on sample collection methods. The USGS National Field Manual for the Collection of Water Quality Samples (Wilde et al., 2014) is a key reference for guidance on this topic, with Table 6-3 summarizing several options for in-stream sample collection. Similar guidance is available for groundwater sample collection in Wilde et al. (2014).

Table 6-2. Minimum Recommended Water Quality-Related Monitoring Parameters for Stream Restoration Projects.

Field Parameters (Can Be Monitored Using Field Probes)	
Temperature	Specific Conductivity
pH	Turbidity
Dissolved Oxygen (DO)	Flow Rate and Depth
Laboratory Parameters (for Nutrient[1] and Sediment Objectives)	
Alkalinity	Nitrate/Nitrite
Total Suspended Solids (or Suspended Sediment Concentration)	Total Kjeldahl Nitrogen
Total Dissolved Solids	Ammonia
Total Phosphorus	Particle Size Distribution[2]
Orthophosphate	Chlorophyll-a (or Other Biological Monitoring)
Total Organic Carbon	*Note: certain metals (e.g., iron, selenium) may also be of interest on a site-specific basis.*

[1]Nitrogen-related parameters can be used to calculate Total Nitrogen and Total Inorganic Nitrogen
[2]Particle size distribution should ideally be monitored for several pre-restoration and post-restoration events under varying flow conditions, even if it is not evaluated for every monitoring event.

Table 6-3. Water Quality Sample Collection Methods for Stream Restoration Monitoring.

Representative Method	Description
Point Sample (Grab)	If the stream depth and (or) velocity is not sufficient to use a depth- and width-integrating method to collect a sample, use the hand-dip method. Open-mouth sample bottles can be used.
Automatic Samplers (Composite)	Automatic pumping samplers (autosamplers) with fixed-depth intake(s) can be used to collect samples at remote sites, from ephemeral, small streams, or from urban storm drains where stage rises quickly. These samplers can be programmed to collect samples under a combination or variety of conditions such as precipitation, stage, or discharge. Samples from automatic samplers or pumps are considered point-integrated samples. Automated point samplers generally do not result in a discharge-weighted sample unless the stream is completely mixed laterally and vertically.
Width-Integrated Sampling and Depth-Integrated Sampling	Isokinetic, depth-integrating methods are designed to produce a discharge-weighted (velocity-weighted) sample; that is, each unit of stream discharge is equally represented in the sample, either by dividing the stream cross section into intervals of equal width (EWI) or equal discharge (EDI). The analyte concentrations determined in a discharge-weighted sample are multiplied by the stream discharge to obtain the discharge of the analyte. (Although recommended by the USGS, these sampling methods are not implemented on many projects due to additional effort/cost.)
Stage Sampler	Single-stage samplers are designed to obtain suspended-sediment samples from streams at remote sites or at streams where rapid changes in stage make it impractical to use a conventional isokinetic, depth-integrating sampler. Single-stage samplers can be mounted above each other to collect samples from various elevations or times as streamflow increases and the hydrograph rises.

For additional information on sampling methods, see USGS National Field Manual for the Collection of Water-Quality Data (USGS 2015)

6.2 Monitoring: Geomorphic Measurements

While direct in-stream water quality measurements are important, many of the benefits of the restoration practices discussed here require assessment of geomorphic indicators. This monitoring is important to ensure long-term channel stability or maintenance of dynamic equilibrium where no directional trends in channel morphology are observed (e.g., alternating deposition and erosion on the channel bed is acceptable, but continued erosion and channel incision indicates a move away from dynamic equilibrium). Bank and bed stabilization is directly tied to channel stability, but all restoration practices depend on maintenance of a relatively stable channel morphology. Riparian buffers and subsurface nutrient removal become ineffective if the channel incises and the groundwater table never intersects the rooting zone. Floodplain reconnection can similarly be limited if incision confines large flows within the channel. Hyporheic exchange is dependent on geomorphic variability of the channel bed. Sustained erosion or deposition could eliminate this surface-subsurface transfer.

Geomorphic monitoring can follow two general approaches. Direct measurement is the most accurate and robust option but also requires the greatest investment. This involves the establishment of monumented cross sections along the stream reach with sufficient survey frequency to observe changes in channel morphology and geometry. Longitudinal surveys can be used to monitor bed complexity and aggradation/erosion through time. These surveys should be of adequate spatial density to capture bedform morphology and thalweg variability (e.g., pool-riffle sequences). Finally, bank heights and angles should be measured (or obtained from cross section surveys) to assess bank stability and erosion through time.

If these direct monitoring and quantification approaches are not feasible, indirect methods such as a rapid geomorphic assessment (RGA) may be suitable. This approach is more qualitative and uses indicators or simple measurements (e.g., bank heights) to assess the geomorphic complexity and stability of a stream channel. A number of RGA methodologies have been developed (e.g., Bledsoe et al., 2010, 2012; Simon and Downs, 1995), including some specifically designed for monitoring stream restoration projects (Starr et al., 2015). Recently, a credit verification approach was outlined for the Chesapeake Bay program that provided recommendations specific to the basic restoration approaches included in this guidance (Schueler and Wood, 2019). The general approach consists of rapid visual assessment of the project reach, with more detailed data collection at any observed problem areas (e.g., eroding banks or disconnected floodplain).

Regardless of the method used (surveys or RGAs), it is important to conduct repeat visits with adequate temporal frequency to allow changes to be observed through time. In addition, continued monitoring throughout the duration of a nutrient reduction credit is essential to ensure channel form remains consistent and the functions of restoration practices are not compromised. This repeat monitoring is also essential for identifying potential required maintenance to ensure continued function of the restored stream. Different restoration projects may have different maintenance requirements depending on the specific techniques used and potential damage sustained (e.g., from floods) since project completion (Moore and Rutherfurd, 2017). Ideally, stream function is restored such that these systems are self-healing; however, this may not always be achieved. Requiring project maintenance may be incorporated into credit verification to ensure continued water quality benefits (Schueler and Wood, 2019).

6.3 Functional Assessment for Nutrient Removal and Stream Restoration Benefits

Functional assessment procedures (FAPs) translate best available scientific knowledge into reasonable predictions of how streams, riparian zones, and wetlands function in different landscape contexts. They can be used to indicate the potential and priorities for nutrient reductions as well as other purposes including flood mitigation and provision of wildlife habitat. Functional assessment enables users to predict the functioning of an ecosystem without the need for extensive and costly data collection. FAPs therefore provide a methodology that can be used by both experts and non-experts to assess ecosystem functions relatively rapidly (Maltby, 2009)

As demonstrated by the discussions above, direct quantification of nutrient removal and avoided loading from stream restoration projects is extremely difficult. While the relationships used are not theoretically complex, quantification of the required parameters is difficult, time-consuming, and relies on methodologies with significant uncertainty. This makes it unlikely that these approaches will be applicable to many crediting programs. While direct quantification should be used where practical, a functional assessment approach will be preferred in many cases, or should at least be considered to supplement quantification.

Functional assessments have a long history of application to restoration and compensatory mitigation of riverine wetlands (Brinson, 1993; Brinson et al., 1995). Briefly, the functional assessment approach requires a number of steps. First, different functions of interest are identified (e.g., riparian denitrification). Next, indicators for these various functions are defined. These functional indicators are surrogate variables that indicate the presence or absence of the function of interest without intensive quantification. Multiple functional indicators may be identified for a single function of interest. These functional indicators are then combined to yield a single index for that function. The functional assessment approach requires a well-established database of function in reference systems to which the restored system can be compared. High functional indicator scores suggest the studied system is functioning similar to the reference while lower values suggest lower function in the restored system. This reference site calibration will be the primary task of any regional crediting program.

Others have attempted to identify the primary functions of interest for stream restoration (Fischenich, 2006; Harman et al., 2012). Functions pertinent to stream restoration for nutrient removal include stream evolution processes, surface/subsurface water exchange, quality and quantity of sediments, and chemical processes and nutrient cycles. For the purposes of this guidance, a set of functional indicators for each restoration practice has been developed, as summarized in Table 6-5. These indicators and suggested direct and indirect quantification methods should not be taken as direct recommendations. Rather, they have potential to serve as useful indicators depending on the specific objectives and needs of a regional crediting program. Furthermore, these indicators do not take into account the location of selected restoration techniques. For example, a restored riparian buffer adjacent to forest may have the same functional capacity as a buffer adjacent to agricultural land, but the expected functional uplift will be much lower for the forested contribution because of lower pollutant loading from the uplands. Floodplain connectivity and in-stream enhancement are similarly dependent on supplied nutrient loads for functional assessment.

The difference between function capacity and functional uplift is important to recognize when developing a regional function assessment framework. Bledsoe et al. (2019) evaluated functional assessment methods for urban stream restoration. They note that many existing tools and methods are unfairly biased against urban stream restoration. Degraded urban streams may have relatively low functional capacity (total functional potential) but have significant potential functional uplift (increase in function). Functional uplift is largely a function of watershed condition, and the authors recommend accounting for several watershed characteristics when assessing this functional lift potential:

- Use effective imperviousness instead of total imperviousness. Effective imperviousness is the impervious surface that directly contributes surface runoff into the stream network. This can be quantified based on road and pipe density in a watershed.
- Account for stormwater controls and differentiate between stormwater controls that simply reduce peak flow rates of large events versus those that address peak flows and runoff volume across a range of storm sizes.
- Assess riparian buffer condition both at the restoration site and in the broader watershed.
- Assess channel stability and geomorphology upstream of the restoration project.

The example indicators provided in Table 6-5 may be used to estimate a baseline functional capacity that can then be modified based on relative supplied pollutant loads. Regardless of the specifics of a functional assessment quantification approach, a higher score (using indicators such as those outlined in

Table 6-5) translates into a higher nutrient reduction credit. These scores are generally proportional to the reference site (0-1, with higher values indicating function more comparable to the reference location). Actual pollutant reduction credits are then estimated based on the pollutant removal quantified at the reference site.

It is essential to develop regional and stream-type specific references because functional *capacity* may vary considerably between systems. For example, a relatively steep, confined stream has significantly less capacity for floodplain connection and nutrient removal than a low gradient coastal plain stream. Similarly, the functional capacity for in-stream nutrient removal will be much less in a stream with a smooth and relatively impermeable bed compared to a stream with large bedforms, in-stream wood, and porous substrate. Accounting for differences in functional capacity between streams and regions is essential. For this reason, potential functions and indicators of interest are simply identified in Table 6-4 for the four restoration techniques, but development of a formal quantification methodology is beyond the scope of this crediting guidance. The methodology will vary depending on local conditions and must be developed regionally.

While these quantification methodologies and regional reference sites will be unique to specific areas, it is important to employ consistent data collection methods and report results in equivalent units to enable inter-site comparison and the development of regional reference site databases. Standardized monitoring protocols for stream restoration have been developed (Table 6-1) and national reporting databases for water quality (NWQMC, n.d.) and stream restoration (WRF, n.d.) exist or are in development. Utilizing these existing resources will facilitate data sharing and consistent reporting which will enhance development of functional assessment approaches for stream restoration monitoring and crediting.

Even with a robust dataset to support the development of a functional assessment system, site-specific watershed characteristics should always be accounted for. For example, a restored site may have indicators suggesting high function, but may have upstream impairment (e.g., acid mine drainage or high salinity from road runoff) that limits functional lift. Functional assessment methods can be misused, and care is needed to ensure the indicators being measured are representative of desired functions (Cole, 2006). This may require more than a single site visit to determine, and potentially the use of ancillary data sources (e.g., hydrologic records) to support site assessment (Stander and Ehrenfeld, 2009). Importantly, it is essential that watershed condition and context is considered to ensure that stream functions are not limited by stressors from outside the restored site (Bledsoe et al., 2019).

Table 6-4. Example Functional Indicators and Descriptions of Potential Direct and Indirect Measures for Each Stream Restoration Practice.

Restoration Practice	Functional Indicator	Description
All Practices	Channel Stability/ Equilibrium	The effectiveness of all restoration practices depends either directly or indirectly on long-term channel stability or maintenance of dynamic equilibrium. Channel morphology and geometry may change in response to variable flows, but there should be no long-term trends in this channel evolution (e.g., continued incision). This is important for ensuring reduced loading from channel erosion, maintaining the integrity and function of the riparian buffer and reconnected floodplain, and preserving the geomorphic variability that drives hyporheic exchange processes. Assessment of channel stability through time requires repeated geomorphic monitoring, as described in Section 6.2.
Bank and Bed Stabilization [P, sediment]	Net Change in Channel Erosion Potential	Quantifying observed changes to channel geometry and morphology post-restoration is important to assess the effectiveness of restoration in halting further channel erosion. This assessment may take the form of a time series of cross-section and longitudinal surveys to monitor changes through time. Even with the same degree of disturbance, different channel types have different potentials for response. The important indicator is erosive power relative to the resistance of the channel material. For example, a fine-grained stream without stabilizing bank vegetation would have much greater erosion potential than a coarser-grained stream with natural or constructed grade controls, which would limit potential channel incision. This indicator lends itself to modeling to determine probabilities of potential channel evolution trajectories through time. This can allow for quantification of total avoided channel erosion and associated pollutant loading. Channel evolution model (CEM; Figure 3-1) stage is also important. Stabilizing a Stage II channel will avoid greater channel change than stabilizing a Stage III-IV channel. The proximity to this transitional threshold is therefore an important indicator. Visual assessment methodologies exist for rapid assessment of channel response potential (e.g., Bledsoe et al., 2010, 2012; Caltrans 2014; Simon and Downs 1995).
	Bank Stability	Bank stability is an important component of determining the success of bank and bed stabilization in preventing erosion. Visual assessment can determine the extent of eroding banks based on bank geometry, presence or absence of well-established vegetation on the bank face, and whether or not blocks of failed bank soil are present.
	Soil Phosphorus Content	Relative soil nutrient content can only be roughly estimated visually based on soil type and knowledge of regional soil phosphorus concentrations. For example, phosphorus concentrations tend to increase with silt-clay content so finer-grained bank soils would be indicative of higher phosphorus content.
Riparian Buffer [P, N, temperature]	Buffer Width [P, N]	Pollutant removal efficiency increases with buffer width. Quantification of buffer width relative to stream width may provide a useful metric of nutrient retention capacity relative to nutrient loads in the stream.
	Groundwater Connectivity	Groundwater interaction with the organic carbon-rich root zone is important for denitrification and other nutrient removal pathways. Entrenchment ratio may be a useful indicator of groundwater connectivity.
	Organic Carbon Availability	Organic carbon is required for denitrification to occur. Organic carbon availability can be assessed qualitatively based on vegetation cover, leaf litter, or detritus.

(continued)

Table 6-4. Continued.

Restoration Practice	Functional Indicator	Description
Riparian Buffer [P, N, temperature]	Deposition Potential [P]	Deposition of sediment-bound nutrients can be an important removal pathway, especially for phosphorus. Deposition potential can be assessed visually based on presence or absence of deposited sediment and debris as well as micro-topographic variability and roughness, which can greatly increase deposition potential.
	Vegetation Height and Density [temperature]	Taller and denser vegetation will provide more shade and reduce water temperatures more than shorter and sparser vegetation. The most important parameter is the fraction of the stream that is shaded. Although this will change throughout the year, a simple metric may be vegetation height relative to stream width.
Floodplain Connectivity [P, N]	Overbank Frequency [P, N]	Frequency of overbank flows is an indicator for floodplain connectivity as its function for nutrient removal depends on hydrologic connection to the stream. Overbank frequency can be determined through a hydrologic analysis or by visual indicators such as deposited debris or water lines.
	Inundated Area [P, N]	The area of inundation is also important for determining floodplain function. Floodplain area relative to watershed area may be a useful indicator.
	Deposition Potential [P]	Deposition of sediment-bound nutrients can be an important removal pathway, especially for phosphorus. Deposition potential can be assessed visually based on presence or absence of deposited sediment and debris as well as micro-topographic variability and roughness, which can greatly increase deposition potential.
	Organic Carbon Availability [N]	Similar to riparian buffers, organic carbon availability is important for biologically mediated nutrient removal. Similar indicators, including vegetation cover, leaf litter, or detritus, can be used.
In-Stream Enhancement [N]	Bed Complexity	Bed complexity is important for forcing hyporheic exchange in streams. This can be assessed quantitatively based on surveys and statistical analysis (e.g., Gooseff et al., 2007; Yochum et al., 2012) or qualitatively by accounting for bedforms, in-stream wood, and constructed features.
	Bed Conductivity	The hydraulic conductivity of the bed material is also important for determining hyporheic zone exchange potential. This will most likely be assessed qualitatively based on bed material grain size and embeddedness.
	Organic Carbon Availability	The importance of organic carbon for nutrient removal processes has already been discussed. Indicators for riparian buffers and floodplains can be used but may not be readily apparent within the stream channel. The presence of organic carbon indicators on the channel margins may suggest sufficient availability within the stream and hyporheic zone even if carbon sources on the channel bed are not observed directly.
	Bed Contact	Contact between the stream flow and channel bed is essential for hyporheic exchange and nutrient retention. The ratio of wetted surface area to flow volume is a suitable parameter that accounts for the relative potential for surface-subsurface exchange.

6.4 Modeling

Given the significant challenges and costs associated with collecting adequate data to verify the success of stream restoration projects, the use of predictive models to assess performance will likely be needed in the foreseeable future. Modeling of stream restoration project benefits would be considered a presumptive approach for quantification or verification of credits. While there are numerous models available that can be used depending on the restoration type and the performance metric being quantified, the choice of model often depends on the level of resolution needed and the associated data and resources available. Often there are too many uncertainties associated with model inputs and simulated processes to justify complex mechanistic models. However, they also have their place and can prove to be quite useful and informative. Table 6-5 summarizes some of the commonly used, publicly available deterministic models for assessing stream restoration projects. This list is not exhaustive. Many other models may also prove useful or more appropriate for a particular application. For a more comprehensive evaluation of models that could potentially be used to support stream restoration project evaluation and crediting refer to Fitzpatrick et al. (2001), Shoemaker et al. (2005), and Stein and Bledsoe (2013b).

Table 6-5. Models and Tools for Assessing Stream Restoration Projects.

Model/Tool	Potential Uses	Reference
HEC-RAS	Estimate water surface profiles, flood inundation areas, bank and channel shear stresses, and frequency of overbank flow when coupled with streamflow gage data or continuous hydrologic modeling. Practices: Bed and Bank Stabilization; Floodplain Reconnection	USACE, n.d.
Bank-Stability and Toe-Erosion Model (BSTEM)	Estimate factor of safety for multi-layer stream banks. Estimate erosion of bank and bank toe based on hydraulic shear stress. Practices: Bed and Bank Stabilization	USDA 2016.
Bank Assessment for Nonpoint Source Consequences of Sediment (BANCS)	Estimate sediment and nutrient load reductions using bank erodibility estimation tools: Bank Erosion Hazard Index (BEHI) and the Near Bank Stress (NBS) methods.	Rosgen 2001a
Channel Evolution and Pollutant Transport System (CONCEPTS)	Evaluate the long-term impact of rehabilitation measures to stabilize stream systems and reduce sediment yield. Practices: Bed and Bank Stabilization	USDA 2020
River Erosion Model (REM)	Quantify sediment and phosphorus loading from channel bed and bank erosion at watershed scales. Incorporates BSTEM and can simulate specific restoration practices. Practices: Bed and Bank Stabilization	Lammers 2018
One-Dimensional Transport with Inflow and Storage (OTIS)	Characterize solute transport including transient storage, sorption processes, and first order decay. Practices: In-stream Enhancement	USGS 2016
MODFLOW	Surface/groundwater interactions. Practices: In-stream Enhancement, Riparian Buffers	USGS 2021, Lautz and Siegel 2006.
SWAT+	Estimate upland pollutant loading and removal in riparian zones. Practices: Riparian Buffers	TAMU, n.d
Shade-a-lator	Quantify shade benefits and solar energy reduction from riparian buffers. Practices: Riparian Buffers	ODEQ, n.d.

6.5 Using the International Stormwater BMP Database Stream Restoration Module as a Monitoring/Reporting Guide

The International Stormwater BMP Database is a long-term project that has grown and evolved over the past 20 years to help document the performance of urban stormwater BMPs. The database has recently expanded to include two performance modules for agricultural practices and for stream restoration practices. With this expansion, the International Stormwater BMP Database has become an integrated repository of available data on the efficacy of non-point source BMPs from a variety of sectors for reducing pollutant loading and improving water quality. The new Stream Restoration Database can be used as a tool to help support stream restoration crediting programs by: 1) providing guidance on project characteristics that should be reported with stream restoration projects; 2) providing a project storage tool for crediting programs; and 3) eventually serving as a resource to support the population of crediting equations with reasonable geographically appropriate values. As the Database grows with additional studies and as researchers and program managers adopt its use, the utility of the Database to water quality crediting programs is expected to increase. Ultimately, the Stream Restoration Database is envisioned as a supporting tool to improve stream restoration designs and/or better target practices to achieve water quality and other restoration goals.

The overall objective of the database structure is to enable consistent reporting and compilation of critical aspects of stream restoration projects. The Stream Restoration Database follows a relational database model that organizes data into 12 tables of information, as illustrated in Figure 6-2. The database is designed so that both direct monitoring and functional assessment metrics can be reported at multiple points in time. A Stream Restoration Database User's Guide, including a list of the recommended reporting parameters, is accessible on the Stream Restoration Database website (WRF, n.d.). Brief descriptions of the content of these tables are also provided below. Clary et al. (2017b) provides a summary report of version 1.0 of the Stream Restoration Database.

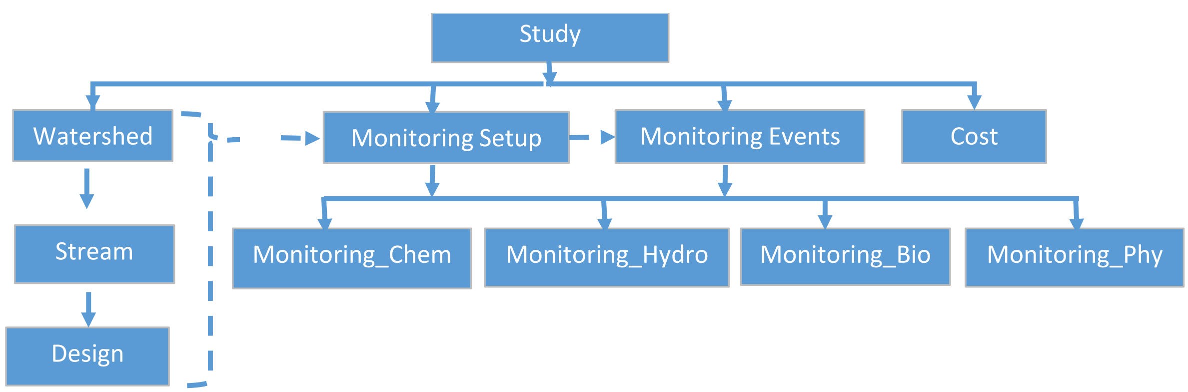

Figure 6-2. Generalized Stream Restoration Database Structure.
Source: Clary et al. 2017b.

6.5.1 Study

The purpose of the "Study" table is to provide basic descriptive information about the project purpose(s), geographical information associated with the project location, relevant reports or publications documenting project conditions, a summary of benefits documented by the researcher, available data types, and identification of documents provided to support the study information (e.g., reports, photos, cross-sections, channel profiles, sampling and analysis plan, calculations, model results, etc.). Information about the study design, duration and scale are also requested.

6.5.2 Watershed

The purpose of the “Watershed” table is to provide basic data about the overall watershed characteristics upstream of the restoration study that influence conditions in the stream. Examples of the types of information (metadata) requested include:

- Drainage area
- Geologic setting
- Upland and floodplain soils
- Watershed slope
- Controls on hydraulic regime
- Extent of watershed served by BMPs
- Imperviousness (total and hydraulically connected)
- Land use

6.5.3 Stream

The purpose of the “Stream” table is to document key physical characteristics of the stream and its valley. Most studies will include at least two records for stream characteristics. These are typically conditions before improvements were implemented (control) and conditions after improvements were implemented (treatment). Alternatively, spatially based “treatment” and “reference” stream conditions may be entered instead of a before-after approach. Multiple combinations of treatments, controls and reference streams are also accommodated. Some studies may also have information on stream conditions corresponding to several temporal phases: before, interim (during establishment of improvements) and established (or “after”). For some network-scale studies, it is also possible that multiple tributaries (individual stream entries) may be described in this table.

- Stream condition (e.g., impairment)
- Valley setting
- Channel geometry/condition (e.g., channel type, bedforms, planform, average bank height, channel width, depth and slope)
- Flow regime information
- Bed and bank materials (e.g., soil bulk density, gradation of materials, P and N content)
- Vegetation types

6.5.4 Design

The purpose of the “Design” table is to document key attributes of the stream restoration design associated with a “Stream” record. The intent of the table is to provide basic design information to enable comparison and identification of features influencing performance. For crediting programs that require that certain design standards be met, the Design table can be used as a tool to document these characteristics and ensure that the design criteria of the local entity were met. Examples of information requested include:

- General Information:
    - Stream restoration practice type
    - Channel length restored
    - Design approach/methods
    - Fluxes managed
    - Year completed
    - Riparian width restored
    - Goals/performance standards

- Description of measures implemented:
    - Bank stabilization measures
    - Bed stabilization measures
    - Riparian buffer reestablishment
    - Floodplain reconnection/reconfiguration
    - Habitat enhancement/features
    - Barrier removal/fish passage
    - Infrastructure protection measures

Information is also requested regarding operation, maintenance and replacement, expected establishment periods, expected stream response time, and expected design life.

6.5.5 Monitoring Approach and Data

The "Monitoring" series of tables includes information on the monitoring setup (related to experimental design), monitoring events and monitoring data. The monitoring setup and monitoring events tables are used to relate the monitoring data to locations and events. Events may include a time series of individual monitoring events or statistical summaries of annual or multi-year data. Examples of monitoring data allowed for each data category include:

- Chemical data: Chemical data can be provided for surface water, groundwater, soil, or other media. Results can be provided as concentrations, loads, or rates. Information is requested on sample type (e.g., composite), sample size, and laboratory methods.
- Hydrologic data: Hydrologic data can be provided for surface flow, groundwater, pore water, and precipitation. Examples of data types include water depth, volume, flow rate, velocity, shear stress, stream power, and others.
- Biological data: Biological data are focused on the fauna present in biological assessments such as biomass, index of biotic integrity (IBI), invertebrate community index (ICI), fish abundance, and others.
- Physical (habitat) characteristics: Physical data generally include habitat-related characteristics, including characteristics such as canopy coverage, stream substrate characterization, erosion rates, and other physical characteristics.

6.5.6 Contacts

Contact information for a stream restoration project is important to record in order to enable follow-up correspondence, accountability for long-term maintenance and even as a resource for contractors who have experience in a particular geographic area. Contact records can be added for the Owner, Sponsor, Researcher, Maintenance Contractor, Designer, Construction Contractor, Monitoring Entity, or other relevant contact.

6.5.7 Costs

The cost table accepts data related to both the stream restoration project and monitoring costs. The cost table is set up in a manner that allows a user to enter multiple records per study related to various costs. The type of cost being recorded is selected from a picklist, with an accompanying field for a narrative description to be provided by the data provider. The cost of stream restoration practices is a significant area of interest nationally, particularly as related to comparison of various improvements that could be implemented in a watershed to achieve water quality and aquatic life goals.

CHAPTER 7

General Considerations for Integrating Stream Restoration Practices into a Crediting Program

While there are multiple components, processes, and administrative requirements for a successful water quality crediting program, four elements that deserve supplemental consideration for stream restoration projects include the applicable credit area, credit multipliers, credit life, tracking and accounting, and banking.

7.1 Applicable Crediting Area

The crediting area is the geographic area within which credits must be generated and applied. This area is typically restricted to the same watershed, but the HUC scale used to define the watershed or additional restrictions on the location of the stream restoration project relative to the project being offset may vary depending upon permit conditions and crediting or trading program requirements. Regulators can determine this crediting area to address areas of environmental concern within their watersheds. For example, regulators can optimize environmental benefits by identifying key target or high priority locations for implementation of stream restoration projects in relation to certain discharge locations and/or pollutant loadings.

7.2 Credit Multipliers (Trading Ratios)

Credit multipliers (also called trading or mitigation ratios) are factors applied to pollutant reduction credits. These multipliers accelerate or reserve credit amounts for the projected net environmental benefits, account for pollutant-specific fate and transport processes within the watershed, and provide a margin of safety against risks and/or uncertainty in terms of measurement or scientific error associated with performance of the stream restoration project. In this guidance, recommendations for accounting for this final source of uncertainty directly via safety factors for each credit calculation are provided. Multipliers ensure that the environmental benefit of a credit-generating project is equivalent to or greater than the reduction that would occur if the buyer installed onsite treatment technology (EPA, 2009).

The minimum trading ratio is typically 1:1 (credits created to credits sold) and can be adjusted by one or more of the following common ratios used in water quality crediting (Sammans et al., 2015):

- **Uncertainty Ratio**: Accounts for potential inaccuracies in estimation methods and/or variability in project performance by reducing the estimated pollutant reduction or credit amount.
- **Reserve Ratio**: Accounts for unforeseen credit losses due to project failure by setting aside a portion of the estimated credits into a reserve pool as insurance. These reserved credits are never released.
- **Retirement Ratio**: Accelerates water quality improvements and environmental gains by setting aside a portion of credits for net environmental benefit and to serve as a hedge against potential environmental degradation.
- **Delivery, Attenuation, or Location Ratio**: Adjusts estimated loads to appropriately convey the impact of the estimated loads at the point of concern. For example, water quality benefits of a project will be higher in the local watershed than a distant receiving waterbody.
- **Equivalency Ratio**: Creates equivalency between different forms of the same pollutant or different types of pollutants that contribute to the environmental stress in multi-pollutant trading programs.

Because this crediting guidance is a framework document that may be adapted to meet a wide range of local and regional objectives as part of crediting programs, specific crediting ratios are not specified in this guidance. Nonetheless, Table 7-1 provides some examples of crediting ratios that have been used in stream restoration and wetland programs in various parts of the country. These trading ratios are often not developed based on rigorous scientific analysis and may be more representative of rule of thumb or best guess estimates. This is due to the difficulty in directly quantifying various sources of uncertainty, which often differ between regions and programs. A more robust approach to quantifying these trading ratios, accounting for the types of uncertainties listed above, is recommended. More rigorous approaches for determining trading ratios have been developed. For example, Zhang (2008) provides a methodology for quantifying an “equivalent trading ratio” incorporating uncertainty in estimated loading rates. Using this approach, trading ratios are higher when a greater portion of total load reduction comes from nonpoint sources due to greater uncertainty in predicting long-term performance of nonpoint BMPs. In essence, the ratio should be equitable, based on sound science and integrate local considerations pertinent to the watershed and receiving water. As suggested by Doyle and Shields (2012), these trading ratios, or the “release” of nutrient credits, can be modified based on monitoring data that shows achievement of project goals. The achievement of project goals may vary over time and, therefore, some programs require a recertification of credits. If a restoration project does not continue to provide the estimated water quality benefits, the credit may expire (see Section 7.3).

Table 7-1. Examples of Pollutant Trading Ratios from Existing Water Quality Trading Programs.

Program	**Non-Point Source to Point Source Pollutant Trading Ratio** (credits are for non-point source projects to off-set POTW/industrial point sources, unless otherwise noted)
Tar-Pamlico Nutrient Trading, North Carolina (NC DENR, 2010)	2.1:1
Great Miami River Watershed Trading Pilot, Ohio (Ohio EPA, 2013)	1:1–3:1 Depending on program-defined eligible buyer status and whether discharge is to an impaired water (require higher ratio).
Minnesota River Basin Trading (MPCA, 2009)	1.1:1 (existing facilities) 1.2:1 (new facilities)
Pennsylvania Nutrient Credit Trading	1.1:1
Chesapeake Bay Watershed Nutrient Credit Exchange	2:1
Piasa Creek Watershed Project (Great Rivers Land Trust, 2013)	2:1
Clean Water Services/Tualatin River, Oregon	2:1
Wisconsin DNR, 2013	2:1
Vermont Environmental Protection Rules (VTDEC 2005)	1:1 (Riparian Buffers); 3:1-10:1 (Channel or Infrastructure Modification)
Willamette Partnership Ecosystem Credit Accounting System (Willamette Partnership, 2013)	Seller ratio dependent on project location. Buyer ratio minimum 1.5:1 to cover risks, impact location.
Long Island Sound Nitrogen Credit Exchange Program (Connecticut Department of Environmental Protection, 2010)	Nitrogen Trade Ratios: Attenuation factors for individual tributary watersheds draining to the Long Island Sound can be developed using mathematical models and then be used to develop appropriate trading ratios. This effort may reduce the uncertainty ratio for a project.
Calculating Credits and Debits for Compensatory Mitigation in Wetlands of Western Washington (WDOE, 2012)	Wetland Mitigation: Considers the type of mitigation, the risk of failure, and the temporal loss of functions as factors in the calculations. Many examples are provided such as a 1.5:1 ratio, but examples are given where that ratio may change.

7.3 Credit Life

Credit life refers to the period of time in which the credits generated from a restoration project are considered valid. Per the Water Quality Trading Policy (EPA, 2003), "credits should be generated before or during the same period they are used to comply with a monthly, seasonal or annual limitation or requirement specified in their NPDES permit." Additionally, verification that the practice is functioning as intended is important. If a practice is no longer functioning as intended, then corrective measures to restore design objectives are needed; otherwise, the credit should be terminated.

Credit life should be determined based on a number of factors, including the temporal and spatial scale of the water quality issue being addressed (e.g., year-round issues or seasonal), the type of processes addressed by the restoration project, and the expected longevity of the applied restoration techniques. More rigorous monitoring demonstrating continued project function may be grounds for longer credit life (Doyle and Shields, 2012).

Examples of credit life vary throughout the country, with a few notable examples including:

- Crediting period or credit life has a duration tied to the impacts being mitigated, and credits must be regularly verified (Willamette Partnership, 2013).

- Credit expires if practice becomes less effective over time and is not maintained or replaced (EPA, 2009).
- Credit has a 12-month term and may be renewed for successive terms (Ohio River Valley, Electric Power Research Institute, 2012).
- Credit has a maximum life of 5-years, although it may be renewed indefinitely if monitoring demonstrates continued function (Chesapeake Bay, Schueler and Stack, 2014).

7.4 Tracking and Accounting

The tracking and accounting of credits, including when credits are active and retired, must be done systematically and transparently. Compliance demonstration for projects in water quality crediting programs involves providing documentation that enough credits have been obtained to offset established water quality mitigation requirements in accordance with the permit-specified or TMDL-specified unit times (e.g., seasonally, annually) and by the implementation schedule due dates. Program-specific legal and regulatory requirements must be clearly understood, and documented as part of the tracking and accounting system.

7.5 Banking

The term banking used here refers to the application of a water quality credit developed in one year to mitigate loading in a subsequent year. Excess credits may be generated if a restoration project is estimated to provide a larger benefit than needed to offset a discharger's current loading. This excess credit could then be applied in the future as long as the life of the credit has not been exceeded. Depending on the specifics of the crediting program, the bank may be administered internally by the individual discharger or by a third-party sponsor. If sponsored, the third party would be responsible for completing the stream restoration projects and dischargers would purchase offset credits directly from the bank. An issue with banking of stream restoration nutrient credits is that water quality concerns may be time dependent and a load reduction during one year may not have the same ecological benefit as a load reduction during a subsequent year. In cases where chronic nutrient loading is of greatest concern and temporal dynamics are less influential, credit banking may be appropriate. However, this determination is a programmatic decision that ideally is based on the dynamics of a specific system, the local water quality improvement goals, and the type and characteristics of dischargers needing credits.

CHAPTER 8

Conclusions and Recommendations for Additional Research

In this final section, general conclusions from this guidance as well as recommended areas of additional research are summarized.

8.1 General Conclusions

This crediting guidance provides a general framework for quantifying the water quality related benefits of a specific suite of stream restoration practices, focusing on sediment and nutrients. The four practices addressed in this guidance include bed and bank stabilization, riparian buffers, in-stream enhancement, and floodplain reconnection. Information in this guidance is appropriate for supporting the initial technical basis of water quality crediting programs for these practices as part of water quality trading programs. General conclusions and caveats that should be considered when incorporating stream restoration into these programs include:

- Stream restoration can provide nutrient removal benefits; however, the benefits are highly site-specific and variable, which leads to substantial uncertainty, especially with respect to denitrification processes.
- The empirical basis for stream restoration as a water quality BMP is improving, but additional research is needed, especially for regions and stream types that are poorly represented in the literature. Similarly, some practices have stronger empirical basis than others, and some practices have inherently higher functional capacity for nutrient removal than others. Currently, the relative magnitude of benefits is also more certain than the absolute magnitude of the benefits. To be scientifically defensible, stream restoration crediting schemes must necessarily acknowledge uncertainty through safety factors and allow updating of assumptions and methods as the empirical basis for quantification improves over time. The Stream Restoration Database will serve as an important repository for data on stream restoration effectiveness and encourage more targeted research in this area.
- Direct measurement of the water quality benefits of stream restoration is very challenging because of the large number of measured parameters, long time period for effects to be observed, and high levels of variability and statistical noise associated with environmental datasets. For this reason, monitoring approaches that incorporate surrogate (proxy) measures are an important aspect of evaluating the benefits of stream restoration practices. Functional assessment approaches developed in the wetland arena provide a logical framework and principles that are transferable to crediting programs for stream restoration. For example, wetland mitigation protocols distinguish among types and potential success of restoration and regional differences when quantifying credits. Developing rapid assessment indicators of stream restoration functions greatly simplifies monitoring and reduces costs.
- This crediting guidance focuses on the science supporting crediting for stream restoration projects; however, there are many policy decisions that must be made, which are not addressed in this report. Examples include trading ratios, incentives for project implementation, and methods for prioritizing watersheds and segments for projects that provide the greatest synergistic benefits. Additionally, credit values should consider stream context, type, and regional/watershed setting (i.e., classification/stratification is important).

- When communities consider implementation of crediting programs for stream restoration projects, the following guiding principles should be recognized:
  - Restoration should be targeted where it is most needed. A watershed approach (12- or 14-digit HUCs) should be applied and efforts prioritized to support broader water quality goals.
  - Restoration approaches must have a sound empirical basis and quantification of credits must be scientifically credible/defensible.
  - Restoration projects should consider and avoid adverse impacts on- and off-site (e.g., downstream sediment starvation, impeded aquatic organism passage, increased flooding risk upstream or downstream).
  - The most beneficial projects will recognize the multiple, synergistic benefits of stream restoration that go beyond pollutant removal and simultaneously provide other functions.
  - Ideally, restoration projects will improve watershed scale continuity and connectivity, restoring long segments and poorly functioning segments between functioning segments.
  - Understanding long-term channel evolution is important to recognize stream processes over time and consider stream evolution that would occur in the absence of intervention (~10-30 year time scale). Restoration efforts should be targeted to areas where the greatest pollutant loading is anticipated. Similarly, time lags in restoration of stream functions are also important to recognize.
  - Non-technical and regulatory requirements must be considered in restoration designs. Examples include regulations related to wetlands, floodplains, water quality, and threatened and endangered species. When evaluating the cost and feasibility of stream restoration projects, these regulatory requirements can be significant considerations affecting the feasibility of a project. Additionally, property ownership, access, stakeholder support, and adequate funding for capital improvements and long-term maintenance are important considerations, among others.
- For entities considering water quality crediting programs for stream restoration, performance assessment and accountability for the credit value over time is important. Evaluation of the performance of a restoration project is best accomplished based on a functional assessment approach.

8.2 Research Needs to Support Water Quality-Related Quantification of Stream Restoration Benefits

Although the empirical evidence of stream restoration's potential to increase nutrient processing and retention has increased recently, significant uncertainties remain that would benefit from additional research. Areas of future research for each restoration technique are described below:

- **Bed and Bank Stabilization**: The largest sources of uncertainty are variable bank phosphorus content and channel response potential (soil bulk density is another, smaller source of uncertainty). In addition, bank nitrogen concentrations are rarely quantified so it is difficult to assess bank erosion potential as a nitrogen source. Future research should examine both phosphorus and nitrogen concentrations in a variety of locations and soil types to provide more generalized information. Additionally, more effort is needed to develop simple yet robust methods and models for estimating channel response potential that have fewer data requirements than existing models (see Wood, 2020 for recommended improvements to the common BANCS method). An example of recent work to address this point is the River Erosion Model (Lammers, 2018).
- **Riparian Buffers**: The empirical basis for the benefits of riparian buffers is strongest for nitrate removal and removal of sediment and particulate phosphorus in surface runoff. However, research is needed to improve nutrient removal quantification methods for both surface and subsurface flow.

In addition, linkages between buffers and in-stream water quality deserve additional attention as recent work has shown little detectable in-stream improvement (Collins et al., 2013).

- **In-Stream Enhancement**: Empirical evidence for enhanced hyporheic nutrient processing is mostly at the scale of a single in-stream structure at a single baseflow discharge. Better understanding of the reach-scale influence of increased geomorphic complexity and flow variability is needed. In addition, the variability of organic carbon fluxes to a stream and its effects on nutrient dynamics deserves further attention.
- **Floodplain Reconnection**: Conceptually, the nutrient-related benefits of floodplain reconnection are clear. However, there have been only limited studies demonstrating significant nutrient retention in restored floodplains. Future research should focus on quantifying restoration effectiveness in a variety of geomorphic regions that differ in terms of frequency and seasonality of overbank flows, among other factors.
- **Other Restoration Practices**: There is a lack of information on how other restoration practices (e.g., dam removal or channel reconfiguration) affect nutrient and sediment processing and removal. Furthermore, more research into the cumulative and interactive effects of all restoration practices, in conjunction with more watershed-based approaches like stormwater controls and land use management, is essential for understanding these complex systems. Finally, long term monitoring is necessary to determine restoration performance over time, especially under changing land use.

In addition to these practice-specific research needs, there are also more general research needs, including:

- A general need for improving the empirical basis for these approaches across regions and stream types poorly represented in the literature (e.g., U.S. southwest, and northeast; ephemeral, intermittent, and braided channels) through 1) consistent data collection methods, 2) stratification by region and stream hydrogeomorphic type, and 3) better accounting for natural variability associated with seasonality, flow regimes, and extreme events. (These research needs are particularly relevant for floodplain reconnection and hyporheic exchange.) Collected empirical data should also be used to help improve modeling of the impact of stream restoration on nutrient dynamics, as this may be necessary approach for projects with limited monitoring.
- There is a need for regional functional assessment procedures for streams to show implementation of a functional design, functions persist over time, and to provide a benchmark for assigning credits. In addition, development of rapid assessment indicators deserves further attention. Further functional assessment research needs are outlined in Bledsoe et al. (2019).
- There is a need to combine both physically based models and statistical approaches into probabilistic and Bayesian network models that facilitate explicit quantification of uncertainty (Stein and Bledsoe 2013b), similar to previous applications for biological integrity of streams (Kashuba et al., 2012)
- Future research focused on techniques for accelerating the establishment of denitrification functions is also needed. For example, additions of carbon sources such as sawdust to streambanks, riparian zones, or in-stream features may have the potential to help "jump start" biogeochemical cycling and reduce time lags in functional performance. However, the feasibility, effectiveness, and sustainability of such techniques have not been well documented to date.
- Streams have some upper limit on nutrient removal and retention, limiting their ability to serve as nutrient sinks. While nutrient saturation in stream systems has received some attention (e.g., Earl et al., 2006), it is important to quantify this saturation point for various stream types and regions as this would serve as an upper limit on available nutrient credits for stream restoration projects.

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